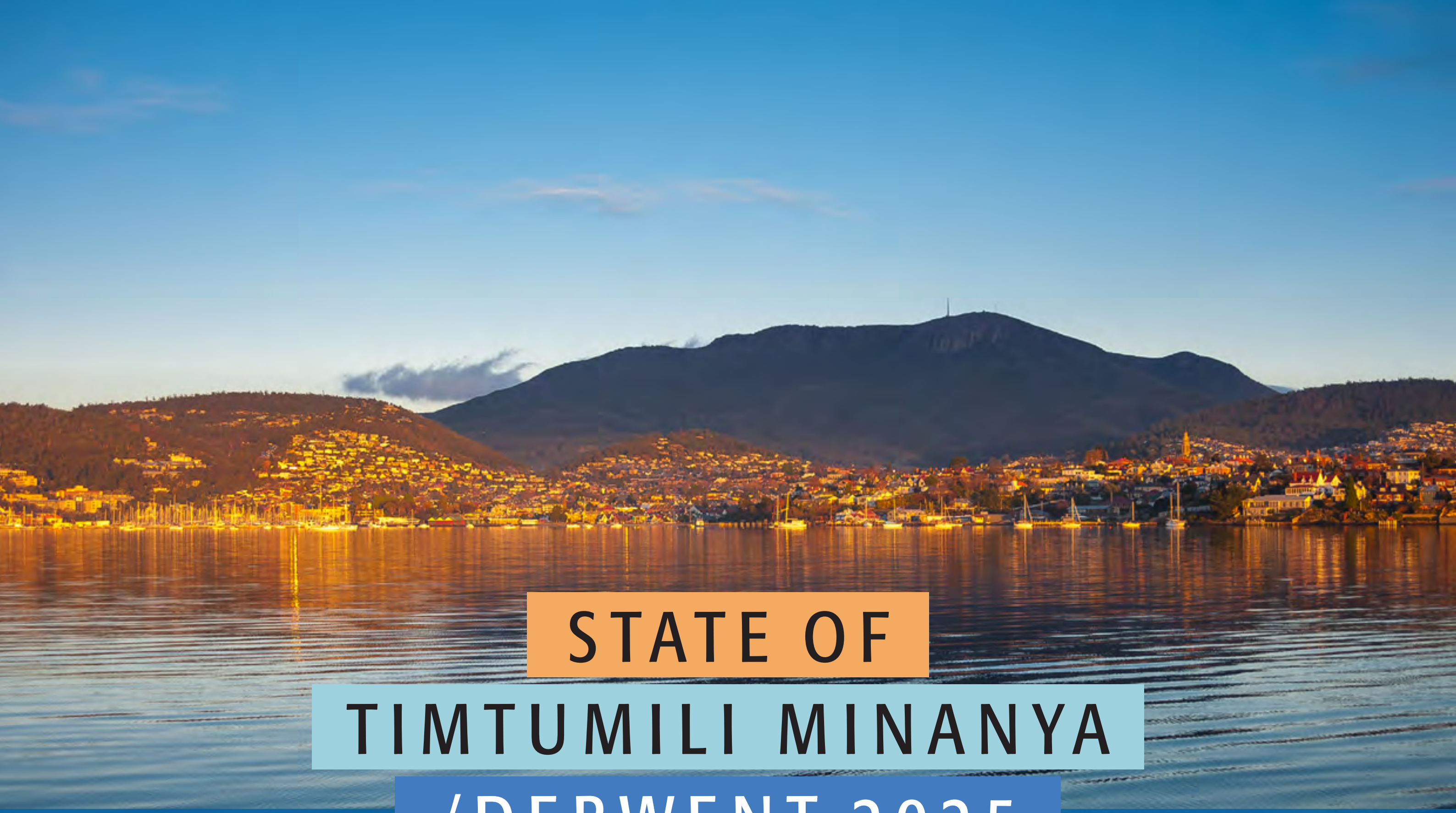

STATE OF TIMTUMILI MINANYA /DERWENT 2025

An update and review of environmental data and activities

Derwent Estuary Program

STATE OF TIMTUMILI MINANYA /DERWENT 2025

An update and review of environmental data and activities

Derwent Estuary Program

Left: Jane Bamford creating ceramic Little Penguin (*Eudyptula minor*) burrows in studio for City of Hobart colonies. Image: Peter Whyte.

The Derwent Estuary Program pays our respect to the traditional and original owners of Lutruwita's land, sea, waterways and sky Country, the Tasmanian Aboriginal people, and to those who have passed before us. We respect all Tasmanian Aboriginal people, their culture and their rights as the first peoples of Lutruwita. We recognise and value Aboriginal histories, knowledge and lived experiences and commit to being culturally inclusive and respectful in our working relationships with all Aboriginal people.

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The Derwent Estuary Program

The Derwent Estuary Program (DEP) is a not-for-profit regional partnership between local governments, the Tasmanian State Government, businesses, scientists, and community-based groups to work together to understand the Derwent Estuary, use science to enhance and protect its values and inform and involve the whole community. The DEP was established in 1999 and has been nationally recognised for excellence in coordinating initiatives to reduce water pollution, conserve habitats and species, monitor river health and promote greater use and enjoyment of the foreshore.

Our partners include Brighton, Clarence, Derwent Valley, Glenorchy, Hobart and Kingborough councils, the Tasmanian State Government, TasWater, Tasmanian Ports Corporation, Boyer Paper Mill, Nyrstar Hobart Smelter, Hydro Tasmania, NRM South and EPA Tasmania.

All photos are by DEP, except where acknowledged.

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Our partners

Tasmanian Government

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- Tasmanian State Government (including the Department of Natural Resources and Environment and the Department of Health)
- Brighton Council
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- Kingborough Council
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- Tasmanian Ports Corporation
- Hydro Tasmania
- NRM South
- EPA Tasmania

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- CSIRO
- La Trobe University
- University of Tasmania, including the Institute for Marine and Antarctic Studies and Centre for Marine Socioecology
- California State University
- Landcare Tasmania
- Australasian Wader and Seabird Study Groups

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Many thanks also to Edward Butler for scientific review and editorial assistance and Kelly Eijdenberg for report design and layout.

Executive summary

About the Derwent Estuary Program

The Derwent Estuary Program (DEP), established in 1999, is a not-for-profit partnership between government, industry, scientists, and the community to be the voice of the Derwent. By working together, we aim to understand the estuary, use science to enhance and protect its values and inform and involve the whole community. DEP coordinates monitoring, supports scientific investigations, and shares findings through regular reports. This 2025 State of Timtumili Minanya/Derwent update summarises estuary health, environmental trends, and recent management actions.

Timtumili Minanya/Derwent

Timtumili Minayana (River Derwent including the Estuary) is Tasmania's largest estuary, spanning nearly 200 km^2 from New Norfolk to the estuary's mouth between Tinderbox and the Iron Pot light. It is deep, stratified upstream (with fresh water at the surface and more saline water below) and well-mixed downstream. It has small tides (approximately 1 m) with an average flushing time of 12 days. The estuary is central to Hobart's identity – ecologically, culturally, and economically – supporting wetlands, seagrass beds, tidal flats, rocky reefs, and species such as the endangered Spotted Handfish. Around 255,000 people – about 40% of Tasmania's population – live along its shores, using it for recreation, boating, fishing, and transport. It also hosts major industries and is Tasmania's fourth busiest port.

Key Environmental Issues that affect the estuary include:

- Metal and metalloid contamination (mercury, zinc, cadmium, lead, copper, arsenic)
- Elevated nutrients, algal growth in localised areas and seasonal low dissolved oxygen in the upper estuary
- Habitat loss and species decline
- Altered flows and fish migration barriers
- Intermittent faecal contamination of recreational waters
- Marine pests and coastal weeds
- Climate change

There are multiple sources of contamination to the estuary with pollutants entering this waterway via:

- **Point sources:** sewage treatment plants, Boyer paper mill, Nyrstar zinc smelter
- **Diffuse sources:** Urban runoff, river inputs, agriculture, aquaculture, contaminated sites, atmospheric deposition, shipping and port activities
- **Sediment remobilisation:** Under certain conditions such as low dissolved oxygen, legacy pollutants can re-enter the water column.

Contaminants associated with these sources include pathogens, nutrients, organic matter, litter, metals, hydrocarbons, and other toxicants.

Catchment (Chapter 3)

The Derwent Estuary catchment covers around one-fifth of Tasmania, with land use dominated by conservation reserves and forestry in the west, and agriculture concentrated in the Jordan and Clyde sub-catchments to the east. Tasmanian rivers are naturally oligotrophic (low in nutrients) and highly sensitive to enrichment. River flow regimes play a major role in nutrient transport, with both seasonal variability and extreme events influencing downstream loads. Monitoring since 2015 indicates declining total nitrogen and phosphorus loads in several rivers; however, dissolved nutrient concentrations continue to exceed EPA guideline values in parts of the catchment, notably the Clyde, Florentine, Tyenna, Ouse, and Derwent. To safeguard the estuary, management priorities include establishing defensible nutrient budgets and caps, setting progressive reduction targets, and maintaining real-time monitoring to capture short-term variability and guide adaptive management.

Pollution sources and contaminant loads (Chapter 4)

Contaminant loads indicate the River Derwent continues to be the primary source of total nitrogen and total suspended solids to the estuary, while sewage treatment plants contribute the majority of dissolved inorganic nitrogen and total phosphorus. Loads were highest in the middle-to-lower estuary reflecting the concentration of point sources. Although annual loads varied significantly, elevated years such as 2017 and 2021 were driven by high rainfall and associated increases in effluent discharge. Groundwater flows from the Nyrstar site remain the largest source of zinc.

Industry actions have delivered progress but also highlighted challenges. TasWater's project to decommission Macquarie Point sewage treatment plant (STP) and upgrade the Selfs Point STP is expected to significantly reduce nutrient loads by up to 58% for total nitrogen and 42% for total phosphorus. Dissolved nutrient loads from Boyer have increased in recent years, prompting installation of in-situ analysers to improve process control and initial results are positive. Nyrstar have significantly increased zinc recovery since commissioning a grout curtain extraction system in 2021, reducing metal inputs to the estuary. However, mercury levels—though low—have risen slightly. Continued monitoring and process optimisation remain priorities.

Water Quality

Stormwater (Chapter 5)

Stormwater runoff from urban and suburban catchments remains a major source of pollution to the Derwent Estuary, transporting sediment, nutrients, pathogens, and litter. The Derwent Estuary Program (DEP) is addressing this challenge through initiatives such as monitoring, erosion and sediment control training, faecal source tracking using emerging technologies, and collaborative clean-up efforts. Supported by the Australian Government's Urban Rivers and Catchment Program, the DEP conducted extensive monitoring across 35 sites between June 2024 and May 2025. Results revealed elevated pollutant levels in urban catchments, particularly in areas undergoing development.

Estuarine water quality (Chapter 6)

The Ambient Water Quality program tracks spatial and temporal trends in estuarine water quality to assess broad scale changes, including potential climate impacts, and interactions between contaminant sources and their effects on the water column. Key environmental drivers—salinity, temperature, dissolved oxygen—showed strong seasonal variability but no significant long-term trends. However, pH decreased in the lower estuary and increased in the middle estuary, suggesting emerging climate-related influences and other pressures that warrant further investigation. Low dissolved oxygen (DO) conditions during summer and early autumn persist in the upper estuary, and DEP continued to investigate DO dynamics through observations of flow, season and tide on DO in bottom waters, as well as a new study to characterise sources of dissolved organic matter in the upper estuary.

Encouragingly, water clarity indicators (Secchi depth, turbidity, and total suspended solids) improved in the lower and middle estuary, supported by ecological surveys showing an expansion of canopy-forming kelp on rocky reef habitats. Water quality also improved in Ralphs Bay, likely due to reduced contaminant loading from Rokeby STP and the now-sewered Lauderdale catchment however, further investigation is needed to understand the role of microphytobenthos (e.g. diatoms that live on the surface of sediments) in nutrient cycling.

Over the monitoring period, nutrient trends varied across the estuary. Total nitrogen increased in both surface and bottom waters of the upper estuary, while total phosphorus declined in bottom and surface waters of the lower estuary and Ralphs Bay, and in bottom waters of the middle estuary. Dissolved nutrients remained elevated in surface waters of the middle estuary and bays, and in bottom waters of the upper estuary. Chlorophyll a showed no zone-wide trends, though it declined significantly at Prince of Wales Bay. Median chlorophyll a concentrations were typically low, indicating the estuary remains mesotrophic (an intermediate level of nutrients). Reduced nutrient loading likely contributed to improvements in water quality, particularly reductions in total ammoniacal nitrogen and phosphorus at Ralphs Bay.

Metal concentrations also showed positive trends. Zinc distribution reflected source inputs and estuarine hydrodynamics, with highest concentrations in New Town Bay and the middle estuary near the Nyrstar zinc smelter. However, zinc declined at 11 sites, signalling ongoing improvement in water quality. These gains are likely due to proactive site management at Nyrstar Hobart, including remediation and treatment of stormwater and groundwater.

Recreational water quality (Chapter 7)

The DEP coordinates a recreational water quality monitoring program at 38 sites—19 swimming and 19 environmental—using enterococci as the key faecal indicator. Water quality varied seasonally, with rainfall strongly influencing contamination levels. While most swimming sites maintained good or fair ratings, a few recorded poor ratings, prompting targeted investigations and infrastructure upgrades. Environmental sites, often located in degraded bays, consistently showed poorer water quality. The addition of daily water quality forecasting and *Bacteroides dorei* testing supports councils respond to human faecal contamination in catchments and keeps the public informed of the status of water quality at Derwent swimming sites.

Metals in fish, shellfish and biota (Chapter 8)

Deployed oysters, wild mussels, and flathead are monitored as bioindicators to evaluate human health risk and track changes in metal bioavailability over time. Elevated concentrations of metals in biota compared to the control site indicate that legacy contamination continues to pose a risk to seafood safety in the Derwent Estuary. Spatial patterns in shellfish (oyster and mussels) were consistent, with concentrations highest near Nyrstar. Elevated metal concentrations in Ralphs Bay, suggest environmental drivers influencing bioavailability. Temporal trends show limited improvement despite reductions in metal inputs over recent decades, underscoring the lasting influence of historical contamination on the food web.

Habitats (Chapter 9)

The assessment of changes in extent – especially for ephemeral habitats like seagrass is limited as in the last five years there has not been any new habitat mapping. However, a recent rocky reef survey shows improved habitat condition in the middle estuary compared with ten years ago. Tube worm matting has declined, replaced by Golden Kelp – indicating better water clarity and environmental health but declines in Bastard Trumpeter and increases in invasive seastars in modified areas were noted. Saltmarsh is a priority habitat and during this reporting period the focus has been on trialling restoration at Windermere Bay, in the middle estuary.

Fauna (Chapter 10)

The estuary supports diverse resident and visiting species of fauna. Monitoring shows stable populations of Spotted Handfish, Little Penguins, waterbirds, and gulls – a positive result amid global declines. Australian Pied Oystercatcher numbers have increased four-fold since 1973, demonstrating benefits of environmental improvements for breeding shorebirds. Migratory species trends reflect global patterns: Humpback Whale sightings have increased, while Southern Right Whales are low in numbers. Declines in migratory shorebirds are linked to habitat loss along the East Asian–Australasian Flyway, underscoring the need to protect critical habitats locally. Routine monitoring remains essential for accurate population assessments and timely management interventions to care for the Derwent's fauna.

Recent and ongoing management

The Derwent Estuary continues to face pressures from nutrient enrichment, metal contamination, habitat loss, and climate-related stressors. Significant progress has been made through collaborative management initiatives, yet ongoing actions are essential to safeguard ecological health and community values. Major initiatives implemented by councils and industry to improve water quality in the Derwent since 2020 include:

- A deep-water outfall and diffuser were installed at Turriff Lodge sewage treatment plant to improve effluent dilution and mixing.
- Nyrstar installed a grout curtain and groundwater extraction system in 2021 and upgraded their stormwater treatment capacity to further prevent contamination reaching the estuary.
- Volunteer programs for Northern Pacific Seastar removal are focused on reducing seastars from areas to support Spotted Handfish breeding.
- A Weed Action Fund grant for karamu control works in 2023–24 enabled the Derwent Catchment Project to access steep riverbanks and treat new infestations exposed by floods.
- Ralphs Bay seagrass restoration trials were initiated by the Institute for Marine and Antarctic Studies in 2024 and investigated viability of transplant and seed-based techniques.
- Storm Bay Modelling and Information System developed by the CSIRO has characterised the primary sources of nutrients into Storm Bay from ocean currents, sediment resuspension, river inputs and delivered a validated model of water quality in Storm Bay.
- The DEP is collaborating with team members in a National Environmental Science Program Climate Systems Project led out of the Institute for Marine and Antarctic Studies, in Hobart to establish baseline understanding and estimates of marine heat waves as an overall hazard in the Derwent Estuary, D'Entrecasteaux Channel, and Huon Estuary system.
- Environmental DNA for biomonitoring conducted by the CSIRO, combined with current regular monitoring programs was used to produce a time-series dataset of microbial, planktonic and eukaryote biodiversity, including an analysis of antimicrobial resistance.

Major DEP initiatives since 2020 have included the preparation and endorsement of the 2025–2030 Strategic Plan. Other key projects include:

- Initiatives funded by the Australian Government's Urban Rivers and Catchment Program including an urban saltmarsh restoration project at Windermere Bay which provides a model for future urban habitat restoration to enhance coastal habitat, stormwater rivulet monitoring, Spotted Handfish survey and habitat installation and a data visualisation tool.
- Production of sediment and erosion control guidelines in cooperation with the Tamar Estuary Esk Rivers Program to support developers and councils in preventing soil erosion from construction sites.
- Trial of real-time water quality analysers in the River Derwent catchment.
- Installation of a flow gauge on the River Derwent above New Norfolk.
- Routine estuarine and recreational monitoring continued including the introduction of a daily beach watch water quality forecast.
- Source tracking of stormwater catchments for nutrients and pathogens.
- Monitoring and management of important species including Little Penguins, Spotted Handfish and water birds including shorebirds, ducks and Black Swans.
- Monitoring of key habitats including rocky reefs, seagrass and saltmarsh.
- Ferry wake monitoring was trialled to detect physical impacts of boat wake waves on the shorelines of the Derwent Estuary.
- Continued monitoring and control of the invasive weeds including karamu and rice grass.
- Initiatives to better understand community values, raise awareness and increase enjoyment of the estuary (community surveys, regular communication and update of the Greater Hobart Trails website).
- Support of University students and researchers.

Many of these initiatives were implemented with support from Australian, State Government and philanthropic grants, and in collaboration with our partners.

Declines in migratory shorebirds are linked to habitat loss along the East Asian–Australasian Flyway, underscoring the need to protect critical habitats locally. Routine monitoring remains essential for accurate population assessments and timely management interventions to care for the Derwent's fauna.

Contents

1

Setting the scene

1 Setting the scene

The Derwent Estuary lies at the heart of the Hobart metropolitan area and is an asset of great natural beauty and diversity (Figure 1.1). It is an integral part of Tasmania's cultural, economic and natural heritage. The estuary is an important and productive ecosystem and was once a major breeding ground for the southern right whale. Areas of wetlands, seagrasses, tidal flats and rocky reefs support many species, including Black Swan, wading birds, Little Penguin, dolphins, platypus and Weedy Seadragon, as well as the critically endangered Spotted Handfish.

Approximately 255,680 people – 40 % of Tasmania's population – live around the estuary's margins (ABS, 2019, 2024). The Derwent is widely used for recreation, boating, fishing and marine transportation, and is internationally known as the home of MONA (the Museum of Old and New Art), the finish-line for the Sydney–Hobart Yacht Race and a gateway to the Antarctic. The Derwent Estuary supports several large industries, including paper and zinc production, boatbuilding and chocolate manufacturing. Upstream, the Derwent supplies approximately 60% of Hobart's drinking water, supports irrigation in southern Tasmania, including the new Greater South East Irrigation Scheme and is an important source of hydro-electric power.

Many pressures affect the Derwent Estuary, in particular:

- Metal contamination of water, sediments and seafood.
- Loss and pollution of estuarine habitats and species through urbanisation.
- Introduced marine pests and weeds.
- Altered river flow regimes and blocked fish migration routes.
- Elevated levels of nutrients and low dissolved oxygen levels in localised areas.
- Climate change.

1.1 Derwent Estuary Program

The Derwent Estuary Program (DEP) is a regional not-for-profit partnership between the Tasmanian Government, local governments, industry, scientists and the community. The DEP shares science to enable decisions for sound environmental management, and to guide the public in their choices around recreation and community activities. The DEP was established in 1999 and has been nationally recognised for excellence in reducing water pollution, conserving habitats and species, monitoring river health and promoting greater use and enjoyment of the foreshore (https://www.derwentestuary.org.au/).

In 2017, the DEP registered as a not-for-profit company limited by guarantee. Guiding the DEP's project priorities is the Strategic Plan 2025–2030.

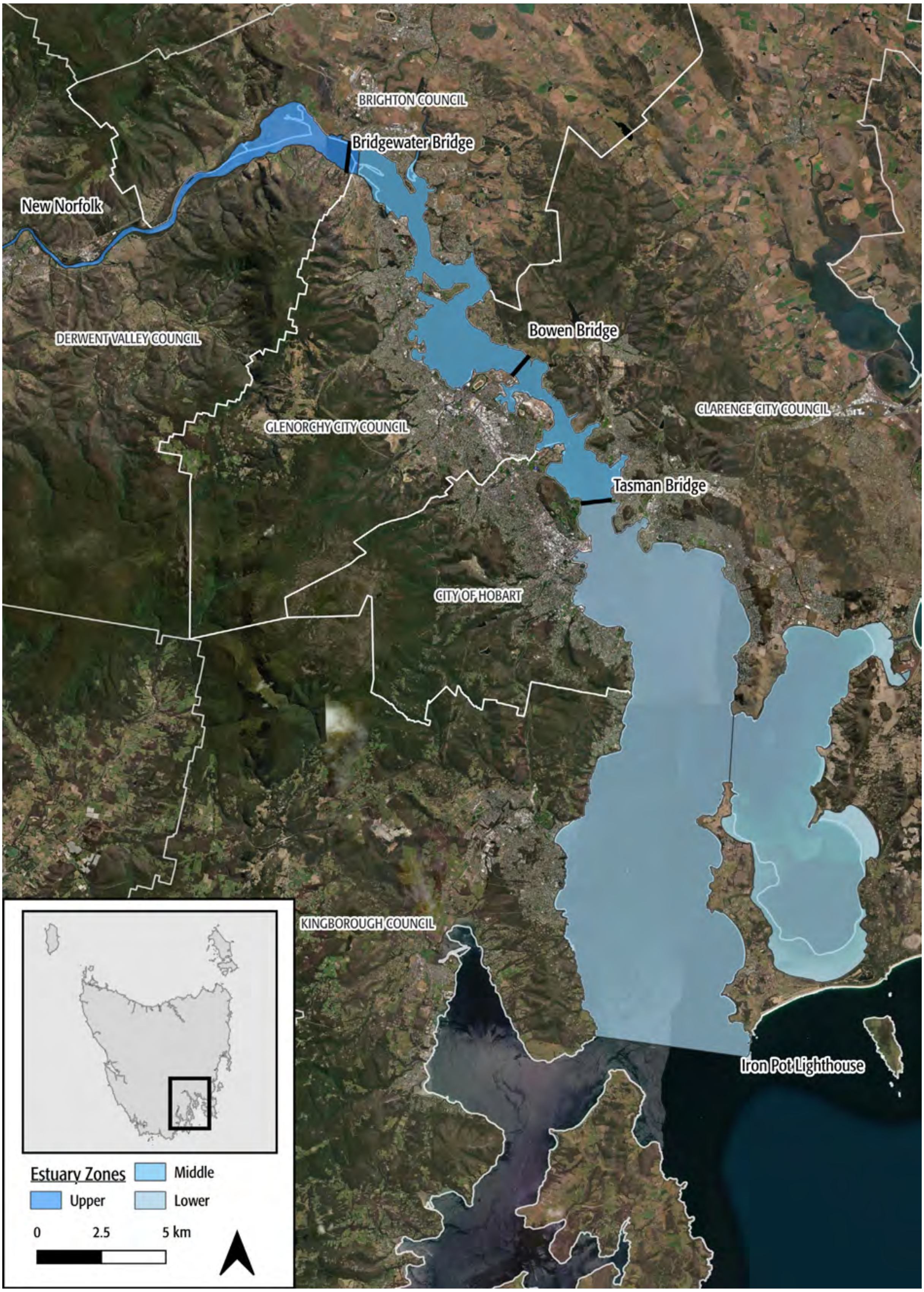
During the period from 2020 to 2025, the DEP's partners have included:

- Tasmanian State Government
- Brighton Council
- Clarence City Council
- Derwent Valley Council
- Glenorchy City Council
- City of Hobart
- Kingborough Council
- TasWater
- Boyer Paper Mill
- Nyrstar Hobart
- Tasmanian Ports Corporation
- Hydro Tasmania
- Institute for Marine and Antarctic Studies/University of Tasmania
- NRM South
- EPA Tasmania

The DEP is underpinned by a comprehensive integrated monitoring program that documents environmental conditions and trends, and that supports scientific research into key issues, such as metals and nutrient processing. Cooperative monitoring arrangements with the State Government, industries, local governments and the scientific community have generated a wealth of new information on water and sediment quality, seafood safety and estuarine habitats and species, which have been analysed and interpreted in this new report.

The DEP still uses the Derwent Estuary Conservation Action Plan (CAP), which was developed in 2013, to guide and prioritise activities to protect and improve key habitats and species (DEP, 2012). The CAP was developed using a framework devised by the Nature Conservancy in the US (DEP, 2015a). The DEP has, and continues to implement recommendations in the CAP, including saltmarsh monitoring and restoration; foreshore weed surveys; monitoring and management of Little Penguins and Spotted Handfish; seagrass surveys in the upper estuary and seagrass restoration trials.

Figure 1.1 Map of the Timtumili Minanya/River Derwent (including the Estuary). Source ESRI Imagery – https://server.arcgisonline.com/ArcGIS/rest/services/World_Imagery/MapServer/tile/{z}/{y}/{x}



1.2 Derwent Estuary values

Values of the Derwent Estuary include intrinsic, cultural and historical values, natural values associated with land, water and biota and socio-economic values. The Derwent Estuary is widely used for a diverse range of commercial, industrial, social and recreational purposes. An important regional management goal is to maximise these benefits, while minimising potential environmental damage and conflicts between users.

1.2.1 Cultural values

Timtumili Minanya, the River Derwent (including the Estuary), is a significant cultural landscape for Tasmanian Aborigines. It has been a central living place and route between the coast and hinterland for around 40,000 years. Today the Palawa people continue to care for country to maintain and cultivate their cultural connections and practices.

The Oyster Bay Tribe, the Mumirimina people (pronounced mu mee ree mee nah) on the eastern shore and the South East Tribe, the Muwinina people (pronounced moo we nin ah) on the western shore inhabited the region surrounding the Derwent Estuary (TAC, 2025). Both tribes utilised the Derwent as a source of food, with shellfish, such as oysters and mussels, being a major part of their diet (Ryan, 1996). The Derwent Estuary shoreline contains a very high density of Aboriginal sites. These sites include shell middens, stone artefact scatters, rock shelters and quarries, which continue to be destroyed by modern development (Aboriginal Heritage Tasmania, 2025).

The Tasmanian Aboriginal Centre's (TAC) Language Program has undertaken both linguistics and historical research on the original languages of Tasmania to retrieve and revive Aboriginal language in Tasmania – *palawa kani* (TAC, 2025). The name of the River Derwent and Estuary in Palawa Kani, is Timtumili Minanya (Milligan, 1859), and in 2015 the TAC, who owns Risdon Cove, chose the name Piyura Kitina for this site, meaning "little native hen"(TAC, 2025). Previously, the Aboriginal names for the River Derwent came from G.A. Robinson's records from the 1830s, where he attempted to give his idea of their sound by dividing words into syllables. The names for the Derwent were noted as: TEETOOMELE MENENNYE, RAY. GHE.PY.ER.REN.NE and NIB.BER.LIN (Plomley, 1990).

At the DEP, we recognise the continued connection of Tasmanian Aboriginal people to the Derwent and are working to develop a better understanding of the cultural dimensions of the Derwent from their perspective. In participating in cultural awareness training, we are better placed to work with Tasmanian Aboriginal people to understand, and respect cultural values and offer opportunities for collaboration to support their aspirations for land and water management. Examples of this cooperation include; DEP staff supporting junior rangers to develop, design and undertake water quality assessments, and employment of Aboriginal businesses to develop culturally appropriate strategies, cultural landscape plans and interpretation of the Derwent's habitats and species. From our engagement we better appreciate why water quality is important for maintaining Aboriginal cultural values. This shared experience enables us to integrate cultural values into our planning and operations. This commitment is reflected in our values and outlined in our most recent Strategic Plan (2025-2030) that states: *We Care. The natural, cultural and community values are at the heart of everything we do.*

European Colonisation

In 1793, Captain Willaumez of the d'Entrecasteaux/ Kermadec expedition entered and surveyed the river, naming it 'Riviere du Nord'. One year later, Commodore Sir John Hayes of the East India Company explored the river further and renamed it Derwent, after the Derwent River in Cumberland, England (Land Tasmania, 2020). The name 'Derwent' is thought to be derived from the Celtic word for 'clear water' (Wiktionary, 2025), or a 'valley thick with oaks' (Wikipedia Foundation Inc., 2025).

Risdon Cove was selected as Tasmania's first European settlement in 1803. Due to unfavourable conditions, the settlement was moved to Sullivans Cove in 1804, where it grew into the City of Hobart. Some of the sites around the estuary with important European heritage values include, Sullivans Cove, Battery Point, Queens Domain, Royal Botanical Gardens, Government House, Mount Nelson Signal Station, Mulgrave and Alexandra batteries, Kangaroo Bluff Battery, the Shot Tower and Batchelors Grave Historic Sites, and the Iron Pot Lighthouse.

The Tasmanian Museum and Art Gallery have excellent resources about the story of Tasmanian Aboriginal people past and present, and the colonisation of Hobart. **https://www.tmag.tas.gov.au/programs_and_learning/learning_resources/online_resources**

1.2.2 Natural values

Estuaries are partially enclosed bodies of water formed where freshwater from rivers and streams flows into the ocean, mixing with seawater. These transitional areas between land and sea are typically protected from the full force of ocean waves, winds and storms by the promontories, islands, reefs and sandy spits that mark an estuary's seaward boundary. The sheltered, tidal waters of estuaries support unique communities of plants and animals, specially adapted for life at the margin of the sea. Estuarine environments are among the most productive on earth, producing more organic matter per year than equivalent areas of forest, grassland or agricultural land. The wetlands that fringe many estuaries also provide fundamental ecosystem services. Water draining from the catchment to the estuary carries sediments, nutrients and other pollutants. As this water flows through marshes and other wetlands, pollutants are filtered out creating cleaner and clearer water – a benefit to both people and marine life. Wetlands also act as natural buffers between the land and the sea, absorbing flood waters and dissipating storm surges.

Numerous habitat types are found in and around estuaries. In the Derwent, these include beaches and dunes, rocky foreshores, saltmarshes and wetlands, mud and sand flats, seagrass meadows, kelp forests, and rocky reefs. Details about these habitats are given in **Section 9**. Innumerable birds, mammals, fish, invertebrates and other animals depend on the estuarine habitats of the Derwent as places to live, feed and reproduce. The Derwent is particularly important for migratory birds, which rely on the estuary as a resting and feeding ground during their long journeys with details summarised in **Section 10**.

The estuary's natural values are closely integrated with the social fabric of the region. People are attracted to the region for many of the opportunities that the estuary offers, including aesthetics, recreational pursuits – such as water sports, yachting, fishing and bird watching – and simply being able to connect with the natural environment.

1.2.3 Community values

In 2024, the DEP surveyed the public (using an online platform) to understand their interest in and interaction with the Derwent Estuary (unpublished). 92 respondents answered questions about what they know about the Derwent Estuary, how they feel about it, what they would like to know and in what form they would prefer their information.

The goals of the survey were twofold: to inform the data visualisation project (funded by the Australian Government's Urban Rivers and Catchments Program), but also to inform DEP's communications messaging and practices.

In general, people shared that they would like access to four types of information:

- Governance – how DEP is managed, and what types of mechanisms are in place to manage industry including agriculture and aquaculture.
- Water quality – updates and improvements in water quality, and how it impacts recreational activities.
- Biodiversity, history and natural environment – what species, natural features and the history of the Derwent.
- Participation – how people can be involved, volunteer, stay updated and discover special features and insights.

For access to clear information around water quality, many respondents said they would like beach watch, and its water quality data, available all year round, and typically the format it is presented in – a snapshot on a map – is all that people require. Respondents also asked for assurances around DEP's independent and unbiased status. Of those who wanted to be provided with more information, many requested lists of actions that link any issues with what is being done to combat it. Most issues raised are not unknown to the DEP, but the request for information on Aboriginal history and background in the DEP was a new theme to community engagement surveys.

The Community Survey revealed that the Derwent Estuary and its health are very important to the majority of people living here. People use the estuary regularly for swimming (62%), boating/rowing/sailing (35%) and fishing (25%). Walking beside the estuary was by far the most popular activity (80%).

1.2.4 Volunteers and education

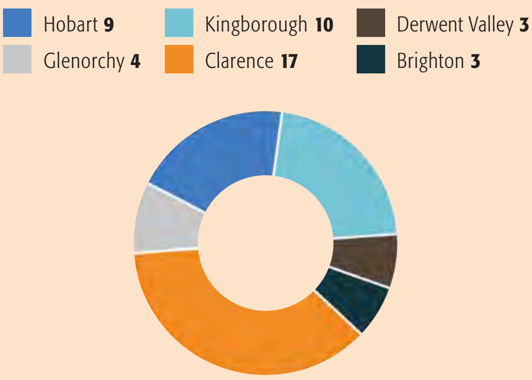
There are numerous Landcare and Coastcare groups that regularly remove weeds, revegetate degraded foreshores, collect litter or conduct wildlife surveys. Landcare Tasmania have provided an overview of this activity which is included in the Case Study: Landcare activity around the Derwent Estuary.

Landcare activity around the Derwent Estuary

Provided by Landcare Tasmania

A total of 46 Landcare Tasmania member groups (38 community groups and eight school associates) work along the Derwent Estuary or its rivulets and tributaries across six municipal areas (Derwent Valley, Brighton, Glenorchy, Clarence, Hobart and Kingborough) (Figure 1.2).

Figure 1.2 Landcare Tasmania members based along the Derwent Estuary by council municipality.



In 2024, a total of 1,617 volunteers contributed 13,678 hours of volunteer labour across 514 events, with a combined volunteer labour value of $642,318.88 (calculation based on The Centre for Volunteering, 2025).

Of the 46 members who volunteer along the Derwent, a total of 28 (61%) work directly in riparian, coastal or wetland environments. Eighteen (39%) members reported that they did not work directly with waterways around the Derwent Estuary (Figure 1.3). However, their work around the Derwent Estuary has indirect benefits on the overall health of the estuary.

In the Derwent Estuary area, the main activities conducted by members are weed control, planting and revegetation, with the focus for members being biodiversity (Figure 1.3).

Figure 1.3 Focus area of work by Landcare Tasmania members along the Derwent Estuary.

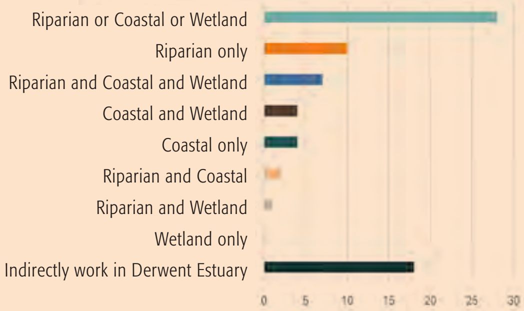


Figure 1.4 Kingston Beach Coastcare group members conducting weeding activities at Kingston Beach. Image: Landcare Tasmania.

In this reporting period the DEP supported tertiary students with research projects focused on the river and estuary. The benefit of this support is mutual. The students gain practical experience working with DEP staff and partner organisations and the DEP benefits from research that fills gaps in our understanding of estuarine science, restoration and economics. Here, we highlight the natural capital accounting PhD project.

This approach, the first of its kind for the Derwent Estuary is a first step in developing a comprehensive set of natural capital accounts to show changes in the health of our natural capital over time and be used to inform decisions on land use, human health, climate-change mitigation and adaptation. Natural capital accounts can also be used to inform economic and political decision-making, policies and business strategies, as well as supporting evidence-based investment, urban planning, rural development, and health and sustainability projects.

Student Case Study: Ecosystem Accounting in the Derwent Estuary

By Le Thuy Duong Ha, PhD Candidate, University of Tasmania (Ha DLT, 2025)

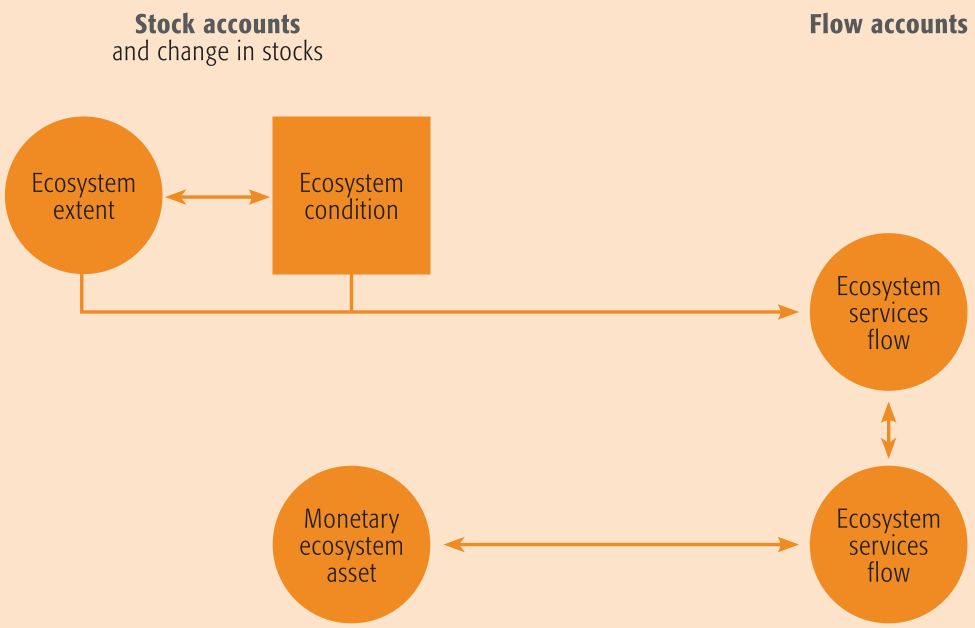
Natural Capital Accounting and the SEEA-EA framework

Natural Capital Accounting (NCA) is considered an umbrella term for all accounting methodologies and frameworks seeking to measure stocks of natural capital assets and flows of benefits (ecosystem services and abiotic flows) provided to the reporting entity and other users (Smith *et al*., 2023). NCA allows countries and organisations to integrate natural assets into their balance sheets, thereby strengthening accountability, enhancing reputation, and advancing sustainability goals (La Notte and Rhodes, 2020).

Ecosystem accounting, a specialised branch of NCA, treats ecosystems as environmental assets by measuring their extent, condition, and benefits (United Nations Statistics Division, 2024). This project adopts the System for Environmental-Economic Accounting – Ecosystem Accounting (SEEA-EA) framework, the globally recognised standard for ecosystem accounting, to guide analysis of ecosystems and their associated services in the Derwent Estuary.

The SEEA-EA framework, adopted by the United Nations in 2021, conceptualises how ecosystems (stocks) generate benefits for humans through ecosystem services (flows) and applies the exchange value principle to ensure consistency with national accounting standards (United Nations Statistics Division, 2024). The SEEA-EA provides a system of integrated, internally consistent accounts, as illustrated in Figure 1.5, comprising the following components: (1) Ecosystem extent accounts that record the total area of different ecosystem assets; (2) Ecosystem condition accounts that measure the biotic and abiotic characteristics of ecosystems; (3) Ecosystem services accounts in physical terms, which record the flows of supply and use of ecosystem services; (4) Ecosystem service accounts in monetary terms, which record the value of flows of supply and use of ecosystem services; and (5) Monetary ecosystem asset accounts, which record the monetary value of all ecosystem assets.

Figure 1.5 Connections between ecosystem accounts (United Nations Statistics Division, 2024) with details for the Derwent accounts provided in tables 1–6.



Natural Capital Accounting for the Derwent Estuary

The DEP has adopted NCA for two reasons. First, NCA provides a reporting language that is already familiar to many of our partners, particularly businesses and government agencies operating within environmental, social, and governance frameworks. Second, NCA offers a platform for communicating the value of our monitoring activities not only in scientific terms but also in economic terms, making this information more accessible and relevant to policymakers, industry, and community stakeholders. In this way, the adoption of NCA is intended to strengthen recognition of the value of the Derwent Estuary and to highlight the importance of ongoing ecosystem monitoring.

The following tables present the ecosystem accounts. Detailed explanations of their construction and accompanying narrative are provided in (Ha *et al*., 2025).

Table 1.1 Ecosystem extent accounts for the year 2007. Ecosystem extent data was derived from Seamap (Lucieer, Lawler, Morffew, *et al.*, 2007) and DEP (2009).

Ecosystem type	Extent (hectares)
Sand and silt	17060
Sand flat and beach	1140
Seagrass	680
Rocky reef	300
Saltmarsh	220
Wetland	130
Unvegetated mud flat	100
Rocky shorelines	90
Cobble reef	30
Kelp forest	30
Total	**19780**

Table 1.2 Ecosystem condition account for seagrass ecosystems in the Derwent Estuary from 2016 to 2019. Seagrass (macrophyte) condition monitoring methodologies and results are presented in **Section 9.5**. Seagrass condition metrics were derived by aggregating data and normalising to a range between 0 and 1.

SEEA ECT Class	Variables	Seagrass			
	Descriptor	Measurement unit	Opening value	Closing value	Change
Structural state	Seagrass cover	Score (0-1)	0.11	0.67	0.56
	Bare sediment cover	Score (0-1)	0.35	0.2	-0.15
	Algae cover	Score (0-1)	0.53	0.14	-0.39

Table 1.3 Condition account for rocky reef in the Derwent Estuary for the year 2010. Rocky reef monitoring methodologies and results are reported in (Barrett *et al.*, 2012a). Rocky reef condition metrics were derived by aggregating data for each zone (lower estuary and middle estuary) and normalising to a range between 0 and 1.

SEEA ECT Class	Variables	Rocky reef			
	Descriptor	Measurement unit	Middle Estuary	Lower Estuary	Total Accounting Area
Compositional state	Fish species abundance	Score (0-1)	0.38	0.17	0.28
	Fish species diversity	Score (0-1)	0.1	0.58	0.34
	Invertebrate & cryptic abundance	Score (0-1)	0.25	0.51	0.38
	Invertebrate & cryptic fish species diversity	Score (0-1)	0.16	0.54	0.35
	Algal species diversity	Score (0-1)	0.08	0.54	0.31

Table 1.4 Global climate regulation service physical and monetary accounts provided by seagrass from 2016 to 2019.

	Accounting entry	Units	Carbon sequestration	Carbon retention
Physical term	Opening stock	tonnes	811	183,740
	Addition to stock		187	42,401
	Reduction to stock			
	Net change in stock		187	42,401
	Closing stock		998	226,141
Monetary terms	Opening stock	A$	27,374	6,201,218
	Addition to stock		6,317	1,431,050
	Reduction to stock			
	Net change in stock		6,317	1,431,050
	Closing stock		33,691	7,632,269

Table 1.5 Recreational fishing service physical and monetary accounts in the Derwent estuary from 2013 to 2018.

	Accounting entry	Units	Recreational fishing
Physical term	Opening stock	Number of fishers	9,557
	Addition to stock		
	Reduction to stock		-2,596
	Net change in stock		-2,596
	Closing stock		6,961
Monetary terms	Opening stock	Fishers' expenditure (A$)	9,633,456
	Addition to stock		2,805,851
	Reduction to stock		
	Net change in stock		2,805,851
	Closing stock		12,439,307

Table 1.6 Fish nursery service physical and monetary accounts provided by seagrass from 2016 to 2019.

	Accounting entry	Units	Fish nursery
Physical term	Opening stock	Fish biomass enhancement (kg)	1,796,288
	Addition to stock		414,528
	Reduction to stock		
	Net change in stock		414,528
	Closing stock		2,210,816
Monetary terms	Opening stock	Value of fish biomass enhancement (A$)	9,403,992
	Addition to stock		2,170,152
	Reduction to stock		
	Net change in stock		2,170,152
	Closing stock		11,574,144

Discussion and conclusion

By implementing NCA for the Derwent Estuary, this project offers several insights. First, it demonstrated how existing monitoring data can be repurposed for NCA. For example, linking seagrass condition indicators with service flows (such as climate regulation and fish nursery functions) showed how improvements in condition can translate into both greater physical service delivery and higher estimated economic value. Second, the project highlighted how NCA can enhance understanding of the ecosystem services provided by the Derwent Estuary, thereby strengthening recognition of their contribution to environmental health and community wellbeing.

At the same time, the project identified some data gaps that limit the full application of NCA. Addressing these challenges requires: (1) more locally based ecological research to expand data availability; and (2) stronger collaboration with government, non-government, and research partners to improve data access.

Regular implementation of NCA encourages data exchange between providers and producers, improves data quality and reliability, and builds trust among stakeholders. Over time, this creates more accurate and accessible information, enabling policymakers to better integrate environmental considerations into development and economic planning.

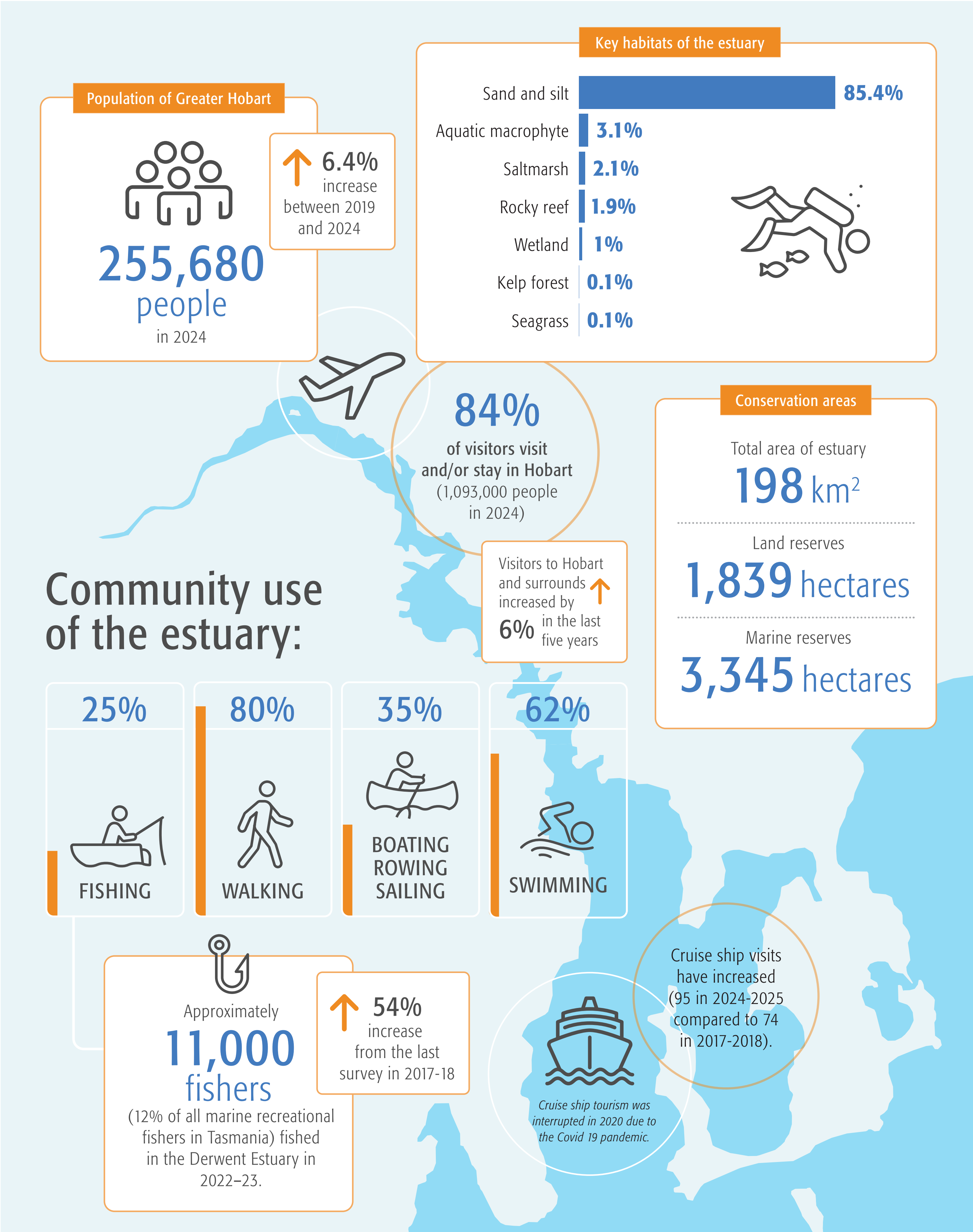
1.2.5 Access

As the condition of the estuary improves, the more interest there is in the estuary for recreation. Walking is by far the most popular way for people to engage with the estuary, and the Greater Hobart Trails website provides information and directions to over 100 tracks in and around Hobart (**https://www.greaterhobarttrails.com.au/**). This website was updated in 2023 with funding support from the Tasmanian Government's Healthy Tasmania grant program and it continues to be popular. The Greater Hobart Trails website is an initiative of the DEP in cooperation with DEP's six member councils, Parks and Wildlife Services and the Wellington Park Management Trust.

1.3 Derwent Estuary uses

The Derwent Estuary is surrounded by Tasmania's largest population centre, and the estuary is widely used for recreation both on and off the water. The estuary is also very much a working waterfront. The Derwent is Tasmania's fourth largest port and is an important regional centre for the shipping of goods. Antarctic support vessels, commercial fishing vessels, cruise ships and visiting military vessels use the Derwent. There are several major water-dependent industries situated on the foreshore, including the Boyer paper mill, the Nyrstar Hobart zinc smelter, Impact Fertilisers and Incat Catamarans, as well as a host of smaller commercial enterprises. The Derwent Estuary is an important tourism resource for Hobart, which is the most visited place in Tasmania. These various uses are indicated in Figure 1.6 and described in the sections following.

Figure 1.6 Features and uses of the Derwent Estuary.



1.4 Industry and commerce

Commercial and industrial access to the estuary and river were critical to the early economic development of the region for local transportation, shipping, water supply and wastewater discharge. This dependence has declined over the past 50 years as other forms of transport have predominated; however, a number of water-dependent commercial activities are still situated along the foreshore. These include:

- Prince of Wales Bay maritime industries precinct (construction, maintenance industries), including Incat, which relies on the estuary for construction and maintenance of vessels;
- Nyrstar Hobart Smelter, which relies on the estuary for shipping, water supply and wastewater discharge;
- Boyer paper mill, which relies on the estuary for water supply and wastewater discharge;
- Selfs Point fuel storage facilities, which rely on the estuary for shipping and refuelling of vessels;
- Cadbury Chocolate factory which relies on the estuary for water supply and wastewater discharge;
- Impact Fertilisers, which relies on the estuary for shipping;
- Domain slipway and other slipway facilities (boat maintenance and limited construction);
- Hobart docks / Tasmanian Ports Corporation (TasPorts) (commercial, tourism and research shipping); and
- Sullivans Cove (commercial fishing and tourism).

In addition to these major industries, there are numerous commercial facilities that support recreational and tourism needs, such as:

- marinas and yacht clubs
- restaurants and cafes
- ferry cruises, cycle, boat and kayak rentals.

1.5 Transportation

Shipping and other marine transportation operations on the Derwent are jointly managed by TasPorts and the Marine and Safety Authority of Tasmania (MAST). The Port of Hobart is Tasmania's southern-most port which supports a diverse range of operations including forestry, bulk minerals, fuel, research vessels, tourism, fishing and recreational activities.

The Derwent River has often been described as one of the world's best harbours, with few rocks, reefs or other hazards. The river has a stable and well-defined channel with a small tidal range and minor to moderate tidal currents. Furthermore, the Derwent has few sedimentation problems that impede navigation and has many safe anchorages with shelter from prevailing winds. As a result, the Derwent River rarely requires dredging to maintain shipping passages to the critical port facilities. Alongside this however, the River Derwent does possess a navigational challenge – the Tasman Bridge.

On January 10th, 1975, the bulk carrier Lake Illawarra collided with Hobart's Tasman Bridge, bringing three unsupported spans and a 127 m section of roadway crashing into the River Derwent, where they remain to this day. This event has gone on to shape many of TasPorts' critical pilotage and maritime practices today at the Port of Hobart.

During 2024–2025, 359 vessels visited TasPorts facilities at the Port of Hobart including bulk commodity, cargo, research, military and cruise vessels. More than 1.6 million tonnes of freight moved through the port, of which, more than 674,000 tonnes were exported and 974,000 tonnes were imported (TasPorts, 2024). Between October and April, Hobart welcomed 95 cruise ships carrying more than 160,000 passengers and 72,000 crew, providing a critical boost to tourism and the local economy. Alongside this, a number of international Antarctic programs visited the southern port for re-supply operations (pers. comm. T. Furlonge, TasPorts, May 2025).

Passenger ferries

Ferry transport has operated in various guises for more than a century in Hobart until the 1980's (Clements, 2017) and was re-established on the Derwent Estuary in August 2021, operating between Bellerive and Brooke Street Pier. The operation began as a one-year fare-free trial to establish demand for the service. The trial was a great success with over 110,000 passengers utilising the transport service in the first year of the trial, with an average of approximately 500 daily users. With the success of the first year, from August 2022 the Tasmanian Government introduced fares for the services equivalent to that of a Metro Tasmania bus fare, using the same Greencard system. The aim of the service is to provide an alternative mode of transportation from the eastern shore to the city centre, easing peak-hour congestion on the Tasman Bridge. As of August 2024, over 360,000 passengers have used the service across the estuary.

Boat wake can have a variety of impacts, including shoreline erosion and sediment resuspension that may affect plants and animals in the estuary and infrastructure located on the shore. The commencement of ferry operation between Bellerive and Hobart, provided an ideal opportunity for the DEP to initiate a program to assess monitoring methods of susceptible local shores from ferry wake. A project was proposed to investigate the usefulness of quick, efficient and low-cost methods, such as photo point monitoring, to track shoreline changes over time with funding support from the Department of State Growth (**Section 9.7.1**).

The lessons learnt, and observations made during this project will be useful when it comes to any expansion of the Derwent Estuary ferry network, as outlined in the River Derwent Ferry Service Masterplan (Tasmanian Government, 2023). It will assist in identifying shoreline types and substrates most prone to erosion owing to wave attack so that monitoring prior to launching additional ferries can be done. Management options for reducing foreshore impacts include ferry design and adjusting boat speed to reduce wave intensity.

1.6 Fishing

The Derwent Estuary supports an extensive recreational fishing industry throughout its length. In the 12 months prior to October 2023, an estimated 130,500 Tasmanian residents aged 5 years or older fished at least once, representing a 27% participation rate in recreational fishing (Tracey S. and Stark K.D., 2024). For recreational fishing, there is an emerging interest in Yellowtail Kingfish and Snapper. There are public health warnings stating that shellfish and Bream (due to its long lifespan) should not be consumed, and only limited consumption of other species because of high concentrations of zinc, cadmium and other metals.

Commercial fishing activity is limited in the Derwent, with the area upstream of Dennes Point to Cape Direction a no-rock lobster potting area, no-gillnetting area, no-setline area and Shark Nursery Area. Commercial fishers may only take scalefish with a special endorsement, with one Danish seine operator endorsed for a whiting "cod-end" net in the Derwent. An average of 37 tonnes of fish was taken for commercial purposes between 2019 to 2024, almost entirely composed of school whiting (Sharples R. *et al.*, 2024; pers. comm. NRE Tas 8 May 2025).

The Derwent is an important regional home port and unloading area for many fishing vessels, including those catching rock lobster, abalone and scalefish.

There are presently no shellfish or finfish farming operations in the Derwent, nor should shellfish collected from any part of the Derwent (including Ralphs Bay) be consumed because of high concentrations of zinc, cadmium and other metals (**Section 8**).

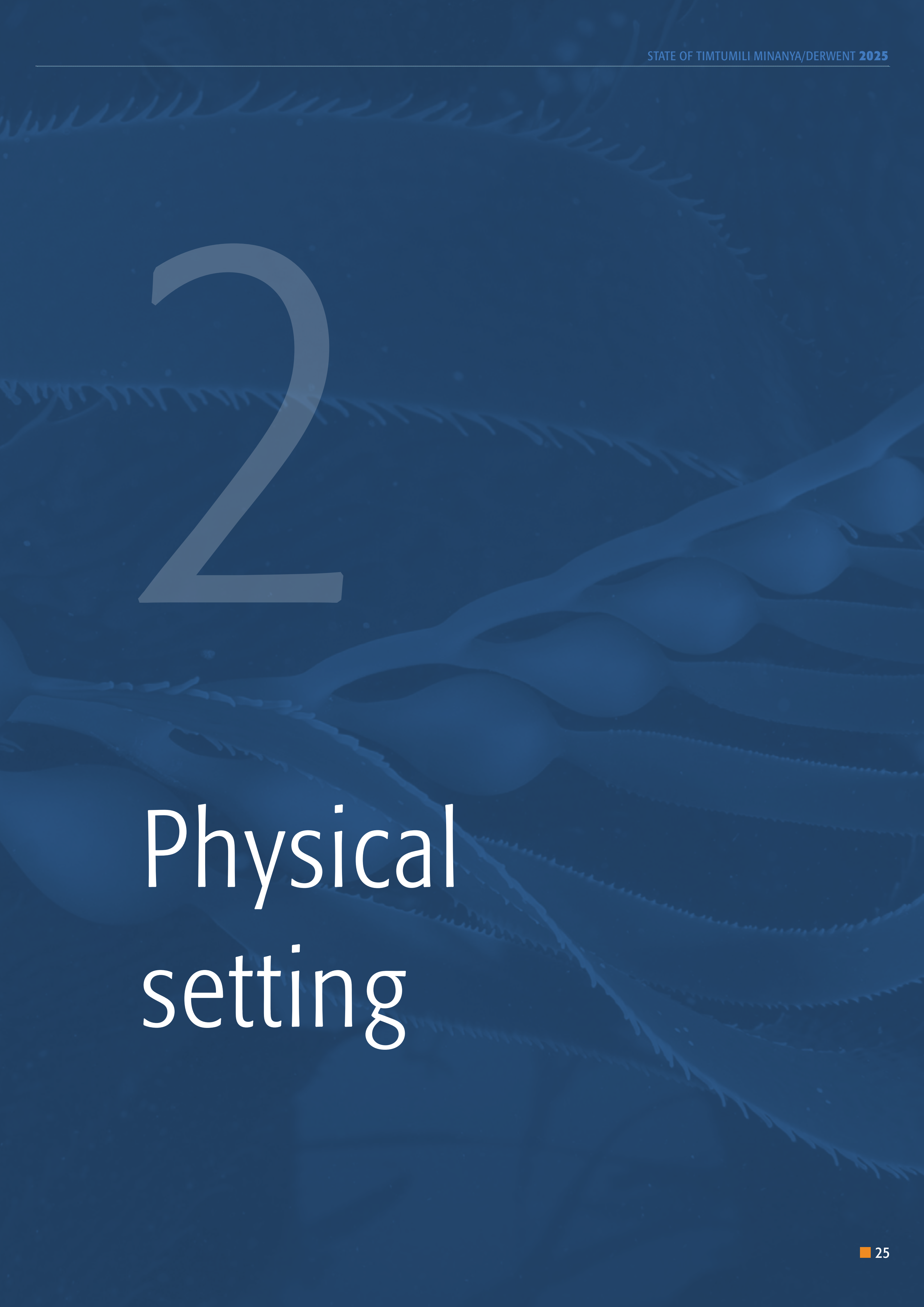
1.7 Research, Education and Antarctic gateway

Hobart is an important centre for research and education, particularly for marine and Antarctic studies. The following research and education centres are located in the area:

- CSIRO Marine Laboratories (Hobart)
- Institute for Marine and Antarctic Studies (Hobart and Taroona)
- Australian Antarctic Division (Kingston)

Several Antarctic icebreakers and other large research vessels are based in Hobart, including the Nuyina, L'Astrolabe and Investigator, and a number of other research vessels visit Hobart on a regular basis.

Antarctic tourism continues to be popular. During the Southern Hemisphere summer, a number of ships depart Hobart for Macquarie Island and the Antarctic continent, carrying scientists and tourists to visit and explore these relatively untouched wildernesses. Operators to Antarctica see Hobart as an important and attractive port, being close to the city and having well-developed infrastructure and suppliers.



2

Physical setting

2 Physical setting

2.1 Derwent Estuary

2.1.1 Estuarine dynamics and zonation

Estuaries represent a continuum of water chemistry from freshwater to saltwater. For the purposes of discussion, it is useful to separate the Derwent estuary into broad zones based on key water quality indicators and geography. The five estuarine zones – upper, middle, middle estuary bays, lower and Ralphs Bay – are each characterised by different physical, chemical and biological conditions. The allocation of ambient water quality monitoring sites to estuarine zones is as follows (Figure 6.1):

Surface waters of the upper estuary are dominated by freshwater inflow from the River Derwent and during periods of low river discharge, an underlying layer of saltwater extends along the bottom of the estuary, often beyond New Norfolk. This type of stratification with freshwater overlying a saltwater tongue is the definitive feature of salt wedge estuaries. Upper estuary sediment is dominated by cobblestones in the narrow upper reaches and quickly changes to silt in the broader expanses around the Motorboat Club between Boyer and Bridgewater. The middle estuary extends from Dogshear Point, southward to the Tasman Bridge. The mid-estuary is the most industrialised and urbanised section of the Derwent and is also occasionally stratified with fresher surface water and saltier bottom waters. Middle estuary bays are more likely to accrete sediment, have more variable water temperatures, salinity and are more likely to support complex habitats, such as seagrasses and rocky reefs, compared to deeper sections toward the mid-estuary channel. Middle estuarine bay sediments are largely silty sand and ecosystems are exposed to many anthropogenic stressors, such as sediment loading and nutrient enrichment.

The lower estuary zone extends southward from the Tasman Bridge to the Iron Pot in the east and Tinderbox in the west, but excludes Ralphs Bay. This zone is generally well mixed and dominated by influence of marine waters, with some influence of upper estuary and middle estuary water particularly in the upper 10 m of the water column. Marine waters off south-eastern Tasmania are a convergence zone between subtropical and sub-Antarctic water masses. Nutrient-poor subtropical waters may be carried along the east coast of Tasmania in warmer months and these occasionally extend into Storm Bay and the mouth of the Derwent estuary. In cooler months, nutrient-rich sub-Antarctic waters enter Storm Bay and the Derwent Estuary (Harris *et al*., 1987a). These dynamics are detected in ambient water quality of the Derwent Estuary, particularly with higher nutrient concentrations detected through winter.

Ralphs Bay supports vast areas of intertidal habitats owing to its broad shallow geography and sediment is principally sandy. Water quality in Ralphs Bay is predominantly influenced by waters originating in middle estuary surface waters and residence times are greater in Ralphs Bay than elsewhere. Fish growth rates and metal accumulation in Ralphs Bay are unique throughout the Derwent (Jones *et al*., 2014), likely owing to its different geographical, sediment and water-quality characteristics.

2.1.2 Morphology and geology

The Derwent Estuary extends about 55 km from New Norfolk at its northern end to the Iron Pot Light at its mouth, and covers an area of 198 km^2. The morphology of the estuary is that of a drowned river valley, which was formed between 6,500 and 13,000 years ago when sea level rose around 60 m to near its current level.

Estuarine bathymetry is illustrated in Figure 2.1. The upper estuary extends from New Norfolk to the Bridgewater causeway, and is characterised by a narrow channel 3-6 m deep, flanked by extensive wetlands and shallow subtidal macrophyte meadows that provide valuable nutrient filtration services to the Derwent Estuary (Wild-Allen *et al*., 2010, 2013b; Wild-Allen and Skerratt, 2011). The middle part of the estuary–between the Bridgewater Causeway and Bowen Bridge–is 1–2 km wide, with a more convoluted shoreline with some rocky headlands and numerous small embayment's. South of the Tasman Bridge the lower estuary widens and is characterised by relatively straight western and eastern shorelines, and a large (> 50 km^2), shallow embayment–Ralphs Bay–on the eastern shoreline.

Average water depths in the lower and middle estuary are in the order of 10 to 20 m, with a maximum depth of 44 m observed immediately south of the Tasman Bridge. The regional geology of the Derwent Estuary is complex, dominated by Jurassic dolerites and Cambrian basalts, with smaller areas of Triassic and Recent sedimentary deposits (Department of Mines, 1976). High resolution geophysical and bathymetric surveys were conducted across the lower Derwent Estuary in 2000 and 2001 to investigate the distribution of Cainozoic sediments and Tertiary volcanic rocks. Magnetic data indicated the location of several previously unknown Tertiary volcanic centres. Seismic reflection profiles recorded a complex sedimentary history aged from late Tertiary to Holocene. This study noted that geologically the Derwent Estuary is known as the Derwent Graben, which is an elongated block of the Earth's crust that has dropped down between two parallel faults, forming a trench. (Roach and Gibbons, 2003).

Coastal landforms along the Derwent foreshore are highly varied and include sandy or muddy intertidal flats, sand and pebble beaches, dunes, rocky shorelines and platforms, steep bluffs and sea cliffs. These landforms have predominantly been shaped by erosional processes as sea level continues to rise. Mapping of the foreshore has been conducted as part of an assessment of coastal vulnerability to erosion from changes in sea level (Sharples, 2006). This information can be accessed via the LIST (**https://maps.thelist.tas.gov.au/**). A recent assessment of the susceptibility of foreshore erosion from boat wake was conducted in 2021 (**Section 9.7.1**).

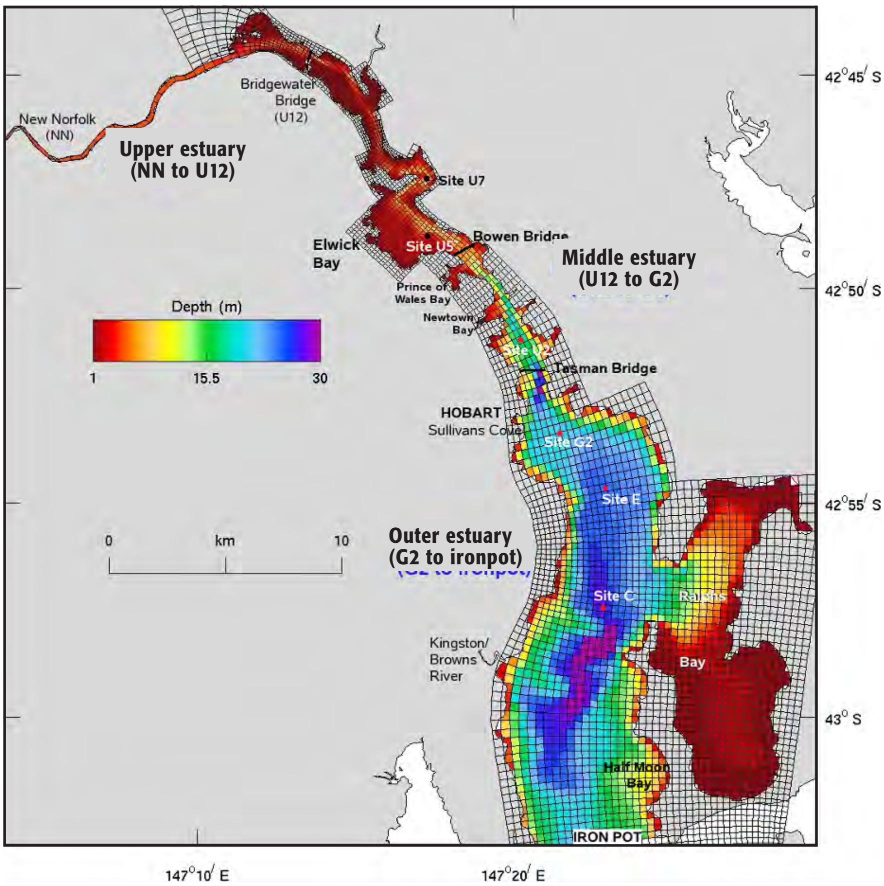
2.1.3 Estuary circulation and coastal oceanography

The mid- to upper-estuary is generally stratified with fresh water overlying a salt wedge, the toe of which is generally located near New Norfolk but may be pushed downstream as far as Bridgewater when flow exceeds 150 m3/s or 13,000 megalitre per day (ML/day) (Davies and Kalish, 1994; Wild-Allen and Andrewartha, 2016). The mid- to lower estuary is classified as partially to well mixed due principally to wind-driven and tidal mixing, and relatively large vertical mass movements occur within the water column.

The average tidal range of the Derwent is slightly greater than one metre, ranging from a minimum of 0.3 m to a maximum of 1.6 m. Tides in the Derwent tend to be asymmetric, in that the diurnal (daily) tide has a slightly greater range than the semidiurnal (twice daily) tide. Hence, Hobart frequently has large variations in the heights of successive tides and occasionally has only a daily tide. Tidal currents are relatively weak, typically in the order of 0.1 to 0.2 m/s. Westerly winds and the Coriolis force deflect the main flow of fresh water from the River Derwent along the estuary's eastern shoreline, while saline bottom water travels slowly up-river. The average flushing period for the estuary is estimated to be about 11 days (Herzfeld *et al.*, 2005) but bottom waters of the upper estuary may be retained for between 20 to 35 days, particularly during low flow (Davies and Kalish, 1994). Flushing times may vary considerably, depending on river flow, wind stress and other variables.

More detailed circulation modelling has been done in specific areas of the estuary, such as the area downstream from Boyer paper mill outfall (DEP, 2015b) and around existing or proposed sewage treatment plant outfalls. A range of scenario simulations exploring plausible future conditions under contrasting levels of urban development and wastewater discharge have also been completed (Skerratt *et al*., 2013).

Figure 2.1 Derwent Estuary bathymetry, modified from (Skerratt *et al*., 2013).



2.2 Storm Bay

The marine waters off southeast Tasmania are known to be an area of convergence between subtropical and sub-Antarctic water masses. Nutrient-poor, subtropical waters are carried along the east coast of Tasmania in summer (extension of the EAC) and the west coast of Tasmania in winter (Zeehan Current), whilst nutrient-rich sub-Antarctic waters lie to the south of Tasmania. These water masses enter outer Storm Bay and the D'Entrecasteaux Channel throughout the year and provide nutrients and plankton that fuels primary production in inshore waters (Harris *et al*., 1987b; Buchanan *et al*., 2013). In the D'Entrecasteaux Channel, the marine nutrient supply is augmented by nutrients in rivers (including the Huon, Esperance, Kermandie, Snug, and Nicholls Rivulet), sewage treatment plants and industrial discharge (including fish-farm waste).

Storm Bay and the D'Entrecasteaux Channel play an important role with respect to the overall circulation and water quality in the Derwent Estuary. Marine water from Storm Bay and the D'Entrecasteaux Channel travel up the bottom of the estuary as far as New Norfolk and gradually mix with overlying freshwater from the River Derwent. Recent modelling suggests that the influx of nutrients from the D'Entrecasteaux Channel into the Derwent is relatively small, as elevated concentrations found in surface waters are typically transported south into Storm Bay (Wild-Allen and Andrewartha, 2016). In 2009, (Wild-Allen and Andrewartha, (2016) found that bottom water from Storm Bay entering the Derwent Estuary had relatively low nutrient content; however, should aquaculture expansion in Storm Bay result in elevated nutrient and/or reduced dissolved oxygen concentrations in bottom waters, then some decline in Derwent Estuary water quality might occur.

2.2.1 Storm Bay Modelling and Information System Project

Provided by Dr Karen Wild-Allen 2025

The FRDC 2017-215 project Storm Bay Modelling and Information System has characterised the primary sources of nutrients into Storm Bay from ocean currents, sediment resuspension, river inputs and delivered a validated model of water quality in Storm Bay suitable for assessing future salmon-farm expansion in the region. The project focus was on water quality in Storm Bay; however, the model also encompasses (with lesser resolution) the broader region from Recherche Bay to the Tasman Peninsula, including the Derwent and Huon Estuaries, the D'Entrecasteaux Channel and Fredrick Henry Bay. The model reliably simulates the hydrodynamic circulation and water quality (temperature, salinity, nitrogen, phosphorus, chlorophyll, oxygen, PAR) from 2015 to present, and has been validated against observations collected throughout the region, including observations in the Derwent Estuary (Wild-Allen *et al*., 2023). Results from model simulations and full documentation of the model design, performance and scenario simulations are available on the information dashboard (**http://stormbaymodelling.csiro.au**) (Figure 2.2).

The Storm Bay model has been used to simulate a range of management scenarios relating to salmon farming in Storm Bay. With increasing nutrient load the model scenarios show an increase in water column nitrogen and chlorophyll, along with a small decline in bottom water dissolved oxygen and light (Figure 2.3). In all scenarios, Storm Bay surface nitrogen levels tended to zero in summer indicating full utilisation by plankton and macrophytes. Nitrogen budget analysis of the scenarios showed that with increasing nutrient load the export of nitrogen from Storm Bay to the ocean and adjacent waterways (including the Derwent Estuary) increased. Preliminary analysis of water quality at the entrance to the Derwent Estuary (site B3) showed some change with increasing farm loads, although as interannual variability in water quality throughout the region is high, it will be difficult to distinguish changes in water quality due to increased fish farms from natural variability using monitoring data alone. The use of continuous sensing systems, model output and ongoing scenario simulations with- and with-out anthropogenic loads could help to distinguish natural variability from fish-farm-induced change in water quality.

Simulation of the hydrodynamic circulation and water quality in southeast Tasmania (the Storm Bay model area) and in Macquarie Harbour, to support ongoing strategic and tactical management of these coastal waters, is currently funded by the Department of Natural Resources and Environment Tasmania (NRE Tas).

Figure 2.2 Model bathymetry and site locations for field observations used for model validation:
l EPA/IMAS/BEMP Aquenal l DEP sites; l Additional EPA/IMAS sites including eastern bays.

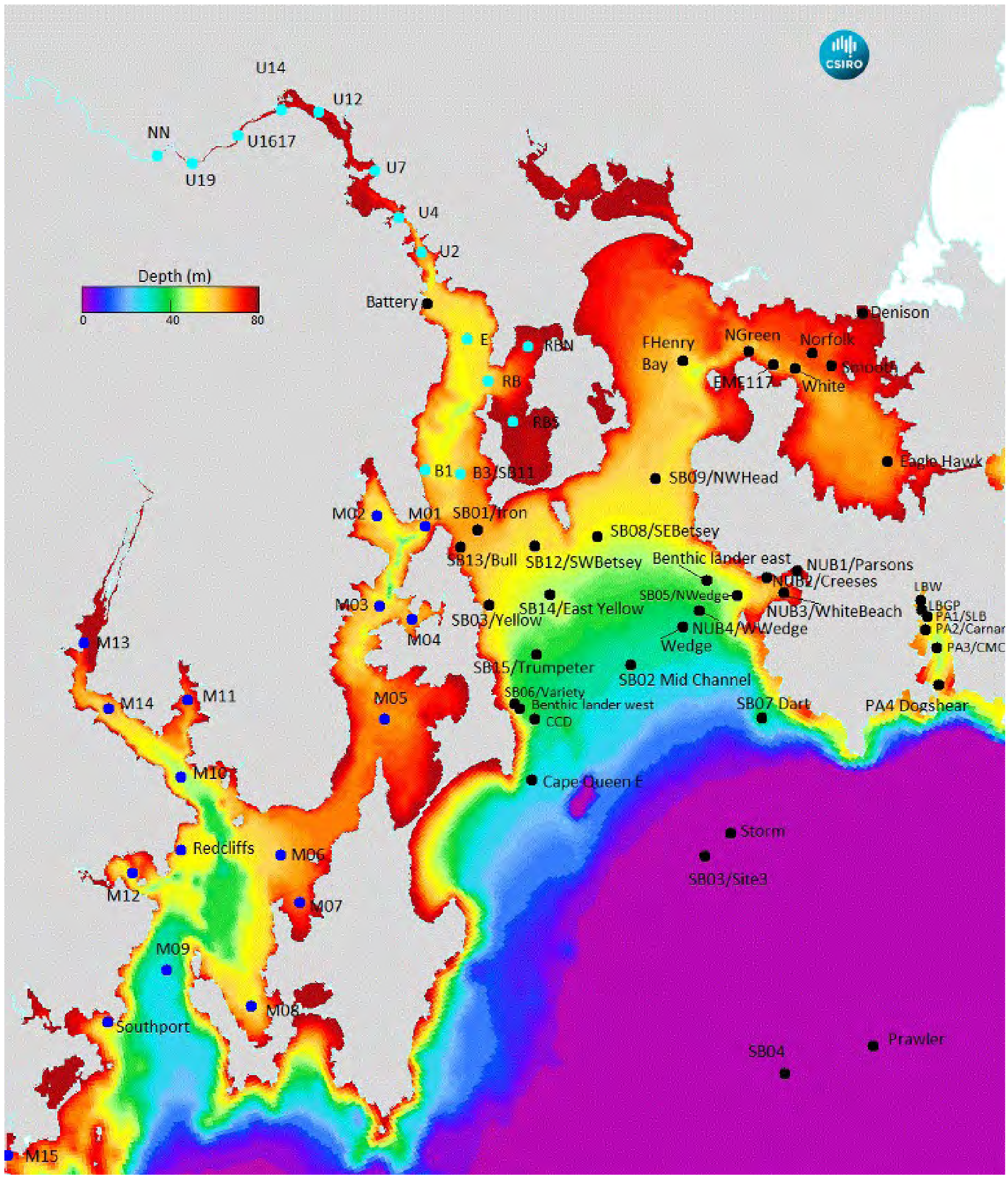
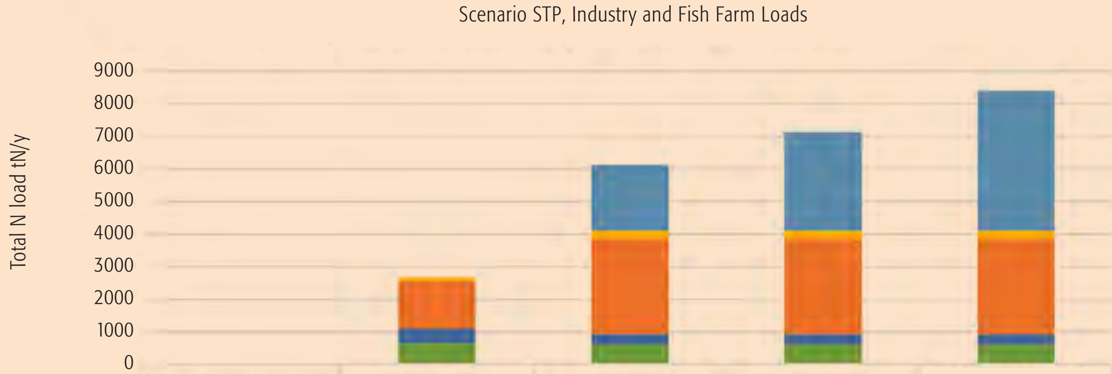


Figure 2.3 Annual anthropogenic total nitrogen load from sewerage treatment plants (STP)+Industry (green) and fish farms in sub-regions of the model (other colours) for each scenario simulation. Note that all post-Storm Bay scenarios had identical 2020 STP+Industry loads and 2020 fish farm loads in Nubeena, D'Entrecasteaux and Huon.



	Pre-Development	Pre-SB (c.2013)	2020+2ktN in SB	2020+3ktN in SB	2020+4ktN in SB
Storm Bay	0	0	2018	3029	4293
Nubeena	0	121	257	257	257
D'Ent	0	1430	2947	2947	2947
Huon	0	456	313	313	313
STP+Ind	0	657	594	594	594

2.3 Climate

The Derwent Estuary region has a cool temperate climate, with a mean maximum temperature range of 12°C in July to 22°C in February. In general, due to topographic influences and the northwest-southeast orientation of the River Derwent valley, katabatic (downslope) winds prevail, blowing from the northwest. However, southerly sea breezes tend to dominate in summer afternoons. Precipitation is monitored by the Bureau of Meteorology at several sites throughout the Derwent. Mean annual rainfall varies across the estuary, with approximately 611 mm a year around Hobart, and about 690 mm further south at Kingston. Rainfall in Hobart is relatively evenly distributed throughout the year at between 39 mm in February and 60.8 mm in October (Figure 2.4 and 2.5).

Environmental conditions in the Derwent Estuary are strongly affected by weather, including wind which is a key driver of estuarine hydrodynamics (Thomson and Godfrey, 1985;Walker and Hunter, 1995). Warm, dry years are often marked by poor estuarine mixing, resulting in low dissolved oxygen, while wet weather brings high surface runoff containing litter, silt, faecal bacteria and oil to the estuary. Climate change is discussed as a principal stressor in Section 2.3.

Figure 2.4 Monthly average rainfall from1882 to 2024 for Hobart at Ellerslie Road (BoM, 2020).

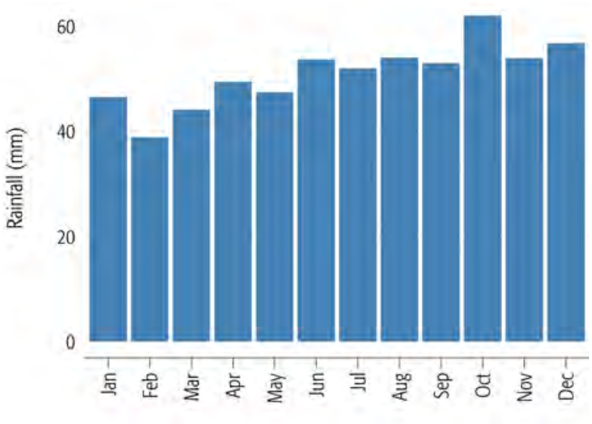
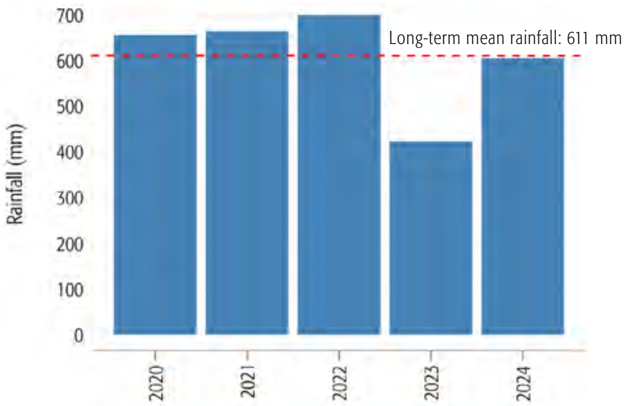


Figure 2.5 Annual rainfall from 2020 to 2024 for Hobart at Ellerslie Road (BoM, 2024a).



2.4 Climate Change

Climate change is characterised by increasing ambient temperatures, variability in weather patterns and a rise in the frequency and severity of extreme weather events—including storms, droughts, bushfires, and floods. A major driver of these changes is the increase in anthropogenic greenhouse gas emissions—primarily from the burning of fossil fuels, industrial activities, and deforestation—which trap heat in the atmosphere and intensify the greenhouse effect. These changes are not only more intense but also less predictable, making it harder for ecosystems, communities and industries to plan and adapt. The consequences are far-reaching, posing escalating risks to the operation and resilience of critical sectors, such as agriculture, aquaculture, hydroelectric power generation, transport, and shipping. For example, prolonged droughts can reduce water availability for irrigation and energy production, while intense storms and sea level rise can disrupt port operations and supply chains.

Climate change will cause significant disruptions to ecosystems, likely resulting in significant losses in biodiversity. Increasing temperatures are causing poleward range shifts in species distributions, or likely extinctions for those that cannot adapt to align with thermal tolerance limits. Sea level rise will impact intertidal ecosystems and changing ocean currents impacting food availability for estuarine and marine organisms. Changed rainfall and runoff patterns will impact plants and ecosystems that rely on these flows and changing fire regimes will impact the resilience of fire affected areas.

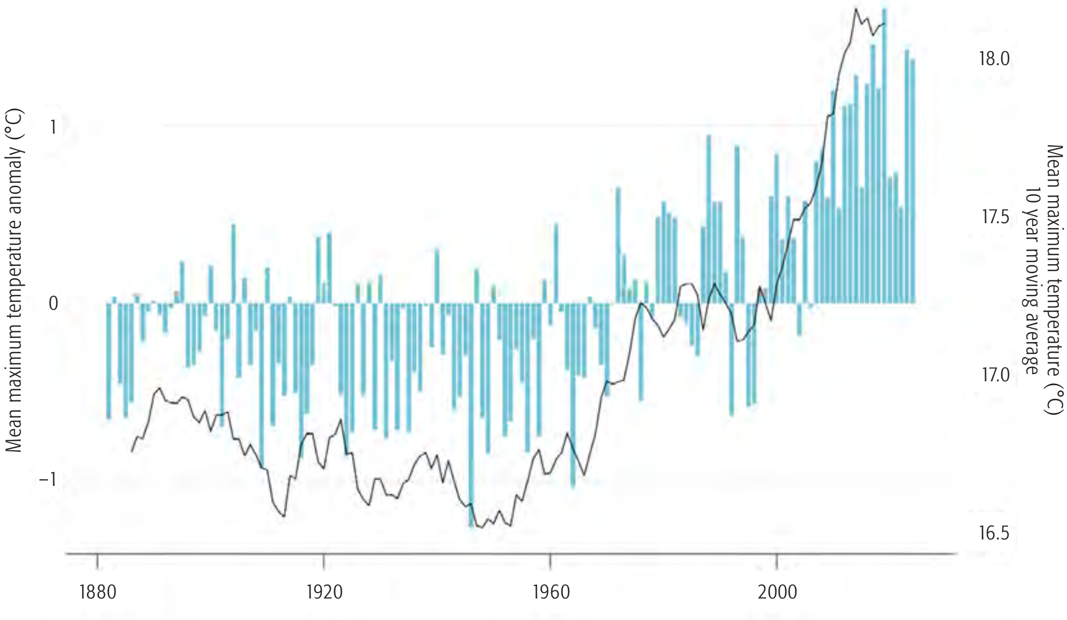
Urban infrastructure is also increasingly vulnerable. Flooding can overwhelm stormwater systems, damage roads and buildings, and lead to costly repairs and insurance losses. Soil degradation—such as drying, cracking, and erosion—can undermine the structural integrity of foundations, pipelines, and transport corridors, particularly in low-lying or reclaimed areas.

Given its broad geographic impact, climate change affects all aspects of the Derwent Estuary's ecology in different ways. Key influences include temperature changes, rainfall patterns, storm intensity, sea level rise, and catchment flows, which are briefly reviewed below. These factors interact in dynamic ways, creating new challenges for estuarine management. A more comprehensive climate change summary specific to the Derwent Estuary is currently being developed and will provide further insights into projected impacts, vulnerabilities, and adaptation strategies.

2.4.1 Temperature

Climate change has led to a gradual but persistent rise in air temperatures globally, with many regions–including Hobart–experiencing warmer average conditions, more frequent heatwaves, and shifts in seasonal temperature patterns. The air temperature of Hobart has warmed, most notably from 1956 onwards (Figure 2.6).

Figure 2.6 Hobart annual mean maximum temperature anomaly (1882-2024), relative to 1882-2023 annual mean maximum temperature. Mean maximum temperature 10-year moving average plotted on the right-hand axis. (Data source BoM, 2024a)



The mean maximum annual temperature for Hobart during the period 1990-2024 was 17.62 °C (+- 0.58 °C). This is 0.56 °C warmer compared to the 1960-1990 period, and 0.91 °C warmer compared to the 1882-1960 period.

The mean maximum annual temperature for Hobart for the period 1982-2024 was 17 °C (+- 3.7 °C), the mean maximum annual temperature for 2024 was 18.4 °C, 1.4 °C warmer compared to the historic record.

Tasmania as a whole has observed an annual average temperature increase of 0.1 °C per decade since 1950. This is below the national average of 0.16 °C per decade for the same period (ACE CRC, 2010).

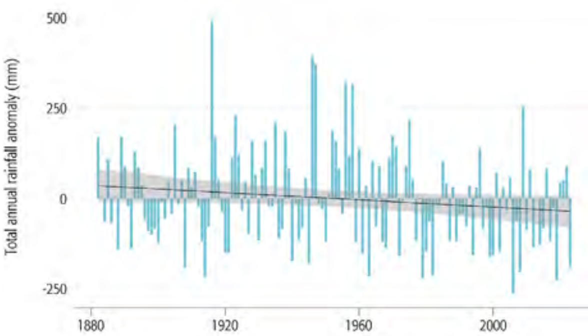
2.4.2 Rainfall and increased storm intensity

Hobart has experienced notable shifts in rainfall patterns and storm intensity, particularly from 1977 onwards, consistent with broader climate change trends affecting Tasmania. Rainfall has generally declined, with 2023 marking a particularly dry year. Hobart (Ellerslie Road) recorded just 422.8 mm, which is 69% of the long-term average of 611.5 mm (BoM, 2024b). This decline reflects a longer-term trend observed since the mid-1970s, with the most significant reductions occurring in autumn. While months like March and June 2023 saw above-average rainfall, most months from August onward were drier than usual.

Storms have become more intense, with extreme rainfall events increasing in severity. Climate projections suggest that rainfall on the wettest days could rise by up to 25%, and extreme events (e.g., those with a 200-year recurrence interval) may deliver 30–40 mm more rain than historical norms. These changes heighten the risk of flooding and erosion, especially during high runoff events.

These patterns reflect the broader impacts of climate change in Tasmania, where annual average rainfall has decreased since 1900, and extreme rain events are projected to become more intense (CSIRO, 2021). Hobart is expected to experience heavier rainfall interspersed with longer dry periods, complicating water management and increasing vulnerability to both drought and flood conditions.

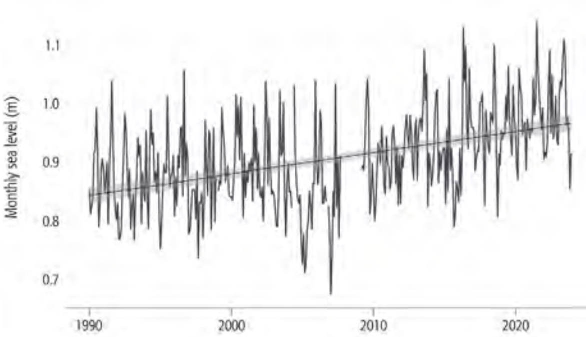
Figure 2.7 Hobart total annual rainfall anomaly (1882-2024), fitted with linear regression line. (Data source BoM, 2024a)



2.4.3 Sea level rise

As a consequence of climate change, rising sea levels have become an increasingly evident trend along Australia's coastline, with long-term monitoring revealing a steady increase in sea level height near Hobart. There has been an observable increase in sea level height of 3 mm/year recorded at the Hobart Port tide gauge, based on linear slope analysis (Figure 2.8). However, it should be noted that this tidal gauge infrastructure is not designed to measure long-term trends in sea level height. The Australian baseline sea level monitoring project was established to accurately investigate changes in sea level height around Australia's coastline with a sea level monitoring network of 14 standard SEAFRAME stations installed. The closest station to Hobart, the Spring Bay SEAFRAME gauge shows an increase in sea level height of 3.44 mm/year (1992–2024).

Figure 2.8 Monthly sea level recorded at the Hobart port tide gauge (1990-2024), fitted with linear regression line. (Data source BoM, 2025)



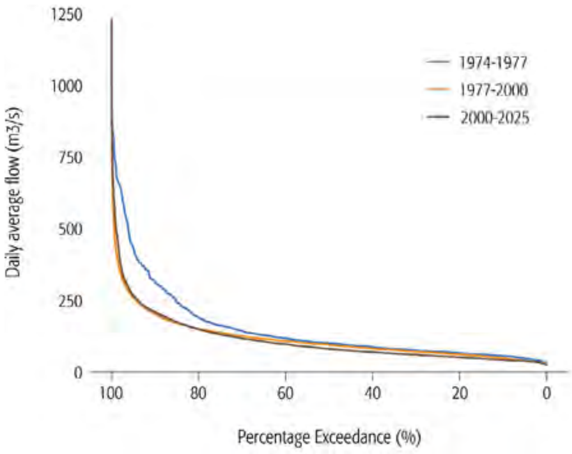
2.4.4 Estuary surface water temperatures

Water temperature in the Derwent Estuary has shown no significant trends over the long-term monitoring period (2007-2024 for Ralphs Bay, upper and lower estuary; and 2011-2024 for the middle estuary). Notably surface water temperatures have not shown increases that would be expected under a warming climate. This is surprising given that the Tasman Sea is considered to be an ocean warming hotspot (Hobday and Pecl, 2014). In estuarine systems land-derived freshwater inflows greatly influence estuarine circulation patterns and heat distribution (Prum *et al*., 2024), therefore freshwater inflows are likely regulating water temperatures within the Derwent Estuary. For a detailed breakdown on Derwent Estuary water temperatures see Section 1.1.

2.4.5 Catchment flows

Recent hydrological data reveals a significant decline in the frequency of high flow events (>200 m^3/s) in the River Derwent, particularly following the onset of a recognised dry period in 1977. Between 1974 and 1977, the probability of exceeding this high flow threshold was 81%, compared to approximately 89% from 1977 onwards, indicating a marked shift in flow dynamics (Figure 2.9). While flow duration curves from 1977–2000 and 2000–2025 show minimal differences, there is a slight reduction in mid to lower flow ranges (<150 m^3/s).

Figure 2.9 Probability exceedance curve of daily average flows in the River Derwent from 1974 to 2025, split into three periods of interest 1974-1977, 1977-2000 and 2000-2025. Probability exceedance curve ranks observations of daily average flow over the period, then converts these ranks to percentages, visualising the distribution of flow observations throughout the period of interest. (Data source NRE Tas, 2025)



These changes are influenced by both climatic variability and long-standing hydrological regulation. The construction of Meadowbank Dam in 1957 initiated significant flow regulation in the lower Derwent, although it did not substantially alter annual yields. However, an approximate 10% reduction in catchment yield occurred in 1964 due to the diversion of upper catchment flows through Poatina into the South Esk catchment.

Davies and Kalish (1994) investigated changes in catchment flow on the Derwent and Huon rivers suggesting that while dam operations have had limited impact on overall yield, the Derwent has experienced a more pronounced decline in high flow events, potentially due to increased evaporative losses and climate-driven changes. These findings underscore the complex interplay between engineered water management and climate variability, with implications for water availability, ecosystem health, and hydroelectric reliability in the Derwent Estuary.

2.4.6 Marine Heatwaves

Marine Heatwaves in the Derwent–Channel–Huon System

Provided by Dr Christopher J Roach and Dr Neil J Holbrook, Institute for Marine and Antarctic Studies, and University of Tasmania

Background

Marine heatwaves (MHWs) are periods of extremely warm water typically defined by temperatures remaining at or beyond the top 10% of historically observed values (for the time of year) for a period of 5 days or more (Hobday *et al.*, 2016). MHWs can be driven regionally by mechanisms, such as advection of warm waters from low latitude to higher latitudes or local surface heating (Holbrook *et al.*, 2019), and coastally may include downwelling of surface waters (Schaeffer *et al.*, 2023).

The interplay between offshore MHWs and shelf and estuarine environments is complex. For instance, shelf-break currents can act as barriers to water moving onshore while some configurations of offshore mesoscale (50–300 km diameter) eddies can inject warm offshore extreme conditions onto the shelf.

MHWs can have significant impacts on marine habitats and species (Smith, Burrows, *et al.*, 2023), including large-scale species mortality, stress leading to decreased reproductive success or increased vulnerability to disease, range shifts of warmer water species into cooler regions, and increased occurrences of algal blooms. These factors can combine to cause ecosystem-scale issues, where loss of keystone species can lead to long-term community changes as seen in Western Australia following the 2011 MHW (e.g. Wernberg, 2021). Ecosystem changes can in turn feed into socioeconomic impacts (Smith *et al.*, 2021).

The Tasman Sea, extending between southeastern Australia and New Zealand, has seen major MHWs in recent years, including the summers of 2015/16, 2017/18 and 2023/24.

Impacts and responses associated with these events have extended onto the continental shelf and into the Derwent Estuary system, including:

- 2015/16: outbreaks of Pacific Oyster Mortality Syndrome (POMS) and mortality of Black Lipped Abalone (Oliver *et al*., 2017); shifts in phytoplankton diversity (Brown *et al*., 2024)
- 2017/18: further POMS outbreaks; reduced salmon aquaculture productivity
- 2023/24: emergency intervention to protect the critically endangered Red Handfish, including temporary housing in captivity followed by release back into the wild post-MHW (Hobday *et al*., 2024).

Studies suggest MHWs are likely to become stronger and more frequent as climate changes (e.g. Frölicher *et al*., 2018), with events in the Tasman Sea comparable to the previously unprecedented scale of the 2015/16 MHW likely to become more common by mid-century (Kajtar *et al*., 2024)

Current Work

The Derwent Estuary Program (DEP) is collaborating with team members in a National Environmental Science Program Climate Systems Project (5.5) led out of the Institute for Marine and Antarctic Studies, in Hobart.

The aim of this project is to establish baseline understanding and estimates of MHWs as an overall hazard in the Derwent Estuary, D'Entrecasteaux Channel, and Huon Estuary system including:

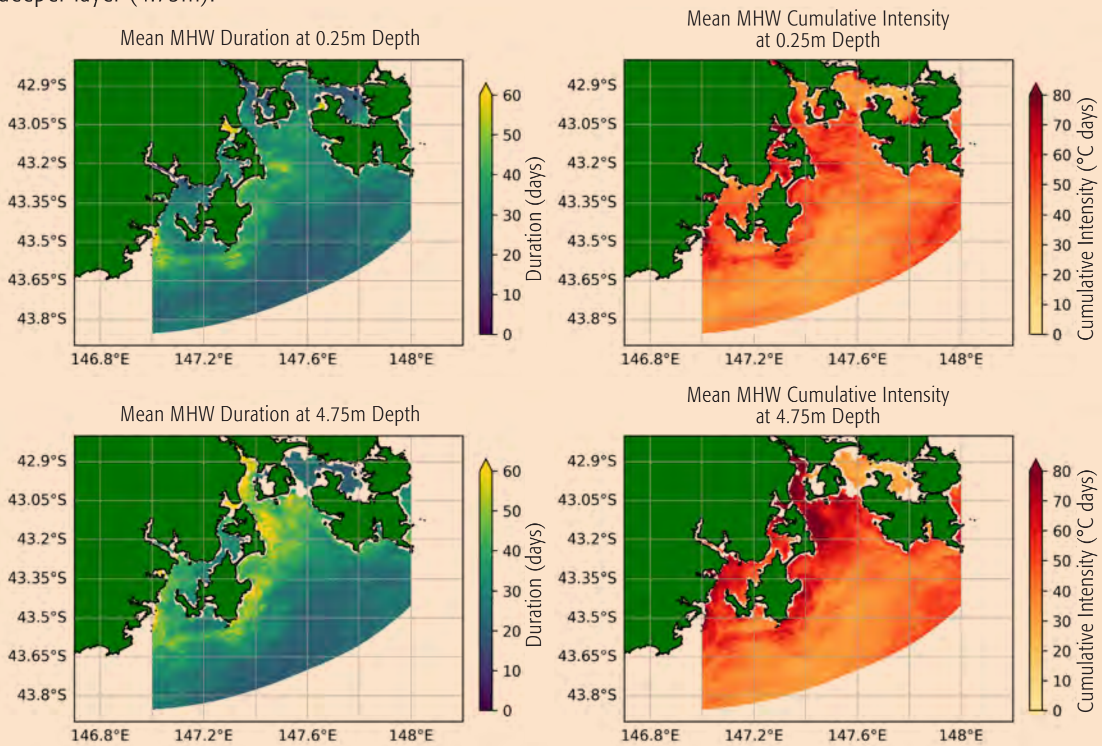
- Establishing how different parts of the system represent a potential MHW hazard, while also identifying potential refugia.
- Identifying the seasonality of MHWs in different parts of the system.
- Describing the vertical structure of MHWs within the regional domain, including across known benthic habitats.
- Investigating the occurrence of compound events where MHWs coincide with other stressors such as low oxygen levels or algal blooms.

The study will involve a detailed analysis of historically forced simulations from two ultra-high resolution coastal ocean models developed by the CSIRO – specifically, ETAS (Oliver *et al*., 2016) a ~2-km resolution model run from 1993 to 2016, and TASSE (Wild-Allen *et al*., 2023) a ~200-m resolution model run from 2015 to the present.

Both models are re-gridded onto a common 250-m grid and bias corrected against *in situ* observations collected by the DEP and the Environment Protection Authority-Salmon Tasmania Broadscale Ecological Monitoring Program.

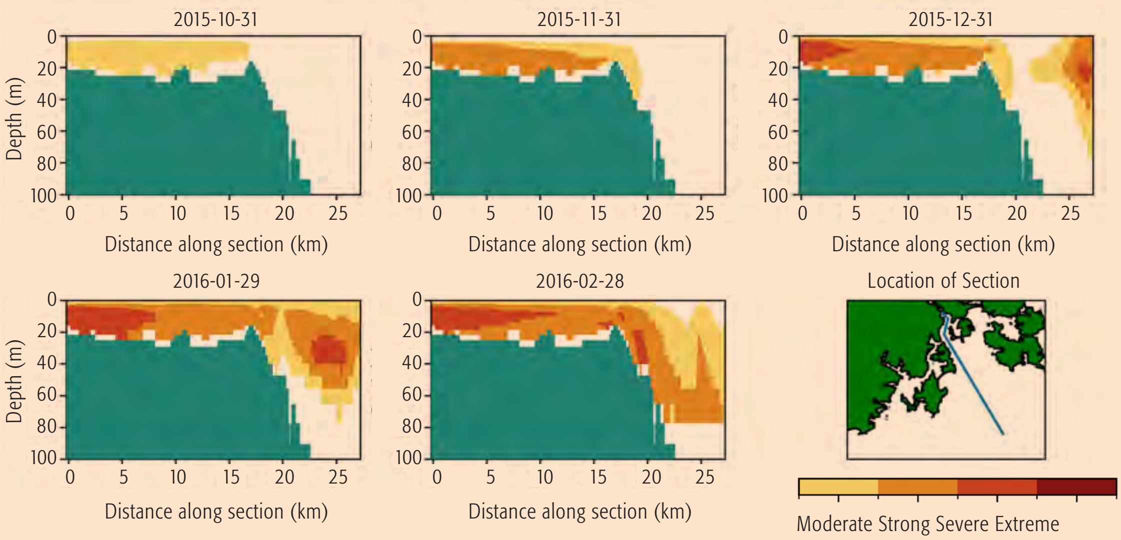
This project is in its early stages, but some preliminary findings are interesting. Figure 2.10 shows the mean duration and cumulative intensity (effectively the combined sum of intensity and duration through the MHW lifetime) of MHWs at the surface and at 4.75 m depth. We see that surface MHWs tend to be longest and most intense along the east coast of Bruny Island and in North West Bay. At 4.75 m depth we see that MHW duration and cumulative intensity strongly increase in the Derwent Estuary. Frederick Henry Bay and Norfolk Bay show a combination of short duration and low cumulative intensity, suggesting this region may serve as refugia for some species.

Figure 2.10 Mean MHW duration and cumulative intensity for the surface-most depth layer (0.25m) and a deeper layer (4.75m).



Another early finding is based on a section along the Derwent Estuary extending into Storm Bay which shows that the 2015/16 offshore MHW driven by southward advection in the East Australian Current (EAC) extension waters did not move into Storm Bay until late-December 2015 but was preceded by extreme heating originating within the estuary (Figure 2.11). This suggests the some of the species and ecosystems badly affected by the 2015/16 offshore MHW may have already been under stress.

Figure 2.11 Evolution of 2015/16 MHW category along a section running from down the Derwent River and into Storm Bay. Note the development of a MHW in the estuary Oct-Dec 2015 before the offshore advective MHW in late Dec 2015.



3

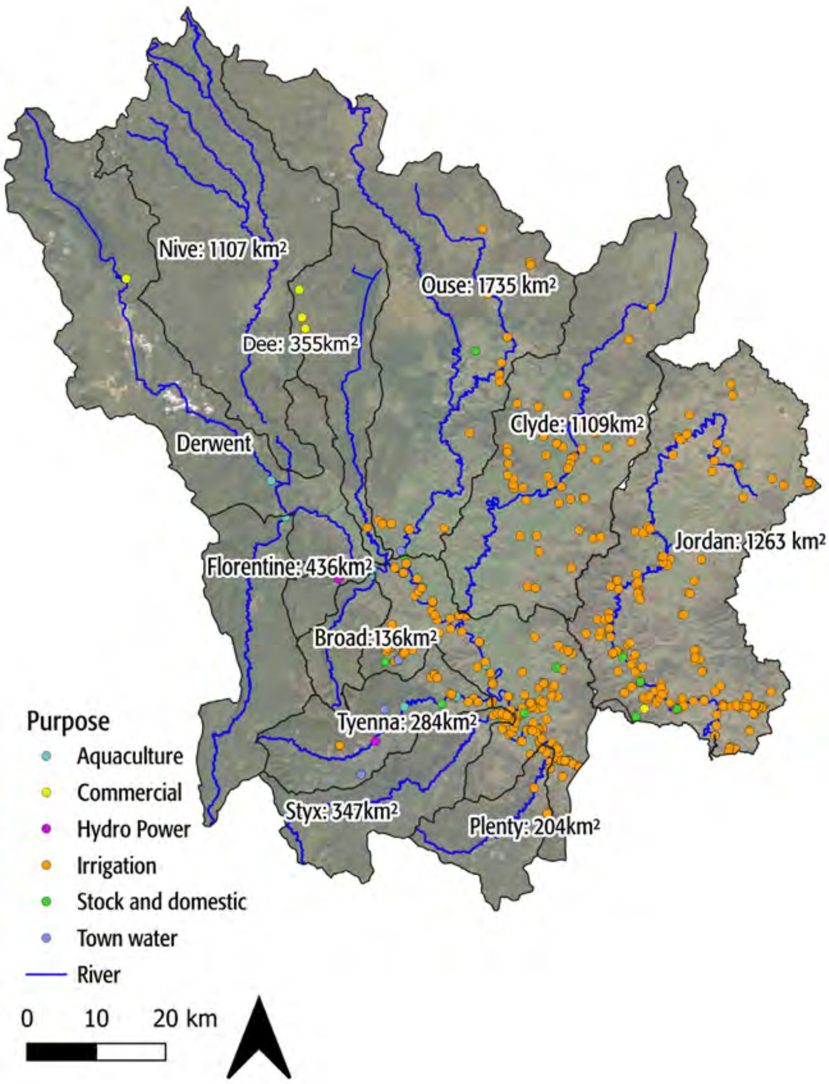
Catchment physical setting

3 Catchment physical setting

The Derwent Estuary catchment occupies approximately 8,900 km² across central and south-eastern Tasmania– around one-fifth of the state. It comprises the Timtumili Minanya/River Derwent catchment (7,764 km²), the Jordan River catchment (742 km²), and adjoining areas immediately fringing the estuary (about 375 km²) (Figure 3.1). The River Derwent – Tasmania's second-longest after the South Esk – rises at Lake St Clair and flows roughly 187 km south to New Norfolk at the head of the estuary. Major subcatchments include the upper Derwent (Florentine River), Ouse and Clyde River catchments, the lower Derwent (Tyenna River), Styx, Jordan, Plenty and Broad catchments (Figure 3.1).

Vegetation across the Derwent catchment mirrors the rainfall gradient, relief, and underlying geology. The wetter western ranges are dominated by wet eucalypt forest and rainforest; the northern Central Plateau supports alpine heathland with wet forest on its southern slopes; and in the drier south, vegetation shifts to remnant native grasslands and open grassy woodlands (DEP, 2015b). For further information about catchment physical settings and land uses, see (DEP, 2015).

Figure 3.1 Derwent Catchment regions and their catchment sizes.



3.1 Catchment land use

Land use varies systematically across the Derwent catchment. The eastern sub-catchments (Clyde and Jordan) are predominantly agricultural, whereas the western and north-western sub-catchments (Tyenna, Plenty, Derwent and Florentine) are dominated by forestry tenures and conservation reserves (Figure 3.2). At the whole-of-catchment scale, conservation land accounts for 76% of the area, forestry 10% and agriculture 11%, with the small remainder in other uses (Figure 3.3). Sub-catchment contrasts are pronounced. For example, the Jordan is 56% agriculture and 23% conservation, while the Tyenna is 97% conservation with 2% forestry and 1% agriculture (Figure 3.3 to Figure 3.7).

Figure 3.2 Land uses in the Derwent Catchment. Data based on 2021 Listmap information (https://www.thelist.tas.gov.au).

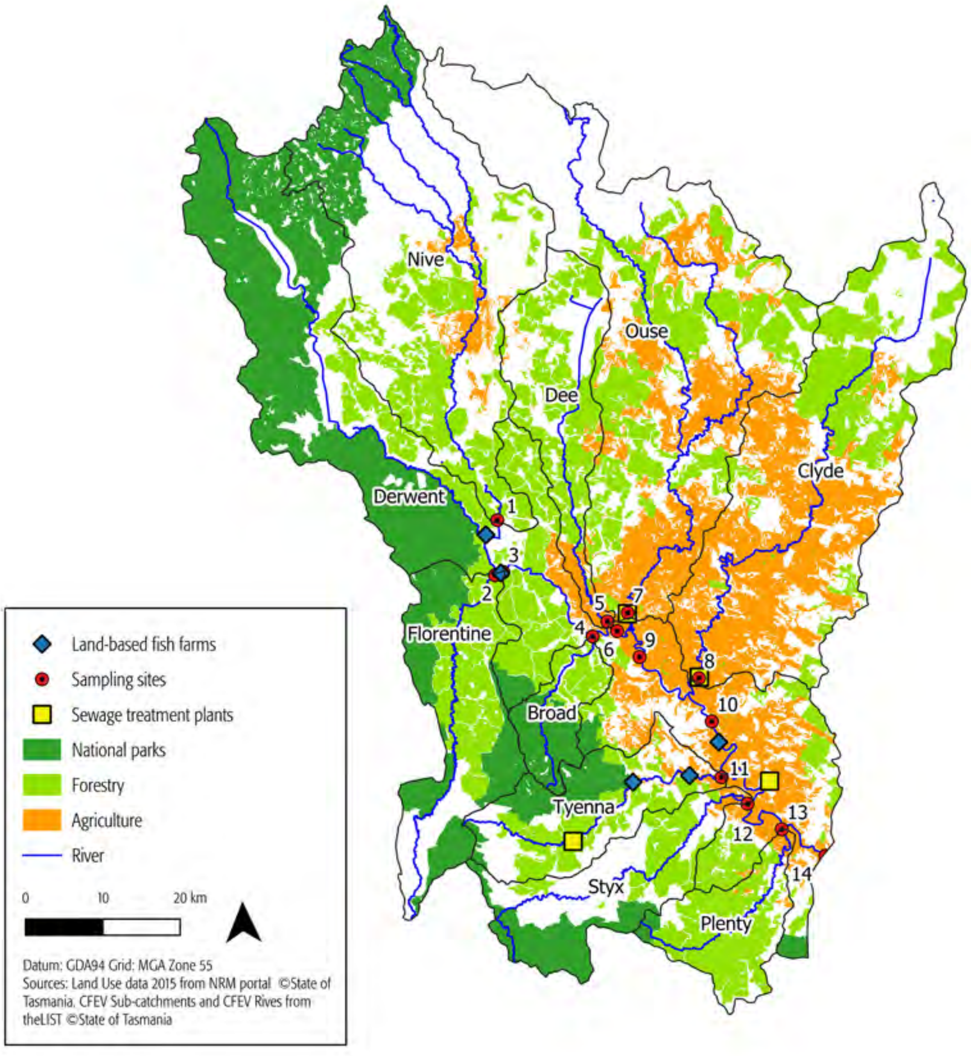


Figure 3.3 Whole Derwent Catchment (including Jordan). Data sourced from The LIST and represents data from 2021.

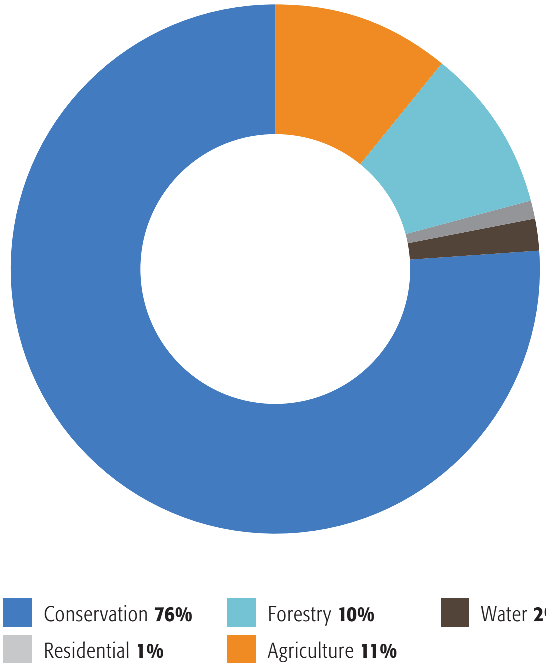


Figure 3.5 Ouse River catchment land use percentages. Data sourced from The LIST and represents data from 2021.

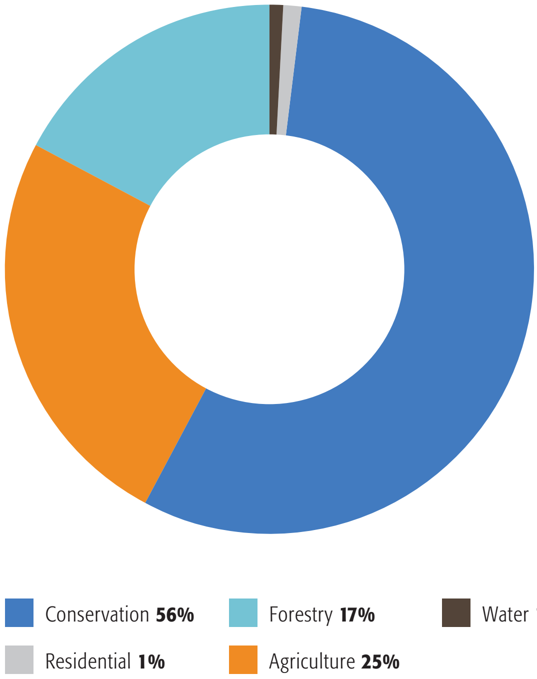


Figure 3.4 Jordan Catchment land use percentages. Data sourced from The LIST and represents data from 2021.

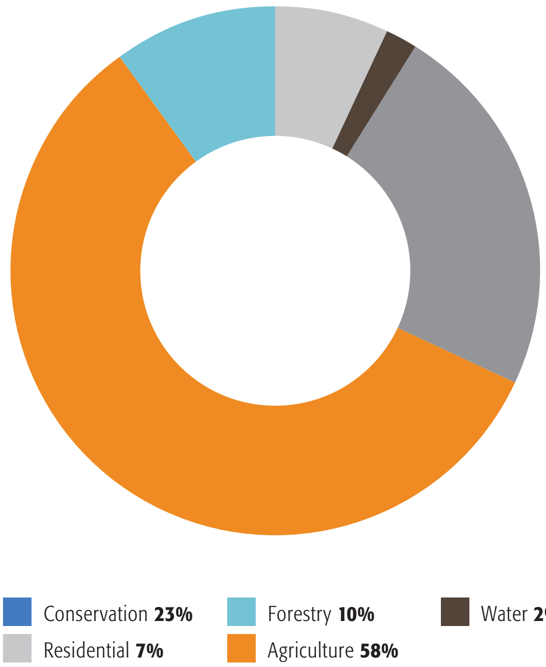
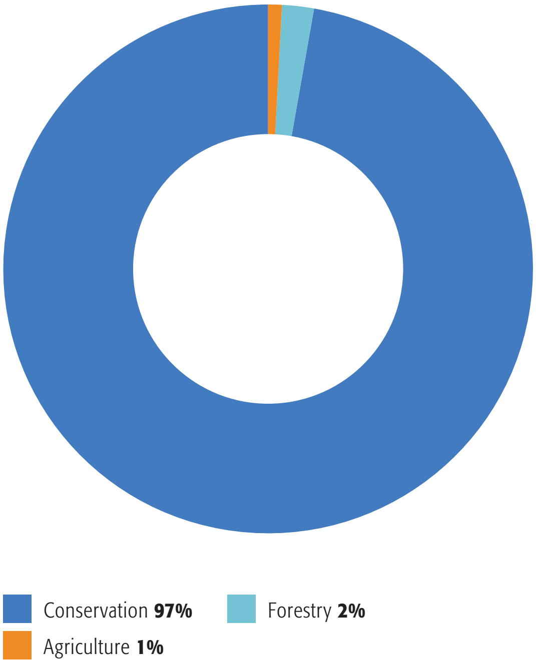


Figure 3.6 Clyde River catchment land use percentages. Data sourced from The LIST and represents data from 2021.

Figure 3.7 Tyenna River catchment land use percentages. Data sourced from The LIST and represents data from 2021.



3.2 Water quality

3.2.1 Tasmanian river status

Most Tasmanian rivers are naturally oligotrophic, draining nutrient-poor, largely forested catchments underlain by siliceous bedrock and leached soils. Consequently, background concentrations of dissolved inorganic nitrogen and reactive phosphorus should be very low, with algal biomass limited (low chlorophyll-a) and water clarity high. This low-nutrient state confers high conservation value but also high sensitivity to enrichment with relatively small additions from wastewater, agricultural runoff, or riparian disturbance pushing systems toward mesotrophy/eutrophy (moderate to high nutrient levels), with increases in periphyton or nuisance blooms. In such systems, concentration "exceedances" that might be minor elsewhere can be ecologically significant, and load-based assessments should be interpreted against a naturally low baseline (Table 3.1). Management therefore prioritises maintaining low nutrient inputs, protecting riparian vegetation, and controlling sediment sources that transport particulate phosphorus downstream.

Table 3.1 Environmental Protection Authority (EPA) Default Guideline Values for (DGV) for the full year for Slightly to Moderately Disturbed (SMD) for river systems within the Derwent Catchment. (TN = total nitrogen, TP = total phosphorous, DRP = Dissolved Reactive Phosphorus). Sourced from the EPA (Catchment DGVs for aquatic ecosystems | EPA Tasmania). All measurements reported in parts per billion (ppb).

River catchment	DGV for Nitrate as N (ppb)	DGV for Nitrite as N (ppb)	DGV for DRP (ppb)	DGV for TN (ppb)	DGV for TP (ppb)
Tyenna river	35	1	3	144	8
Ouse	2	1	3	250	10
Clyde	5	1	4	370	9
Jordan	5	1	4	370	9
Florentine	2	1	3	250	10

3.2.2 River hydrology and flow regime

Flow at New Norfolk is currently calculated by combining average daily flow sampled below Meadowbank dam (BOM) and at Tyenna, Newbury Rd (NRE). Tyenna data is collected every 15 minutes. Observations are averaged for a daily flow rate and multiplied by a factor of 4.89 representing input from catchments that are not measured (Proemse *et al*., 2022a).

This is calculated using the following equation:
Modelled average daily flow rate @ New Norfolk (m^3/s) = A + 4.89 × B
A = average daily flow at Derwent below Meadowbank gauge (m^3/s)
B = average daily flow at Tyenna gauge (m^3/s)
The following equation is used to calculate daily river discharge:
Average daily river discharge (GL) = A × 86 400 × 1000 / 1000000000
A = Modelled average daily flow rate @ New Norfolk (m^3/s)
Nutrient loads were estimated using the following equation:
Nutrient load (t/month) = [(C × D)/1000]
C = concentration of nutrients (µg/L), as measured in surface water at New Norfolk
D = monthly discharge (GL/month)

The flow regime of the rivers in the Derwent's catchment are essential to support healthy river quality. The River Derwent's flow, for example is highly modified by extensive regulation of tributaries due to hydroelectric power generation, irrigation, and extraction for municipal, industrial and aquaculture purposes. The Derwent River has a long-term mean annual flow (1974-2013) of 91.1 m^3/s or 7,900 ML/day. The typical seasonal trend is for higher flows and greater flood frequencies in the second half of the calendar year, and lower flows during the months of January through March (DEP, 2015c).

Importantly, river discharge is a primary determinant of nutrient mass load for two reasons: (1) as water moves through a catchment it entrains nutrients and other contaminants; and (2) mass load is calculated as the product of concentration, discharge, and time (load = concentration × discharge × time). Accordingly, understanding the catchment's flow regime over the previous five-year reporting period provides essential context for interpreting nutrient loads. As shown in Figure 3.8, average monthly discharge is relatively consistent from January to June, while variability increases from July to November. The year 2021 is notable for exhibiting elevated flows across most months, while September 2024 recorded a pronounced peak of approximately 380 m^3/s, attributable to an extreme flood event that occurred in early September 2024 (see also **Section 6.2**). Figure 3.9 also demonstrates the flow regime compared to annual rainfall within the catchment in the last seven years, with flow rate spikes corresponding to higher rainfall events. Further details about Derwent River flow over the reporting period can be found in **Section 6.2**. A new flow gauge has been installed on the River Derwent (below Plenty River) and is expected to be producing flow data before the end of 2025.

Figure 3.8 Average monthly flow on the River Derwent below Meadowbank.

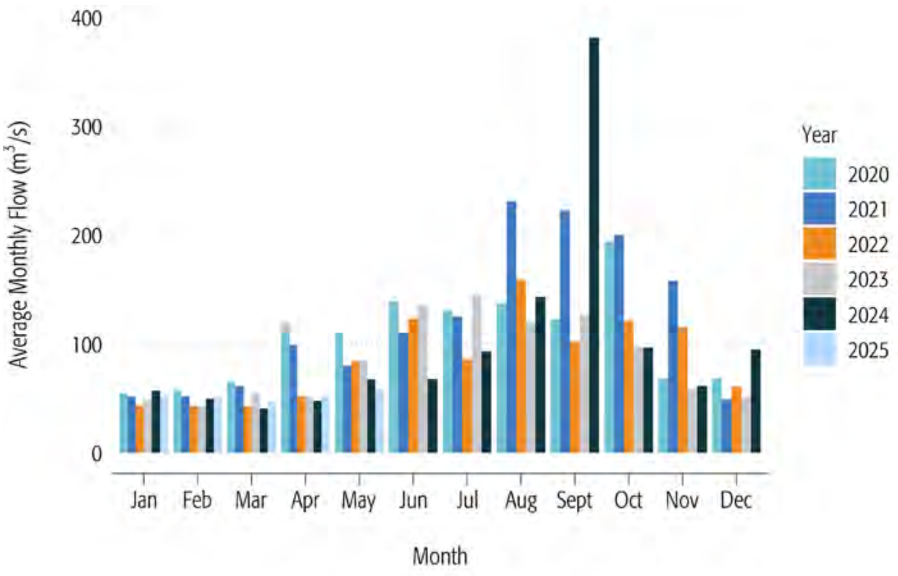
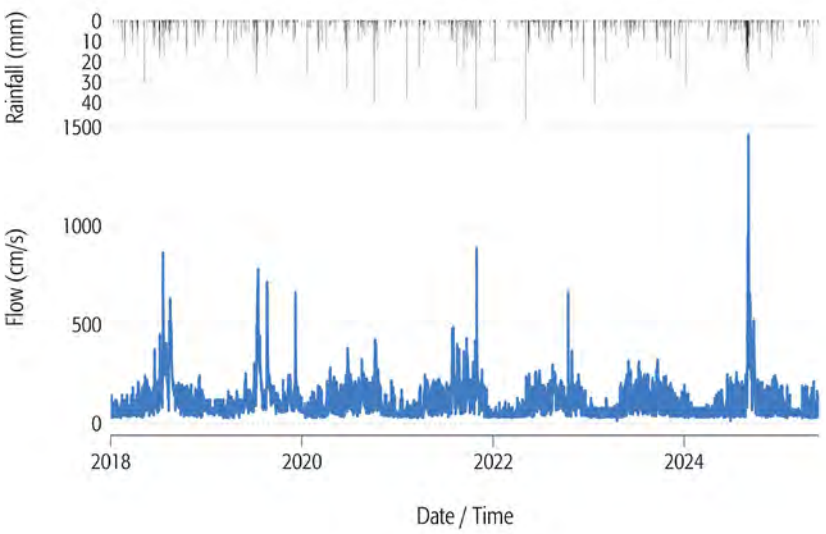


Figure 3.9 Derwent River flow below Meadowbank dam, with rainfall sourced from Ouse fire station (Bureau of Meteorology).

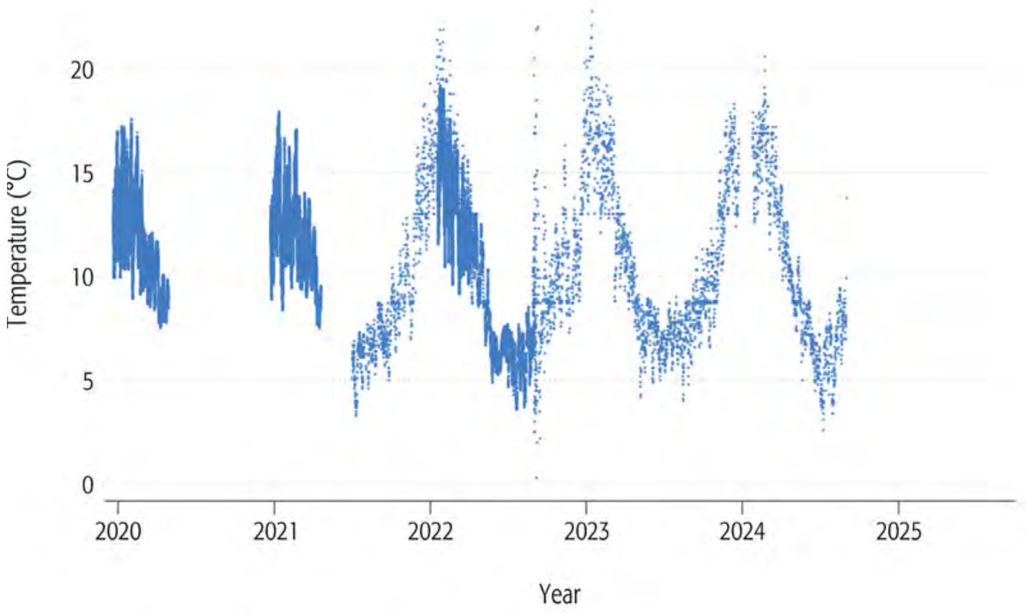


3.2.3 River temperature – case study

Water temperature is a master variable that influences dissolved oxygen, metabolic rates, nutrient cycling and habitat suitability. To examine temporal patterns, we analysed a near-five-year, high-frequency record from the Tyenna River. Temperatures were recorded with HOBO loggers during the initial period and with an Eco Detection in-situ nutrient analyser temperature probe used in subsequent measurements.

Across the monitoring period, water temperature ranged from 0.3 to 22.8 °C (mean 11.6 °C). As shown in Figure 3.10, summer maxima increased in 2022 and 2023 relative to preceding summers, and this pattern is evident in both instrument types.

Figure 3.10 Tyenna River temperature profile from 2019 to 2025. Note that HOBO pennate loggers were used from 2019 to 2022 (data only available over summer months) and *in situ* sensors from Eco Detection field sensors were used from 2021 onwards (continuous data).



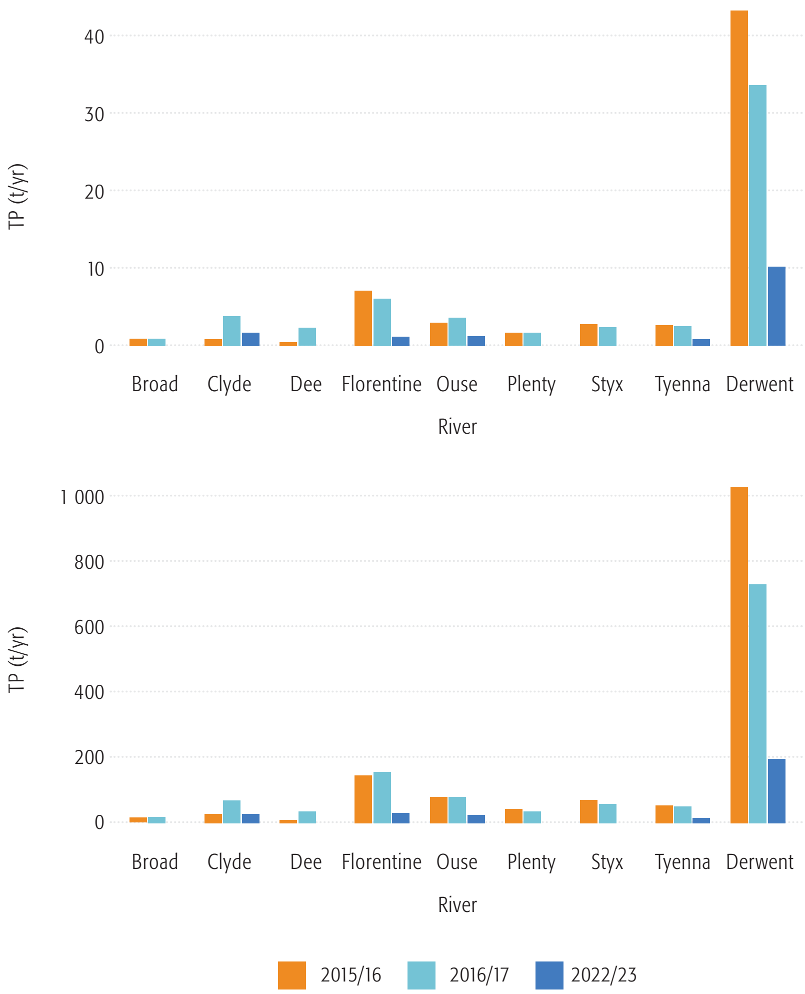
3.2.4 Total nutrients monitoring

Since the 2020 State of the Derwent update (DEP, 2020a), water-quality parameters like dissolved oxygen (DO), pH, turbidity and salinity–and nutrient species (total nutrients, nitrate + nitrite (NOx), and dissolved reactive phosphorus) have been monitored across the catchment at varying frequencies. Total nutrient sampling was undertaken in 2015–2016, 2016–2017 and again in 2022–2023; other years focused on dissolved species and physicochemical parameters.

In downstream receiving environments (lakes, the estuary and nearshore marine waters), nutrient loads, expressed as annual totals of total nitrogen and total phosphorus are more ecologically informative than dissolved nutrient concentrations alone. While annual load estimation for every river is recommended, it is currently cost-prohibitive to cover all sub catchments.

As shown in Figure 3.11, total nitrogen and total phosphate loads in the Derwent mainstem declined between 2015 and 2023, with similar reductions evident in the Tyenna and Florentine. The overall trend of declining total nitrogen and total phosphate does not align with nutrient data collected within the upper estuary which shows a slight increase in the upper estuary regions (see **Section 6.1.4**). Across the different sub catchments the range of total nitrogen is between 1.235 and 1,031.1 tonnes/year. The range of total phosphate within the sub catchments is between 0.5 and 43.1 tonnes/year within the catchment within the reporting period.

Figure 3.11 Total phosphate (TP) and total nitrogen (TN) mass load concentrations for rivers within the Derwent catchment. Note that monitoring programs were from July to June.

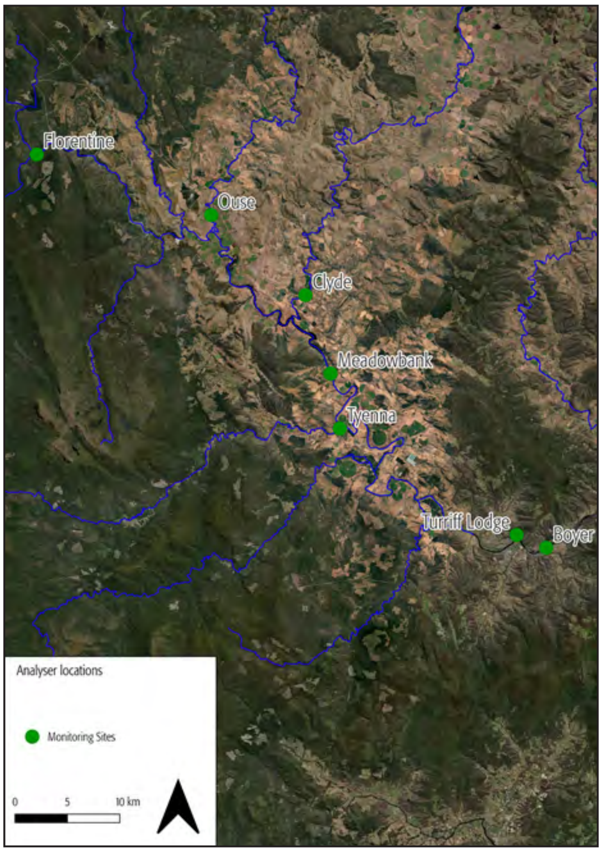


Dissolved nutrients

It is important to measure both total nutrients (total nitrogen, total phosphorus) and dissolved forms (e.g., nitrate, nitrite, ammonium, and dissolved reactive phosphorus) because they provide complementary insights into nutrient risk, sources, and management. Dissolved ammonium (NH_4^+) is the most rapidly assimilated form of nitrogen, particularly relevant to discharges from aquaculture and agriculture, whereas nitrate (NO_3^-) and nitrite (NO_2^-) typically require reduction to ammonium or active transport into cells before uptake. Amino acids and other labile dissolved organic nitrogen (DON) can be taken up at intermediate rates, while more refractory DON and particulate organic nitrogen (PON) are assimilated slowly or transported downstream before becoming bioavailable. In contrast, total nitrogen and total phosphorus measurements integrate both dissolved and particulate fractions, thereby capturing the full nutrient load that may eventually affect downstream estuaries. Dissolved fractions highlight near-term eutrophication risks (e.g., algal bloom triggers), while total nutrient loads provide the necessary basis for annual nutrient budgets and long-term change detection. Relying solely on one measure risks overlooking critical processes: measuring only dissolved nutrients would miss delayed releases from particulate and organic pools, while measuring only totals would mask short-lived but ecologically significant bioavailable spikes.

In September 2021, the Derwent Estuary Program (DEP) embarked on a new water quality monitoring program 'Using revolutionary real-time analyser technology to inform best practice environmental river management in Tasmania', funded by The Ian Potter Foundation and DEP stakeholders (Hydro Tasmania, TasWater, the Environmental Protection Agency, Meadowbank Vineyard) (Proemse and Black, 2025). Seven Eco Detection, real-time, water quality monitoring systems to measure nitrate, nitrite, phosphate, chloride, carbonate, sulphate and fluoride, as well as other key analytes and parameters were deployed across the Derwent catchment (Figure 3.12). Full details of the study can be found in the following report (Proemse and Black, 2025). Deployment of these Eco Detection *in situ* analysers enabled measurement of dissolved NOx (nitrate and nitrite) and phosphate in near real time (Figure 3.13 – Figure 3.15).

Figure 3.12 Location of real-time nutrient analysers within the Derwent catchment.



Over the four years that the analysers have been deployed in the catchment, the river that demonstrates the highest nitrate values is the Clyde. The Clyde demonstrated high nitrate (>750 ppb) over summer months with elevated nitrate levels throughout the rest of the year (Figure 3.13). The Florentine, Tyenna, Ouse and Meadowbank also had elevated nitrate levels above the EPA's Default Guideline Values (DGV).

Figure 3.13 Nitrate concentrations in rivers within the sub catchment of the Derwent catchment. Grey bars indicate maintenance periods where no data was collected. There are some additional gaps in data where analysers were moved between locations.

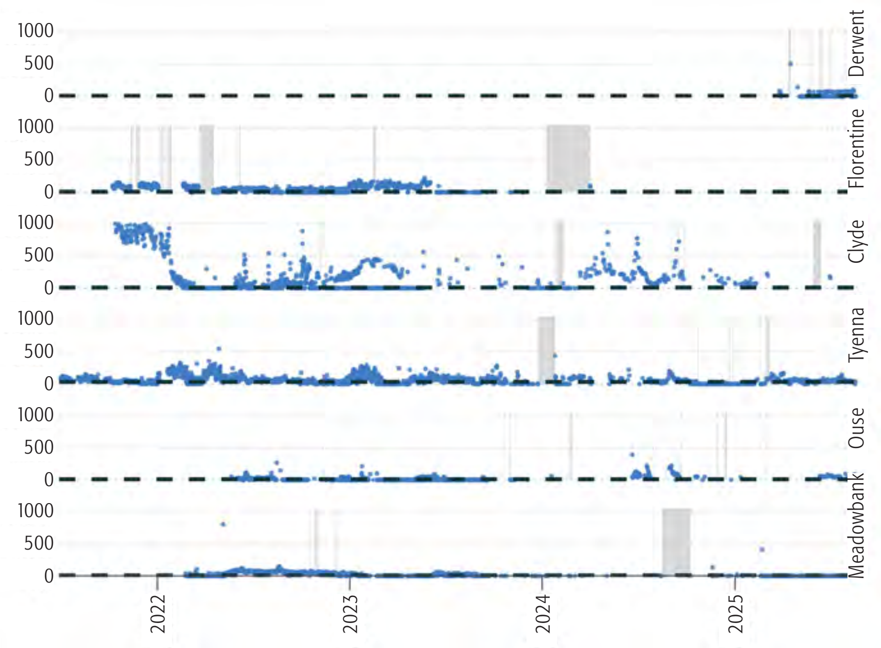


Figure 3.14. Nitrite concentrations in rivers within the sub catchment of the Derwent catchment. Grey bars indicate maintenance periods where no data was collected. There are some additional gaps in data where analysers were moved between locations.

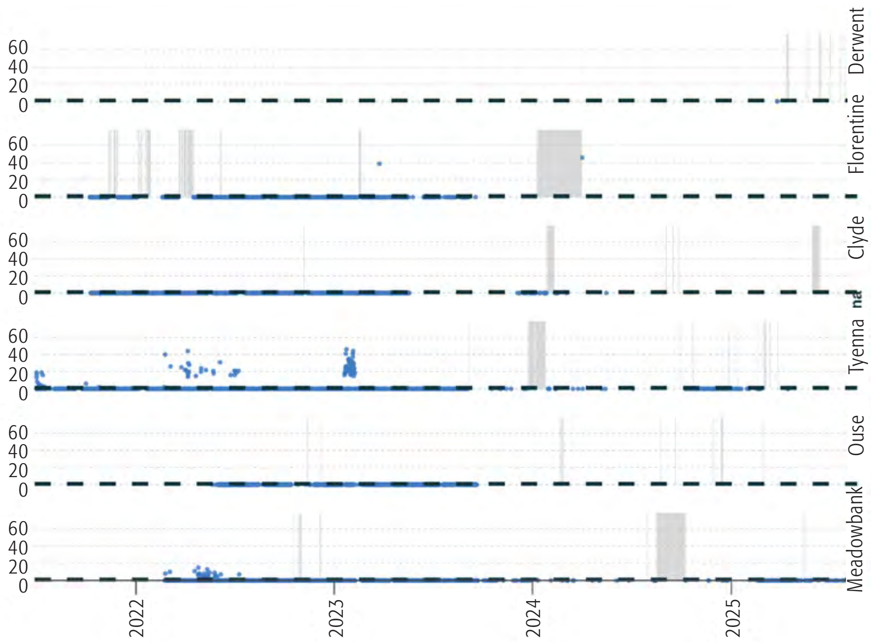
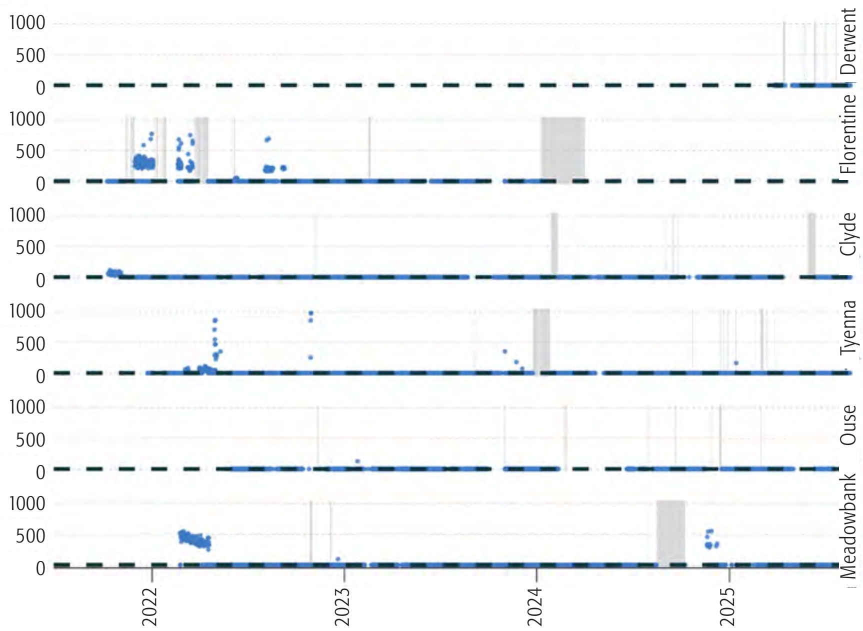


Figure 3.15 Phosphate concentration in rivers across the Derwent catchment. Grey bars indicate maintenance periods where no data was collected. There are some additional gaps in data where analysers were moved between locations.



A nutrient budget for the Derwent catchment

A nutrient budget was developed for the Derwent catchment in 2020 that identifies the relative percentages each industry contributes (Proemse *et al*., 2022a). This budget identified that diffuse sources from agriculture dominated the budget with about 17% and 47% of total nitrogen and total phosphorus respectively. By comparison fish hatcheries contributed 8% and 24% for total nitrogen and total phosphorus respectively (DEP, 2020).

Further work is required to set nutrient caps for the Derwent catchment which requires interrogation of current dissolved and total nutrients within each sub catchment in the Derwent catchment. These nutrient caps need to provide a total allowable limit for not only total nutrients but also dissolved nutrients. Current data needs to explore mass load budgets in tonnes/year, ideally also putting mass load budgets in context of catchment size (i.e., tonnes/year/ha). Currently the only comparison that can be made is to compare to the EPA's DGVs to current values (Table 3.1). Annual dissolved nutrient mass load estimates have been calculated for each river catchment (Proemse B. and Black J., 2025) and are compared in Table 3.2.

Table 3.2 Dissolved nutrient mass load values for the year 2022-2023.

River	Nitrate + Nitrite (tonnes/year)	Phosphate (tonnes/year)
Clyde River	7.224	*ID
Florentine River	10.263	1.307
Derwent River	61.61	4.394
Ouse	2.665	*ID
Tyenna	7.127	4.081

*ID= insufficient data

3.2.5 Recommendations

Nutrient levels in the Derwent catchment are higher than desirable(DEP, 2015c). A previous report has recommended developing a nutrient budget–potentially including load caps–for the River Derwent (Coughanowr *et al*., 2015). Progress toward this goal could involve setting catchment-wide, aspirational nutrient-reduction targets and/or developing a transparent, defensible methodology for deriving budgets/caps that estimates the carrying capacity of rivers across the catchment. It is recommended that collectively land users within the catchment aspire to make annual reductions in dissolved mass loads each year until they are closer to DGV values within each catchment.

4

Pollution sources and contaminant loads

4 Pollution sources and contaminant loads

In urban areas, effluent discharges can affect aquatic and estuarine ecosystems by introducing contaminants. These include nutrients, pathogens, and toxicants (such as metals and hydrocarbons), as well as emerging micropollutants like pharmaceuticals and personal care products, pesticides and industrial chemicals (Carey and Migliaccio, 2009; Yang *et al.*, 2017).

In Timtumili Minanya/River Derwent, including the Derwent Estuary, contaminants enter the system from a variety of sources. These are broadly classified into point sources and diffuse sources:

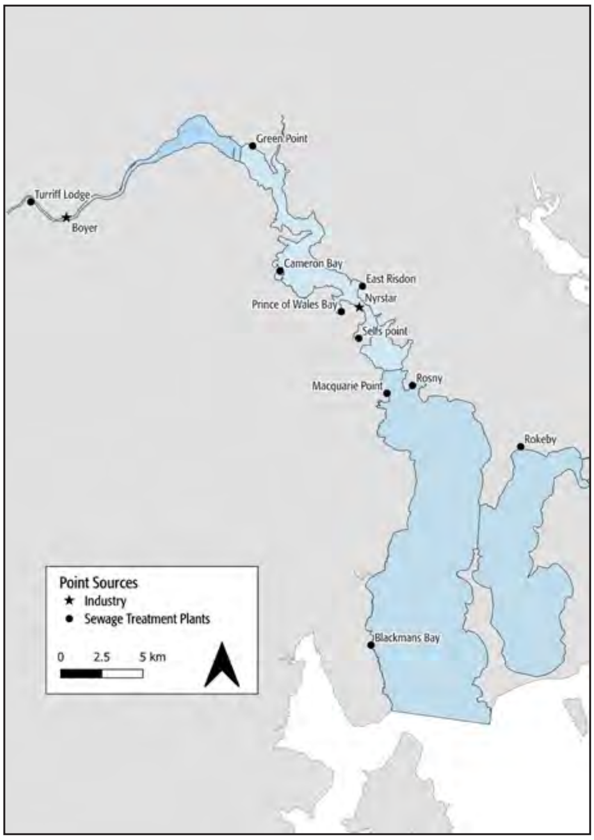
- Point sources are identifiable, localised discharges, including:
  - Sewage treatment plants (STPs), which release treated effluent.
  - Major industrial facilities, such as the Boyer Paper Mill and Nyrstar Hobart smelter.
- Diffuse sources are more widespread and harder to trace, including:
  - Catchment inputs from the River Derwent and Jordan River.
  - Stormwater runoff from urban areas.
  - Marine inputs from Storm Bay.
  - Airborne pollutants, landfills, aquaculture operations, and ports and marinas.

In addition to external inputs, sediments within the estuary can act as a secondary source of pollution. Under certain environmental conditions, these sediments may release previously deposited contaminants back into the water column.

Figure 4.1 illustrates the location of key point sources within the Derwent Estuary.

The following section provides an overview of contaminant loads discharged from the major sources in the Derwent Estuary including STPs, Boyer, Nyrstar Hobart, stormwater and the Derwent River for the period 2015–2024.

Figure 4.1 Derwent Estuary point sources. Circles mark STP locations and stars mark the location of other major industries.



4.1 Sewage treatment plants

Elevated nutrient discharge has led to eutrophication of waterways and is considered one of the greatest threats to ecosystem health (Bricker *et al*., 2008; Smith *et al*., 1999). Increased nutrient supply can lead to increased chlorophyll and biomass production that can disturb natural ecological balance in marine systems. The adverse impacts of eutrophication have been well documented and include harmful algal blooms, smothering of submerged aquatic habitats by bloom forming algae, hypoxia and subsequent dead zones, habitat degradation, changes in food webs and loss of biodiversity (Anderson *et al*., 2002; Akinnawo, 2023). The nutrients primarily responsible for these impacts are nitrogen and phosphorus, particularly biologically available forms, such as dissolved inorganic nitrogen and dissolved reactive phosphorus. Pathogens at levels above public health guideline values also pose risks to human health.

TasWater operates ten Level 2 STPs along the Derwent Estuary foreshore which discharge directly to the estuary (Figure 4.1). Four of the plants discharge to recycled water schemes: Green Point (Bridgewater), Cameron Bay, Rokeby and Rosny. Level 2 STPs operate under Environment Protection Notices (EPNs) and are regulated by the Environment Protection Authority (EPA) under the provisions of the *Environmental Management and Pollution Control Act 1994* (EMPCA). TasWater reports STP performance, compliance and areas for improvement in Annual Environment Reviews which are available on the TasWater website (**https://www.taswater.com.au/customers/businesses/environmental-reporting**).

Monitoring

In accordance with EPNs, TasWater sample effluent weekly at Derwent Estuary STP outfalls, except for Turriff Lodge, Green Point, East Risdon and Rokeby which are sampled monthly. The parameters monitored include nutrients (dissolved and total phosphorus, ammonium, nitrate and nitrite, total nitrogen), biochemical oxygen demand, total suspended solids, pathogen indicator organisms (E. coli & enterococci), oil and grease and residual chlorine.

Treatment

The type of effluent treatment, and consequently effluent quality, varies between plants. Seven of the treatment plants operate at the secondary level, which involves removal of solids and organic matter, while three plants (Selfs Point, Rokeby and Blackmans Bay) operate at the tertiary level, removing solids, organic matter, and nutrients. Most of the STP effluent is discharged into the middle-to-lower estuary, with two plants (Macquarie Point and Prince of Wales Bay) accounting for nearly 70% of the total nitrogen discharged into the estuary in 2024 (Figure 4.2).

Contaminant loads

In this section we calculate contaminant loads to summarise changes in effluent discharge and effluent reuse over time and report data for the period 2015–2024.

The following steps were followed to calculate annual contaminant loads for all Derwent Estuary STPs:

Calculate monthly contaminant load using the following equation:

$L_m = (C_m \text{ x } V_m)/1000$

where:

L_m = monthly contaminant load (t)

C_m = monthly average concentration (mg/L)

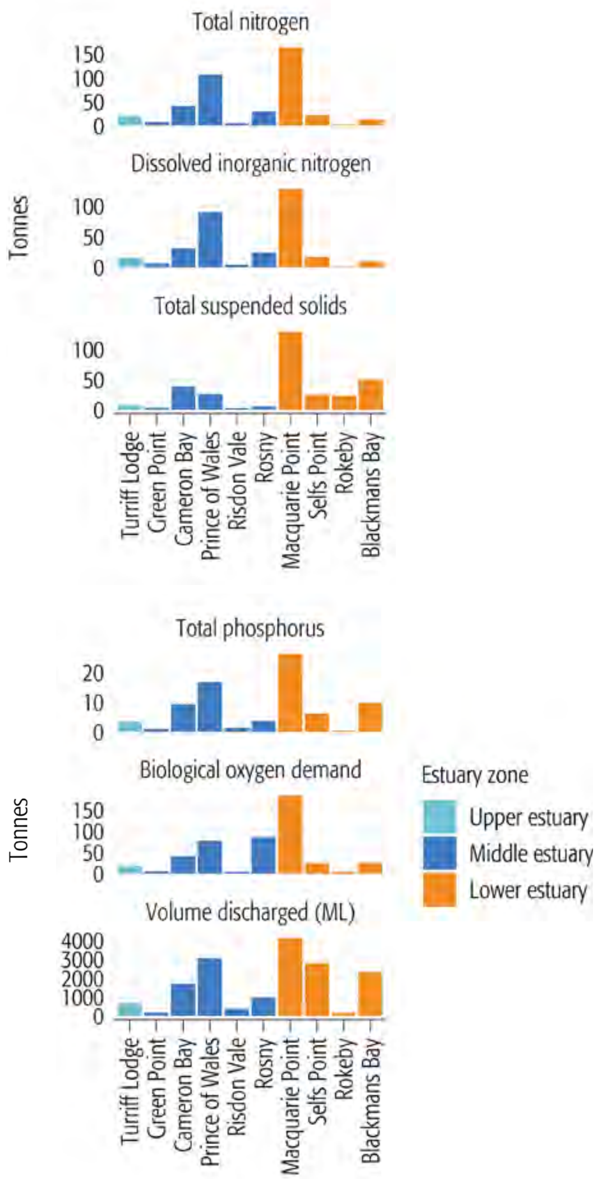
V_m = total monthly discharge volume (ML)

This value was extrapolated to an annual load (t/year).

Spatial trends

The majority of effluent discharge occurred in the middle and lower estuary between Cameron Bay and Selfs Point, which discharges to Blinking Billy Point. In 2024, three middle and lower estuary STPs – Macquarie Point, Prince of Wales Bay and Selfs Point – accounted for 60% of the total discharge volume and contributed more than 70% of the total nitrogen load to the estuary. Among these, Macquarie Point was the largest contributor, accounting for approximately 40% of the total nitrogen discharged in 2024 (Figure 4.2).

Figure 4.2 Annual contaminant loads (t) and volume (ML) discharged from Derwent Estuary STPs in 2024.



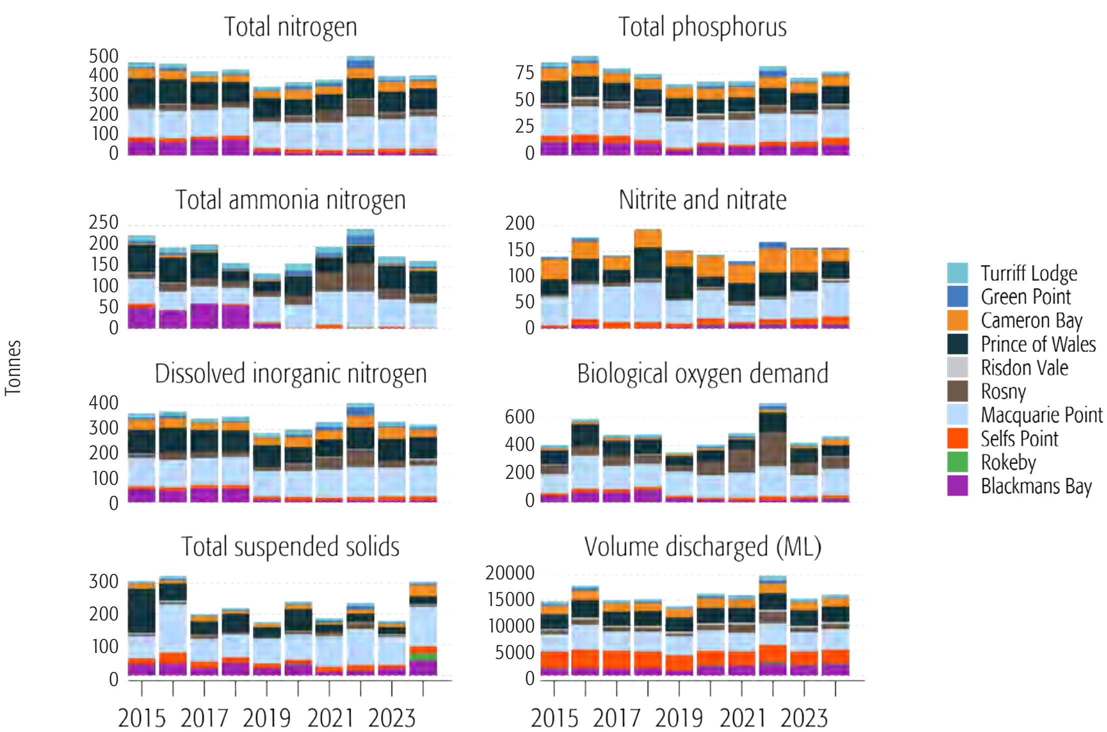
Temporal trends

While contaminant loads in 2024 were generally within the average range, historical data show periods of both increase and decrease throughout the reporting period. Elevated loads in 2022 across the estuary were largely driven by above-average rainfall, which led to storage systems reaching capacity and a consequent reduction in reuse. This resulted in increased effluent volume and, consequently, higher loads discharged to the estuary. Increased discharge during this period was most notable at Rosny and Green Point STP (Figure 4.3).

Between 2018 and 2019, total ammoniacal nitrogen (ammonia + ammonium ion) and total nitrogen loads decreased significantly due to optimisation efforts targeting ammonia removal at the Prince of Wales Bay STP. However, these changes led to a corresponding increase in nitrate + nitrite and enhanced denitrification to nitrogen gas, which in turn increased aeration demand and raised the risk of sludge flotation in the secondary clarifier. As a result, operations were reverted to standard settings in 2021 (Figure 4.3).

The commissioning of the Blackmans Bay STP in 2019 contributed to a substantial reduction–approximately 80%–in nitrogen discharge from the facility. Since this upgrade, contaminant loads (estuary combined) have remained below average (Figure 4.3).

Figure 4.3 Interannual comparison of annual contaminant loads (t) discharged from all Derwent Estuary STPs in for the period 2015–2024. Volume discharged is in megalitres.

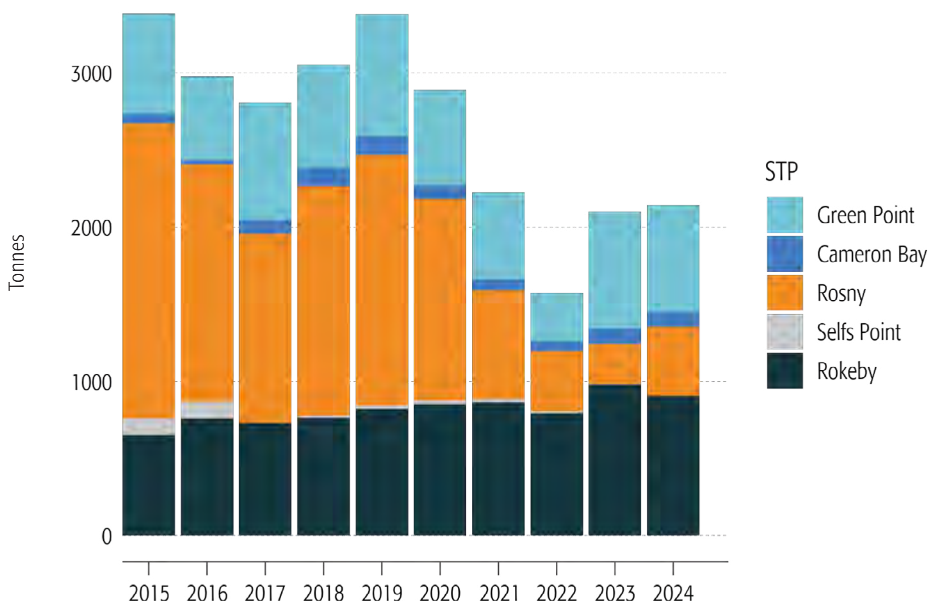


Reuse

Effluent reuse has declined significantly since 2020, leading to increased discharge volumes to the estuary. In 2022, prolonged rainfall events reduced demand for recycled water, resulting in reduced reuse at Green Point and Rosny STP. While reuse at Green Point STP returned to average in 2023, reuse volume at Rosny has remained low (Figure 4.4).

Ongoing variability in reuse at Rosny STP is due to periodic saltwater intrusion into the low-lying pipes servicing the Rosny plant. When saltwater enters the network, effluent becomes unsuitable for agricultural reuse, requiring disposal to the Derwent instead. This increased discharge volume is likely to impact downstream water quality.

Figure 4.4 Volume of effluent reused at Derwent STPs.



Management

Due to the high volume of effluent discharge, ambient nutrient concentrations are typically highest in the middle-to-lower estuary, particularly in poorly flushed bays where water circulation is limited. While overall nutrient loads tend to be greatest in the middle and lower estuary, smaller treatment plants that discharge into poorly mixed or nearshore environments may have a disproportionately greater ecological impact due to the sensitivity of these receiving waters. TasWater recognises the challenges of managing these environmental impacts and continues to assess and optimise the performance of its STPs to minimise harm.

Management actions and incidents for the reporting period include:

- Macquarie Point STP to be decommissioned, and a new pumpstation at the Macquarie Point site will direct all flows to an upgraded Selfs Point STP. This project is expected to result in a significant reduction of mass loads of nutrients (total nitrogen 58% and total Phosphorus 42%) discharged to the Derwent Estuary. An overview of the Selfs Point transformation project is provided in the following section.
- A new outfall was installed at Turriff Lodge STP in April 2023. The STP previously discharged into constructed wetlands which were not improving effluent quality and posed a risk to public health. The new outfall extends 14 metres into the river to a depth of six metres and a new diffuser improves effluent dilution and mixing. A 12-month ambient monitoring program was conducted to assess effluent dilution, water quality and biological conditions in the Derwent River, upstream and downstream of the STP discharge. Results from this study are presented below.
- The chlorine dosing system at Rosny STP is currently being replaced with a UV disinfection system and the project is due for completion in September 2025. Following completion, recycled water supply to Clarence customers will resume. This upgrade will eliminate the risk that chlorine has on the receiving environment and will significantly reduce the levels of bacteria being discharged to both the Derwent Estuary and the recycled water scheme.
- The Arm End Irrigation Scheme was established to provide adequate and reliable supply of irrigation water primarily to the Arm End Golf Course. The project involves the construction and operation (not by TasWater) of a Class A Recycled Water Treatment Plant which will take effluent from the Blackmans Bay STP and pipe it across to the South Arm reuse scheme. Up to 2.6 ML of effluent per day could be diverted from the estuary to the recycled water scheme. Note, this reuse scheme is not operated by TasWater.
- In late December 2024, a significant spill occurred in the River Derwent, originating from the Cameron Bay STP. Due to the scale of the incident and the potential area affected, the Tasmanian Department of Health issued a public health advisory against swimming in the Derwent during the impacted period. To support the response, the DEP provided direct sampling

through the Beach Watch program, ensuring up-to-date enterococci data was collected for all swimming sites under the advisory. All sites returned compliant results, except for one that had previously failed, allowing the advisory to be lifted. Following the incident, the EPA launched an investigation into the causes and potential prevention strategies to avoid similar events in the future. At the time of writing this report, this investigation is ongoing.

- In 2019 Blackmans Bay STP was upgraded to tertiary treatment and capacity increased to accept process waste from Snug, Electrona and Margate. The upgrade has resulted in a significant improvement in effluent quality, with a reduction in annual loads of dissolved inorganic nitrogen of 80%.

Selfs Point transformation project

The Selfs Point Sewer Transformation project will amalgamate sewage treatment from the existing Macquarie Point STP and existing Selfs Point STP into a single modern plant. The new STP will enable treatment of all predicted flows from the Selfs Point and Macquarie Point catchments until the year 2054, which equates to 24.9 ML/day average dry weather flow (ADWF).

The project commenced in 2024 with the development of the new STP at the existing site. Construction works associated with the upgrade and decommissioning works to extend into 2027. The new project will include construction of:

- New inlet works
- A new bioreactor
- A new 8.5 ML treated effluent storage tank
- A new sludge treatment system
- A new odour treatment system
- Replacement and relocation of the existing local Selfs Point effluent outfall pipeline.

The project will result in a net environmental benefit to the Derwent Estuary by reducing the amounts of nitrogen and phosphorus entering the estuary from the current Selfs Point and Macquarie Point STPs (combined flows) by 58% and 42%, respectively. The project will also prevent non-emergency discharges from the Macquarie Point effluent outfall, and will, as far as practicable, discharge dry weather effluent flows to the Blinking Billy effluent outfall which provides superior mixing and dilution of effluent, with seven wet-weather discharge events per year, approximately 1.4% of the total annual effluent volume, at the local Selfs Point effluent outfall.

Overall, the project is expected to be of significant benefit to the community, support growth, improve the aquatic environment and enable the progression of the development of Macquarie Point, a key project for Hobart and Tasmania.

Turriff Lodge STP post new outfall commissioning ambient monitoring

Ambient water quality and biological monitoring to assess the performance of the new outfall and to determine potential impacts within the Derwent Estuary receiving environment was completed in April 2024. Key findings of the investigation are summarised below:

- The behaviour of the effluent plume aligned with the new outfall hydrodynamic modelling predictions. In stratified river conditions the plume rises to the level of the halocline (i.e., mid water) and disperses upstream (incoming tide) or downstream (outgoing tide). In unstratified freshwater conditions, the plume rises to the surface and dilutes rapidly through the water column.
- The effluent discharge had no observable impact on salinity, pH or turbidity in the receiving environment in surface, middle or bottom waters. Dissolved oxygen was impacted by stratification of the water column: dissolved oxygen was high when there was no stratification (freshwater) but was low in stratified bottom waters, particularly over the warmer months.
- The impact of the STP effluent discharge into the upper Derwent Estuary was either low or negligible for measured ecosystem stressors compared to background conditions. Nutrients (ammonia, nitrate, total phosphorus and dissolved reactive phosphorus) were occasionally elevated around the outfall but diluted to background levels within 50 m upstream and 25 m downstream of the outfall. These elevations in the vicinity of the outfall were not consistent with elevations of these nutrients in the STP effluent at the time.
- Total and dissolved organic carbon concentrations were highest following high rainfall and increased river flow (in mixed, unstratified freshwater conditions). Total organic carbon and dissolved organic carbon were occasionally slightly elevated around the STP outfall, but not consistently greater than background concentrations.
- Occasional elevations in metal concentrations (aluminium, copper, iron, manganese and zinc) above background concentrations and exceeding the marine ANZG (2018) toxicant default guideline values were observed in middle and bottom waters in the immediate vicinity of the outfall (5m) and aligned with elevations in the in the effluent.
- There was no measurable impact associated with chlorinated disinfection by-products around the outfall.
- Pathogen indicator organisms enterococci and *E. coli* concentrations occasionally exceeded the EPA low risk recreational guideline values in middle and bottom waters in the receiving environment but there was no evidence of a consistent impact from the STP

discharge with levels likely related to urban and agricultural inputs from the township of New Norfolk.

- Biological monitoring during peak growth in spring, summer and autumn suggested species assemblages and abundance of algae were generally similar between control and impact zones, with a slightly greater relative abundance of cyanobacteria in the impact zone.

Overall, the effluent discharges are considered to pose a low risk to the protected environmental values in the immediate vicinity of the STP outfall due to adequate mixing and dilution within the Derwent River Estuary.

4.2 Boyer Paper Mill

The Boyer Paper Mill (Boyer) is located at Boyer on the northern bank of the upper Derwent Estuary, approximately 4 km downstream from New Norfolk. Norske Skog Boyer was purchased by Boyer Capital in 2025 and henceforth will be referred to as Boyer.

Boyer is a Level 2 industrial premises, operating under an EPN, and is regulated by the EPA under the provisions of EMPCA.

An overview of operations, treatment process, monitoring contaminant loads and management actions is provided below.

Operations

The mill has been operating since 1941 and produces newsprint, specialty newsprint and lightweight coated papers with a paper production capacity of 140,000 tonnes of newsprint and 120,000 tonnes of lightweight coated magazine paper.

Since October 2009 paper has been made from 100% thermo-mechanical pine pulp of which > 99% is Forest Stewardship Council and Programme for the Endorsement of Forest Certification certified. This change resulted in a 98% reduction in resin acids discharged to the estuary (DEP, 2015c). Hydrogen peroxide is the brightening agent used at the mill.

The mill operates an on-site water treatment plant, a secondary effluent treatment plant (SETP) and a small sewage treatment plant. A coal-fired boiler supplies most of the thermal energy to the site, and solid wastes (e.g. wood wastes, water and effluent treatment plant biomass, and ash from the coal-fired boiler) are either reused, recycled or disposed of at the Boyer Mill solid waste landfill.

Treatment

In 2008, a biological SETP was commissioned with the key objective of reducing the organic carbon load discharge to the river. The SETP operates a primary clarification step which removes solid material (largely wood fibre) through gravity clarification. From there the effluent is treated through a microbiological process whereby naturally occurring microorganisms digest most of the remaining organic material. The microbiological material is then settled out in a second clarifier, also by gravity clarification. Most of the microorganisms are recycled in the system and a few are wasted to keep a stable population.

The SETP has been effective in reducing organic carbon with a 66% decrease in dissolved organic carbon, 95% reduction in biological oxygen demand and a 40% decrease in total suspended solids (Ross *et al*. 2010; DEP 2015). Whilst organic carbon loads were reduced, the shift in particulate matter discharged from the mill changed from refractory wood fibre particulate to that dominated by microorganisms that spill over from the secondary treatment process. This particulate matter is more labile and biologically available resulting in increased respiration rates and subsequent release of ammonia from sediment at sites in close proximity to the outfall (Ross *et al*. 2010). Given this challenge, Boyer have continued efforts to reduce total suspended solid loads in the combined effluent stream (CES).

The change in treatment also resulted in increased discharge of dissolved nutrients as the secondary treatment process requires the addition of nutrients to sustain the biological secondary treatment process (Edward *et al*., 2008; Barnett, 2013).

Monitoring

Emission sources from the site are monitored on a regular basis, in accordance with EPA permit conditions. Boyer monitor the CES for contaminant concentrations and physicochemical parameters. Temperature, pH, total suspended solids and flow are sampled in the CES daily whilst biological oxygen demand, resin acids, nutrients (total nitrogen, total phosphorus, nitrate + nitrite, ammonia and dissolved reactive phosphorus) are sampled weekly and E. coli and total recoverable hydrocarbons are sampled monthly.

Contaminant loads

Contaminants originating from the site, with respect to estuarine water quality, are nutrients (dissolved), organic matter, suspended solids, wood extractives (resin acids), aluminium, sulfur, faecal bacteria, and air emissions associated with the coal-fired boiler. The majority of these contaminants enter the Derwent estuary via the CES.

Boyer was historically a source of zinc and mercury to the estuary. The chlor-alkali plant (Caustic Chlorine Plant), which used a mercury-pool cell, was a significant source of mercury to the upper estuary and was decommissioned in 1993. Zinc was also released from the site due to the former use of zinc hydrosulphite as a brightening agent.

In this section we calculate contaminant loads to summarise changes in effluent discharge over time and report data for the period 2015–2024.

The following steps were followed to calculate annual contaminant loads:

Calculate load on each day a pollutant concentration is sampled (weekly):

$L_d = Cd \times Vd$,

where:

L_d = daily contaminant load using 24 h composite sample taken every week (kg)

C_d = concentration of the pollutant in the 24 h composite sample (mg/L)

V_d = total daily discharge volume (ML)

This value was extrapolated to monthly and annual loads.

Temporal trends

Over the reporting period, nutrient loads, particularly nitrogen, have been variable. Nitrogen loads have remained above average over the past three years (Figure 4.5). Loads discharged during 2018 are considered 'optimal' in comparison to other years.

Variability, especially in ammonia, is largely driven by the organic load coming into the SETP. The production of speciality newsprint paper can result in up to double the organic load for secondary treatment, requiring increased nutrient dosing to optimise the treatment process. The commissioning of the in-situ analysers in mid-2024 has resulted in encouraging early signs, with ammonia in the CES declining significantly since commissioning (Figure 4.6).

Totals suspended solids were significantly elevated in 2021 due to filamentous bacteria outbreaks that disrupted biological treatment performance, leading to biomass (microbiology and small amounts of organic matter) overspilling from the SETP. Since 2021, total suspended solids have remained below average for the reporting period.

Figure 4.5 Interannual comparison of annual contaminant loads (t) to the Derwent Estuary from the Boyer combined effluent stream (CES).

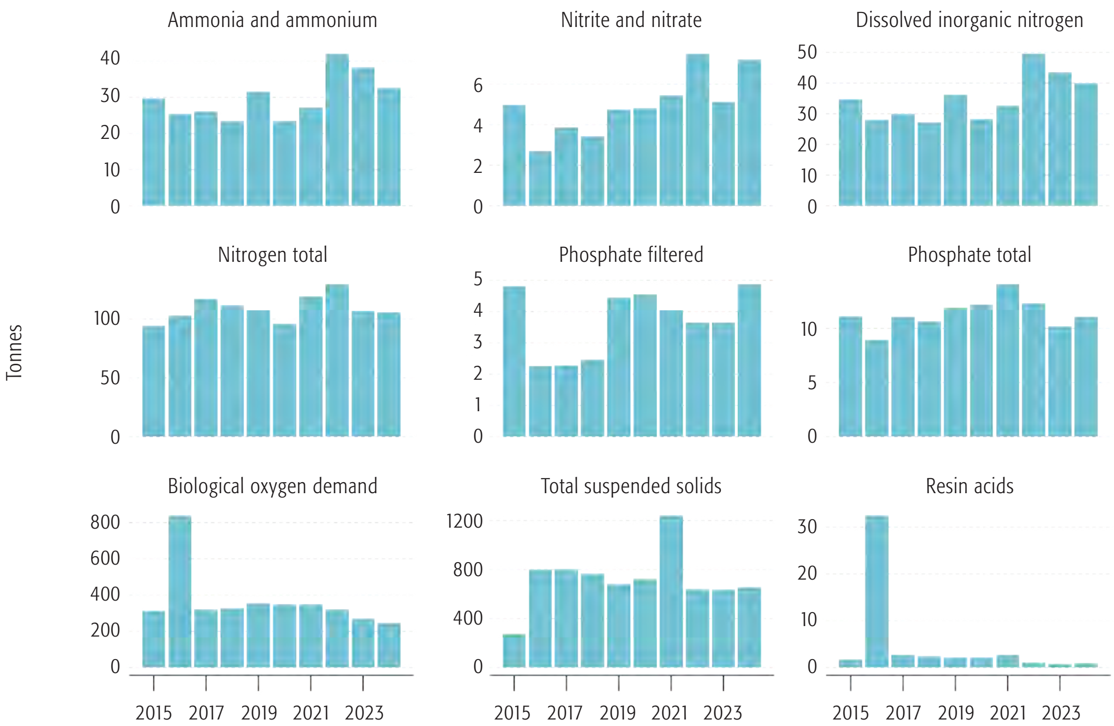
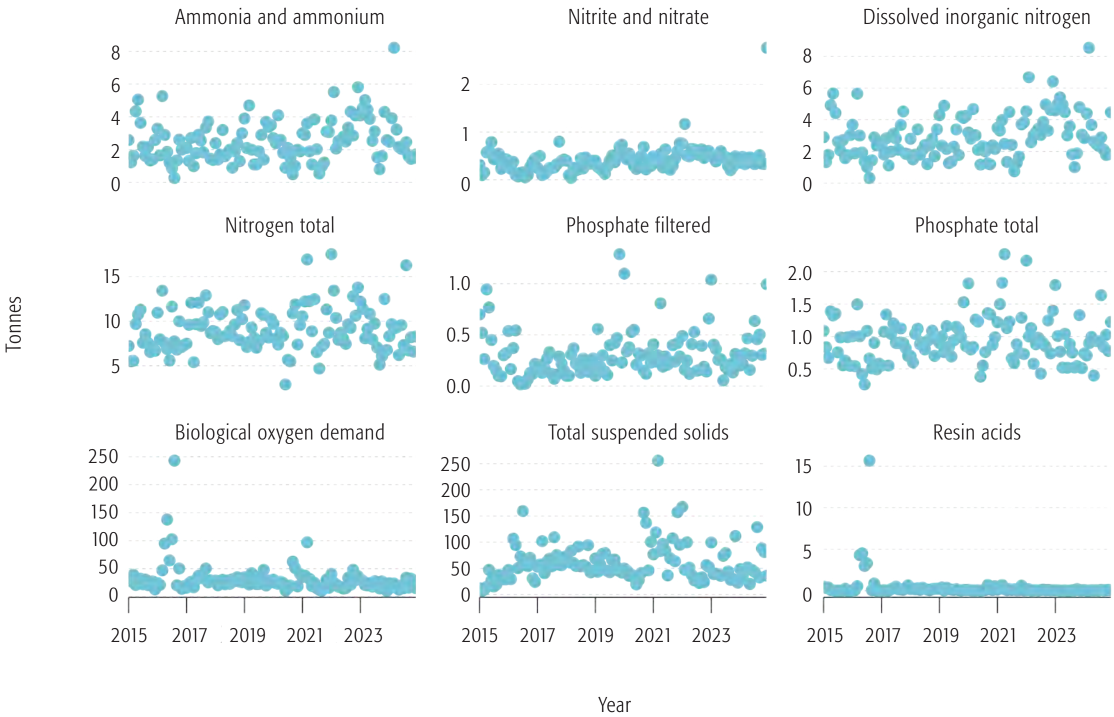


Figure 4.6 Monthly contaminant loads (t/month) discharged to the Derwent Estuary from the Boyer combined effluent stream (CES).



Management

In 2024, Boyer applied for and was granted government funding for improvements to the SETP. This was used to purchase, install and commission, new *in situ* analysers for chemical oxygen demand (COD) and ammonia nitrogen.

These analysers have given Boyer greater control over the urea dosing system based on the incoming organic load and outgoing residual ammonia. The objective is to have a better understanding of the fluctuations within the effluent treatment plant and efficiently treat the effluent entering the plant. The analysers have been operational since mid-2024 and have showed promising results (Figure 4.6). It is hoped that this will help return loads, particularly ammonia and total suspended solids, to optimal levels, as represented by loads discharged in 2018.

4.3 Nyrstar Hobart Smelter

The Nyrstar Hobart zinc smelter has been operating since 1917 and is one of the world's largest and most efficient producers of zinc metals and alloys, with an annual production capacity of 280,000 t of marketable metal. Nyrstar is a Level 2 industrial premises, operating under an EPN, regulated by the Environment protection Authority (EPA).

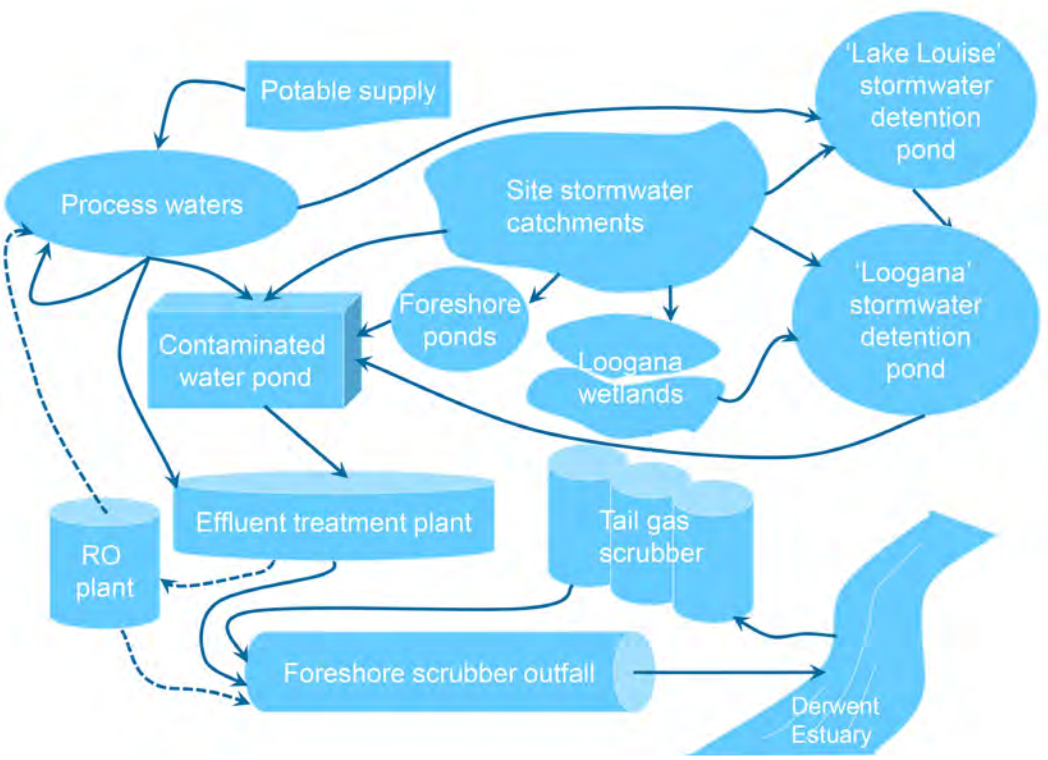
An overview of contaminants, sources, monitoring programs and management actions is provided below. For more information, see the 'Nyrstar Hobart Triennial Public Environment Report 2022–2024' (Nyrstar Hobart, 2025).

Contaminants and sources

Historically the Nyrstar Hobart site has been a major source of contamination to the Derwent Estuary and the local airshed. Contaminants associated with zinc production include metals and metalloids–particularly zinc, mercury, cadmium, lead, copper, and arsenic–as well as fluoride, particulates, sulfur oxides/sulfates, and some nutrients. As part of the zinc and acid-making process, estuarine water is drafted for air emission control. The water is utilised to clean acid plant exhaust gas prior to emission, by removing sulfur compounds. This process acidifies the water prior to discharge into the Derwent Estuary.

These pollutants have entered the Derwent Estuary through multiple pathways, including the foreshore outfall via the combined effluent stream (CES), groundwater, surface and stormwater (now managed via a closed-circuit system), and air and dust emissions. In recent years, the majority of metals entering the Derwent Estuary have originated from contaminated groundwater. Process waters and pathways to the Derwent are presented in Figure 4.7. More details about treatment processes and process waters can be found in the Nyrstar's Public Environment Report (2025).

Figure 4.7 Nyrstar Hobart's process and stormwater system. Recycled water flows that came online in 2016 shown with a dotted line (Source: Nyrstar Hobart Triennial Public Environment Report, 2022 – 2024).



Monitoring

Emission sources from the Nyrstar site are monitored on a regular basis in accordance with EPA permit conditions. Under the EPN Nyrstar monitors the CES daily for zinc, cadmium, lead, mercury and flow rate. Other analytes are measured biannually for EPA and National Pollutant Inventory (NPI) reporting. They include arsenic, total suspended solids, fluoride, iron, manganese, ammonia, total nitrogen, total phosphorus, beryllium, cobalt, and nickel.

Nyrstar also operate an extensive groundwater monitoring program including water level monitoring at approximately 106 bores twice a year and groundwater sampling at least once every two years. This frequency has been increased and is conducted using a risk-based approach where high risk bores are sampled biannually, moderate risk sampled annually and low risk biennially.

Contaminant loads

In this section we calculate contaminant loads to summarise relative contributions and changes in effluent discharge over time and report data for the period 2016–2024.

The following steps outline the approach used to calculate annual contaminant loads discharged to the estuary via the outfall, groundwater and loads (zinc and cadmium) recovered from contaminated groundwater.

Calculate daily contaminant loads using the following equation:

$L_d = C_d \times V_d/1000,000,000$

Where:

- L_d = estimated daily contaminant load (t)
- C_d = concentration of the pollutant (mg/L)
- V_d = total daily discharge volume (L)

This value was then extrapolated into an annual load (t/year).

Annual contaminant load to the estuary via groundwater was estimated using the following equation:

$L = L_m - (L_r - L_a)$

Where:

- L = estimated annual contaminant load to the estuary via groundwater (t/year),
- L_m = modelled annual load to the estuary via groundwater prior to installation of the grout curtain extraction system in 2021. The modelled load is 120 t/year
- L_r = annual load (t) recovered from groundwater in the current year, calculated using above equation (applied to monthly as opposed to daily data).
- L_a = average annual load recovered from groundwater prior to 2021. This value is 80 t/year.

This approach estimates how much additional contaminant load (zinc and cadmium) was removed due to the grout curtain system. It then subtracts this from the modelled groundwater load to the Derwent Estuary to provide an annual estimate. For example, in 2022, 120 t of zinc was recovered from groundwater. The estimated load to the estuary in 2022 is therefore: 120 – (120 – 80) = 80 t/year.

Groundwater contamination

Legacy contamination of groundwater at this site remains the greatest ongoing source of metals to the Derwent Estuary. This contamination stems from over 100 years of industrial activity, with key sources including: leakage of process solutions in operational areas; infiltration of contaminated surface water; percolation through stockpiled feedstocks and residues; and leaks from above- and below-ground storage tanks and pipelines.

Extensive groundwater monitoring has shown a general improvement in groundwater quality across the site over the last two reporting periods (2020–2024). Notably, concentrations of heavy metals in wells located along the foreshore are generally lower than in other wells across the site (Nyrstar Hobart, 2025). This monitoring program has facilitated the identification and elimination of numerous contamination sources, as well as the acceleration of remedial efforts in known hotspots.

Groundwater recovery

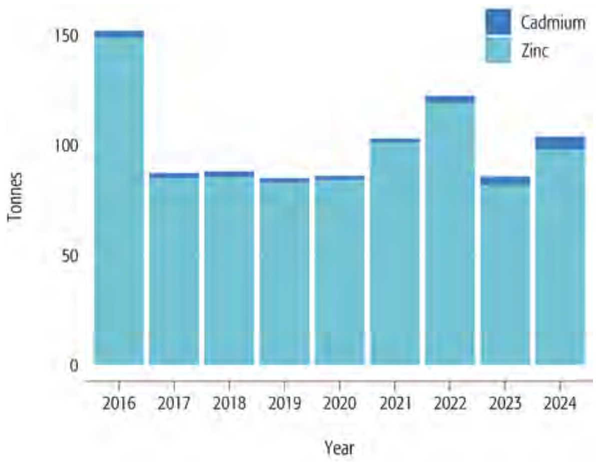
Zinc and cadmium are recovered from groundwater via 10 extraction systems targeting contaminant hotspots across the site, including the grout curtain extraction system installed in 2021. Zinc and cadmium are reported here as they are generally present in elevated levels relative to Australian and New Zealand Environment and Conservation Council (ANZECC) Guidelines for Fresh and Marine Water Quality. Groundwater recovery has improved since the installation of the grout curtain extraction system, with optimal performance occurring in 2022 (120 t; Figure 4.8). Prior to 2021, the average load recovered was 80 t. Above average recovery in 2016 was due to increased recovery volume during the period.

Groundwater loads to the estuary

Prior to the installation of the grout curtain extraction system in July 2021, modelling estimated a contribution of approximately 120 t per year (Figure 4.12). The increased capacity provided by the new system has likely reduced the volume of metals reaching the Derwent via groundwater flows. In 2022, zinc input via groundwater was estimated as 80 t (Figure 4.12; Figure 4.8).

However, while the grout curtain system has improved zinc recovery from groundwater, its performance has fallen short of initial expectations largely due to heavy fouling and subsequent pump system failure which have reduced its overall capacity to operate at design efficiency.

Figure 4.8 Annual estimated zinc and cadmium loads (t/year) recovered from groundwater at the Nyrstar Hobart zinc smelter.

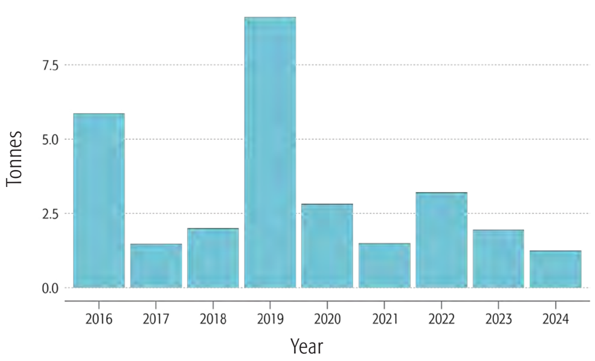


Atmospheric emissions

Over fifteen stacks are located on the Nyrstar Hobart site, and a number of methods are used to capture airborne contaminants to reduce adverse effects on human health and the environment. These include wet scrubbing, baghouses, chemical absorption towers and electrostatic precipitators. These stacks, which rely on gas cleaning processes to meet air quality standards, are monitored in accordance with the EPN, with requirements varying from continuous automatic monitoring to six-monthly testing.

Based on NPI reporting, Nyrstar's air and fugitive emissions account for the second largest proportion of metal loads from the site (Figure 4.12). Average annual air emission of zinc for the reporting period was 3.2 t/year, with loads declining over the past 5 years (Figure 4.9). An estimate of loads falling into the Derwent estuary from air emissions (stacks and dust) has not been determined. NPI air emission data has been included to provide context only.

Figure 4.9 Annual estimated zinc air emissions (t/year) at the Nyrstar Hobart zinc smelter.



Outfall

Effluent discharge and monitoring

Nyrstar discharges about 60–100 ML/day of aqueous effluent at a monitored outfall point. Approximately 95% of this effluent consists of saltwater taken from the Derwent and passed through a scrubbing system to remove residual sulfur dioxide from tail gas exiting the acid plants. The remaining 5% is treated wastewater, groundwater and stormwater discharged from the Effluent Treatment Plant (ETP), which forms part of the CES. Contaminants discharged and monitored in the outfall are discussed in the 'monitoring' section.

Of the daily metals monitored, only zinc and mercury are typically present above laboratory reporting limits. Here, we report loads for zinc and mercury only.

Zinc and mercury loads

Annual zinc loads ranged from 0.9 to 6.5 tonnes, with an average of 2.3 tonnes. Discharges have declined since 2021, with 2024 levels below average for the reporting period. The elevated annual load in 2016 is not representative of general operations (Figure 4.10). Mercury loads increased in 2020 and 2021, and although reduced in 2022–present, are currently 1.8 times higher than 2017 levels (Figure 4.10; Figure 4.11). The increase is linked to decreased efficiency of the electrostatic mist precipitator (EMP), which allows higher carryover of metalliferous particulates into the acid plant and, subsequently, the acid plant exhaust.

Nyrstar is actively working to reduce mercury discharges through ongoing EMP and tail gas scrubber maintenance to achieve optimum discharge loads.

Figure 4.10 Annual estimated zinc loads (t) discharged to the estuary from the Nyrstar Hobart zinc smelter outfall.

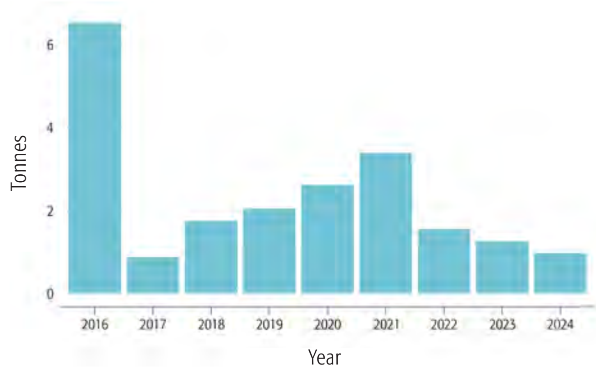
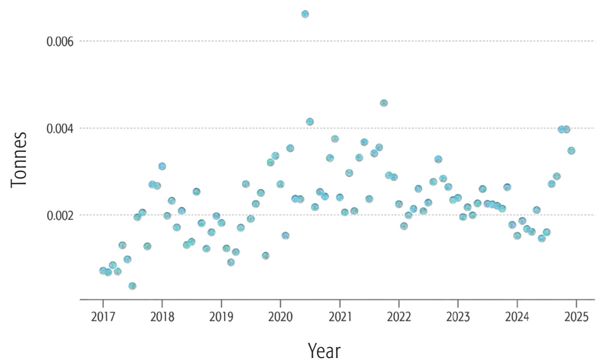


Figure 4.11 Monthly estimated loads (t) of mercury discharged to the estuary from the Nyrstar Hobart zinc smelter outfall.



Stormwater

Nyrstar Hobart operates with a fully closed-circuit stormwater system, with all stormwater being captured on the site and directed to the ETP for treatment prior to discharge into the Derwent Estuary.

A stormwater overflow event occurred on 6 May 2022 resulting in discharge of 0.5 t of zinc to the estuary. The incident was a result of a heavy rainfall event, with 100 mm of rain recorded at the site during the storm event. A similar event occurred on 10 May 2018 resulting in zinc discharge of 1.1 tonnes following a storm event recording 133 mm of rainfall at the site (Figure 4.12). Whilst the site's infrastructure can manage heavy rainfall, intense rainfall over a brief period can overwhelm the site's pumping and treatment capabilities.

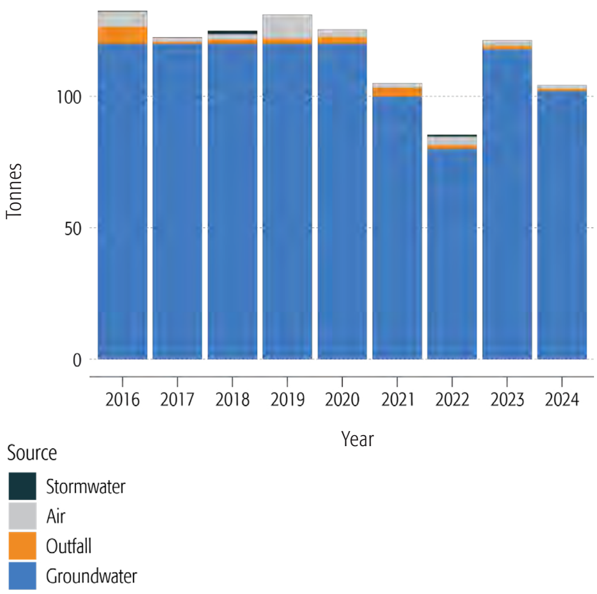
Nyrstar contaminant load summary

During the reporting period, groundwater was the dominant source of zinc to the estuary, contributing an average of approximately 110 tonnes per year (Figure 4.12). Other sources included:

- Air emissions: 3.2 t/year (not assumed to be directly deposited to the estuary; included for reference only)
- Effluent Treatment Plant discharge (via foreshore outfall): 2.3 t/year
- Stormwater overflow events: 0.3 t/year

These figures highlight the significance of subsurface pathways in zinc transport compared to other sources.

Figure 4.12 Annual comparison of estimated zinc loads originating from groundwater, the foreshore outfall, air emissions and stormwater overflows from Nyrstar Hobart in t/year.



Management

Nyrstar Hobart have completed significant site remediation works in recent years to reduce contaminants entering the Derwent Estuary. Actions include:

- Reduce mercury discharges through ongoing EMP and Tail Gas Scrubber maintenance.
- Construction of a 760-m long grout curtain coupled with an up-gradient groundwater collection system, isolating the most contaminated section of the site from the Derwent Estuary, was completed in July 2021. The upgraded groundwater extraction system was commissioned in October 2021. Groundwater extraction bores allow for the extraction and removal of metals from contaminated groundwater. Modelling indicated at project conception that this new system will substantially increase the recovery of metals from groundwater, preventing their ultimate migration into the Derwent Estuary.
- Modification to the stormwater system onsite to ensure that all captured stormwater is directed to retention basins and the site's purpose-built water treatment plant. The treatment plant can treat approximately 300 kL per hour of water, dramatically reducing the heavy metal content of the water prior to discharge into the Derwent Estuary.
- Sealing the floor of the electrolysis basement, formerly a significant source of ongoing metal contamination to groundwater. This project was completed in 2018.
- Ongoing repairs to site bunds, ensuring process solution is prevented from contaminating the underlying groundwater.

Planned actions

Looking forward, some of the key focuses for the site in relation to estuarine health will be the optimisation of existing groundwater extraction systems, preventing further groundwater contamination through asset maintenance as well as assessing options for the installation of new groundwater extraction systems.

4.4 River Derwent catchment

The River Derwent is a significant source of contaminants in the estuary, particularly total nitrogen and TSS (Figure 4.15). Previous monitoring has indicated that primary sources of nutrients in the catchment are diffuse sources (agriculture, forestry) and point sources (aquaculture) (Proemse *et al*., 2018). Early warning signs of nutrient stress, including the growth of algae leading to taste and odour issues in Hobart's water supply and filamentous algal blooms smothering seagrass in the upper estuary, have raised concerns about the water quality of the River Derwent upstream of New Norfolk. Catchment land use and pressures are discussed in detail in Section 3.1.

Whilst previous monitoring has focused on nutrients, there are significant knowledge gaps in the catchment (and estuary) for emerging contaminants including per- and polyfluoroalkyl substances, microalgal toxins, pesticides and pharmaceuticals and personal care products.

4.5 River flow and contaminant loads

River Derwent contaminant load estimates are presented to provide an indicative value of the contribution of loads to the estuary from the catchment. River loads were calculated using monthly water quality monitoring data (AWQ) and flow data at New Norfolk.

River flow calculations at New Norfolk are detailed in 3.2.2 River hydrology and flow regime.

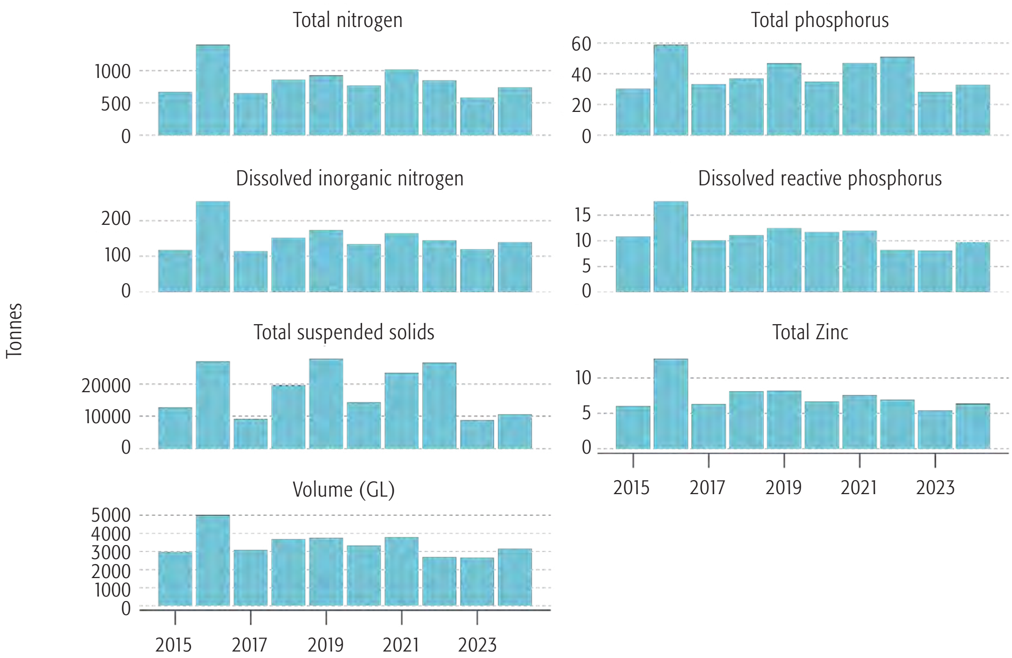
Derwent River trends

Loads are based on monthly observations and should be viewed as indicative only due to uncertainties based on: variability in nutrient concentrations associated with rainfall/runoff events and other discharge events; low sampling frequency; data below the level of reporting; and variability in flow. Thus, data shown is provided for general contextualisation.

Contaminant loads in the River Derwent vary significantly from year to year, largely driven by interannual fluctuations in river discharge (Figure 4.13). In 2024, all measured parameters were below the long-term average for the reporting period. Notable observations include: elevated total suspended solids in 2022 despite below-average flow volumes; dissolved reactive phosphorus concentrations typically sit at the limit of reporting, meaning observed values more likely reflect changes in flow rather than actual phosphorus levels; and elevated contaminant loads in 2016, attributed to above-average annual discharge volumes.

Accuracy of contaminant loads will be significantly improved with the commissioning of the flow gauge at the head of the estuary. See **Section 3.2.2** for more details.

Figure 4.13 Interannual comparison of calculated river discharge volume and the calculated loads of nutrient, organic carbon, suspended solids and zinc.



4.6 Stormwater loads

Stormwater loads provided are based on MUSIC models based on data collected in 2003 and on an average rainfall year (DEP, 2010a). These loads are provided as an estimate only and do not represent interannual variability or recently data. Stormwater contaminants, sources, recent monitoring data and management actions are discussed in detail in **Section 5**. The DEP plan on updating stormwater catchment loads by remodelling stormwater catchments using data collected in 2024–2025.

4.7 Nutrient budgets, marine nutrients and denitrification

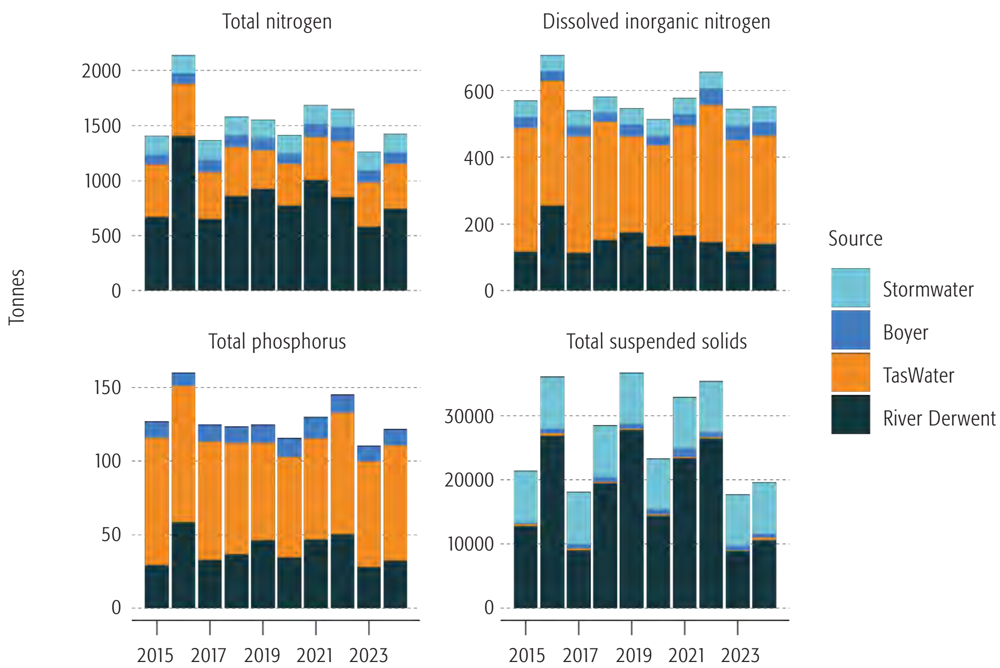
Notably, marine sources of nutrients are missing from the current contaminant load study. The DEP plans to produce a complete nutrient budget for the estuary. Wild-Allen *et al*. (2013) presented a budget using a 3D biogeochemical model that incorporated inputs from the Derwent River, stormwater, industrial discharges, and STPs. While the model included marine influx and export as boundary conditions, DEP currently lacks modelled values for these marine fluxes, as well as for denitrification rates. Acquiring this data is essential to quantify both nutrient inputs and losses, and to develop a full nutrient budget. DEP will collaborate with CSIRO, who are actively updating the Storm Bay model, to generate these figures. This approach builds on Wild-Allen *et al*. (2013), who showed that in 2003, marine sources contributed the largest nitrogen input, with a similar amount exported, and that denitrification played a critical role in maintaining estuarine water quality.

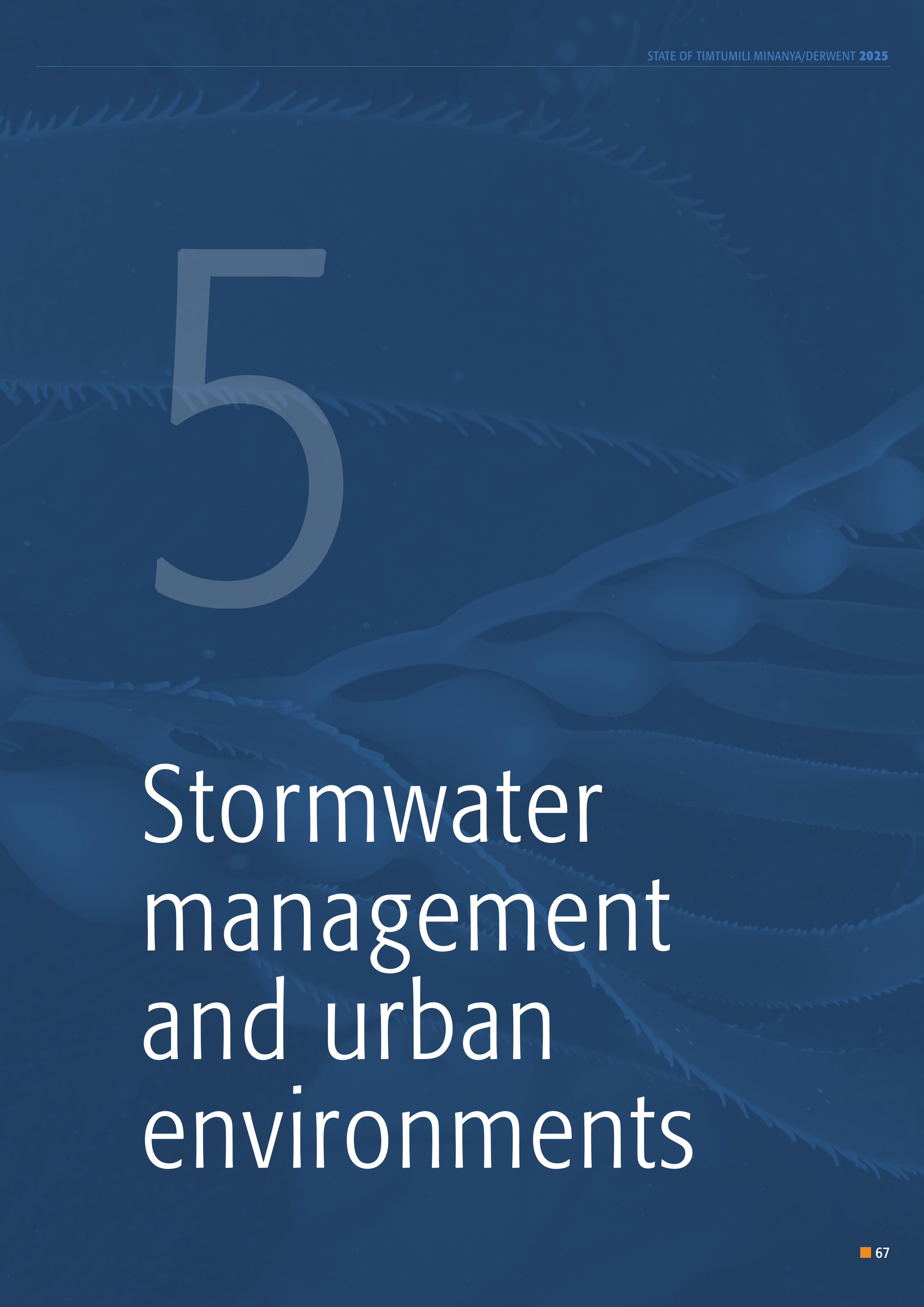
4.8 Derwent Estuary contaminant load summary

This summary of contaminant loads from 2015 to 2024 highlights the temporal variability in nutrient and sediment inputs to the Derwent Estuary from key sources: stormwater, Boyer, TasWater, and the River Derwent. The River Derwent is consistently the primary source of total nitrogen and total suspended solids, while STPs are the dominant source of dissolved inorganic nitrogen and total phosphorus.

Overall, contaminant loads were elevated in 2017 and 2021, reflecting periods of increased discharge. Total suspended solids have dropped significantly in recent years due to a substantial reduction in river loads (Figure 4.14). Though not displayed here, groundwater flows form Nyrstar site are the greatest source of zinc to the estuary. Contaminant loads are typically greatest in the middle-to-lower estuary, where the density of point sources is highest. These patterns underscore the importance of accounting for both river and point-source contributions. However, marine nutrient fluxes and denitrification remain unquantified, limiting the completeness of the current load summary. Another source not quantified in this report is effluent discharged from Cadbury at Claremont. Addressing these gaps will allow for the development of a full nutrient budget and help guide future management strategies.

Figure 4.14 Estimated contaminant loads (t/year) entering the Derwent Estuary from point and diffuse sources including the River Derwent, TasWater sewage treatment plants (STPs), stormwater systems, and Boyer. River Derwent loads are calculated using flow data from Meadowbank and monthly surface water quality data at New Norfolk. Stormwater loads are based on MUSIC modelling based on average rainfall year, and do not reflect interannual variability. This figure provides a comparative overview of source contributions and supports the identification of dominant inputs across years.





5 Stormwater management and urban environments

5 Stormwater management and urban environments

5.1 Stormwater and the Derwent Estuary

Stormwater is surface runoff generated by rainfall or snowmelt that flows across impervious land surfaces, such as roads, rooftops, and pavements. As it travels, stormwater picks up pollutants including pathogens (e.g., faecal bacteria), nutrients (such as nitrogen and phosphorus), hydrocarbons (from vehicle emissions and spills), heavy metals (like lead, zinc, and copper), sediments, and litter. These contaminants are often washed directly into waterways without any form of treatment.

In most urban catchments around Hobart, stormwater is conveyed through a network of kerbs, gutters, storm drains, and underground pipes, eventually discharging into the River Derwent. This untreated runoff can significantly impact the health of urban streams and the estuary, contributing to downstream flooding, erosion of stream banks, and degradation of aquatic habitats and recreation areas, such as swimming beaches.

The Derwent Estuary receives stormwater from 57 urban and suburban catchments, which are drained by 13 major rivulets and numerous large outlet pipes. These rivulets act as conduits for stormwater, channelling it from residential, commercial, and industrial zones into the estuary. Key pollution sources include:

- Construction sites, which contribute sediment and debris.
- Roads and car parks, which introduce hydrocarbons, heavy metals, and litter.
- Industrial and commercial areas, which may discharge chemicals and waste.
- Eroding stream banks, which add sediment and nutrients.
- Occasional sewer cross-connections, which can introduce untreated sewage.

Stormwater pollution is a major environmental concern for the Derwent Estuary. It threatens water quality, aquatic life, and public health, particularly due to:

- Litter, which can smother habitats and harm wildlife.
- Faecal bacteria and pathogens, which pose risks for recreational water use and shellfish harvesting.
- Total Suspended Solids (TSS), which reduce light penetration, smother benthic organisms, and carry attached pollutants.

Inadequately designed or poorly maintained stormwater systems can exacerbate these issues by increasing the volume and velocity of runoff, leading to flash flooding, infrastructure damage, and stream-channel instability. Beyond physical impacts, the chemical and biological pollutants in stormwater can alter nutrient cycles, reduce oxygen levels, and disrupt the ecological balance of the estuary.

The DEP has continued to play an important role in coordinating stormwater initiatives within the region, as well as providing capacity building opportunities. Key activities include:

- Continuation of the Stormwater Taskforce (SWTF): The SWTF is a collaborative working group comprising specialists from local councils, State Government, and TasWater. It meets quarterly to share management ideas, exchange experiences, discuss challenges, and review management priorities.
- Development of the Erosion and Sediment Control – The Fundamentals for Development in Tasmania: This resource was produced collaboration with the Tamar Estuary and Esk Rivers (TEER) program, to support best practice erosion and sediment control on construction sites in Tasmania. It provides clear explanations of the what and why of erosion and sediment control, and more importantly, the how–offering practical steps that users can take to mitigate sediment impacts. The guide also includes example plans to support implementation.
- Stormwater and urban rivulet pollution monitoring: Funded by the Australian Federal Government's Urban Rivers and Catchment Program (URCP), the DEP's 2024–2025 stormwater monitoring initiative aims to assess contaminant concentrations in Hobart's stormwater systems and enhance the existing legacy dataset. Over a 12-month period (June 2024 – May 2025), the program monitored 35 sites across the estuary, covering upper, lower, urban, and rural catchments.
- Stormwater management training: Facilitation of specialist training courses that addressed key knowledge shortages highlighted by DEP partners. The DEP coordinated Water Sensitive Urban Design maintenance training in collaboration with Optimal Stormwater, council GPT training and erosion and sediment training for council and industry staff.

5.2 Derwent Estuary stormwater monitoring program 2024–2025

Commenced during 2024, the Derwent Estuary Program's (DEP) stormwater monitoring program collected baseline pollution data from rivulets across the region. The DEP previously conducted stormwater monitoring during base flow conditions, targeting pollutants, such as sediment, faecal bacteria, nutrients, and metals. Building on earlier studies the current monitoring program expands both the temporal scope and site coverage across the estuary.

The Derwent Estuary Program's 2024–2025 stormwater monitoring initiative aims to assess contaminant concentrations in Hobart's stormwater systems while enhancing our existing legacy dataset. Over a 12-month period (June 2024 – May 2025), the program monitored 35 sites across the estuary, encompassing upper, lower, urban, and rural catchments, captured in Figure 5.1 and Table 5.1. This study is a repeat of stormwater monitoring conducted in 2002–2005 and 2010–2011 and captured samples from base flow conditions and targeted pollutants, such as sediment, faecal bacteria and nutrients (Milne, 2005; DEP, 2011). Monthly aseptic grab samples were collected by local councils and the DEP to evaluate the current state of stormwater pollution.

During the monitoring period, seven key water quality analytes were measured to evaluate the ecological health, and potential pollutant loads at each site.

These analytes included:

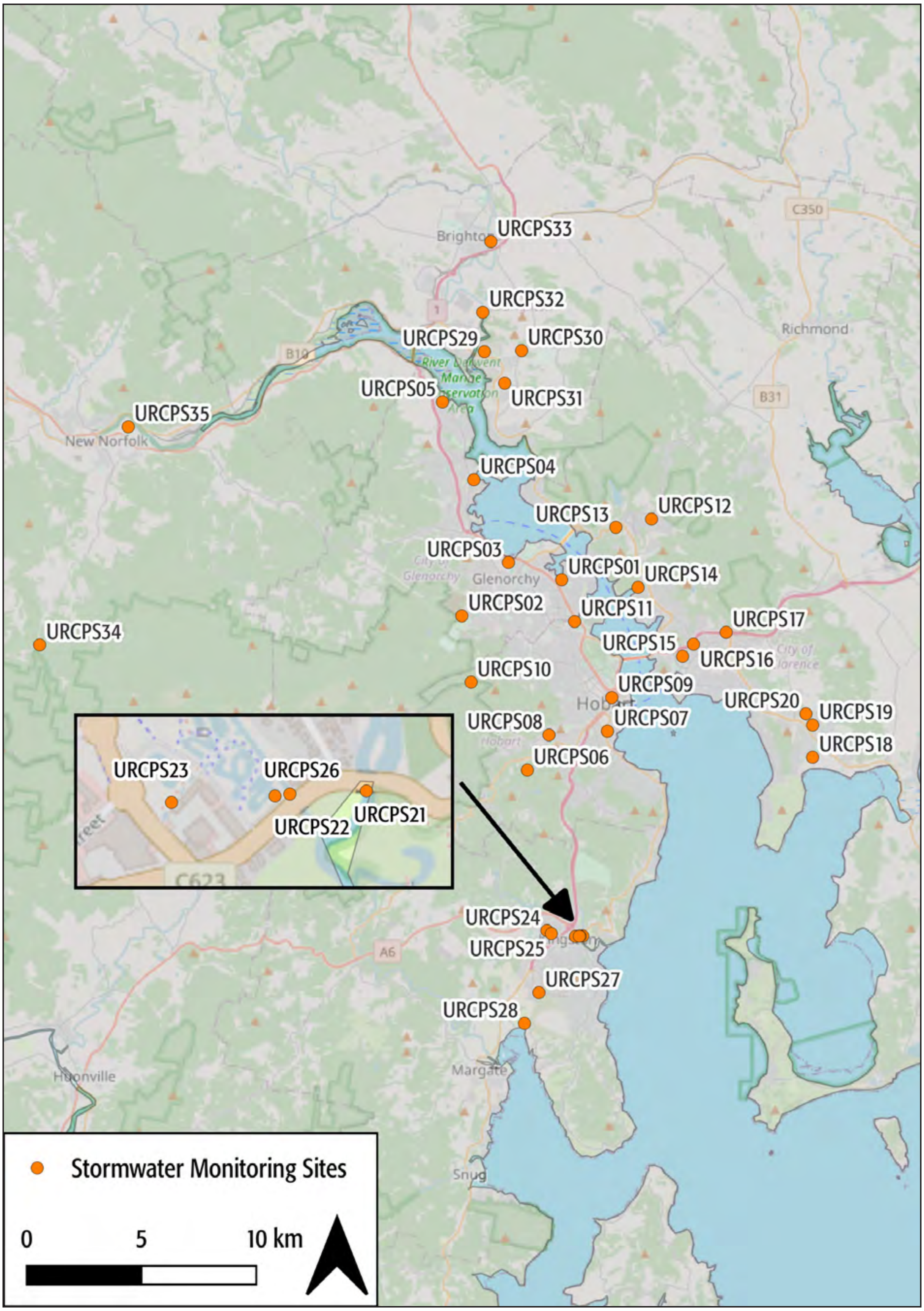
- Total Suspended Solids (TSS): Indicative of particulate matter in the water, which can affect light penetration and aquatic habitats.
- Turbidity: A measure of water clarity, closely linked to TSS and often used as a proxy for sediment and pollutant transport.
- Total Nitrogen (TN) and Nitrate + Nitrite: Nutrient indicators that can contribute to eutrophication and algal blooms if present in excess.
- Total Phosphorus (TP): Another key nutrient that, is a target analyte of WSUD devices.
- Enterococci: A microbial indicator used to assess the presence of faecal contamination and potential public health risks.
- E. coli: Added to the monitoring profile in July 2024, this bacterium serves as a complementary faecal indicator, particularly relevant for recreational water quality assessments.

Metals and hydrocarbons were excluded from this program due to the high cost of analysing samples across multiple sites. Additionally, the method used to collect water samples (grab sampling) may not be ideal for detecting metals, given their low concentrations in the water column and the limitations of laboratory detection. Contaminants, including metals, are often found at significantly higher concentrations in sediments due to their tendency to bind with particulate matter. Therefore, a dedicated study focusing on sediment analysis may be considered in the future.

Table 5.1 Breakdown of Stormwater Monitoring sites sampled monthly between June 2024 and May 2025.

Council	Site ID	Site Location
Glenorchy City Council	URCPS01	Prince of Wales Bay Outfall
	URCPS02	Humphreys Rivulet upper site
	URCPS03	Humphreys Rivulet lower site
	URCPS04	Faulkners Rivulet lower site
	URCPS05	Goulds Lagoon
City of Hobart	URCPS06	Sandy Bay Rivulet upper site
	URCPS07	Sandy Bay Rivulet lower site
	URCPS08	Hobart Rivulet 5m below tip SW outfall
	URCPS09	Hobart Rivulet lower site
	URCPS10	New Town Rivulet upper site
	URCPS11	New Town Rivulet lower site
Clarence City Council	URCPS12	Risdon Creek upper site
	URCPS13	Risdon Creek lower site
	URCPS14	Faggs Creek
	URCPS15	Kangaroo Bay lower site
	URCPS16	Kangaroo Bay mid site
	URCPS17	Kangaroo Bay upper site
	URCPS18	Clarence Plains Rivulet lower site
	URCPS19	Clarence Plains Rivulet mid site
	URCPS20	Clarence Plains Rivulet upper site
Kingborough Council	URCPS21	Kingston Rivulet
	URCPS22	Whitewater Creek 1
	URCPS23	Browns River
	URCPS24	Whitewater Creek 2
	URCPS25	Whitewater creek 3
	URCPS26	Kingston Wetlands
	URCPS27	Coffee Creek lower site
	URCPS28	Coffee Creek upper site
Brighton Council	URCPS29	Cove Creek lower site
	URCPS30	Cove Creek upper site
	URCPS31	Tivoli Green Wetlands
	URCPS32	Jordan River lower site
	URCPS33	Jordan River upper site
Derwent Valley Council	URCPS34	Lachlan River upper site
	URCPS35	Lachlan River lower site

Figure 5.1 URCP Monitoring Sites 2024-2025.



The results from this monitoring program are presented in the report card below. Each site's 80th percentile value for key parameters is compared against both the current Tasmanian EPA default guideline values (DGVs) and the previous Australian New Zealand Environmental and Conservation Council (ANZECC, 2000) guidelines which were used in the DEP monitoring from 2005 and 2010. The Tasmanian EPA's DGVs for the Derwent Estuary/Bruny Catchment (EPA Tasmania, 2021) have replaced the older ANZECC guidelines. These DGVs are based on the 80th percentile of historical monitoring data, and the same statistical method has been applied to each site in this program.

Scoring Method

Each site received:

- 1 point for each parameter within the reference value (either the DGV or recreational water 'poor threshold').
- 0.5 points if the parameter was within ±10% of the reference value.

The maximum score per site is 5 out of 5, indicating all assessed parameters met the reference criteria. This scoring system aligns with previous DEP stormwater monitoring programs. For sites previously monitored, an additional score using the ANZECC guidelines (ANZECC, 2000) has been calculated to allow comparison across monitoring periods.

Key Findings

Urban catchments showed the highest levels of stormwater pollutants, especially in areas with active development. Natural or undisturbed sites had significantly lower pollutant loads. The most common pollutants in urban areas were sediment-related, particularly turbidity and total suspended solids (TSS). Sites such as Cove Creek (URCPS29 & 30), Tivoli Green (URCPS31), Risdon Creek (URCPS12 & 13), and Faggs Creek (URCPS14) had elevated turbidity and TSS levels, reflecting the impact of ongoing construction.

Environmental Impacts

High sediment levels reduce water clarity and oxygen levels, harming aquatic ecosystems and diminishing recreational and aesthetic value (Figure 5.2). A noticeable decline in waterbug diversity (see below) was observed in areas affected by sedimentation. High sediment turbidity follows expected patterns, with elevated levels observed at:

- Risdon Rivulet (URCPS12 & 13)
- Cove Creek (URCPS29 & 30)
- Clarence Plains Rivulet (URCPS14)
- Jordan River (URCPS32 & 33)

These results are significantly higher than both reference datasets.

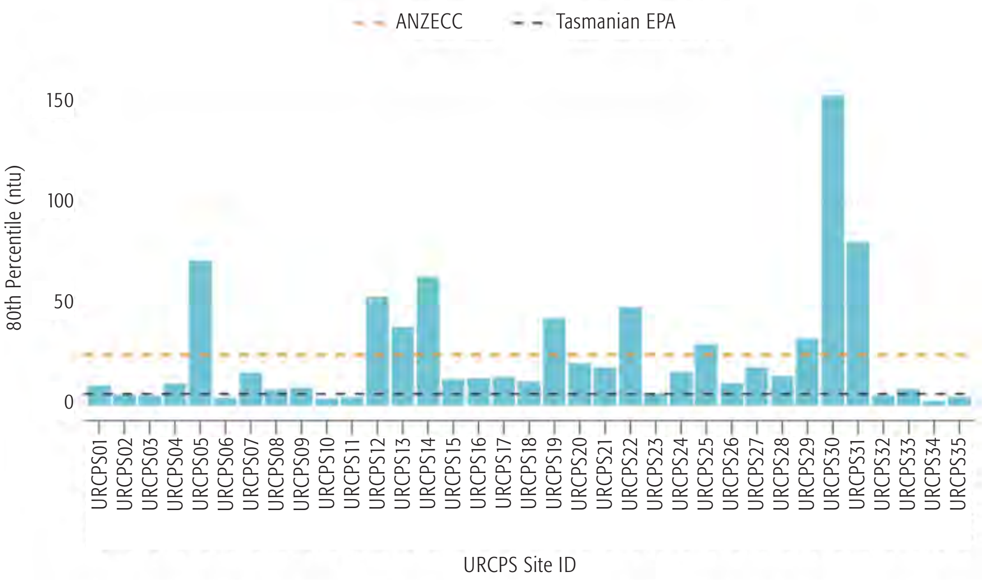
When compared to the previous ANZECC default guideline value (DGV) of <25 NTU and the more stringent Tasmanian EPA DGV of 5.6 NTU, many sites exceed the Tasmanian threshold. The stricter Tasmanian DGV results in more frequent exceedances.

However, New Town Rivulet, Humphreys Rivulet, and Lachlan River consistently remain within acceptable limits under both guidelines, indicating better water quality at these locations.

Cove Creek is of particular concern due to high turbidity and total suspended solids (TSS). This is likely influenced by dispersive soils and a large area of exposed, erosion-prone agricultural land in the upper catchment.

In more established urban areas like Hobart (URCPS06 – 11) and Glenorchy CBDs (URCPS01 – 05), sediment impacts were less severe. However, these sites showed elevated nutrient and bacterial levels, likely due to aging or compromised wastewater infrastructure. High nutrient levels can lead to algal blooms, while the presence of faecal bacteria poses public health risks, especially in recreational areas.

Figure 5.2 Turbidity 80th percentile for 2024 – 2025 monitoring sites. ANZECC (2000) and Tasmanian EPA (2021) Default Guideline Values displayed on plot for comparison.



Waterbug Survey

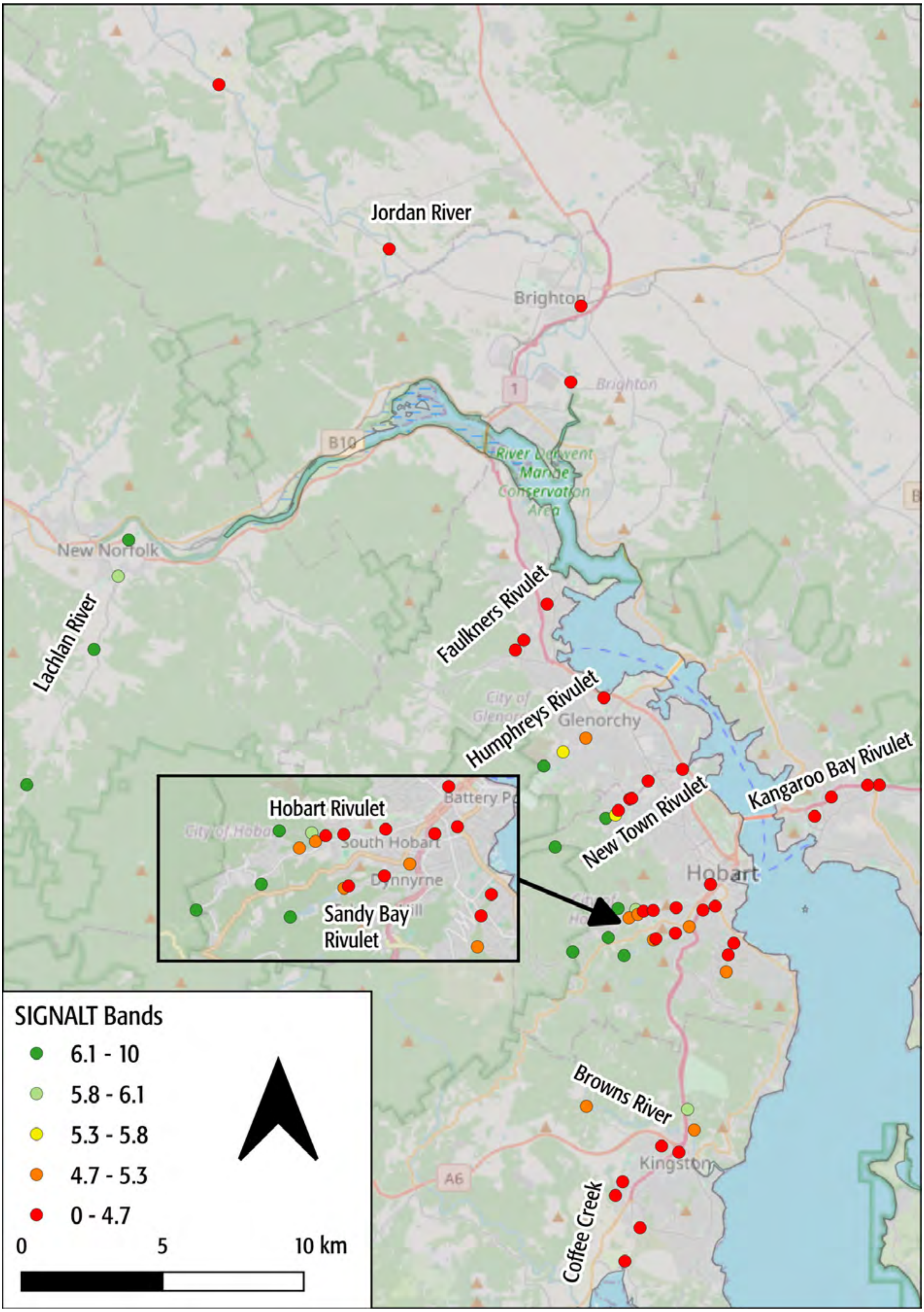
To complement the vast array of water quality information obtained throughout this program a survey of waterbugs at 56 sites across the various tributaries entering the Derwent Estuary during spring 2024 was included (Figure 5.3). Samples allowed for the health of the waterway to be assessed using waterbug data as they provide an insight into the waterway's long-term exposure to impacts such as pollution and sedimentation. The presence of more sensitive waterbugs indicates lesser impacts, whereas sites populated only by "tolerant" organisms suggest impacted waters. Tolerant waterbugs are those that have resilience to pollution, sediment and other impacts.

Across both the water quality and waterbug data the following trends were observed.

- Quality declines across both metrics within the urban zone, with the most degraded sites typically being right before they enter the estuary.
- Sites within peri urban and rural use see trends in both metrics remain relatively stable throughout the catchment.
- Point sources and land use have the most profound effect on both metrics within any given catchment.

Further information regarding the waterbug survey can be found in the supplementary report "Tributaries of the Derwent Estuary – Waterbug survey Spring 2024", (Gooderham, in preparation).

Figure 5.3 All 56 sites sampled as part of the Derwent Estuary tributaries waterbug sampling project. Colours use weighted SIGNALT, higher scores (green) indicate sites in better condition.



This study provides a comprehensive and timely update on stormwater quality across the Greater Derwent Estuary, highlighting the complex interplay between land use, urban development, and waterway health. By expanding the monitoring network and incorporating biological indicators, such as waterbugs, the program offers a more holistic understanding of ecosystem responses to pollution.

The findings reinforce the urgent need for improved sediment and nutrient management, particularly in rapidly developing catchments, and underscore the value of emerging technologies in identifying contamination sources. Moving forward, a coordinated approach that integrates regulatory reform, community education, and on-ground best practices will be essential to safeguarding the estuary's ecological integrity and ensuring it remains a valued asset for future generations.

For more details and to access the final report, please visit our website.

5.2.1 2024–2025 Report card

The results of the 2024–2025 Stormwater Monitoring Program are presented in Figure 5.4 . This "report card" format has been used previously in DEP programs conducted between 2002–2005 and in 2010–2011.

A key update in this report is the adoption of the Tasmanian EPA's default guideline values (DGVs) for the Derwent Estuary/Bruny Catchment (EPA Tasmania, 2021), replacing the previously used ANZECC (ANZECC, 2000) guidelines. These DGVs are based on the 80th percentile of monitoring data, and this same statistical approach has been applied to each site in the current program.

The report card includes results for the following analytes:

- Turbidity
- Total Suspended Solids (TSS)
- Total Phosphorus
- Total Nitrogen

In addition, Enterococci levels are assessed using the Poor Water Quality category from the Tasmanian Recreational Water Quality Guidelines (DoH Tas, 2007). This is based on the geometric mean of the collected sample results. Site data for *E Coli* and Dissolved Nitrogen were not included as part of this report card as there is currently no appropriate metric to assess these results.

Each site was awarded a point for each parameter that was within the analytes reference value (either the DGV value or Rec water 'poor threshold'). Where a parameter was within +/- 10% of the reference value, 0.5 points were awarded for that analyte. The maximum score available was 5/5 – indicating all assessed parameters at the site were within the range specified by the reference value. This scoring system is closely aligned to the system used in previous DEP Stormwater monitoring programs.

Please note for sites that have been duplicated from the previous monitoring programs an additional score based upon the previous ANZECC (ANZECC, 2000) has also been generated to allow comparisons between monitoring periods.

Figure 5.4 2024 – 2025 Report Card.

		Water Clarity		Nutrients		Faecal Bacteria				
Site	Site ID	Total suspended solids	Turbidity	Total Nitrogen	Total Phosphorous	Enterococci *	TAS EPA Score	ANZECC Score **	2010 – 2011 Score ^	2002 – 2005 Score ^
ANZECC ** Reference Value	**5 mg/L**	**25 (NTU)**	**0.5 mg/L**	**0.05 mg/L**	**230 MPN/100mL**					
TAS EPA Reference Value	**11 mg/L**	**5.6 (NTU)**	**0.67 mg/L**	**0.02 mg/L**	**500 MPN/100mL**					
Upper Rivulet Sites										
Humphreys Rivulet	URCPS02	6	5	0.33	0.01	134	**5/5**	**4/5**	4/5	5/5
Sandy Bay Rivulet	URCPS06	19	3.6	1.09	0.02	121	**3/5**	**3/5**	3/5	4/5
Hobart Rivulet	URCPS08	11.4	7.8	0.76	0.05	1100	**0.5/5**	**2/5**	4.5/5	5/5
New Town Rivulet	URCPS10	6	3.2	0.21	0.01	418	**5/5**	**3/5**	5/5	5/5
Risdon Creek	URCPS12	80.4	54	3.20	0.17	1465	**0/5**			
Kangaroo Bay Rivulet	URCPS17	8.8	14	1.99	0.11	492	**1.5/5**	**1/5**	1.5/5	2/5
Clarence Plains Rivulet	URCPS20	30	20.9	5.16	0.13	1638	**0/5**	**1/5**	1/5	4/5
Coffee Creek	URCPS28	19.6	14.4	1.0	0.11	154	**1/5**			
Cove Creek	URCPS30	340	154	2.38	0.23	1585	**0/5**	**0/5**	0/5	1.5/5
Jordan River	URCPS33	17.2	8.1	1.84	0.08	917	**0/5**			
Lachlan River	URCPS34	4	2.1	0.22	0.01	110	**5/5**	**5/5**	5/5	-
Middle Rivulet Sites										
Kangaroo Bay Rivulet	URCPS16	14.2	13.6	2.2	0.09	703	**0/5**			
Clarence Plains Rivulet	URCPS19	53.8	43.2	3.8	0.12	1004	**0/5**			
Whitewater Creek – UF	URCPS24	20.4	16.6	1.5	0.15	1993	**0/5**			
Whitewater Creek – LF	URCPS25	48.2	30.4	1.6	0.09	825	**0/5**			

Water Quality Score improved from previous monitoring

Water Quality Score declined from previous monitoring

Site failed to meet any Tasmanian EPA DGV Thresholds

^ Highlights when sites have been previously monitored by the Derwent Estuary Program in either 2002–2005 & 2010–2011 and their corresponding report card score

* Enterococci reference values are based on the geometric mean of the 2024–2025 monitoring data and are used for comparison with previous monitoring programs. The Hazen percentile has not been included due to the limited number of data points

** ANZECC score for 2024–2025 monitoring sites generated if site has been previously monitored, allowing for long term comparison. Score is based upon the previous ANZECC guideline values (ANZECC, 2000)

		Water Clarity		Nutrients		Faecal Bacteria				
Site	**Site ID**	**Total suspended solids**	**Turbidity**	**Total Nitrogen**	**Total Phosphorous**	**Enterococci ***	**TAS EPA Score**	**ANZECC Score ****	**2010 – 2011 Score ^**	**2002 – 2005 Score ^**
ANZECC ** Reference Value	**5 mg/L**	**25 (NTU)**	**0.5 mg/L**	**0.05 mg/L**	**230 MPN/100mL**					
TAS EPA Reference Value	**11 mg/L**	**5.6 (NTU)**	**0.67 mg/L**	**0.02 mg/L**	**500 MPN/100mL**					
Lower Rivulet Sites										
Prince of Wales Bay	URCPS01	10	9.7	3.18	0.21	3887	**1/5**			
Humpherys Rivulet	URCPS03	10	4.8	0.49	0.01	611	**4/5**	**4/5**	3/5	3/5
Faulkner's Rivulet	URCPS04	12.8	10.8	1.58	0.06	1184	**0/5**	**1.5/5**	1.5/5	2/5
Sandy Bay Rivulet	URCPS07	24.6	16.2	3.26	0.17	1157	**0/5**	**1/5**	2/5	2.5/5
Hobart Rivulet	URCPS09	12	9	1.54	0.12	1436	**0.5/5**	**1/5**	1/5	
New Town Rivulet	URCPS11	5.4	3.92	0.65	0.01	1044	**4/5**	**3/5**	2/5	4.5/5
Risdon Creek	URCPS13	37	39	1.69	0.07	286	**1/5**			
Faggs Creek	URCPS14	69.6	63.8	6.24	0.26	2638	**0/5**			
Kangaroo Bay Rivulet	URCPS15	15.6	12.9	2.03	0.10	1013	**0/5**	**1/5**	0/5	1.5/5
Clarence Plains Rivulet	URCPS18	17	11.6	1.21	0.11	1978	**0/5**	**1/5**	2/5	1/5
Kingston Rivulet	URCPS21	24.6	19	1.71	0.09	1214	**0/5**	**1/5**	0/5	0/5
Whitewater Creek	URCPS22	43.8	48.6	1.29	0.09	1100	**0/5**	**0/5**	0/5	0.5/5
Browns River	URCPS23	8.8	5.8	0.41	0.02	906	**3.5/5**	**3/5**	2.5/5	4/5
Coffee Creek	URCPS27	21.2	19	0.94	0.07	473	**1/5**			
Cove Creek	URCPS29	57.8	33.2	11.9	0.18	436	**1/5**	**0/5**	1/5	
Jordan River	URCPS32	7.6	4.8	1.38	0.08	467	**3/5**			
Lachlan River	URCPS35	7.8	4.1	0.32	0.01	210	**5/5**	**4/5**	5/5	4/5
Wetland Sites										
Goulds Lagoon	URCPS05	106.8	71.6	3.3	0.37	1078	**0/5**			
Kingston Wetlands	URCPS26	25.8	11.1	1.1	0.07	246	**1/5**			
Tivoli Green Wetland	URCPS31	198	80.8	6.1	0.48	1247	**0/5**			

5.3 Waste and sustainability

Over the last five years, waste management in the greater Derwent region has significantly improved, with a focus on reducing environmental impact, increasing investment in the sector, improving business and community education for behavioural change and moving towards a circular economy. There are five operating waste management facilities in the greater Derwent region, and multiple historic landfill sites (DEP, 2015b). These sites pose risks to surrounding environments through leachate, surface runoff, and wind-blown litter. To address this, the *Environmental Management and Pollution Control Act* is now used to regulate major landfills as Level 2 activities. Under this framework, the Tasmanian EPA sets permit conditions to ensure these sites operate according to best environmental practices. Smaller public and private landfills are also regulated under the same legislation, typically by local councils. However, identifying and managing all small-scale operations remains a challenge.

In 2023 the Tasmanian Government released the *Tasmanian Waste and Resource Recovery Strategy 2023-2026*. This strategy outlines objectives and actions to implement to transition the state to a more circular economy. The Tasmanian Government has implemented several actions from this strategy, and local councils have taken additional proactive steps, including:

- **Landfill Levy** – The introduction of Tasmania's landfill levy is a long-term strategy to improve waste management and resource recovery by charging a fee on all waste sent to landfill. Its goal is to divert reusable, recyclable, and repairable materials away from landfill, encouraging more sustainable practices. Waste recovered through Resource Recovery Facilities is exempt from the levy. Revenue generated is reinvested into the sector to support industry growth, create jobs, and fund community education programs focused on recycling, reuse, and litter prevention.
- **Recycle Rewards** – Recycle Rewards is a Tasmanian Government initiative designed to reduce litter and support Australia's circular economy by offering a 10-cent refund for eligible drink containers returned to designated refund points, starting from May 2025.
- **FOGO Collection service** – Several councils across the Derwent region have implemented FOGO (Food Organics and Garden Organics) collection services to significantly reduce the amount of organic waste ending up in landfill. These programs allow households to separate food scraps and garden waste from general rubbish, enabling the organic material to be processed into compost or mulch. This not only helps lower greenhouse gas emissions from decomposing organic waste in landfills but also supports the creation of valuable soil products for agricultural and landscaping use.
- **Single use plastic ban Hobart** – In 2020, City of Hobart Council introduced the Single-Use Plastics By-Law, banning single-use plastic takeaway packaging to reduce plastic waste and encourage sustainable alternatives. Unlike the statewide plastic bag ban, this by-law represents a broader shift away from single-use plastics. Many events and venues in Hobart now use compostable or biodegradable alternatives, such as paper, cardboard, and plant-starch-based products, which are widely available and already in common use.
- **Soft Plastics Recycling** – In 2025, the City of Clarence launched several soft plastics recycling collection points to help reduce the amount of this difficult-to-recycle material ending up in landfill. Twelve month soft plastics recycling trials have also been established in the municipalities of Hobart and Kingborough.

5.3.1 Community clean-ups

Public education and clean up events play a key role in prioritising and influencing the reduction of litter entering the Derwent Estuary. Several community groups and motivated individuals regularly volunteer to clean up the estuary.

The DEP coordinates annual Clean Up Australia Day events throughout the Derwent Estuary. Notably the DEP focused clean-up efforts at Prince of Wales Bay (PWB) from 2021–2024, a bay that has historically accumulated high loads of litter. Clean-up efforts at Prince of Wales Bay have been a collaborative effort with DEP partners and many PWB business. In 2023, DEP coordinated two other cleaning events in collaboration with community groups and local councils to target rubbish at Cleburne Point and Shag Bay, with a significant amount of waste being removed.

These events are a great example of collaboration between local business, industry and the community and have helped contribute to litter load reduction in PWB, however, there is still much work to be done.

5.4 Faecal source tracking

Stormwater contaminated with human and animal faeces contain pathogens which pose a significant risk to beach goers. A key source of contamination is sewage discharge from sewerage infrastructure via spills, leaks caused from cracked or blocked pipes or direct cross-connections to the stormwater system. Local councils conduct a range of monitoring and source tracking activities to locate and rectify sewer intrusions to stormwater (DEP, 2020b).

In 2020, the DEP published a Faecal Source Tracking Framework and Toolkit to assist council to conduct effective investigations. Since the implementation of these resources, several partner councils have utilised the tool kit to help identify and rectify stormwater issues within their local catchments, particularly where nearby recreational sites have been impacted.

Emerging technology for Faecal Source Tracking within Urban Stormwater Systems

Faecal source tracking within urban stormwater systems presents a significant challenge that councils and water utilities regularly contend with. The presence of sewage traces in stormwater can adversely affect surrounding natural and recreational environments.

To support critical stakeholders, we partnered with ZiP Diagnostics to evaluate their field-deployable *Bacteroides dorei* test–a highly specific human faecal indicator bacterium (FIB)–as part of our 2024–2025 Stormwater and Rivulet Monitoring Program.

As part of this program, a subset of 17 sites was monitored monthly over six months, with results assessed alongside collected Enterococci and E. coli data. In addition to these routine monitoring efforts, stormwater samples were also collected and analysed during the Beachwatch program. Notably, samples from stormwater sites that tested positive for *Bacteroides dorei* also exhibited elevated levels of Enterococci and *E. coli.* This strong correlation across both programs suggests that human faecal contamination is likely a primary contributor to microbial pollution in these systems.

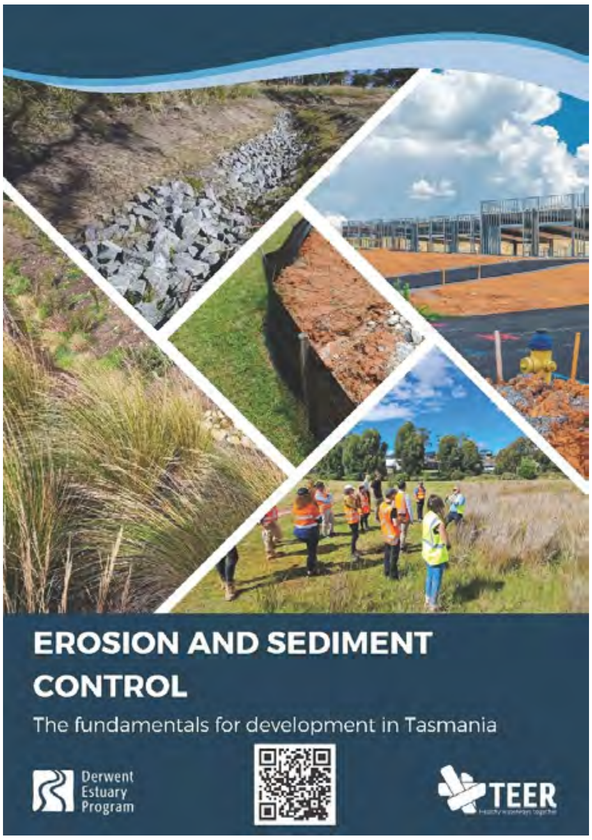
Overall, the findings demonstrate that the *Bacteroides dorei* test is a promising tool for rapidly identifying human faecal contamination in stormwater systems. This enables councils and regulators to trace sources more efficiently and prioritise remediation efforts. The approach builds upon previous methods developed at the DEP using ammonia test kits, offering stakeholders more direct insight into pollution sources and ultimately supporting better management of recreational water quality.

By incorporating complementary tools to detect non-human sources, this method allows for a more comprehensive assessment of recreational water quality – particularly valuable for targeted investigations of human pollution in stormwater and catchment areas.

5.5 Erosion and sediment control – The fundamentals for development in Tasmania

As Tasmania continues to grow, the demand for land development increases–bringing with it the critical challenge of protecting the state's waterways from sediment pollution. In response, the Tamar Estuary and Esk Rivers (TEER) Program and the Derwent Estuary Program (DEP), in collaboration with local councils, industry professionals, and the Environment Protection Authority (EPA), have released a new guidance document (TEER and DEP, 2023) (Figure 5.5).

Figure 5.5 Erosion and Sediment Control fundamentals document produced in collaboration with the TEER Program.



Overview of the Fundamentals Resource

The Fundamentals Resource is a comprehensive guide developed over two years through collaboration between the Derwent Estuary Program (DEP) and the Tasmanian Erosion and Sediment Control Reference Group (TEER), with contributions from both northern and southern stormwater working groups. It has undergone extensive expert review, including input from Terry Clark (IECA), Murray Powell (Optimal Stormwater), and the Tasmanian EPA. Grounded in current IECA best practices, the document provides a practical foundation for understanding erosion and sediment control (E&SC), its significance, and its application on development sites. Structured in two parts– 'What and Why' and 'How'–it explains the environmental risks of sediment runoff, outlines Tasmania's regulatory context, and offers detailed guidance on planning, installing, and maintaining E&SC measures. The resource introduces a modernised format, replacing outdated factsheets with a full-colour, user-friendly layout, including an A3-sized E&SC Plan template to support site-specific planning and compliance.

Enhancements such as clearer language, reduced jargon, and colour-coded sections improve accessibility for a broad audience, including those with limited technical backgrounds. New content includes emerging technologies like spray-on soil stabilisers, sustainable sediment controls such as mulch filter berms and rock filter dams, and a classification system based on sediment size and site complexity. Designed to support planners, engineers, contractors, regulators, and others involved in land development, the resource promotes best-practice E&SC aligned with environmental regulations, helping protect Tasmania's waterways. The full document is available on the Derwent Estuary Program website.

During the construction of the new Bridgewater Bridge, best-practice erosion and sediment control (E&SC) measures were consistently applied through a dynamic and evolving Erosion and Sediment Control Plan managed by the project proponents. This proactive approach ensured that environmental safeguards kept pace with the changing conditions of the large-scale infrastructure project.

The Derwent Estuary Program's (DEP) Stormwater Taskforce had the opportunity to visit the site and observe the E&SC measures in action. This site visit provided valuable real-world examples of effective planning and implementation, offering practical insights that taskforce members could apply to their own development sites across Tasmania.

Figure 5.6 New Bridgewater Bridge Project – Image provided by the Department of State Growth 2025 – https://www.bridgewaterbridge.tas.gov.au/current_work/images_and_video

6

Ambient water quality

6 Ambient water quality

6.1 Ambient water quality monitoring program

The Derwent Estuary Program (DEP) coordinates a monthly ambient water quality (AWQ) monitoring program throughout *Timtumili Minanya*/River Derwent, including the Derwent Estuary. The AWQ monitoring program is a collaboration between the DEP, Tasmanian Government, the Environment Protection Authority (EPA), Nyrstar Hobart and Boyer Paper Mill (Boyer). The program commenced in 2007, formalising monitoring that had been underway since 1972.

The primary objectives of the program are to track spatial and temporal trends in estuarine water quality, and to support the assessment of interactions between point and diffuse sources of pollution and their effects on the water column.

This chapter presents findings based on data collected during the 'reporting period' (2020–2024), illustrated through bar charts, and examines long-term trends over the 'monitoring period' (2007–2024) using scatter plots and trend analysis.

6.1.1 Methods

Field sampling

Physicochemical, transparency, nutrient, phytoplankton biomass and metal parameters were sampled at 29 sites on the third Tuesday of each month (Figure 6.1). Samples were collected by Boyer (upper estuary), Nyrstar Hobart (middle estuary, middle estuary bays), and the EPA and DEP (lower estuary, Ralphs Bay). For AWQ reporting the estuary separated into the following five zones (Figure 6.1):

- Upper estuary (NN, U19, U16/17, U14 and U12)
- Middle estuary (U2, U3, U4, U5 and U7)
- Middle estuary bays (CB, GB, LB, NTB 1, NTB 2, NTB 5, NTB 13 and PWB)
- Lower estuary (SC, KB, G2, E, C, B1, B3 and B5)
- Ralphs Bay (RBN, RB and RBS)

At each site, physicochemical parameters of the water column, including temperature, salinity, conductivity, pH, dissolved oxygen (DO), and turbidity were measured using multiparameter sonde (Hydrolab MS5, YSI EXO3). Parameters were measured at < 0.5 m below the surface, every metre to 10 m, every 5 m to the bottom, and within 1 m of the bottom.

Teflon pole samplers (< 0.5 m below the surface) and a Niskin bottle or pump sampler (within 1 m of the seabed) were used to collect water samples for the analysis of total nitrogen total Kjeldahl nitrogen, total phosphorus, total ammoniacal nitrogen, nitrate + nitrite, dissolved reactive phosphorus, non-purgeable organic carbon, true colour, total suspended solids and total zinc.

Integrated water column samples for chlorophyll a were collected from the surface to 10 m depth using a weighted Lund tube (Talling and Lund, 1957). Water clarity was measured using a black-and-white Secchi disc.

Lab analysis

All laboratory analyses were conducted by NATA-accredited laboratory Analytical Services Tasmania (AST).

Over the data collection period, there have been updates to the methods used by AST for calculating and validating limits of reporting (LoRs). This has resulted in subsequent increases to reporting thresholds. A summary of changes to LoRs is provided in Table 6.1.

To ensure that any stepwise changes in LoRs do not introduce inconsistencies in long-term data trend analyses, all historical LoRs have been standardised to reflect the post-2014 reporting limits. This allows for a consistent analytical baseline across the entire dataset, ensuring that trends observed are not artifacts of reporting limit changes. Values below the LOR were assigned the LOR value.

In March 2024, AST replaced the existing flow injection analysers (FIA) with segmented flow analysers (SFA). This change in analysis resulted in a significant difference in total Kjeldahl nitrogen results (and total nitrogen) between the two methods. The SFA analysis eliminated positive interference due to a salinity mismatch in FIA analysis of total Kjeldahl nitrogen resulting in significantly lower results when analysed by SFA. This has been demonstrated in analyser commissioning data and the DEP AWQ dataset. As such, total nitrogen and total Kjeldahl nitrogen data collected after March 2024 have been excluded from this analysis. Approaches to deal with this change in analysis are being discussed with AST and the EPA.

Exceptions

The following is a list of exceptions to the sampling regimes described above:

- Zinc data collected by Nyrstar prior to September 2011 has been omitted as samples were analysed at a different lab and are not considered comparable;
- Sampling for the full suite of parameters at sites NTB05, NTB01, Prince of Wales Bay (PWB), U3, GB, U5, G2 and KB commenced in November 2010;
- Sites B5, C, RB, RBS, CB and LB were sampled for physicochemical parameters and Secchi depth only;
- No turbidity data was collected for the upper and middle estuary;
- Site U12 was moved upstream to 'U12 temp' due to the New Bridgewater Bridge constructions from 2023–2024;
- Total nitrogen and total Kjeldahl nitrogen were excluded from analysis from March 2024 onwards due to changes in lab equipment from FIA to SFA;
- All Nyrstar sampled sites were also sampled for total cadmium, copper lead, mercury and iron.

Quality assurance and quality control

Quality Assurance (QA) was achieved through the consistent use of standard operating procedures that were used by all sampling teams. The DEP facilitated regular inter-calibration exercises with sampling teams to maintain consistency in sampling techniques and validate performance of multiparameter sondes. All samples were analysed by AST, a NATA accredited lab, ensuring uniformity and reliability in analytical results.

Quality Control (QC) techniques were used to verify the effectiveness of QA procedures. For each survey, duplicate seawater samples were taken from one randomly selected site. Field blanks, supplied by AST were handled using the same procedures as nutrient sampling (filtered and unfiltered) to detect any contamination during sample collection. Trip blanks were also taken on the vessel, but the containers were left unopened to identify if sample contamination is due to storage or transport effects.

Database

Data was processed, uploaded and stored to Enviro Data Vault, a secure Software-as-a-Service platform. Data is maintained and managed by the DEP ensuring centralised oversight, version control, and long-term storage. Authorised users can access and export data ensuring consistency and traceability. Data is made available upon request, subject to a data sharing agreement.

Presentation, analysis and guidelines

Water quality in the Derwent Estuary was assessed against the Draft Derwent Estuary Default Guideline Values (DGVs). The EPA developed the DGVs for assessing water quality for the protection of aquatic ecosystem values for each functional zone (EPA Tasmania, 2021b). The DGVs are based on 80th or 20th percentiles (depending on parameter) of data collected between 2009–2013. They represent reference conditions intended to be maintained to preserve ecological health. DGVs were developed in accordance with the National Water Quality Management Strategy (**https://www.waterquality.gov.au/anz-guidelines**)(Table 6.2). Further discussion is required around the development of aspirational water quality targets that reflect longer-term management goals and aim to improve ecosystem condition, rather than simply maintain current status.

Water quality was assessed against the DGVs using the median for the reporting period (2020–2024) for each combination of zone, parameter and depth. Box plots displaying median, 25th and 75th percentiles are presented to assess water quality parameters against Derwent Estuary DGVs. For each boxplot, whiskers extend to the most extreme data points within 1.5 times the interquartile range from the first and third quartiles. Values outside this range are considered outliers and were excluded from the whiskers and plot display. Exceedances suggest poor water quality outcomes and should be used to trigger further investigation.

Data were analysed for temporal trends using the Correlated Seasonal Mann-Kendall method (Hirsch and Slack, 1984). Lower and upper-estuary sites were analysed from 2007–2024 as data were continuous during this period. Middle estuary sites were analysed from 2011–2024 due to a period of missing data from 2008–2011. Missed monitoring events (one-off) were filled with previous data point as this analytical method requires no missing data. The dataset was modified to account for stepwise change in LoRs in 2014 (Table 6.1).

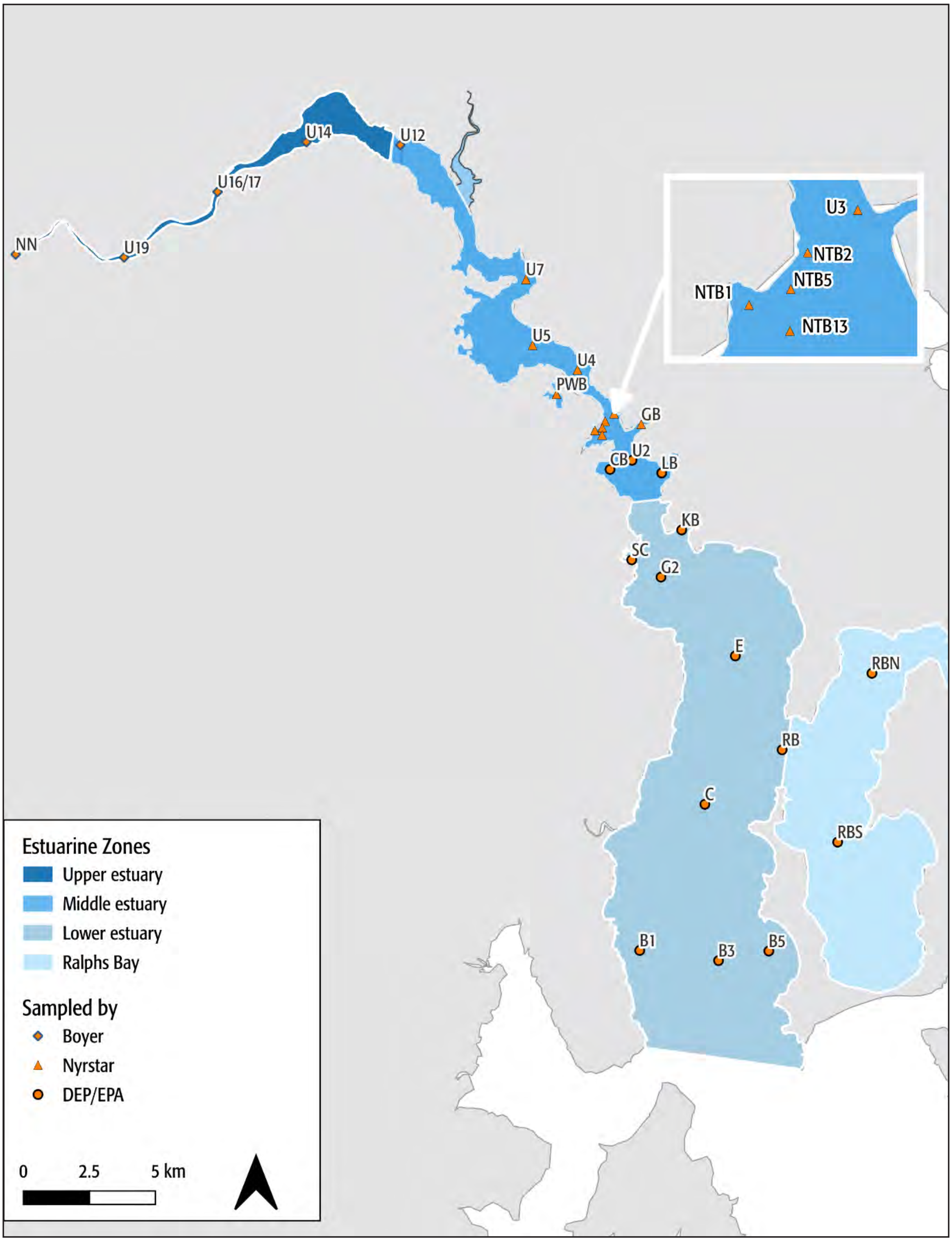
A conservative approach was taken to raise LoRs to post 2014 values. The results of the Correlated Seasonal Mann-Kendall trend analysis are summarised in (Table 6.3).

Scatter plots showing the average concentration for each parameter and zone are presented and fitted with a trend line indicating presence or absence of a significant trend.

Table 6.1 Changes to limits of reporting for parameters analysed at Analytical Services Tasmania (AST), post 2014.

Test	Pre 2014	Post 2014	Units	Measurement uncertainty
Ammonia	2	5	µg/L	±10% for values greater than 25 µgN/L
±5 µg/L for values less than or equal to 25 µgN/L				
Nitrate + nitrite	2	2	µg/L	±10% for values greater than 10 µgN/L
±2 µg/L for values less than or equal to 10 µgN/L				
Dissolved reactive phosphorus	2	3	µg/L	±10% for values greater than 15 µgP/L
±3 µg/L for values less than or equal to 15 µgP/L				
Total Kjeldahl nitrogen	40	100	µg/L	±18% ±100 µgN/L, whichever is greater
Total nitrogen	40	100	µg/L	±18% ±100 µgN/L, whichever is greater
Total phosphorus	5	10	µg/L	±15% ±10 µgP/L, whichever is greater
Total zinc	1	2	µg/L	± 25% for values greater than 2 µg/L

Figure 6.1 Map showing the location of ambient water quality monitoring (AWQ) sites in the Derwent Estuary. Sampling organisation is indicated with shapes: circle = Derwent Estuary Program and Environment Protection Authority, diamond = Nyrstar Hobart, triangle = Boyer).



6.1.2 Physicochemical parameters

Results

Salinity

During the reporting period (2020–2024), hereafter referred to as the reporting period, median salinity of the Derwent Estuary ranged from freshwater (S = 0) in surface waters at New Norfolk to seawater (S = 34) in bottom and surface waters in the lower estuary. Salinity in bottom waters varied little and was typically high except for the upper estuary which was highly variable particularly in winter when the River Derwent flows are high and the salt wedge is displaced further downstream. Surface-water salinity in the Derwent Estuary is variable (high in summer, lower in winter) due to seasonal variability in rainfall, catchment inflows from the River Derwent, and storm events (Figure 6.2).

Over the monitoring period (2007–2024 for the upper estuary, lower estuary, and Ralphs Bay; and 2011–2024 for the middle estuary and middle estuary bays), hereafter referred to as 'the monitoring period', a zone-wide increase in salinity was detected in upper estuary salinity (Figure 6.2).

Water temperature

During the reporting period, median water temperature ranged from 12.4 °C (NN surface) to 14.8 °C (CB surface). Water temperature showed strong seasonality with all sites warmer in the summer than in the winter. Surface waters in the upper estuary tended to be cooler and increased downstream, whilst the opposite was true for bottom waters. Temperature in upper estuary waters was the most variable. There were no exceedances of DGVs for water temperature during the reporting period (Figure 6.3).

Over the monitoring period, no zone-wide trends were detected for water temperature (Figure 6.3).

Dissolved oxygen

During the reporting period, median Dissolved Oxygen (DO) ranged from 10.6 mg/L in surface waters at NN, declining gradually downstream to 8.7 mg/L at the estuary mouth (B1). Dissolved oxygen showed strong seasonality in surface and bottom waters with high concentrations in winter and spring and lower concentrations in summer and autumn. DO in bottom waters varied greatly from 3.8 mg/L at upper estuary site U19 to 8.9 mg/L at RBS. Particularly low levels were observed in bottom waters at NN and U19, which experienced seasonal hypoxia in summer and autumn. At U19, low DO (<3 mg/L) is an annual occurrence and can persist for up to 3 months a year. Median values fell within the DGVs for DO during the reporting period (Figure 6.4).

Over the monitoring period, zone-wide declines in surface-water DO was observed at middle estuary and middle estuary bays whilst bottom water DO increased in Ralphs Bay (Figure 6.4). All trends were statistically significant based on the correlated seasonal Mann-Kendall test.

pH

During the reporting period, median pH ranged from 7.0 in upper estuary bottom waters to 8.1 in lower estuary surface waters and pH generally increased towards the lower estuary. Median pH values fell outside the DGVs at all middle estuary bay sites in surface and bottom waters during the reporting period (Figure 6.5).

Over the monitoring period, zone-wide declines in pH were observed in the lower and upper estuary, whilst pH increased in the middle estuary and middle estuary bays displayed a slight increase (Figure 6.5).

Discussion

In the Derwent Estuary, salinity, water temperature, DO and pH are key environmental drivers that shape habitat distribution and influence the rates of biological and chemical processes. These parameters exhibit strong seasonal variability, driven by the interplay between marine inflows/outflows and freshwater inputs.

Water temperature and salinity

No significant changes in water temperature or salinity were observed during the monitoring period. Notably, surface water temperatures have not shown increases that would typically be expected under a warming climate. This is not uncommon in estuarine environments where high seasonal variability, freshwater inflows and tidal mixing can mask or buffer long-term warming trends. Long-term studies show that water temperatures are rising in Australian estuaries, with some warming at rates faster than oceans (Scanes *et al*., 2020; Ridgway and Ling, 2023). The frequency of marine heatwaves (MHWs) is also expected to increase with ongoing climate change (Frölicher *et al*., 2018; Nardi *et al*., 2025). MHWs have already impacted the Derwent Estuary in 2015/16, 2017/18, and 2023/24 (see **Section 2.4.6**). These warming events pose significant ecological threats, including elevated risks of harmful algal blooms (Gobler, 2020) and degradation or loss of macrophyte meadows (Duarte *et al*., 2018; Strydom *et al*., 2020). The DEP are currently working with Institute of Marine and Antarctic Studies (IMAS) to better understand the dynamics and impacts of MHWs in the Derwent Estuary. See **Section 2.4.6** for more details.

Dissolved oxygen

DO is a critical parameter influencing estuarine health and is essential to supporting many forms of life. Over the monitoring period, a significant increase in DO in bottom waters in Ralphs Bay is likely representative of a general improvement in water quality conditions, discussed further **Section 6.1.4**. While a decline in DO was observed in the middle estuary and middle estuary bays, concentrations typically remain within healthy levels (~6–8 mg/L), therefore this trend was not investigated further.

Seasonally low DO (< 3 mg/L) and hypoxia (< 2 mg/L) have been well documented in the upper Derwent Estuary (Davies and Kalish, 1994; Sheldon and Pope, 2019; DEP, 2020). Low DO in the upper estuary has been shown to drive the release of ammonium from sediments, following the breakdown of organic material (Abell *et al*., 2014). There is also considerable risk of release of metals from sediments to overlying waters(Banks, Ross, Keough, Eyre, *et al*., 2012; Liu *et al*., 2019; Jaiswal and Pandey, 2020), though this risk is less pronounced in the upper estuary where metals in sediment and bottom waters is low (DEP, 2015; Macleod and Coughanowr, 2019). The DEP has expanded its monitoring program in the upper estuary to better understand dynamics between DO and catchment flow, as well as sources of organic matter contributing to hypoxic conditions. Further monitoring includes continuous logging of bottom water oxygen levels, intermittent DO profile surveys, and pilot studies trialling novel dissolved organic matter characterisation methods. Results of upper estuary dissolved oxygen monitoring are reported in **Section 6.2** and the pilot study is reported in **Section 6.3**.

pH

pH, an index of hydrogen ion activity (acidity – alkalinity), governs numerous chemical and biological processes in water, including nutrient cycling, metal solubility, and carbonate equilibria. Under climate change, rising atmospheric CO_2 increases its partial pressure (pCO_2), enhancing dissolution into surface waters and lowering estuarine pH (ocean acidification). Estuarine pH is also influenced by other drivers, including oxidation of acid sulfate soils (natural or disturbed by drainage and dredging) that release sulfuric acid; eutrophication and organic-matter respiration that elevate CO_2; nitrification of ammonium from wastewater and runoff, which generates acidity; upwelling or groundwater inputs rich in CO_2; freshwater inflows that modify total alkalinity and buffering capacity; temperature effects on CO_2 solubility; and mixing/stratification that can trap low-pH bottom waters. Conversely, inputs that increase alkalinity (e.g., carbonate minerals or liming) can raise pH. These physical, chemical, and biological processes interact across daily to seasonal timescales, driving variability around the long-term acidification trend. The response of an estuary to ocean acidification is dependent on its buffering capacity and biological processes (Cai *et al*., 2020; Hall *et al*., 2023), with the upper estuary (fresher component) being lower in alkalinity and hence having a lower capacity to buffer changes in pH.

In the Derwent Estuary, zone-wide changes over the monitoring period were observed, with pH decreasing at the marine (lower) and riverine (upper) reaches of the estuary. Decreases in the lower estuary are likely driven by climate and oceanic processes (acidification) consistent with Australian and global studies. pH decreases in the lower estuary (-0.001 pH units per year) were consistent with expected pH decreases (0.0017 – 0.0020 per year) due to ocean acidification (Carstensen and Duarte, 2019; Scanes *et al*., 2020). However preliminary analysis indicates that climate change is unlikely to explain the rate of change observed in the upper estuary with a large rate of change of -0.023 pH units per year. Data collected between 2011 and 2013 was flagged as possibly impacted by poor pH sensor quality and could be contributing to the unexpected observed rate of change. Middle estuary sites displayed an opposite positive trend with pH increasing at a rate of 0.008 pH units per year.

Butler (in prep.) details a robust methodology for assessing whether observed changes in pH in the Kanamaluka/Tamar estuary were likely driven by climate change, increased organic inputs, increased drainage from acid sulfate soils or acid mine drainage, industrial discharges, or lower pH precipitation across the catchment. The DEP intend to apply this methodology and set of hypotheses to determine the key drivers of pH changes in the upper Derwent Estuary.

Figure 6.2 Spatial and temporal variation in salinity values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in salinity for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly salinity for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant ($p<0.05$) and dashed is not significant ($p>0.05$).

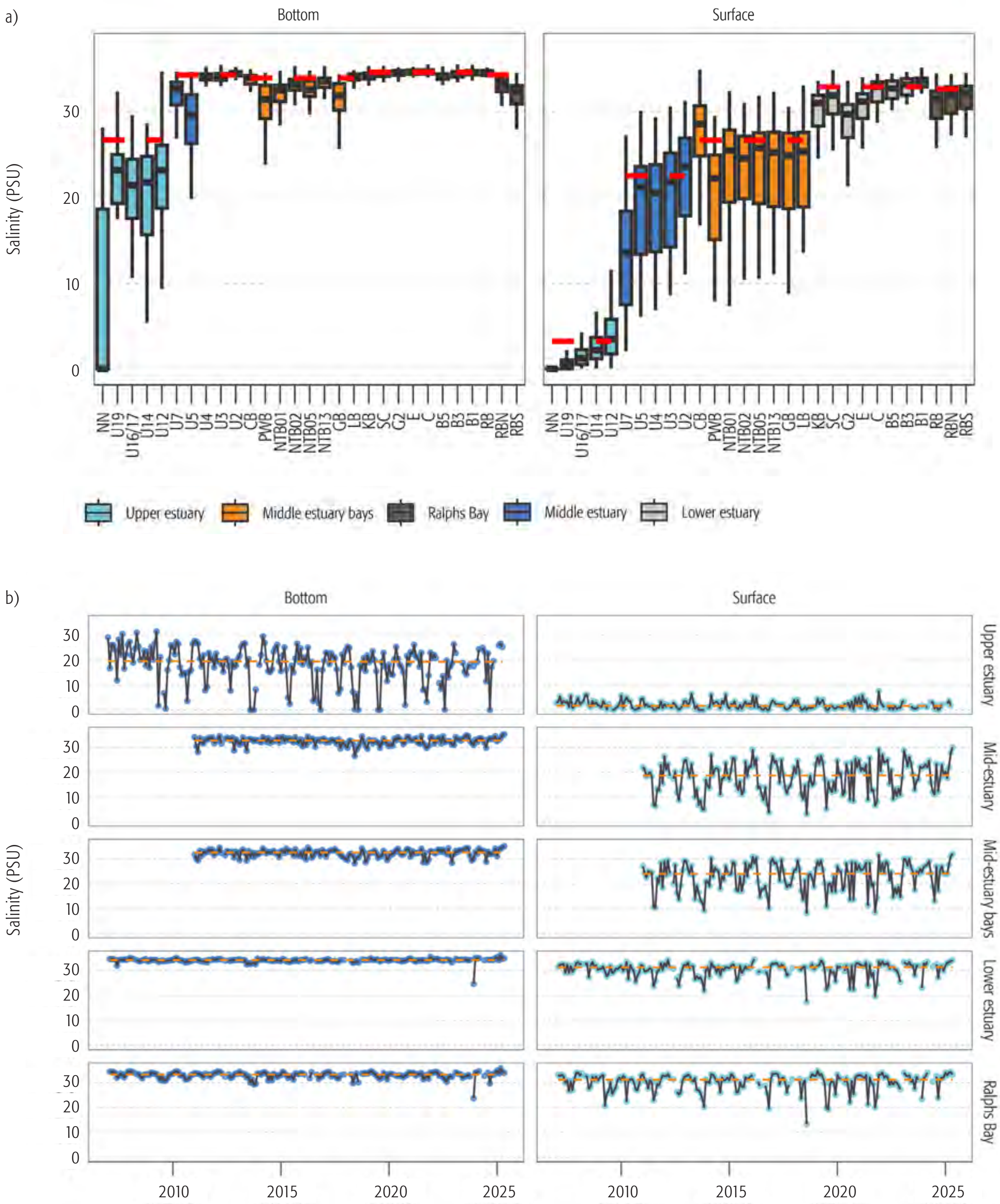


Figure 6.3 Spatial and temporal variation in water temperature (°C) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in temperature for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly temperature for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).

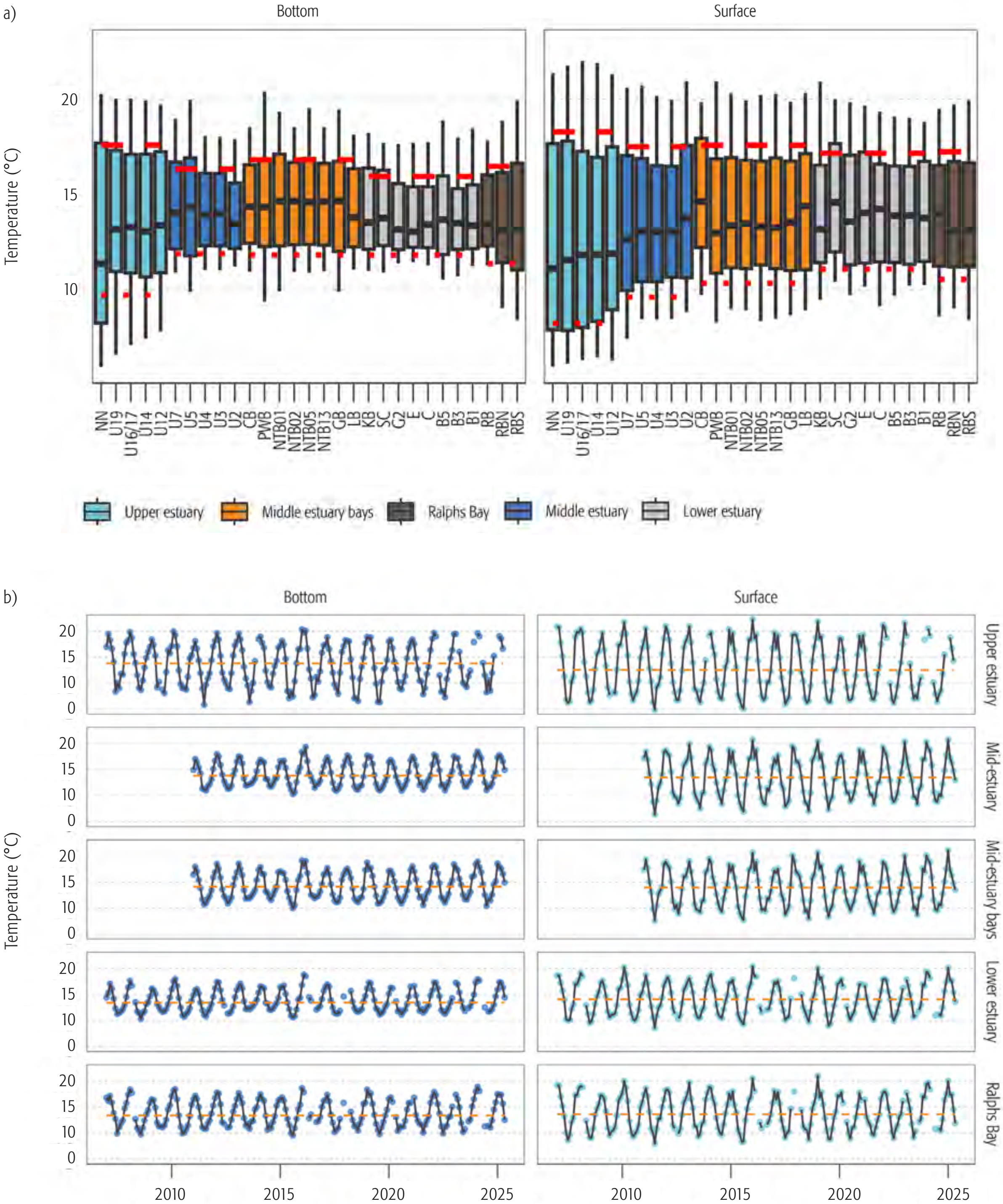


Figure 6.4 Spatial and temporal variation in DO (mg/L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in DO for each site aggregated for the period 2020 – 2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly DO for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).

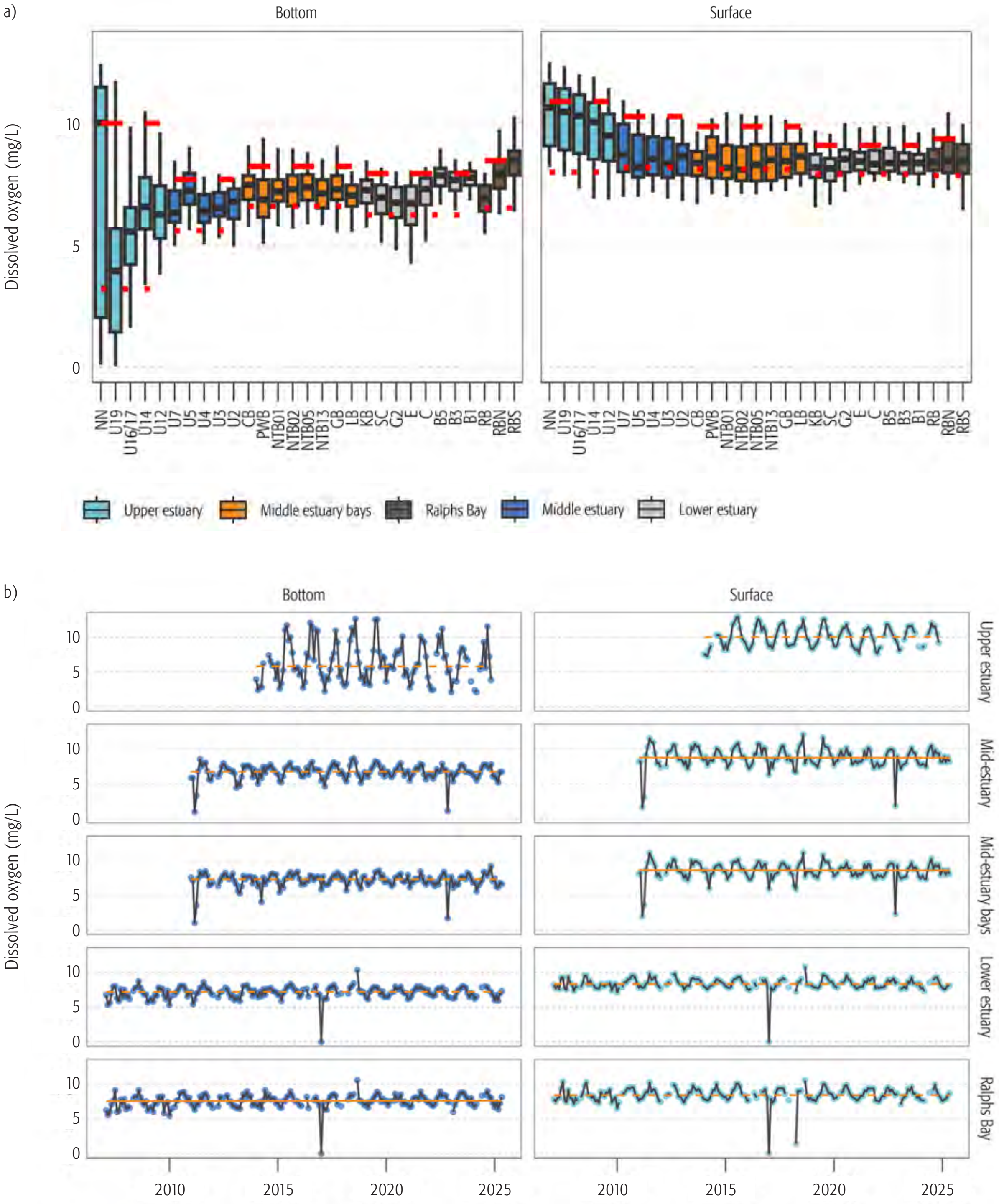
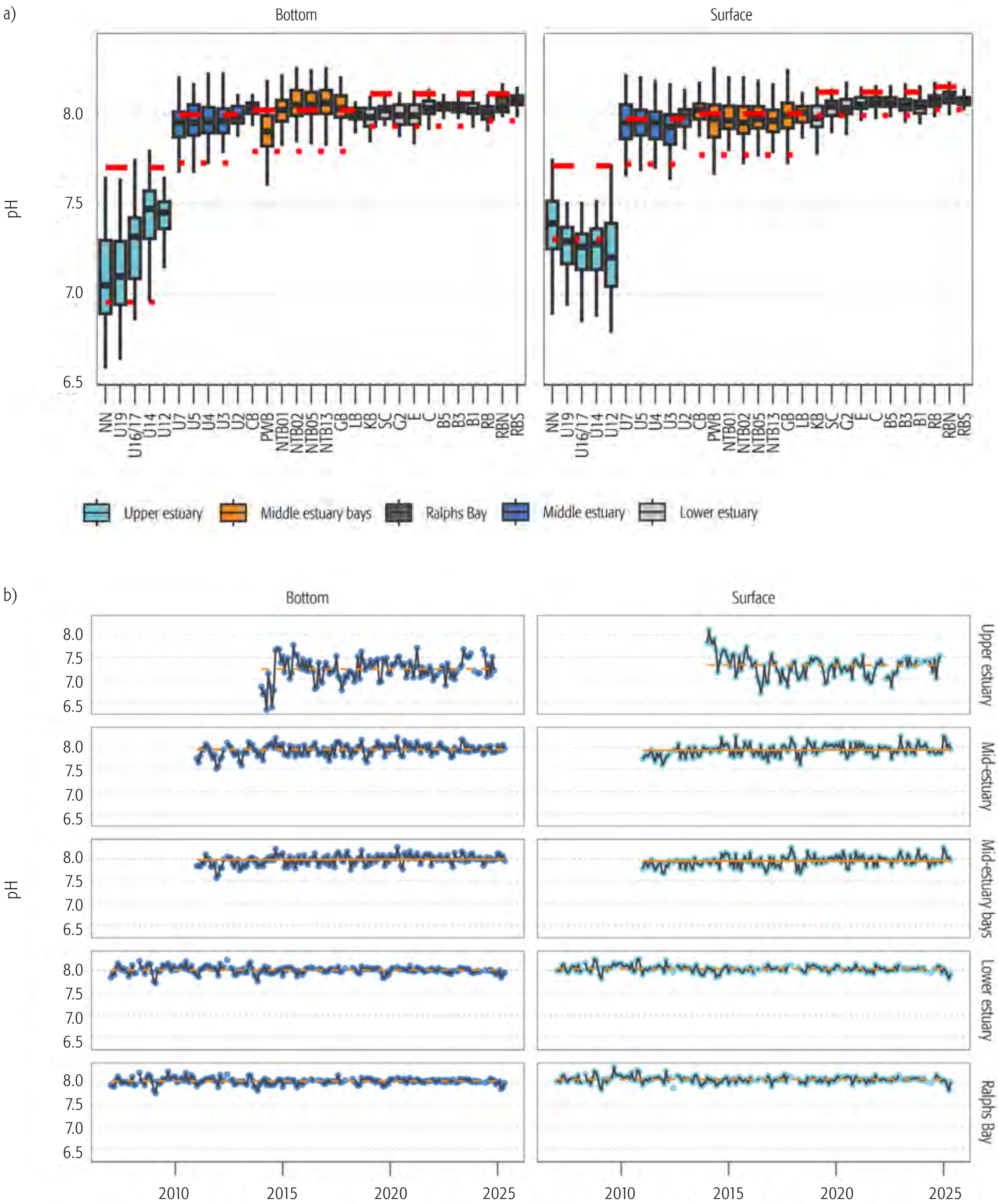


Figure 6.5 Spatial and temporal variation in pH values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in pH for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly pH for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).



6.1.3 Water clarity

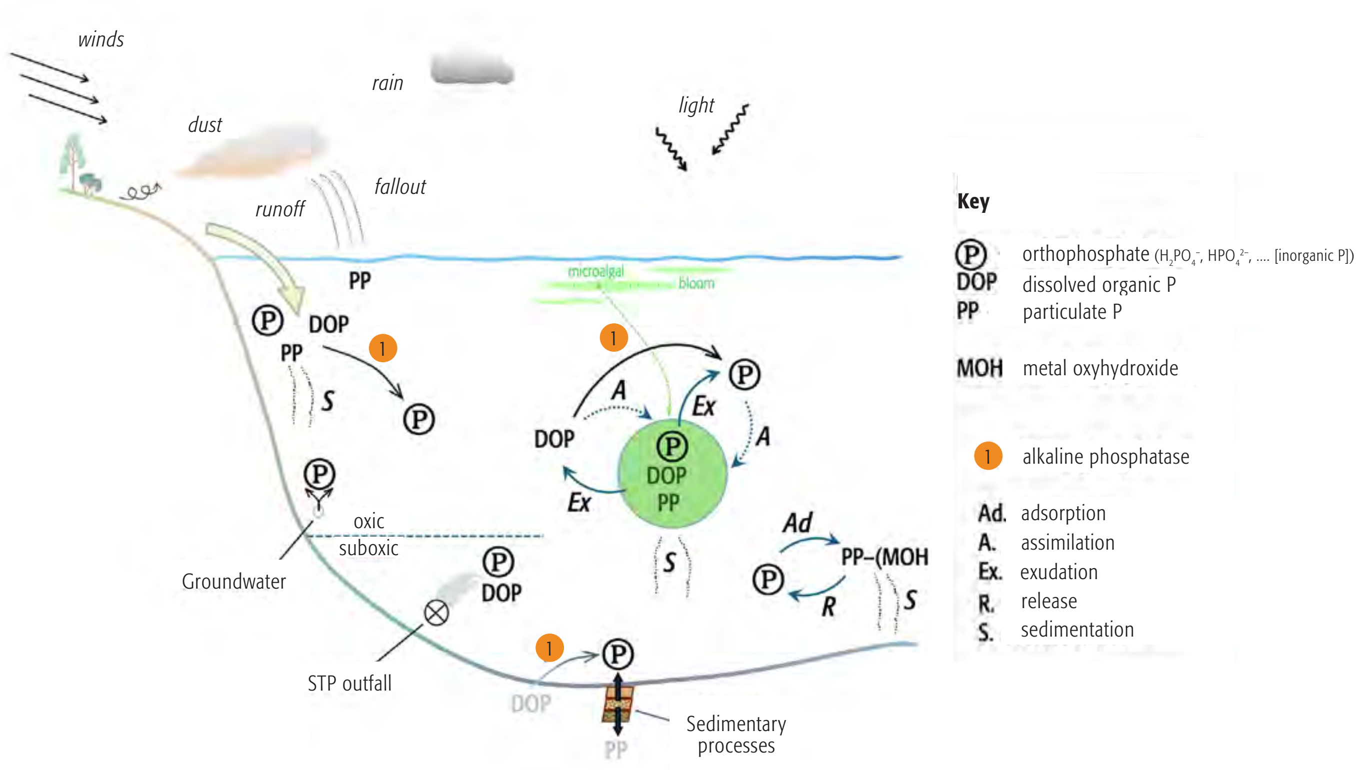
The optical properties of water include clarity, turbidity and colour and are determined by attenuation of light as it passes through the water column (Figure 6.10).

Optical properties of water are important for the following reasons:

- attenuation of light can limit the growth of aquatic plants reliant on photosynthesis, such as phytoplankton, seagrass, macrophytes, macroalgae and benthic microalgae;
- changes in colour can alter the spectrum of light available for photosynthesis and illumination;
- high levels of totals suspended solids can clog fish gills, smother filter feeding organisms and alter benthic substrates;
- impact on fish and birds' ability to see and hunt prey;
- degrades aesthetic values.

The DEP monitoring program measures optical properties using Secchi depth, turbidity (DEP sites only), true colour and total suspended solids.

Figure 6.6 Schematic diagram of phosphorus cycling in the Derwent Estuary. Numbered arrow is a biogeochemical reaction involving forms of P. Processes tagged with a letter are transfer/transport mechanisms. The green disk represents a microalgal cell.



Results

Secchi depth

During the reporting period, Secchi depth ranged from 1 m in the middle estuary (U7) to 6 m in the lower estuary (B1) and typically improved downstream. Secchi depth exhibited some seasonality with depth greatest in late summer to early autumn, likely due to reduced river flows resulting in lower total suspended solids and colour. In marine waters, greater Secchi depth in autumn is likely due to less wind-driven mixing and reduced microalgal biomass. DGVs were exceeded at lower estuary sites (B1, B5) and in the middle estuary, indicating improvements in water clarity (Figure 6.6).

Over the monitoring period, Secchi depth increased significantly in the lower estuary and Ralphs Bay (Figure 6.6).

True colour

During the reporting period, median values for true colour ranged from 43 colour units (CU) in the upper estuary (NN surface) to 5 CU at lower estuary sites which routinely recorded values at detection limit. True colour was greatest in the upper estuary and gradually declined downstream and was much higher in freshwater influenced surface waters. True colour exhibited strong seasonality with highest observations recorded during winter months when river flows are high (Figure 6.7).

Over the monitoring period, zone-wide increase in true colour was detected at all zones (Figure 6.7).

Total suspended solids

During the reporting period, median total suspended solids concentrations ranged from 2 mg/L at many sites (LoR) to 6 mg/L in bottom waters in the middle estuary (U12). Total suspended solids were typically higher in bottom waters, particularly at U7 and U12. Values were typically greatest following high rainfall and strong stormwater and river inflows (Figure 6.8).

Over the monitoring period, zone-wide decline in total suspended solids was observed in bottom and surface waters in the middle estuary and middle estuary bays (Figure 6.8).

Turbidity

During the reporting period, turbidity was typically very low with median values ranging from 0.5 in the lower estuary (B1 surface) to 1 NTU at the uppermost site (U2). Turbidity was typically lower in bottom than in surface waters.

Over the monitoring period, zone-wide decline in turbidity was detected in bottom waters of the lower estuary.

Discussion

Water clarity improved in the Derwent Estuary over the monitoring period, particularly in the lower and middle estuary, with all measured water quality parameters –Secchi depth, turbidity, and total suspended solids – showing improving trends.

These improvements are supported by other lines of evidence including ecological surveys. Rocky reef biodiversity surveys conducted in 2024 demonstrated significant improvements in rocky reef habitat with a dramatic increase in percent cover of algae at several sites, spanning from Bedlam Walls North to Battery Point– locations that were largely dominated by tubeworm matting in 2010, representing an alternate stable state to kelp beds. One of the most notable observations was that the common kelp *Ecklonia radiata* expanded its distribution northwards into areas where it had not been previously recorded. *Ecklonia radiata* expanding its range up estuary, indicates significant improvements in water clarity, and healthier subtidal habitats (see **Section 9.3.1**).

Stormwater monitoring across Greater Hobart catchments indicated a decline in total suspended solids between the two most recent monitoring periods–2010 and 2025. TSS concentrations were generally lowest at sites discharging to the middle estuary (see **Section 5.2**). This spatial pattern likely reflects reduced land disturbance, such as fewer subdivisions, and may also indicate improvements in catchment management practices over time. These improvements include the increased use of gross pollutant traps and the implementation of erosion and sediment control measures.

Anecdotal observations suggest that the MONA ferry wake may impact water clarity in the Derwent Estuary, with speculation that frequent vessel movement could be contributing to reduced visibility. Foreshore monitoring by the DEP at Kangaroo Bay indicated that regular ferry operations (Derwent Ferries) are likely to have caused minor coastal erosion. However, during the monitoring period, the impacts of ferry wake disturbance were outweighed by those of large storm events (see **Section 9.7.1**). Middle estuary water quality monitoring was conducted on Tuesdays, when the MONA ferry does not operate, which may limit the detection of wake-related impacts on water clarity. However, this would only be the case at middle estuary sites and has no bearing on improvements in the lower estuary.

Figure 6.7 Spatial and temporal variation in Secchi depth (m) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in Secchi depth for each site aggregated for the period 2020–2024. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean Secchi depth for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).

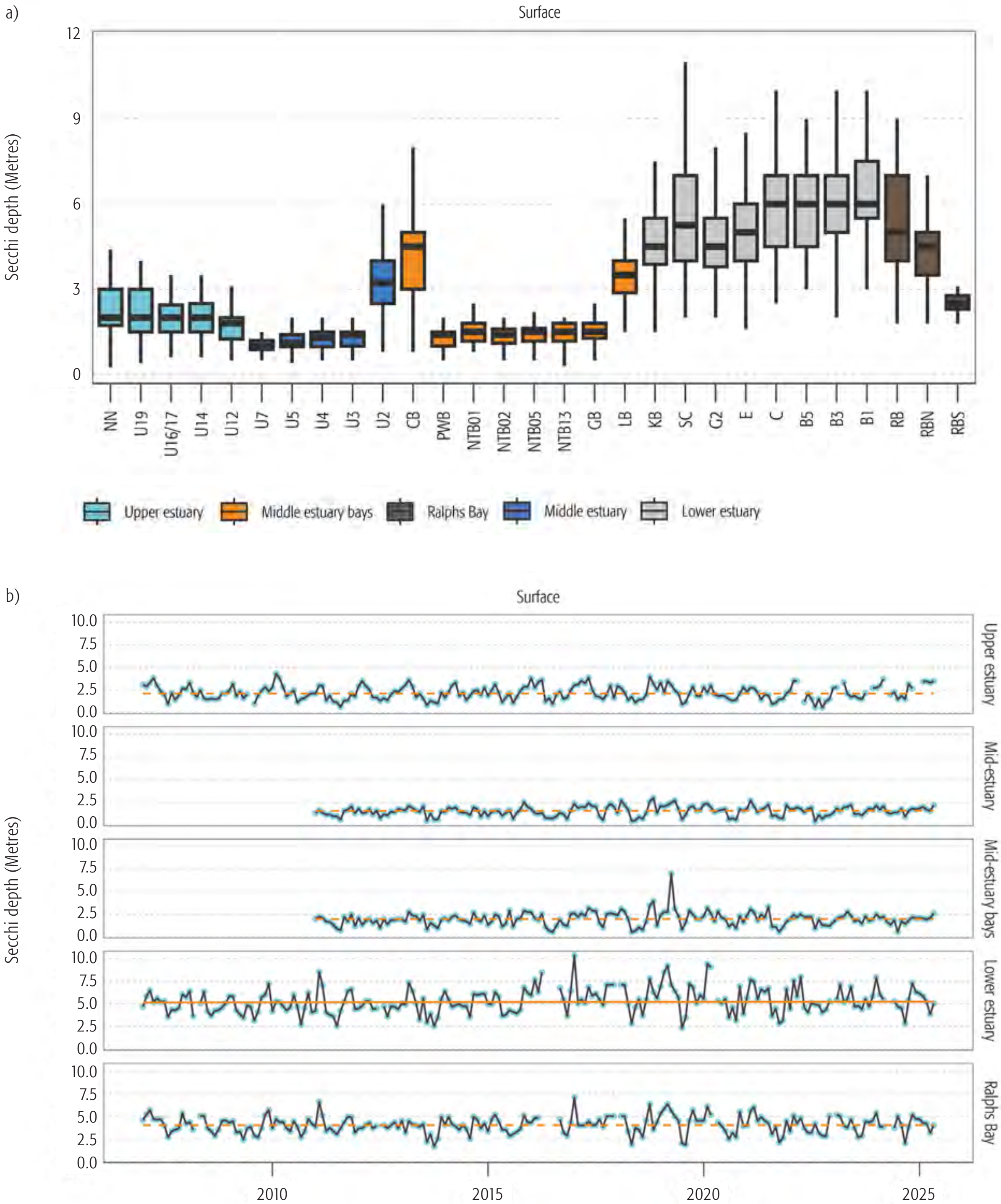


Figure 6.8 Spatial and temporal variation in true colour (CU) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in true colour for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly true colour for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).

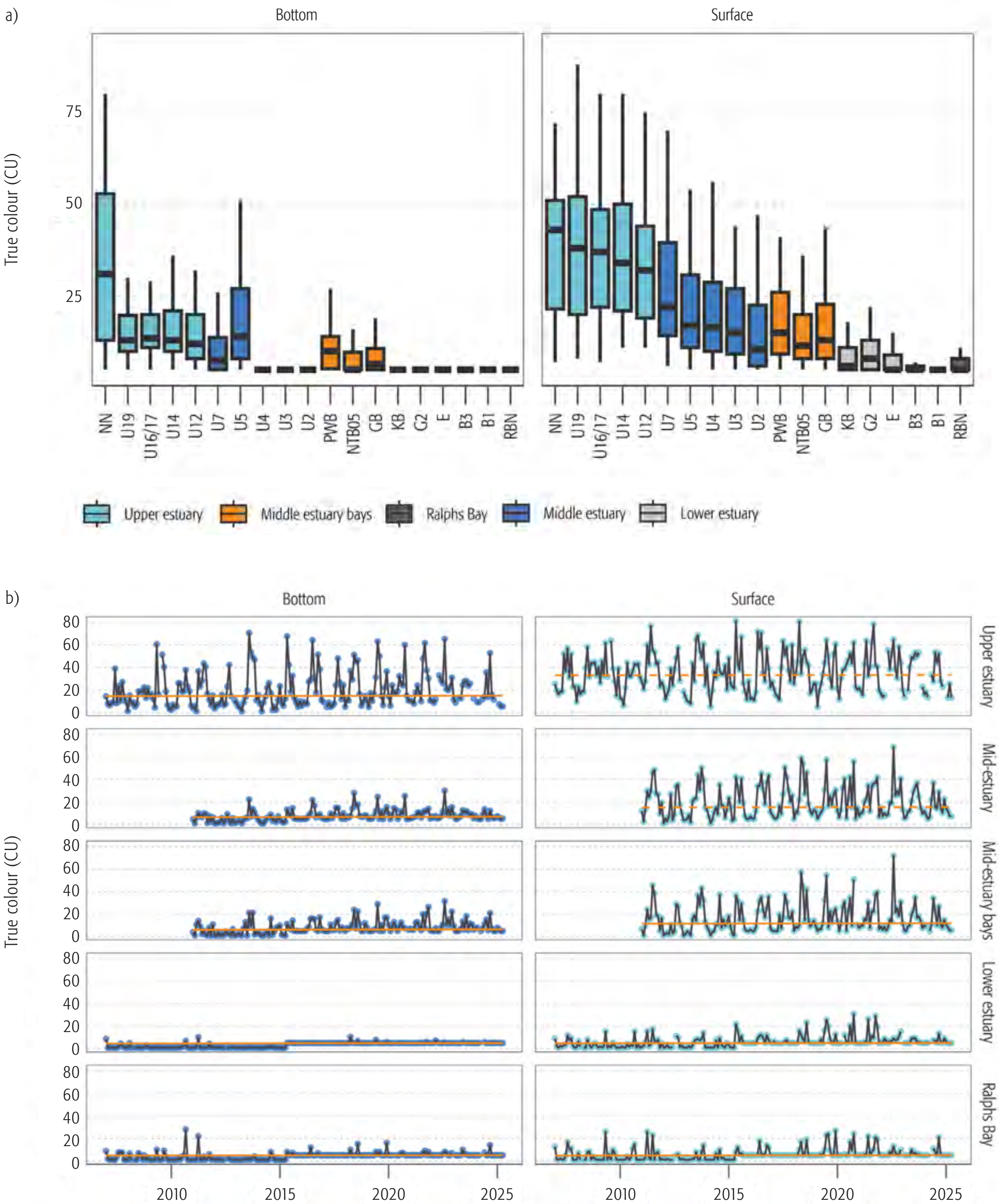
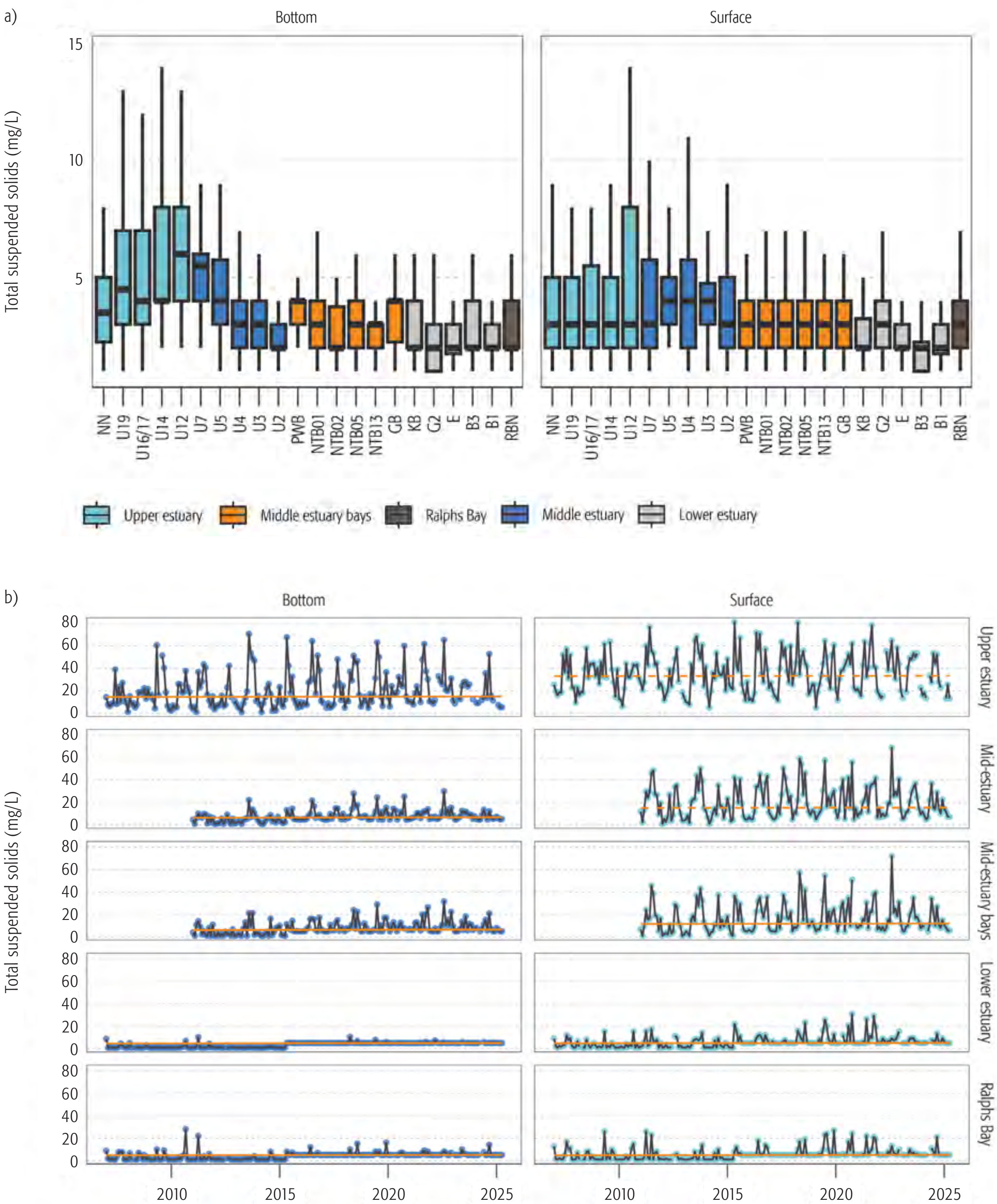


Figure 6.9 Spatial and temporal variation in total suspended solids (mg/L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in total suspended solids for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly total suspended solids for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites Sen's slope trend line fitted where line is significant ($p<0.05$) and dashed is not significant ($p>0.05$).



6.1.4 Nutrients and chlorophyll a

Nutrient enrichment in urban waterways is largely driven by increasing population density and associated industrial activity, including discharges from sewage treatment plants (STPs), industrial sources, agricultural and stormwater runoff. Inputs contribute to nutrient concentrations that exceed natural background levels. Excessive nutrient inputs can lead to eutrophication, which is recognised as one of the most significant threats to coastal ecosystem health (Howarth *et al*., 2000; Bricker *et al*., 2008).

Early signs of nutrient enrichment include elevated nutrient and chlorophyll a levels, algal blooms, increased turbidity, and oxygen fluctuations. These changes can lead to serious ecological impacts including formation of hypoxic zones (low DO), degraded water quality, loss of biodiversity, fish kills and harmful algal blooms. Beyond ecological impacts, eutrophication also diminishes aesthetic and recreational values including fishing success, tourism, and real estate value (Hoagland *et al*., 2002; OzCoasts, 2020).

In the Derwent Estuary, many of these impacts have been observed including elevated nutrients in the middle to upper estuary, seasonally recurrent hypoxia in the upper estuary (**Section 6.2**), macroalgal blooms contributing to dieback of upper estuary macrophyte beds (**Section 9.5**), fish kills (ABC News, 2015) and harmful microalgal blooms (often associated with toxins and poisonings).

Nutrient and chlorophyll a concentrations are key indicators of the nutrient enrichment and eutrophic status of a waterway. Chlorophyll a, a plant pigment, is used as an indicator of algal biomass. The primary nutrients of concern are nitrogen and phosphorus, which cycle through various chemical forms from short-lived dissolved species like ammonia and nitrate, to particulate-bound forms, or assimilated into organisms such as phytoplankton. For a more detailed overview of nutrient cycling in aquatic systems see DEP (2015).

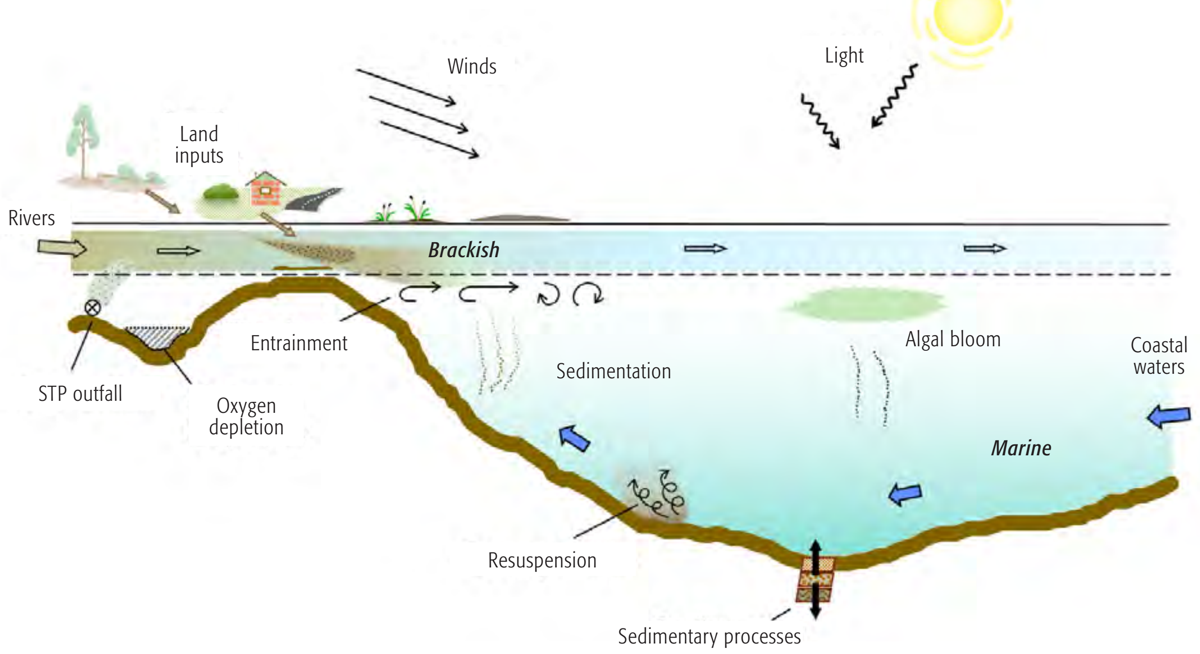
Circulation, temporal and spatial variability in Derwent Estuary nutrients

Temporal

Nutrient inputs to the Derwent Estuary vary both seasonally and spatially. In winter and spring, elevated nutrient concentrations are observed throughout the estuary due to a combination of seasonally enriched marine water and increased riverine inputs. In contrast, during summer and autumn, both marine and riverine inputs are typically low, and surface nutrient concentrations are further depleted by biological productivity (Wild-Allen *et al*., 2013, 2023). These seasonal fluxes are amplified, particularly in the middle estuary, by year-round anthropogenic nutrient loading.

The River Derwent is a primary source of nutrients to the estuary (Wild-Allen and Andrewartha, 2016). There is a strong correlation between total nitrogen at New Norfolk

Figure 6.10 Conceptual diagram of natural and pollution sources impacting the water column optical properties–colour and particle load–in the Derwent Estuary. Colour can be derived from catchment water containing plant matter or from human activities (e.g. colourants in waste streams). Small particles–organic or mineral–are derived from diffuse sources (e.g. land runoff, atmospheric deposition, aquatic microorganisms cells), and direct discharges (point sources), such as stormwater pipes and sewage treatment plant outfalls. The DEP monitoring program measures optical properties using Secchi depth, turbidity, true colour and total suspended solids.



(NN) and River Derwent flow, with total nitrogen levels typically highest in winter, coinciding with peak flow, and lowest in summer (DEP, 2015). Catchment mass loads indicate that agriculture is the largest source of nutrients in the catchment, with loads highest during winter. Land-based fish farms also are a significant contributor particularly for dissolved nutrients in summer (Proemse *et al*., 2022a).

In late winter, intrusions of nutrient-rich subantarctic bottom water occur due to the relaxation of the Zeehan Current, delivering high levels of dissolved inorganic nitrogen to the estuary (Wild-Allen *et al*., 2013, 2023). Recent biogeochemical modelling in Storm Bay also shows that bottom water nitrogen concentration can be elevated in summer, associated with East Australian Current eddies, although this process exhibits strong interannual variability (Wild-Allen *et al*., 2023).

Spatial

In the upper estuary, surface water nutrient concentrations are typically low. However, bottom water concentrations of total ammoniacal nitrogen are often elevated. This has been attributed to the release of nutrients from sediments via mineralisation, especially during summer when DO levels are low (Banks *et al*., 2012; Abell *et al*., 2014).

Nutrient concentrations are generally highest in the middle estuary, where treated effluent from STPs, industrial discharges, and stormwater runoff augment the natural seasonal inputs. This is particularly evident in poorly flushed sites such as PWB where concentrations are consistently elevated.

In the lower estuary, nutrient concentrations are typically low, especially in surface waters, reflecting greater dilution, better flushing, and reduced local inputs.

Results

Total nitrogen

During the reporting period, median total nitrogen values ranged from 210 µg/L in surface waters of the upper estuary (U19) to 430 µg/L in bottom waters of the upper estuary (U19). Total nitrogen was typically higher in bottom than in surface waters. Total nitrogen in bottom waters typically decreased downstream whilst nitrogen in surface waters increased downstream. DGVs for total nitrogen were exceeded at an upper estuary site (U19 bottom) and a middle estuary bay site (PWB surface) (Figure 6.9).

Over the monitoring period, total nitrogen concentrations increased significantly in both surface and bottom waters of the upper estuary (NN, U16/17, U14, U12) (Figure 6.9).

Total ammoniacal nitrogen

During the reporting period, median total ammoniacal nitrogen values ranged from 5 µg/L (LoR) in bottom and surface waters of the lower estuary to 120 µg/L in bottom waters of the upper estuary (U19). In the upper estuary, total ammoniacal nitrogen concentrations in bottom waters were higher than in surface waters. DGVs were exceeded at lower estuary (U19 bottom) and middle estuary bay (PWB bottom) sites (Figure 6.10).

Over the monitoring period, total ammoniacal nitrogen concentrations declined significantly in surface waters at Ralphs Bay (RBN) and increased significantly in the surface waters of the upper estuary (NN, U16/17, U14). Significant decreases were also detected at individual sites (U2, U4 and B1); however, these trends were not evident in the aggregated zone-wide analysis (Figure 6.10).

Nitrate + nitrite

During the reporting period, median nitrate + nitrite concentrations ranged from to 2 µg/L in surface and bottom waters of Ralphs Bay to 78 µg/L in surface waters of the middle estuary bays. Nitrate + nitrite concentrations were typically higher in surface than in bottom waters, with lower estuary sites being the exception. Nitrate + nitrite exhibited strong seasonality, particularly in the lower and upper estuary. The middle estuary bay DGV for nitrate + nitrite was exceeded at PWB in bottom and surface waters (Figure 6.11).

Over the monitoring period, nitrate + nitrite concentrations increased significantly in surface waters of middle estuary bays, and bottom and surface waters of the upper estuary (Figure 6.11). Nitrate + nitrite declined in bottom and surface waters of Ralphs Bay.

Total phosphorus

During the reporting period, median total phosphorus ranged from 10 µg/L in surface waters of the upper estuary (NN) to 63 µg/L in bottom waters at a middle estuary bay (PWB). Total phosphorus was typically higher in bottom than in surface waters, particularly in the upper estuary. Surface water concentrations increased towards the middle estuary, and slightly toward the mouth of the estuary. There were no exceedances of DGVs for total phosphorus during the reporting period (Figure 6.12).

Over the monitoring period, total phosphorus declined significantly in bottom and surface waters of the lower estuary and Ralphs Bay. Total phosphorus also declined significantly in bottom waters of the middle estuary (Figure 6.12).

Dissolved reactive phosphorus

During the reporting period, median dissolved reactive phosphorus ranged from 3 µg/L in surface waters of the upper estuary (NN) to 29 µg/L in surface waters at PWB. Dissolved reactive phosphorus was typically higher in bottom than in surface waters. In surface waters, dissolved reactive phosphorus gradually increased towards the middle estuary, before declining towards the estuary mouth. DGVs for dissolved reactive phosphorus were exceeded in all estuary zones (except Ralphs Bay) in surface and bottom waters (Figure 6.13).

For the monitoring period, dissolved reactive phosphorus increased significantly in bottom and surface waters of the lower estuary, middle estuary and middle estuary bays (Figure 6.13).

Chlorophyll a

During the reporting period, chlorophyll a concentrations ranged from 1.1 µg/L in the upper estuary (U16/17) to 2.7 µg/L in the middle estuary (U2). Median chlorophyll a concentrations were typically low in the upper estuary and increased towards middle estuary bay sites, before decreasing towards the estuary mouth. In lieu of no draft DGVs currently set for the Derwent, the 80th percentile for the period 2009-2013 was applied. There were no exceedances of DGVs for chlorophyll a for the reporting period (Figure 6.14).

No zone-wide trends were detected for chlorophyll a for the monitoring period. Chlorophyll a did decline significantly at PWB (Figure 6.14).

Non-purgeable organic carbon

During the reporting period, median non-purgeable organic carbon concentrations ranged from 0.5 mg/L in bottom waters of the lower estuary (B1) to 4.8 mg/L in the upper estuary (U19). Median non-purgeable organic concentrations were typically high in the upper estuary and decreased towards the lower estuary and concentrations were typically greater in surface than in bottom waters.

Over the monitoring period, there was a significant decrease in non-purgeable organic carbon concertation in bottom waters of the lower estuary, and a significant increase in surface waters of the upper estuary (Figure 6.15).

Discussion

Nutrients and phytoplankton biomass

Nutrient concentrations in the estuary over the reporting period follow previously documented spatial and seasonal patterns (DEP, 2015, 2020). Dissolved nutrients remained elevated in surface waters of the middle estuary and bays and in the bottom waters of the upper estuary. PWB (middle estuary bay) and U19 (upper estuary) were significantly affected, with concentrations exceeding the DGVs for the majority of nutrient indicators.

Seasonal trends were consistent with prior observations, including elevated nitrate + nitrite and dissolved reactive phosphorus from marine sources in later winter/early spring, and elevated nitrate + nitrite from catchment sources in winter.

Upper estuary

Signs of nutrient stress were particularly evident in the upper estuary, where significant increases in nitrogen were observed. Elevated levels at the head of the estuary (NN) and downstream of industrial discharges suggest the influence of multiple sources. Surface waters at site NN showed elevated concentrations of all nitrogen species and non-purgeable organic carbon (NPOC), indicating increased organic matter loading and dissolved nutrient availability from the catchment. This is not supported by recent catchment nutrient loads, which suggest a decrease in nitrogen over the last five years (see **Section 3.2.4**). Further analysis is required to better determine the relationship between nitrogen increases in surfaces waters at NN and catchment flows and nutrient delivery. Catchment water quality and sources are discussed in detail in **Section 3.2.4**.

An increase in true colour and non-purgeable organic carbon in the upper Derwent Estuary is indicative of elevated levels of organic matter entering the system from catchment runoff and point sources. These inputs reduce water transparency and increase nutrient availability. Previous studies have shown that high true colour—largely driven by coloured dissolved organic matter can significantly inhibit light penetration, limiting phytoplankton growth and facilitating the downstream transport of nutrients to the middle estuary (Hallegraeff and Westwood, 1994; Wild-Allen *et al*., 2013). This dynamic has important implications for macroalgal growth, particularly around the extensive macrophyte beds in the upper estuary, where dense macroalgal blooms are commonly observed during spring and summer and can lead to significant smothering and dieback of macrophyte beds (**Section 9.5**).

Previous studies have reported increased dissolved nutrient concentrations in surface waters downstream of Boyer (U16/17, U14, U12) following the upgrade to secondary effluent treatment in 2009 (Ross *et al*., 2012). This trend persisted over the reporting period with a significant deterioration in water quality at these sites. In mid-2024, Boyer installed *in situ* analysers in the combined effluent stream to improve real-time monitoring and management of discharges. These upgrades aim to reduce ammonia levels and minimise total suspended solids entering the estuary. Early results show promising improvements in both dissolved nutrient concentrations and total suspended solids discharged from the site (see **Section 4.2**).

Bottom-water ammonia and hypoxia

Total ammoniacal nitrogen concentrations in bottom waters were consistently elevated during the reporting period, with notable exceedance of the DGV at site U19. Previous studies have linked elevated ammonia in bottom waters of the upper estuary to release from sediments, driven by increased organic matter deposition and biological oxygen demand.

These conditions reduce DO levels and inhibit nitrification (favouring denitrification), resulting in greater ammonia flux from sediments into the water column under low oxygen conditions (Ross *et al*., 2012; Abell *et al*., 2014). Continued loading of organic matter to bottom waters exacerbates seasonal hypoxia observed in the upper estuary.

Particulate matter from the Boyer secondary effluent treatment plant, dominated by microbial content, is considered the main driver of this process at sites downstream of Boyer (U16/17, U14, U12) (Abell *et al*., 2014). However, the extent to which Boyer effluent impacts sites upstream of its effluent discharge point (U19 and NN) remains unclear. A recent pilot study using novel characterisation techniques found that bottom water dissolved organic matter in the upper estuary (upstream of Boyer) was most similar to riverine sources, rather than STP or Boyer effluent (see **Section 6.4**). Further investigation is needed to better characterise dissolved organic matter sources and understand the pressures contributing to poor water quality outcomes in the upper estuary.

Middle estuary nutrient dynamics and management

In the middle estuary, nutrient concentrations exceeded DGVs for all measured parameters at site PWB. These exceedances are likely driven by effluent discharge from the PWB STP located at the mouth of PWB, combined with poor circulation and longer water residence times within the bay.

In 2018, an optimisation program was implemented by TasWater to reduce ammonia concentrations in the PWB discharge. While this led to a zone-wide decline in ammonia, it was accompanied by increases in nitrate + nitrite. The program was later discontinued, as operational demands exceeded the design capacity of the treatment system (**Section 4.1**). These changes likely explain the observed decrease in total ammonia nitrogen and concurrent increase in nitrate + nitrite in the middle estuary over the monitoring period.

The Selfs Point Sewer Transformation project, scheduled for completion in 2027, is expected to significantly reduce nutrient loads discharged (from Selfs Point and Macquarie Point combined) to the estuary—by approximately 58% for nitrogen and 42% for phosphorus (see **Section 4.2**). These reductions are anticipated to improve estuarine water quality and enhance local habitat condition, including rocky reefs, macrophyte beds, and wetlands.

Phosphorus trends in the estuary

During the monitoring period, total phosphorus concentrations showed a declining trend in the lower estuary and Ralphs Bay, while dissolved reactive phosphorus concentrations increased in the lower and middle estuary.

The widespread decline in total phosphorus in the lower estuary may reflect reductions in marine-derived phosphorus over time. In contrast, the increase in dissolved reactive phosphorus in the middle estuary is likely driven by elevated releases from local STPs. STPs employing secondary treatment typically remove particulate phosphorus (which contributes to total phosphorus) but often leave dissolved reactive phosphorus elevated, as it remains in solution unless removed through chemical precipitation or advanced tertiary treatment.

Contaminant load data indicate possible increases in phosphorus discharges over the past three years, particularly from Selfs Point STP, where total phosphorus concentrations were elevated between 2022 and 2024. However, TasWater does not monitor dissolved reactive phosphorus in effluent discharges, making it difficult to determine whether observed shifts in estuarine phosphorus are driven by increases in the dissolved fraction or changes in total load.

Phytoplankton biomass

Chlorophyll a is the most common indicator of phytoplankton biomass and is used to assess eutrophication status of a water body. In the Derwent, median chlorophyll a concentrations were typically low with all sites below 3 µg/L, suggesting trophic status in the Derwent Estuary is currently mesotrophic (Smith, 1998; Wild-Allen *et al*., 2013a). This assessment is based on mean data from the five-year monitoring period, whereas the CSIRO methodology uses annual mean chlorophyl a concentrations. Previous studies have demonstrated that phytoplankton biomass in the upper estuary is likely regulated by low light availability due to freshwater from the catchment that is coloured by humic substances (tannins) (Hallegraeff and Westwood, 1994; Wild-Allen *et al*., 2013a). Increased true colour over the reporting period is likely to have enhanced this process, and may have significant implications for downstream habitats, such as macrophyte beds that experience significant pressure from macroalgal blooms in spring and summer.

Microphytobenthos (MPB) are microscopic photosynthetic organisms that live on the surface of aquatic sediments and can contribute significantly to a systems primary production, particularly in shallow bays and lagoons. MPB have significant ecological importance including the formation of a sediment cap, retention of nutrients in sediment and prevention of nutrient release to overlying waters (Underwood and Kromkamp, 1999; Lake and Brush, 2011). However, the importance of MPB to primary production in the Derwent Estuary is poorly understood, particularly in shallow bays such as Ralphs Bay. It is possible that MPB activity may explain lower nutrient concentrations observed in Ralphs Bay over the reporting period. Further monitoring to understand the role of MPB in nutrient dynamics in this system is recommended.

Improvements in Ralphs Bay

Water quality in Ralphs Bay improved significantly over the monitoring period across multiple nutrient-related indicators. Historically, poor water quality in the bay has been attributed to local point sources (notably the Rokeby STP discharge), estuarine circulation patterns, extended water residence times, and morphological features such as shallow depths, which influence water temperature and nutrient dynamics. Widespread loss of seagrass in the bay was documented by Rees (1993) and attributed to poor water quality.

Since 2010, the Rokeby STP (which discharges to deep water outfall at Tranmere) has operated as a 100% reuse facility and may have contributed to reduced local sources of nutrients over the period. Another significant action during the reporting period, was the transition of Lauderdale from septic to sewered system in 2013/14 which likely also contributed to a reduction in nutrient sources.

Reduced nutrient loading over the monitoring period is likely to have contributed to significant improvements in water quality, particularly reductions in total ammoniacal nitrogen and total phosphorus. As a result of improved water quality over time, Ralphs Bay has been selected as a suitable site for seagrass restoration trials, detailed in **Section 9.5.2**.

Ralphs Bay

Figure 6.11 Spatial and temporal variation in total nitrogen (µg /L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in total nitrogen for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly total nitrogen for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant ($p<0.05$) and dashed is not significant ($p>0.05$).

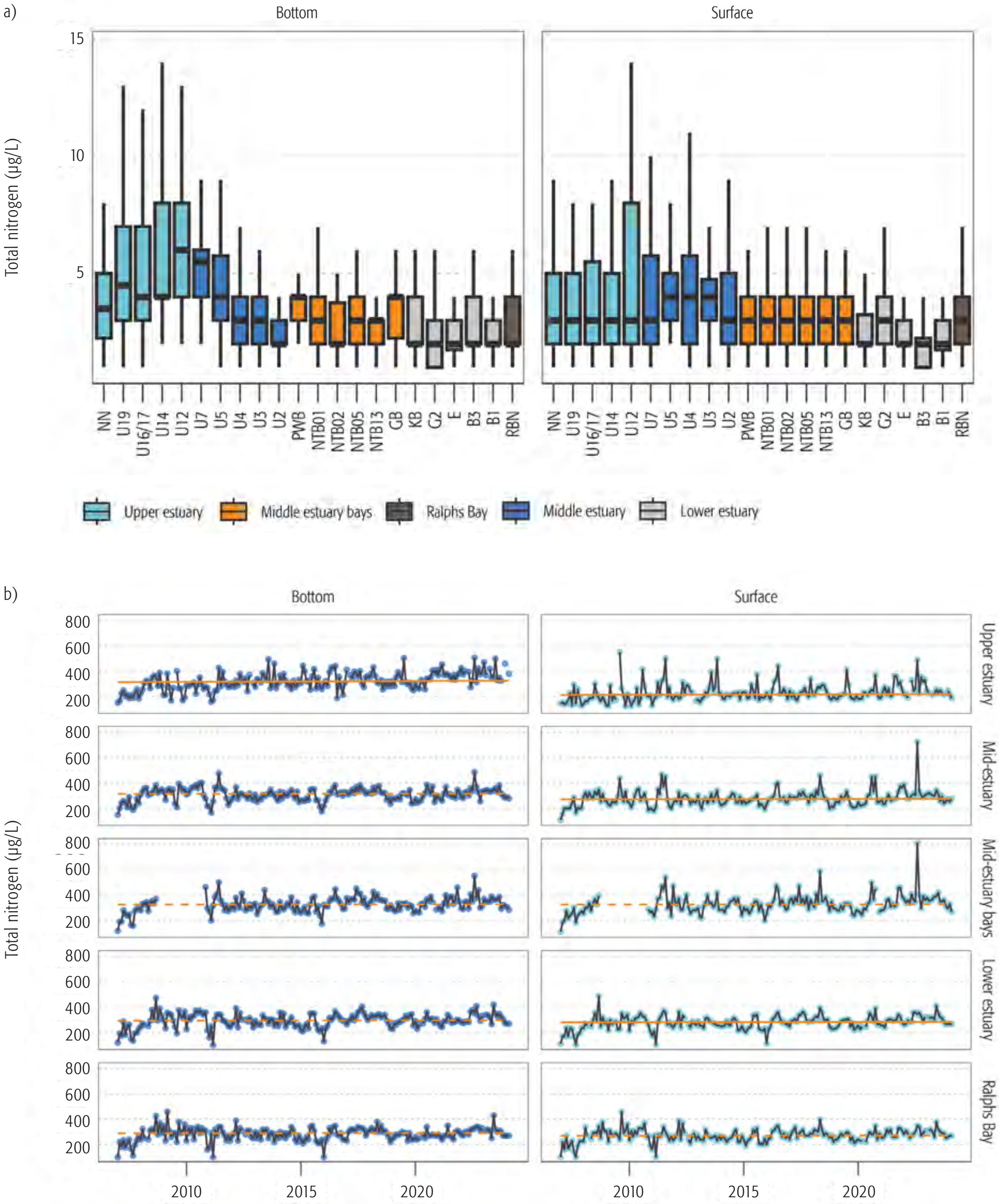


Figure 6.12 Spatial and temporal variation in total ammoniacal nitrogen (µg /L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in salinity for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly total ammoniacal nitrogen for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).

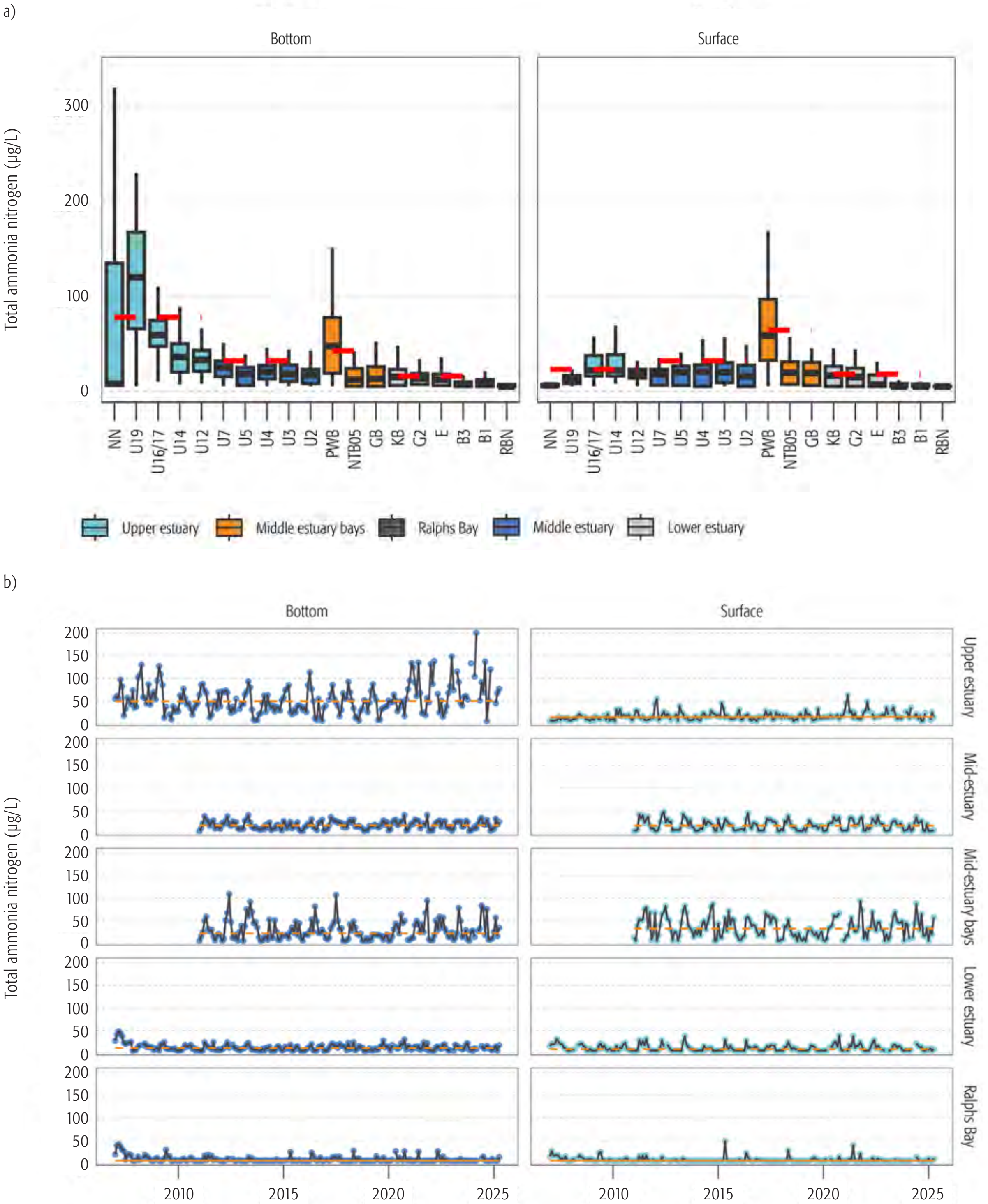
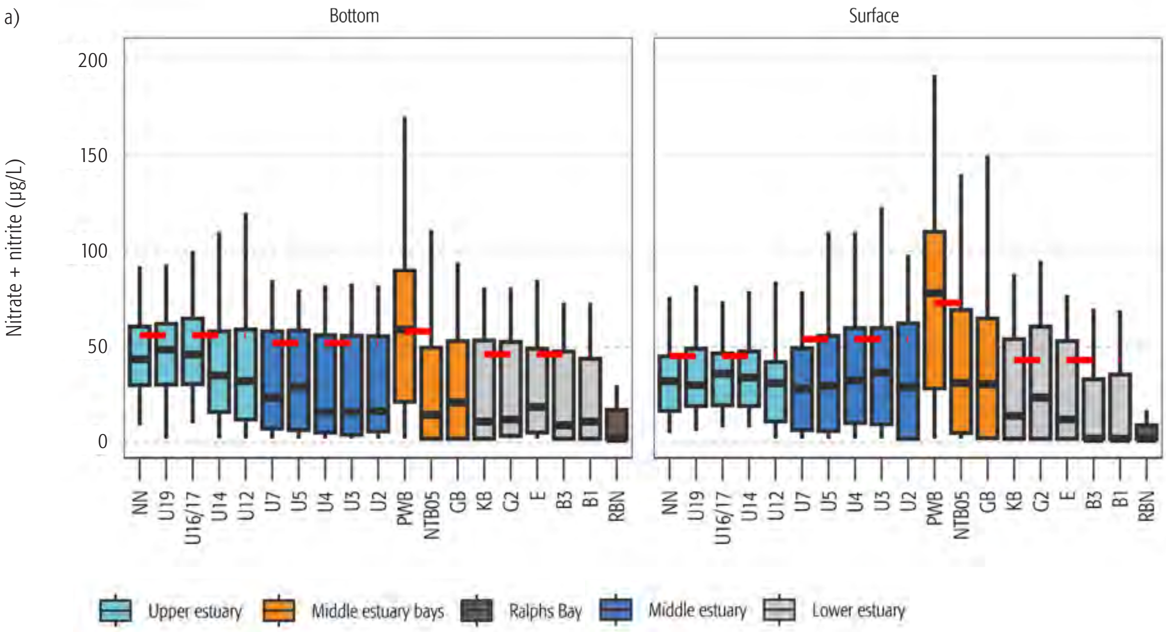


Figure 6.13 Spatial and temporal variation in nitrate + nitrite (µg /L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in nitrate + nitrite for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly nitrate + nitrite for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).

a)



b)

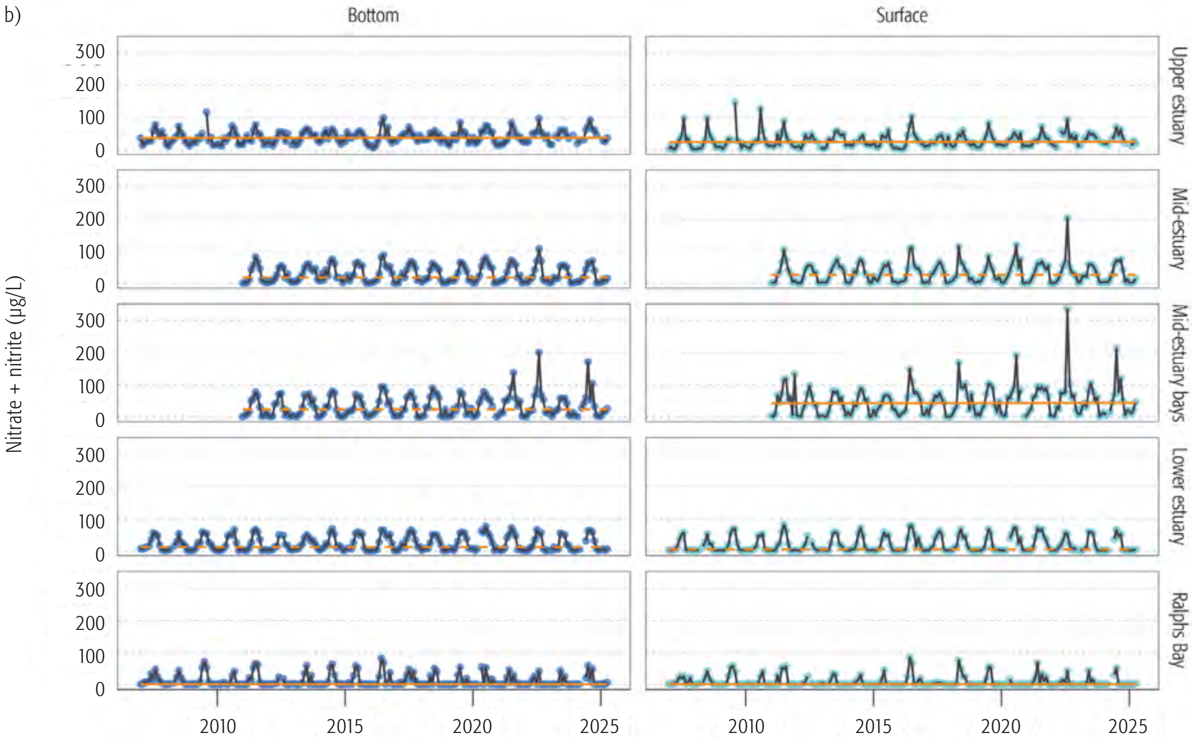


Figure 6.14 Spatial and temporal variation in total phosphorus (µg /L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in total phosphorus for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly total phosphorus for upper estuary, middle estuary, middle estuary bays, lower estuary and ralphs bay sites. Sen's slope trend line fitted where line is significant ($p<0.05$) and dashed is not significant ($p>0.05$).

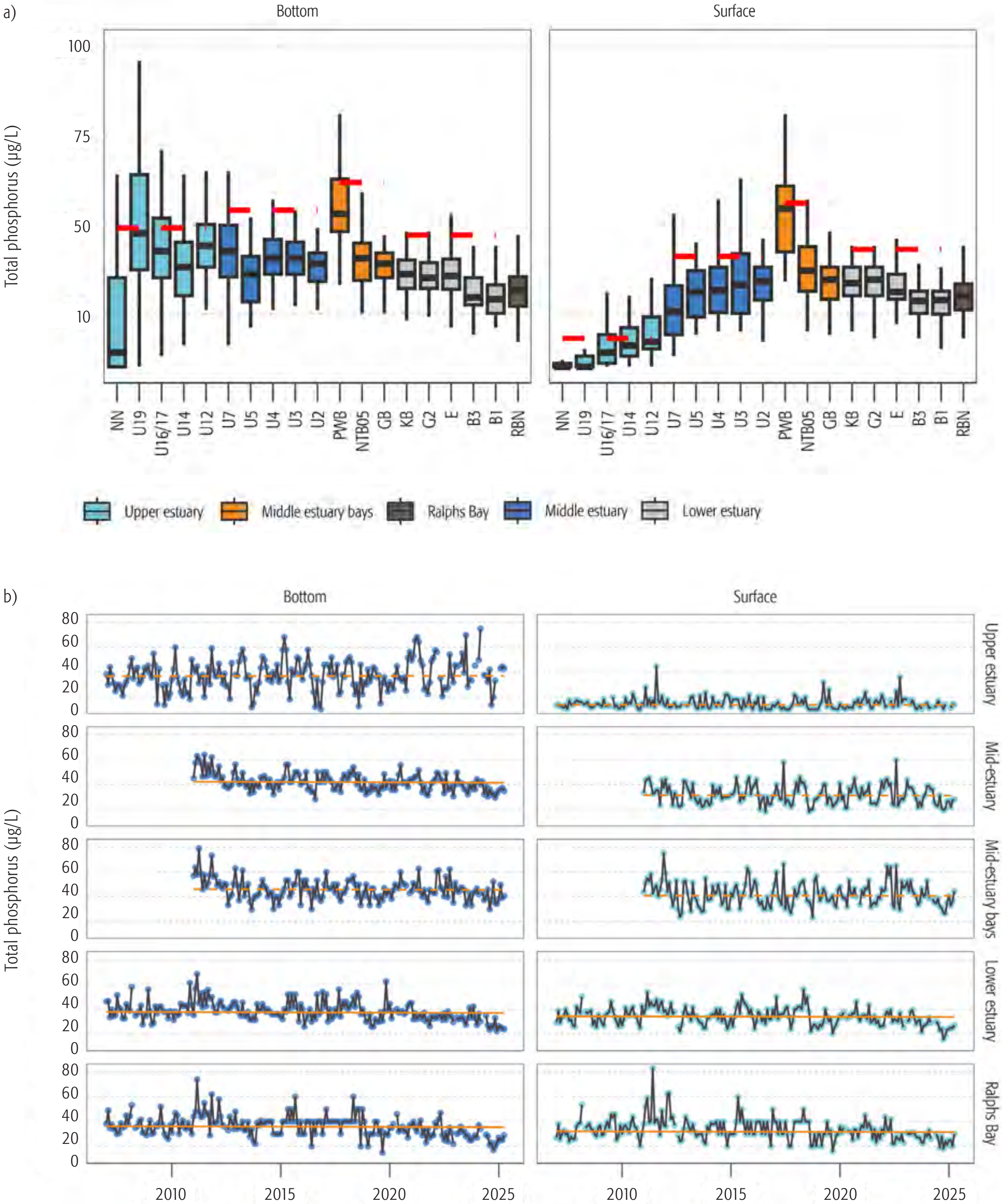


Figure 6.15 Spatial and temporal variation in dissolved reactive phosphorus (µg /L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in dissolved reactive phosphorus for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly dissolved reactive phosphorus for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant ($p<0.05$) and dashed is not significant ($p>0.05$).

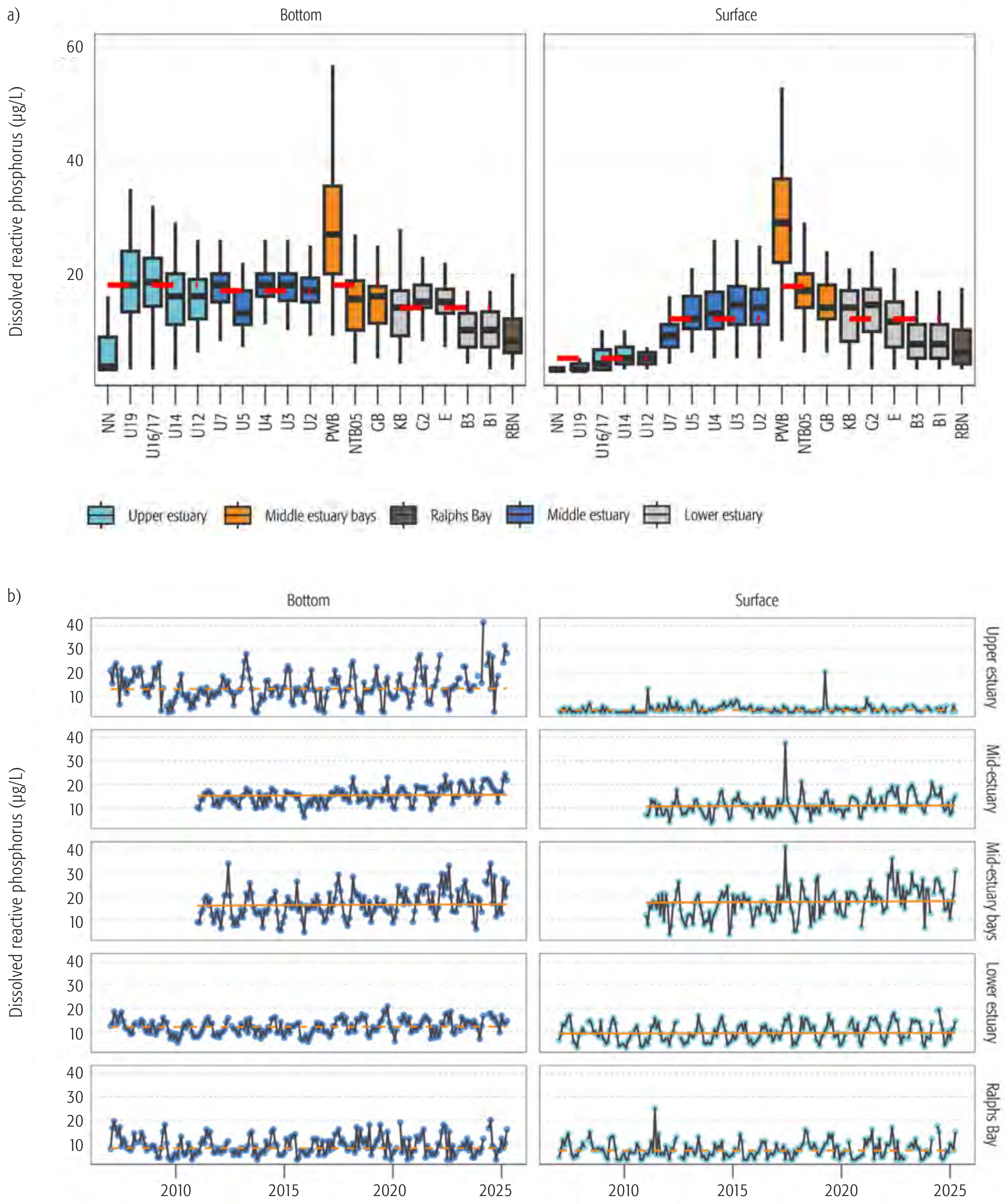


Figure 6.16 Spatial and temporal variation in chlorophyll-a (μg /L) values for integrated samples (10 m) collected throughout the Derwent Estuary. a) variation in chlorophyll-a for each site aggregated for the period 2020–2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly chlorophyll-a for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant ($p<0.05$) and dashed is not significant ($p>0.05$).

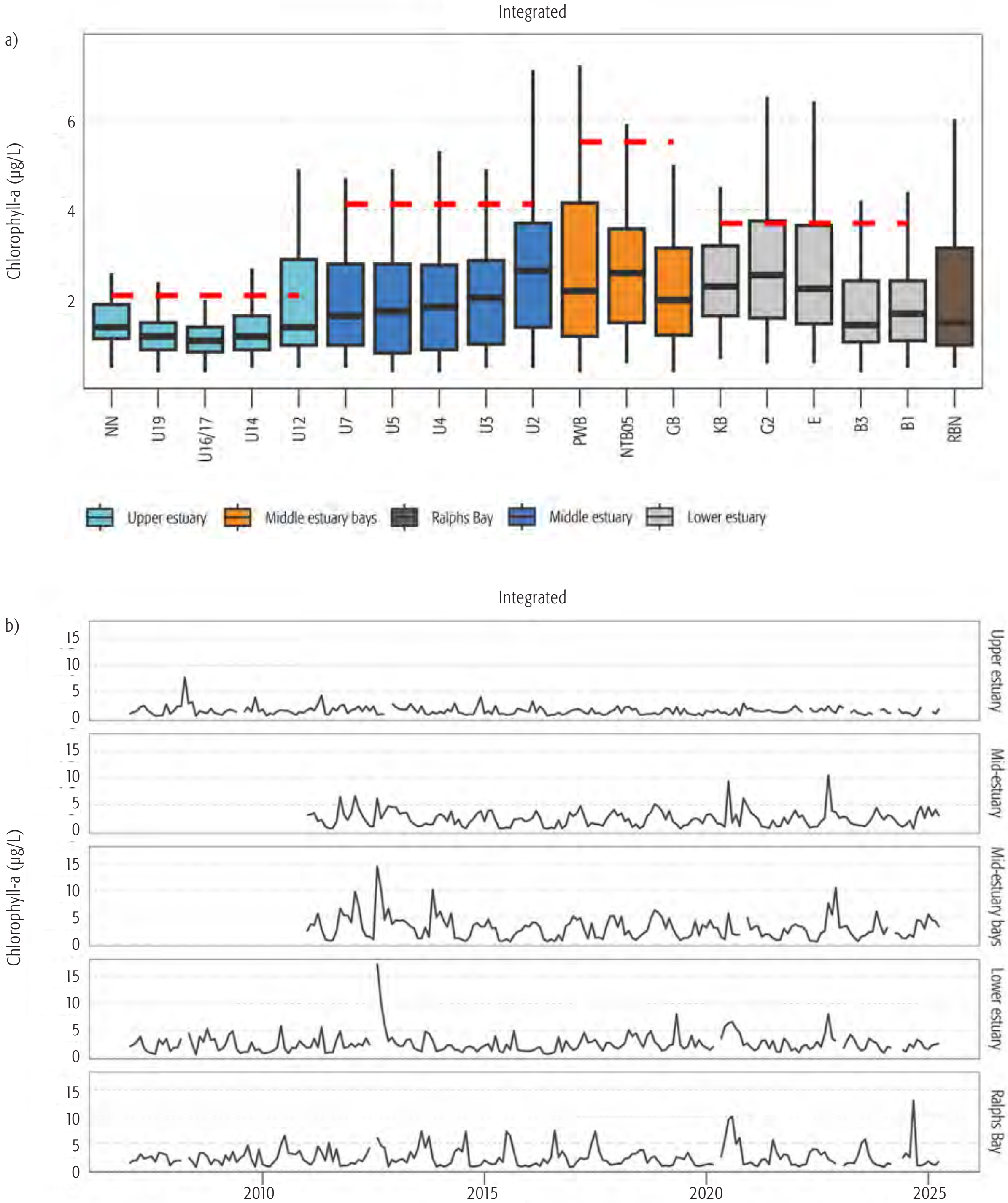
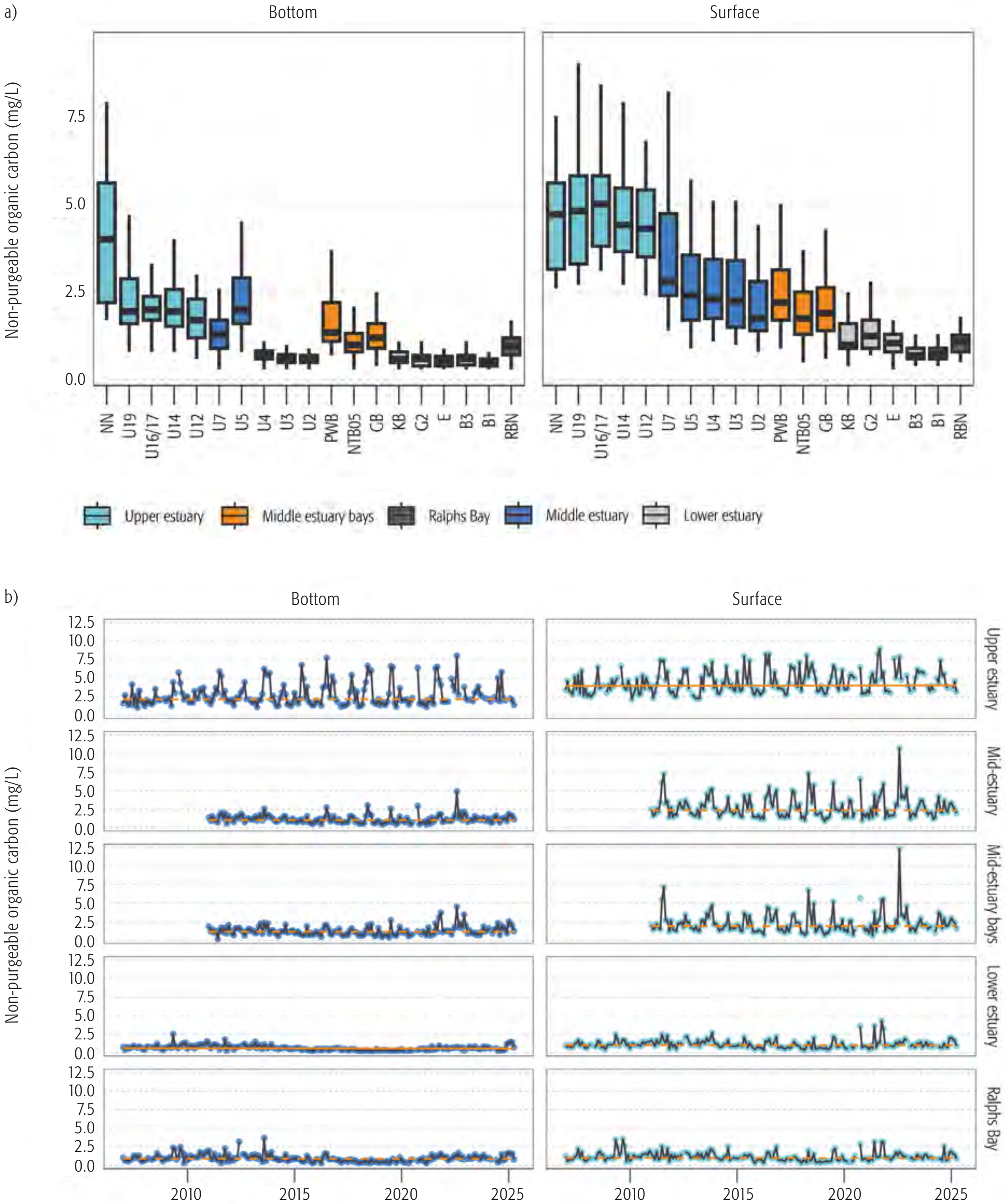


Figure 6.17 Spatial and temporal variation in non-purgeable organic carbon (mg/L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in salinity for each site aggregated for the period 2020 – 2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly salinity for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant ($p<0.05$) and dashed is not significant ($p>0.05$).



6.1.5 Metals

Metals in aquatic systems are derived from both natural and human sources. Industrial processes with poor environmental management practices can be a significant anthropogenic source of metal contamination (Bloom and Ayling, 1977). Metals are persistent in the environment and can be toxic to aquatic organisms, even at low concentrations (ANZG, 2018). Metals are readily adsorbed to the surface of fine particulate matter and tend to accumulate in the bottom sediments of aquatic ecosystems, particularly in areas where water moves slowly, allowing particles to concentrate and settle, known as accretion zones. Aquatic organisms can accumulate metals directly from surrounding water and sediments or via their food source (Wang and Fisher, 1999; Oros, 2025).

Mercury, cadmium, lead and the metalloid arsenic in particular, pose a human health risk, with exposure causing sensory, visual, auditory and kidney functional impairment in adults and neurotoxic effects in infants or developing foetuses (Hutton, 1987; Ullrich *et al*., 2001; Jaishankar *et al*., 2014). An approximate order of decreasing toxicity of common metals is:

Mercury>Cadmium>Copper>Zinc>Nickel>Lead>Chromium>Aluminium>Cobalt;

however, toxicity can vary significantly between different organisms and the chemical species in which the metals occur are particularly important when considering toxicity (Kennish, 1996).

Metal speciation, the process of changes from one metal species or form to another, is influenced by biological, physical and chemical properties of the environment, principally the composition and activity of bacterial communities, temperature, salinity, pH and the concentration of DO and organic matter (Ullrich *et al*., 2001). Inorganic species of mercury have relatively low toxicity to biota but are readily converted to more toxic forms, such as methylmercury. Methylmercury is rapidly absorbed by aquatic organisms and exerts a toxic effect at trace concentrations (Koos and Longo, 1976; Ullrich *et al*., 2001).

The main sources of metal contamination to the Derwent Estuary are principally historical, and while modern environmental management practices have markedly improved, the legacy of former practice still affects the Derwent today, and will most likely persist for many decades. The zinc smelter at Lutana began discharging metallurgical liquid effluent containing heavy metals to the Derwent Estuary when it was established in 1917. A huge amount of work has been conducted by current and former owners of the zinc smelter to reduce ongoing sources of contamination and to remediate onsite legacy contamination.

Currently, contaminated groundwater is the most significant source of metal contamination to the estuary, with smaller contributions from the outfall on site and air emissions (**Section 4.3**).

The paper mill at Boyer also discharged heavy metals to the estuary in the past, especially mercury, which was historically used as a slimicide, and in association with the chlor-alkali plant, which closed in 1993. Zinc was also discharged from this site due to the former use of zinc hydrosulphite as a brightening agent which ceased in the early 1980s.

In past monitoring, a suite of metal species, both total and dissolved were sampled. However, the suite of analytes was reduced to total zinc, as most metals were below the LoR. Total zinc was retained as a key indicator, as it serves as a proxy for metals often co-occurring in zinc ores (e.g. cadmium, lead, mercury and to a lesser extent, copper). It is a poor proxy for metals with different sources (e.g. nickel, cobalt, chromium). Previous comparison of total and dissolved zinc showed that dissolved zinc accounts for the majority of total zinc – 85% in surface waters and 77% in bottom waters (DEP, 2015b).

One limitation of the current approach is the use of total zinc concentrations to assess against DGVs which are set for dissolved zinc concentrations (ANZG, 2018)). This is likely to result in overreporting of guideline exceedances during the reporting period.

Results

During the reporting period, median total zinc concentrations ranged from 2 µg/L in bottom waters at the mouth of the estuary (B1) to 52 µg/L in surface waters of middle estuary bays (NTB05/New Town Bay). Total zinc was typically higher in surface waters and bays of the middle estuary. Total zinc was also elevated in bottom waters of the middle to upper estuary, up to Bridgewater. DGVs were exceeded in bottom waters at New Town Bay (NTB05) and at G2 (Figure 6.16).

Over the monitoring period, total zinc concentrations decreased significantly at Ralphs Bay (bottom and surface waters), in the middle estuary (bottom waters) and upper estuary (bottom and surface waters) (Figure 6.16).

Discussion

The spatial distribution of zinc in the estuary reflects both source inputs and estuarine hydrodynamics and is consistent with previously observed patterns (DEP, 2015c). In surface waters, zinc concentrations were typically highest in New Town Bay and throughout the middle estuary, due to close proximity to the Nyrstar zinc smelter. The primary pathway of zinc to the estuary is currently via groundwater (**Section 4.3**).

In bottom waters, two zones of elevated concentrations were identified: the middle estuary bays (PWB and New Town Bay), and the middle to upper estuary, particularly around Bridgewater (U12). Elevated concentrations in the middle estuary bottom waters are likely due to the proximity of these bays to the smelter. In contrast, high concentrations further upstream are likely a result of estuarine circulation patterns, where saline bottom waters transports contaminants upstream(Wild-Allen *et al*., 2013a).

Subtidal soft sediments of the upper and middle estuary are dominated by silt sediments, whereas the lower estuary is more sand dominated (Lucieer, Lawler, and Pender, 2007). Finer sediments are known to accumulate metals better than coarser-grained particles, due to their surface-to-volume ratio and organic matter content, which is further exacerbated in the upper Derwent by industry effluent (Jones *et al*., 2003; Mucha *et al*., 2004; Buyang *et al*., 2019). The physical structure of these benthic sediments has likely contributed to the accumulation of metals in these silty sediments, compared to coarser sand sediments of the lower estuary.

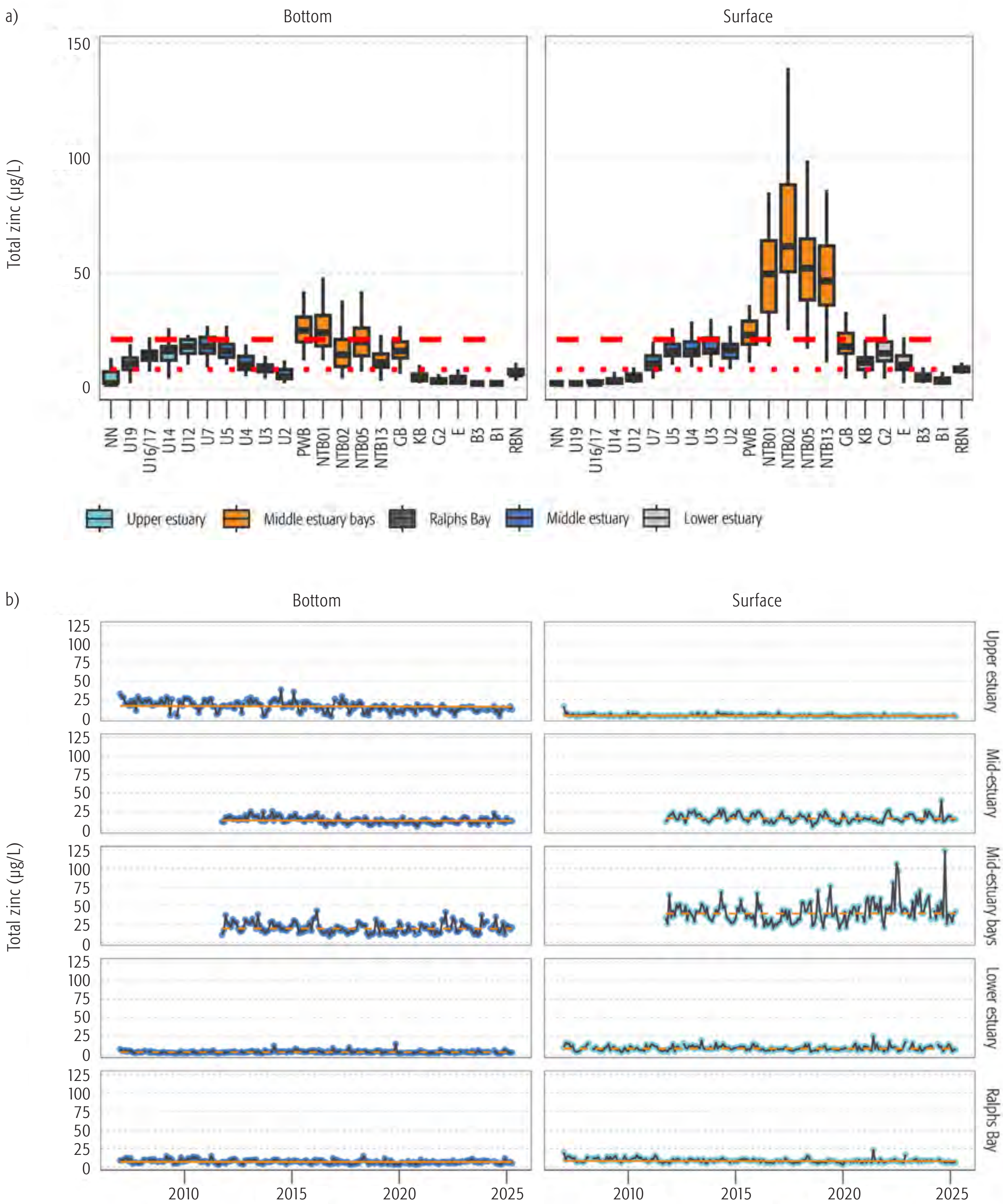
In addition, shallow banks dominated by macrophytes may slow water flow and serve as accretion zones for contaminants and fine sediments. Further, extensive surface area of macrophyte roots and fine sediments in this zone, coupled with high concentrations of organic matter, are likely to be an effective sponge for metals and other contaminants. Other potential sources such as remobilisation of zinc from bank sediments by acid sulphate soils or legacy contaminated sediments have also been considered. However, zinc concentrations in surface sediments along this stretch of the of the estuary are low, suggesting remobilisation from surface sediment may not be a major contributor (DEP, 2020a).

Declining zinc concentrations at 11 sites throughout the estuary indicate ongoing improvement in water quality with regard to zinc. These findings are consistent with sediment core studies that have documented decline in zinc accumulation since peak production at the smelter in the 1970s (Macleod and Coughanowr, 2019; DEP, 2020a; Stevens *et al*., 2020; Hughes *et al*., 2022). No decline at the estuary entry points (surface water at New Norfolk and bottom water at B1) indicates that declines are likely due to estuarine processes.

Likely contributors to observed improvement include proactive site management at Nyrstar Hobart such as site remediation, interception and treatment of onsite stormwater and groundwater (**Section 4.3**). In addition, continued burial of contaminated sediments beneath cleaner material may reduce the potential for remobilisation of legacy metals into the water column and aquatic food web. River discharge could be related to declining zinc at upper estuary sites, however, previous reports found no significant relationship between the two (DEP, 2020a). Given the continued decline in zinc at these sites, we recommend that this analysis be revisited.

Though trend analysis over the monitoring period (2007–2024) suggests regional declines (middle and upper estuary), all sites have shown an increase in zinc in both surface and bottom waters in recent years. This is evident at middle estuary bay sites, particularly New Town Bay where all sites (NTB02, NTB05, NTB13) have experienced increasing zinc contractions since 2017. Concentrations were particularly elevated in 2021 and 2022, possibly linked to above average rainfall and increased groundwater flows that year. Widespread zinc increases in recent years is contrary to observed improvements in groundwater discharge and little change in effluent discharge over the same period (**Section 4.3**). These increases require further attention, starting with New Town Bay.

Figure 6.18 Spatial and temporal variation in total zinc (µg/L) values for bottom and surface samples collected throughout the Derwent Estuary. a) variation in total zinc for each site aggregated for the period 2020– 2024. Boxplots show the median, 25th and 75th percentiles. Draft DGVs for each zone plotted as dashed black line (EPA Tasmania, 2021b). b) mean monthly total zinc for upper estuary, middle estuary, middle estuary bays, lower estuary and Ralphs Bay sites. Sen's slope trend line fitted where line is significant (p<0.05) and dashed is not significant (p>0.05).



6.1.6 Management and recommendations

Water Temperature and MHWs

- Despite climate change projections, water temperature in the estuary has remained relatively stable.
- Ongoing collaboration with IMAS is recommended to better understand the dynamics of MHWs in the Derwent Estuary, particularly the interaction between offshore and catchment-driven heat events.
- Identification of thermal risk and refugia zones may inform future management actions, such as the relocation of vulnerable species to more suitable habitats.

pH Trends

- pH levels have decreased at the estuary's entry points, aligning with climate change projections.
- However, the rate of pH decline in the upper estuary exceeds expected trends, suggesting additional contributing factors. A robust methodology (Butler, in prep.) should be applied to investigate potential drivers such as acid mine drainage, acid sulfate soils, and organic enrichment.
- In contrast, pH levels in the middle estuary have increased, warranting further investigation into the underlying causes.

Water Clarity

- Transparency has improved in the lower to middle estuary, supported by multiple water quality indicators and corroborated by rocky reef habitat surveys.

Dissolved Oxygen, Organic Matter, and Nutrients

- Elevated concentrations of organic matter and nutrients, alongside reduced DO have been observed in PWB and upper estuary sites, highlighting the need for further investigation into contributing sources and ecological impacts.
- There is a need to better understand the relationship between DO, flow regimes, and organic matter sources in the upper estuary to inform management strategies.

Flow Monitoring and Nutrient Dynamics

- There is a clear need for flow monitoring at the head of the estuary to accurately quantify catchment nutrient loads. Improved understanding of these loads will help clarify their influence on declining water quality in the upper estuary.

Effluent Management

- Continued support for partner organisations implementing real-time effluent analysis is encouraged to enhance discharge quality into the estuary.
- Poor water quality in PWB highlights the continued need to prioritise and resource STPs that discharge to poorly flushed bays (e.g. PWB and Cameron Bay). To reduce environmental risk and improve water quality, TasWater are considering a range of options including STP upgrades, outfall relocation and STP decommissioning.

Metal Concentrations

- While metal concentrations have declined at many sites, recent data suggest a potential increase across the estuary.
- Targeted analysis is recommended, particularly in New Town Bay, to assess changes in metal-related water quality.

Table 6.2 Physicochemical indicators and draft DGVs for aquatic ecosystems of the Derwent Estuary (EPA, 2021)

		Dissolved oxygen		Salinity	pH		Water temperature		Turbidity	Chlorophyll a	Total ammoniacal nitrogen	Nitrate + nitrite	Total nitrogen	Total phos-phorus	Dissolved reactive phosphorus
	Units	**mg/L**					**°C**		**NTU**	**µg/L**	**µg/L**	**µg/L**	**µg/L**	**µg/L**	**µg/L**
	limit	**upper**	**lower**		**upper**	**lower**	**upper**	**lower**							
Upper	Surface	10.2	7.1	29.4	8.0	7.4	18.1	9.5	10.6	ND	13.9	43.7	328	31	14.2
	Bottom	6.9	5.4	36.2	8.0	7.6	16.3	12.2	10.1	ND	19.0	67.8	360	45	19.1
Middle	Surface	9.1	7.5	33.1	8.1	7.7	17.3	10.6	10.5	ND	10.4	49.3	330	30	13.7
	Bottom	7.1	5.8	36.5	8.1	7.7	16.0	12.2	9.4	ND	15.8	60.9	268	46	18.3
Middle bays	Surface	9.3	7.7	30.7	8.1	7.9	17.7	10.8		4.1	33.2	49.4	310	45	17
	Bottom	6.9	5.8	34.1	8.0	7.9	16.2	11.8	ND	ND	18	10	305	45	15
Lower	Surface	8.7	7.5	35.4	8.1	7.8	17.4	11.7	9.4	ND	7.2	56.1	328	34	13.0
	Bottom	7.6	6.2	36.6	8.1	7.7	15.7	12.2	12.4	4.7	17.5	57.2	246	43	18.0
Ralphs Bay	Surface	9.3	7.7	32.7	8.1	8.1	17.3	10.4	ND	ND	15.0	20.8	310	41	12
	Bottom	8.4	6.2	34.4	8.1	8.1	16.4	11.4	ND	ND	28.8	53.8	342	54	16

Table 6.3 Results of the AWQ Correlated Seasonal Mann-Kendall (CSMK) trend analysis showing all significant trends, magnitude, and strength for each parameter across zones and depths. Metrics include Z-statistic, p-value, Mann-Kendall score (S), variance of S, Sen's slope, and Kendall's tau.

Zone	Depth	Parameter	P-value	Z-statistic	S	Var(S)	Sen's Slope	Tau
Lower estuary	Bottom	Dissolved reactive phosphorus	0.00	2.85	421.00	21803.67	0.0157	0.193
Lower estuary	Bottom	Non-purgeable organic carbon	0.02	-2.38	-537.00	50994.33	-0.0022	-0.317
Lower estuary	Bottom	pH	0.03	-2.16	-259.00	14340.33	-0.0003	-0.149
Lower estuary	Bottom	Total phosphorus	0.00	-2.94	-529.00	32344.33	-0.0488	-0.297
Lower estuary	Bottom	True colour	0.00	3.79	881.00	54001.00	0.0228	0.601
Lower estuary	Bottom	Turbidity	0.01	-2.76	-479.00	30053.00	-0.0094	-0.259
Lower estuary	Surface	Dissolved reactive phosphorus	0.01	2.46	410.00	27872.00	0.0134	0.124
Lower estuary	Surface	pH	0.04	-2.08	-259.00	15502.33	-0.0002	-0.109
Lower estuary	Surface	Secchi depth	0.02	2.42	335.00	19203.67	0.0056	0.158
Lower estuary	Surface	Total phosphorus	0.01	-2.52	-440.00	30533.33	-0.0395	-0.232
Lower estuary	Surface	True colour	0.00	3.44	612.00	31635.33	0.0267	0.347
Middle estuary	Bottom	Dissolved reactive phosphorus	0.00	3.11	455.00	21345.67	0.0295	0.293
Middle estuary	Bottom	Total phosphorus	0.01	-2.46	-317.00	16559.00	-0.0555	-0.267
Middle estuary	Bottom	Total suspended solids	0.01	-2.46	-356.00	21028.00	-0.0234	-0.313
Middle estuary	Bottom	Total zinc	0.02	-2.35	-307.00	17045.67	-0.0362	-0.292
Middle estuary	Bottom	True colour	0.01	2.82	369.00	17155.00	0.0294	0.263
Middle estuary	Surface	Dissolved oxygen	0.02	-2.35	-159.00	4574.33	-0.0019	-0.051
Middle estuary	Surface	Dissolved reactive phosphorus	0.00	3.13	387.00	15331.00	0.0278	0.240
Middle estuary	Surface	pH	0.04	2.06	201.00	9551.67	0.0007	0.172
Middle estuary	Surface	Total copper	0.01	2.67	308.00	13327.33	0.0000	0.342
Middle estuary	Surface	Total suspended solids	0.02	-2.36	-301.00	16243.67	-0.0154	-0.248
Middle estuary bays	Bottom	Dissolved reactive phosphorus	0.00	3.18	377.00	14081.00	0.0387	0.220
Middle estuary bays	Bottom	pH	0.01	2.46	232.00	8868.67	0.0007	0.214
Middle estuary bays	Bottom	Total suspended solids	0.01	-2.45	-384.00	24638.67	-0.0205	-0.317
Middle estuary bays	Bottom	True colour	0.01	2.84	340.00	14343.33	0.0366	0.296

Zone	Depth	Parameter	P-value	Z-statistic	S	Var(S)	Sen's Slope	Tau
Middle estuary bays	Surface	Dissolved oxygen	0.02	-2.36	-160.00	4610.67	-0.0018	-0.056
Middle estuary bays	Surface	Dissolved reactive phosphorus	0.00	3.31	394.00	14197.33	0.0509	0.281
Middle estuary bays	Surface	Nitrate + nitrite	0.03	2.21	194.00	7734.00	0.0696	0.083
Middle estuary bays	Surface	pH	0.01	2.62	224.00	7328.00	0.0007	0.201
Middle estuary bays	Surface	Total suspended solids	0.03	-2.20	-298.00	18381.33	-0.0148	-0.238
Middle estuary bays	Surface	True colour	0.04	2.04	174.00	7256.67	0.0303	0.110
Ralphs Bay	Bottom	Dissolved oxygen	0.01	2.82	349.00	15270.33	0.0027	0.133
Ralphs Bay	Bottom	Nitrate + nitrite	0.01	-2.47	-164.00	4402.00	0.0000	-0.086
Ralphs Bay	Bottom	Total ammonia nitrogen	0.04	-2.02	-219.00	11720.33	0.0000	-0.160
Ralphs Bay	Bottom	Total phosphorus	0.02	-2.41	-465.00	37169.67	-0.0476	-0.271
Ralphs Bay	Bottom	Total zinc	0.03	-2.22	-339.00	23299.00	-0.0102	-0.204
Ralphs Bay	Bottom	True colour	0.00	3.43	683.00	39719.00	0.0211	0.480
Ralphs Bay	Surface	Nitrate + nitrite	0.04	-2.09	-173.00	6842.33	0.0000	-0.122
Ralphs Bay	Surface	Total ammonia nitrogen	0.01	-2.46	-319.00	16837.00	0.0000	-0.226
Ralphs Bay	Surface	Total phosphorus	0.01	-2.82	-458.00	26479.33	-0.0500	-0.274
Ralphs Bay	Surface	Total zinc	0.01	-2.71	-440.00	26342.67	-0.0122	-0.240
Ralphs Bay	Surface	True colour	0.00	3.33	577.00	29979.00	0.0238	0.365
Upper estuary	Bottom	Nitrate + nitrite	0.02	2.44	231.00	8973.67	0.0455	0.095
Upper estuary	Bottom	Total nitrogen	0.00	3.33	481.00	20863.00	0.5217	0.305
Upper estuary	Bottom	Total zinc	0.00	-3.21	-532.00	27486.00	-0.0552	-0.337
Upper estuary	Bottom	True colour	0.01	2.83	288.00	10378.00	0.0392	0.141
Upper estuary	Surface	Nitrate + nitrite	0.00	3.15	333.00	11159.67	0.0583	0.130
Upper estuary	Surface	Non-purgeable organic carbon	0.01	2.46	258.00	11012.00	0.0051	0.135
Upper estuary	Surface	pH	0.01	-2.66	-491.00	34129.00	-0.0017	-0.319
Upper estuary	Surface	Total ammonia nitrogen	0.01	2.81	330.00	13758.00	0.0293	0.166
Upper estuary	Surface	Total nitrogen	0.01	2.77	322.00	13536.67	0.2707	0.215
Upper estuary	Surface	Total zinc	0.03	-2.23	-274.00	15130.67	-0.0043	-0.164

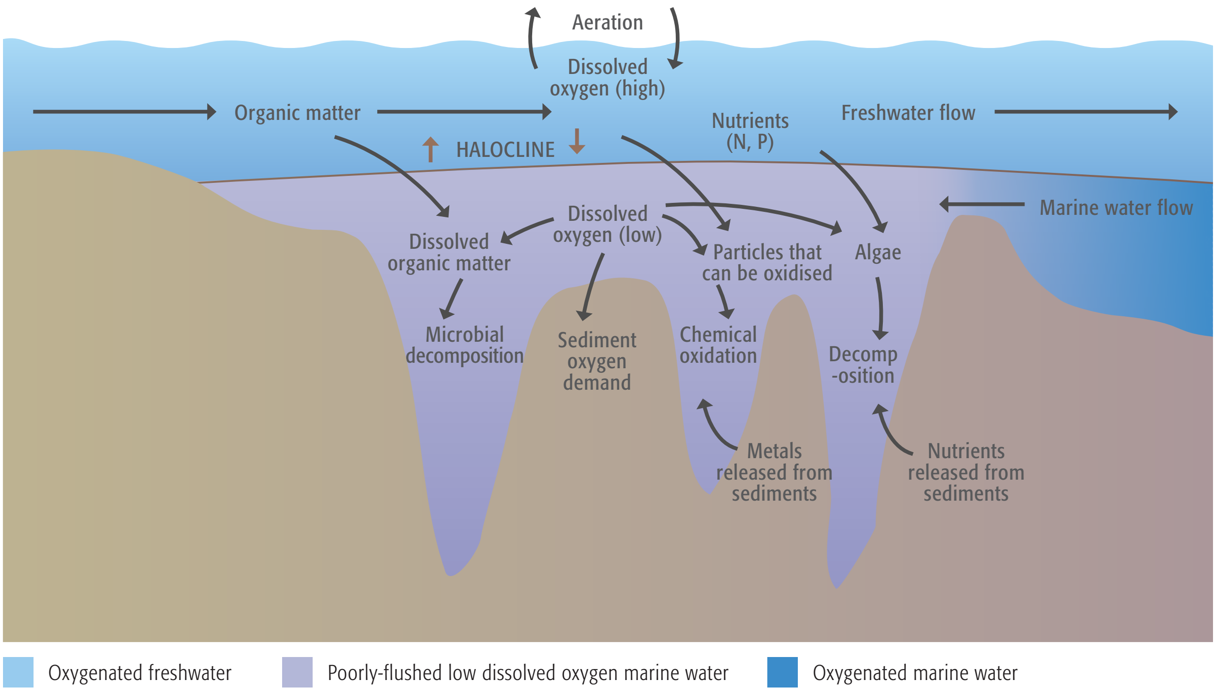
6.2 Upper estuary hypoxia

The Derwent Estuary behaves as a salt wedge estuary, with denser marine bottom waters propagating upstream, overlayed by surface freshwater flowing downstream (Herzfeld *et al.*, 2005). The upper reaches of the estuary, north of the Bridgewater Bridge are mostly shallow (2–4 m depth), however there are several deeper holes (6–15 m depth) that occur in this region. In summer and autumn, when flows in the River Derwent are low, the salt wedge travels further upstream, and the upper reaches of the estuary can become strongly stratified (Herzfeld *et al.*, 2005). Seasonal salt wedge stratification is a common natural occurrence in microtidal estuaries, where limited flow and tide-induced turbulence mean there is little vertical mixing occurring in the water column (Geyer and Farmer, 1989; Kurup *et al.*, 1998). In the upper Derwent Estuary, these stratified marine bottom waters can get trapped in the deep holes for extended periods, resulting in decreases in dissolved oxygen (DO) from biological and chemical processes, leading to low DO and hypoxic conditions (severely low DO concentration) developing.

A 'run-of-river' hydropower scheme is located along the River Derwent. Water is stored at the top of the scheme and passes through at least six power stations before flowing out through the Derwent Estuary. Meadowbank dam is the last impoundment in this scheme, located 48 km upstream of New Norfolk, with several large tributaries flowing into Lake Meadowbank. River Derwent flow is the principal regulator of the location of the salt wedge in the upper Derwent Estuary (Herzfeld *et al.*, 2005). Dissolved organic matter (DOM) input from terrestrially derived sources (i.e. plant material, animal manure, soil etc.) is brought down the catchment from runoff, with additional organic carbon loading from Boyer paper mill effluent. Increased nutrient inputs from catchment agricultural runoff, forestry and land-based fish farms, as well as sewerage and water treatment plants can lead to excessive algal growth in the upper estuary (Proemse *et al.*, 2022). Microbial organisms' break down this DOM and decomposing algae in the water column, consuming oxygen, termed biological oxygen demand (BOD). Higher water temperatures decrease oxygen solubility in water, as well as increase BOD and can result in more rapid development of hypoxic conditions (Buzzelli *et al.*, 2002; Du *et al.*, 2018). Chemical oxygen demand (COD) is the consumption of DO through the oxidation of particles and compounds in water. Prolonged hypoxic conditions in bottom waters can influence nutrient and metal cycling across the sediment-water interface; altering sediment and water biogeochemistry which can further enhance oxygen consumption and have detrimental environmental consequences (Jeong *et al.*, 2025; Wong *et al.*, 2010).

If hypoxic conditions persist for prolonged periods, it can have detrimental impacts on aerobic organisms (Gray *et al.*, 2002), enhance nutrient cycling (Banks, Ross, Keough, Macleod, *et al.*, 2012; Zhu *et al.*, 2017), cause metals to

Figure 6.19 Conceptual diagram of processes occurring in the upper Derwent Estuary that consume dissolved oxygen and result in low dissolved oxygen conditions developing below the halocline. The diagram depicts freshwater flow from upstream of New Norfolk, bathymetry of deep holes located in the upper estuary, to approximately the Bridgewater Bridge at the southern boundary, with marine water flowing up from the estuary mouth.



release from sediments to overlying water (Banks, Ross, Keough, Eyre, *et al*., 2012; Liu *et al.*, 2019; Jaiswal and Pandey, 2020), and alter the structure and function of benthic communities (Levin *et al*., 2009). A conceptual model of DO dynamics in the upper Derwent Estuary is presented in Figure 6.19.

Upper Derwent Estuary stratification and DO dynamics have previously been investigated (Rochford, 1951; Ritz and Buttermore, 1984; Thomson and Godfrey, 1985; Davies and Kalish, 1994; Sheldon and Pope, 2019). A preliminary investigation for source tracking of DOM to upper estuary surface and bottom waters has also recently been conducted (see **Section 6.3**). Biological and chemical oxygen consumption, combined with extended residence time of bottom waters result in DO depletion and persistent hypoxia. This section aims to further describe DO dynamics in the upper estuary through observations of flow, season, and tides on DO in bottom waters and water column profiles in the upper estuary, using data collected from 2019–2025.

Methods

Dissolved oxygen

Thresholds for DO concentrations used in this analysis (Gray *et al*., 2002; Rabalais et al., 2002; Vithana *et al*., 2019) (Table 6.4).

Table 6.4 Thresholds for dissolved oxygen (DO) concentration classifications in water referred to in this analysis.

DO condition	DO concentration (mg/L)
Low DO	< 3
Hypoxia	< 2

Seasonal *in situ data* logger

A HOBO data logger was seasonally deployed during summer and autumn in the upper estuary to monitor salt wedge and bottom-water DO dynamics. The logger was installed at Boyer wharf site at approximately 4.5 m depth (Table 6.5). This site was selected for logger deployment as it is located near one of the deeper holes and has solid infrastructure to tether the logger to (Figure 6.20). Deployed HOBO data logger at this site measured DO, salinity and temperature at hourly intervals. The logger was cleaned and calibrated in field during deployment, approximately every 3 months to ensure accuracy of data.

Due to the differing logger deployment durations between each season, the data was cut into a period of overlapping time during summer and autumn to make comparisons between seasons.

Table 6.5 HOBO data logger deployment dates at the Boyer wharf site. Logger was deployed to bottom waters at approximately 4.5 m depth.

Logger deployment season	Deployment date	Retrieval	Comparison period
Logger_2025	14/02/2025	23/04/2025	17/02/2025–23/04/2025
Logger_2022	17/02/2022	25/06/2022	17/02/2022–23/04/2022
Logger_2020/21	10/11/2020	27/04/2021	17/02/2021–23/04/2021

Flow

Flow data used in this analysis is modelled River Derwent flow calculated at TasWater Bryn Estyn water treatment plant. Hourly flow data for River Derwent below Meadowbank Dam and Tyenna River were extracted from the Department of Natural Resources and Environment Tasmania Water Information Web Portal (NRE Tas, 2025). River Derwent flow at Bryn Estyn was calculated as per Proemse et al., (2022), where *A* is the flow below Meadowbank Dam, and *B* is the flow at Tyenna River:

$$\text{Modelled Flow} = A + 4.89^{*} \times B$$

*Extrapolated flow for the River Derwent below Tyenna based on catchment size (including Plenty, Broad and Styx tributaries).

Flow conditions in the Derwent discussed throughout this section are related to the distribution of flow conditions recorded in the River Derwent (Table 6.6).

Table 6.6 Thresholds for flow conditions in the River Derwent used throughout this section, related to the distribution of flow conditions recorded.

Flow condition	Flow range (m^3/s)
Low flow	20–50
Moderate flow	50–90
High flow	90–400
Flood	>400

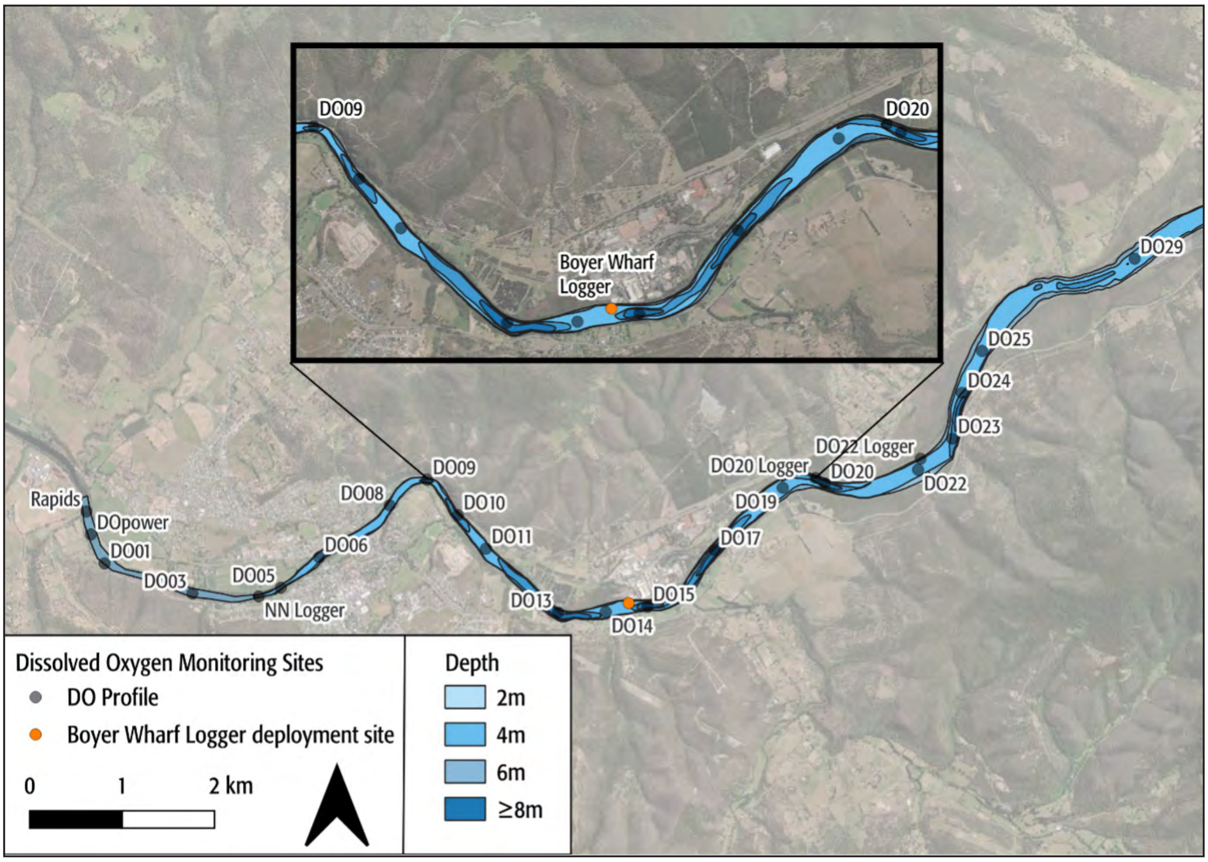
Tide

Water level data for New Norfolk was extracted from the Tasmanian Tide Monitoring Project (Palmer, 2023). Data was collected at six minute intervals from January 2021 to August 2022, which coincides with Logger_2020/21 and Logger_2022 deployment dates.

Profiles

Depth profiling was conducted intermittently during spring (2019), summer (2020) and autumn (2020, 2022, 2023 and 2025) in the upper Derwent Estuary from the approximate upper limit of salt wedge incursion (upstream of New Norfolk), to downstream of the last deep hole in the upper estuary (Figure 6.20). Physio-chemical parameters of the water column were measured using a calibrated YSI EXO3 multi-parameter water quality sonde. Measurements were collected at the surface (approximately 0.3 m), and 1-m intervals to the bottom.

Figure 6.20 Location of dissolved oxygen depth profiles and Boyer wharf logger deployment site in the upper Derwent Estuary, with approximate bathymetry showing location of the deeper holes.



Ocean Data View plots

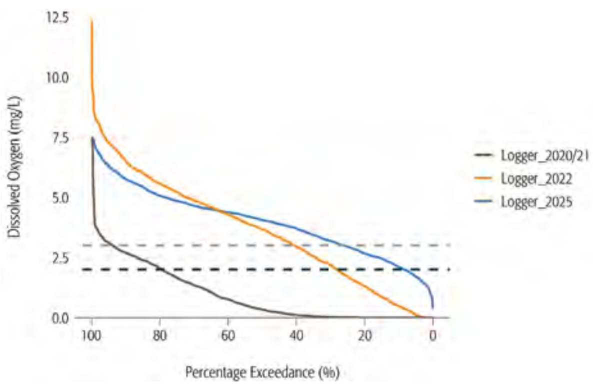
Ocean Data View (ODV) is a software package that visualizes oceanographic, coastal and estuarine physio-chemical data related to geographical coordinates and bathymetry data. Methods for data handling and plot generation are described in Schlitzer (2002).

Results

Deployed logger observations

During the Logger_2020/21 comparison period, hypoxia was recorded 79% of the time. During the comparison period Logger_2022 hypoxia was recorded 28% of the time, and 8% of the time in Logger_2025 (Figure 6.21).

Figure 6.21 Probability exceedance curve of dissolved oxygen (DO) recorded at Boyer site for the comparison period for each seasonal logger deployment. Probability exceedance curve ranks observed DO concentrations over the period, then converts these ranks to percentages, visualising the distribution of observations that exceed low DO values. Low DO threshold (<3 mg/L) is denoted with light grey dashed line and hypoxic threshold (<2 mg/L) with dark grey dashed line.



During Logger_2020/21 deployment season, mean monthly DO was observed to decline from above 3.5 mg/L in late spring and early summer to 0.6 mg/L in March, before increasing in April (Figure 6.22). The rate of DO decline to reach hypoxic conditions is longer in November and April, compared to February and March, when mean monthly DO conditions are very low (<0.8 mg/L) (Table 6.7). Hourly DO concentrations are also more variable in February and March, compared to November and April, where hourly DO concentrations can be observed to decline at a relatively steady rate. The DO decline trend was generally found to be linear, however in February 2021 the rate of decline was observed to be exponential (Figure 6.23).

Figure 6.22 Mean monthly dissolved oxygen recorded at Boyer wharf site during Logger_2020/21 deployment season. Black dashed lines indicate mean monthly flow in the River Derwent during this period.

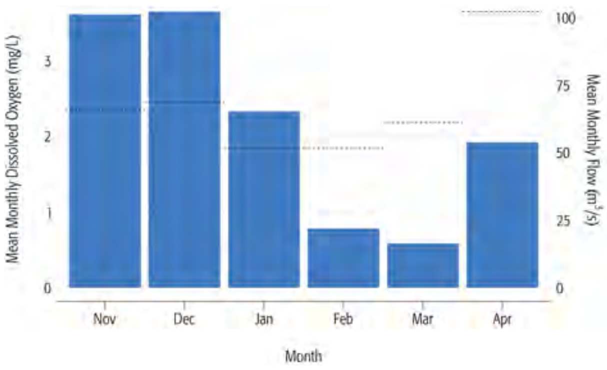
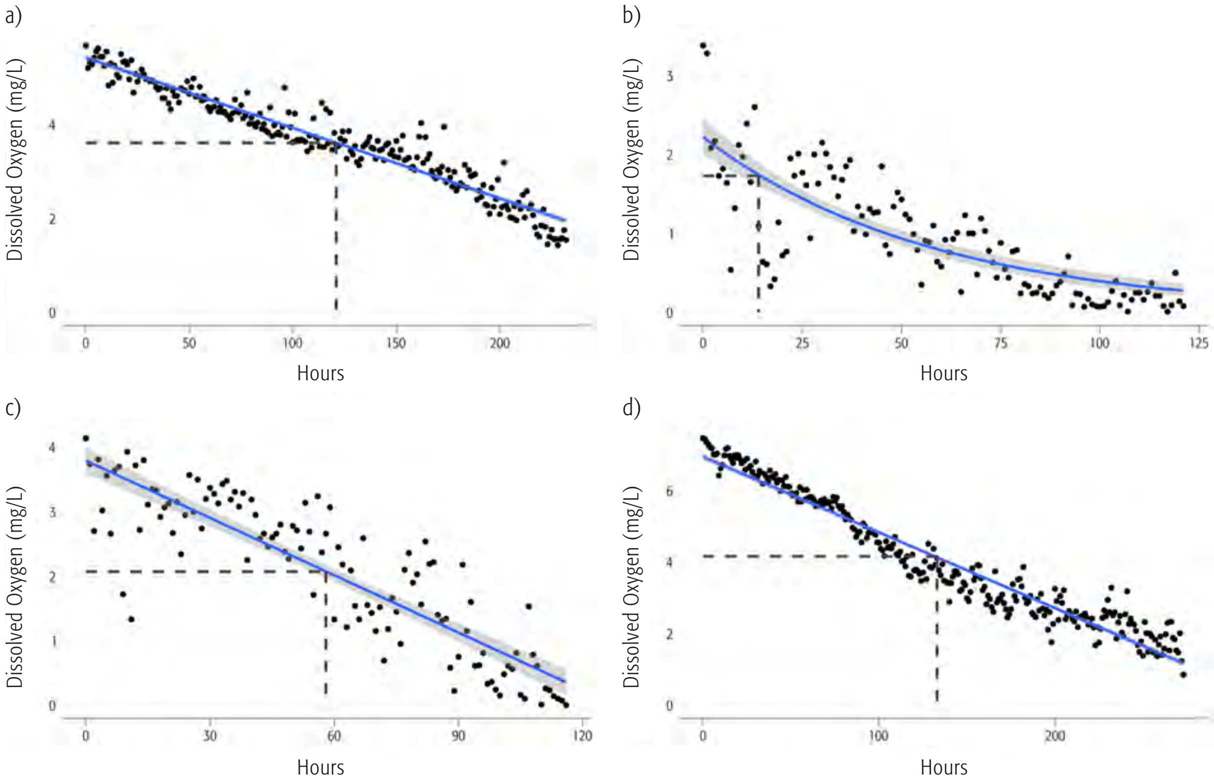


Table 6.7 Time taken for dissolved oxygen (DO) in bottom waters at Boyer wharf site to reach hypoxic concentrations from more elevated levels of DO, during different months during the logger deployment season.

Month	Initial DO concentration (mg/L)	End DO concentration (mg/L)	Time to half concentration (hours)	Time to end concentration (hours)
November	5.68	1.54	121	232
February	3.38	0.08	14	121
March	4.14	0.2	58	116
April	7.46	0.85	133	273

Figure 6.23 Scatterplot of hourly dissolved oxygen (mg/L) of bottom waters at Boyer wharf site from more elevated DO concentrations declining to hypoxic concentrations in (a) November, (b) February, (c) March and (d) April. Scatterplots are fitted with a trend line to visualise the trend in DO decline over the period. Black dashed lines indicate the time taken for DO to reach half the initial DO concentration.



Seasonal flow observations

The River Derwent experiences its lowest monthly flows during late summer and early autumn (January to March). Larger flow events can occur in December, such as those that occurred in December 2019 and 2024. Spring records the largest variability in seasonal rainfall in south-east Tasmania, reflected in River Derwent flows, which can be well above and well below average. The period 2019–2025 has recorded lower mean monthly flows compared to 1990–2024 mean, for most months (Figure 6.24).

Mean seasonal flow for spring 2020 was 130 m^3/s, and mean spring seasonal flows for 2021 and 2024 were 193 m^3/s and 182 m^3/s respectively, with December 2024 flows being above average (Figure 6.25).

Figure 6.24 Mean monthly flow River Derwent at Bryn Estyn for the period January 2019 to May 2025. The dashed line corresponds to the mean monthly flows for the period 1990-2025.

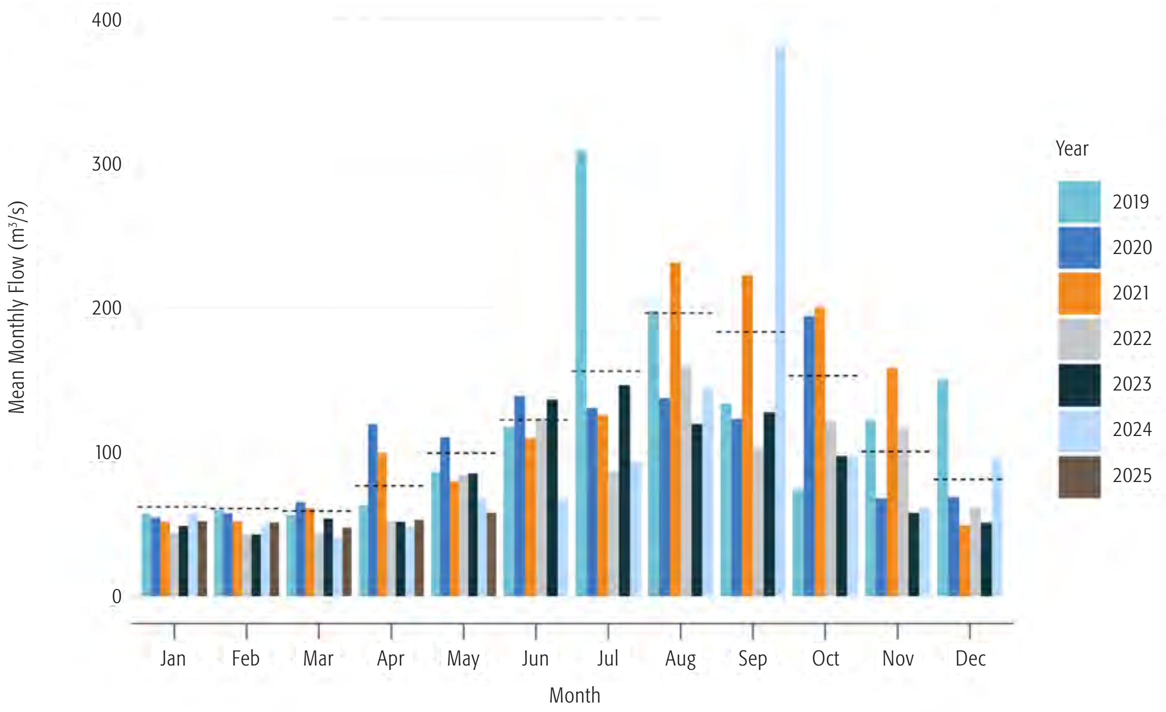
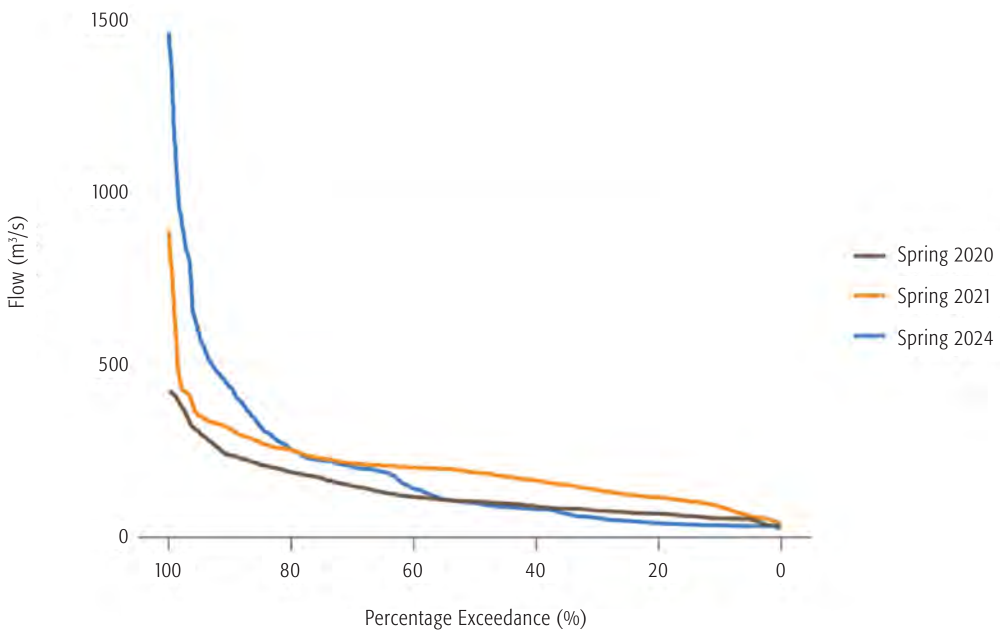


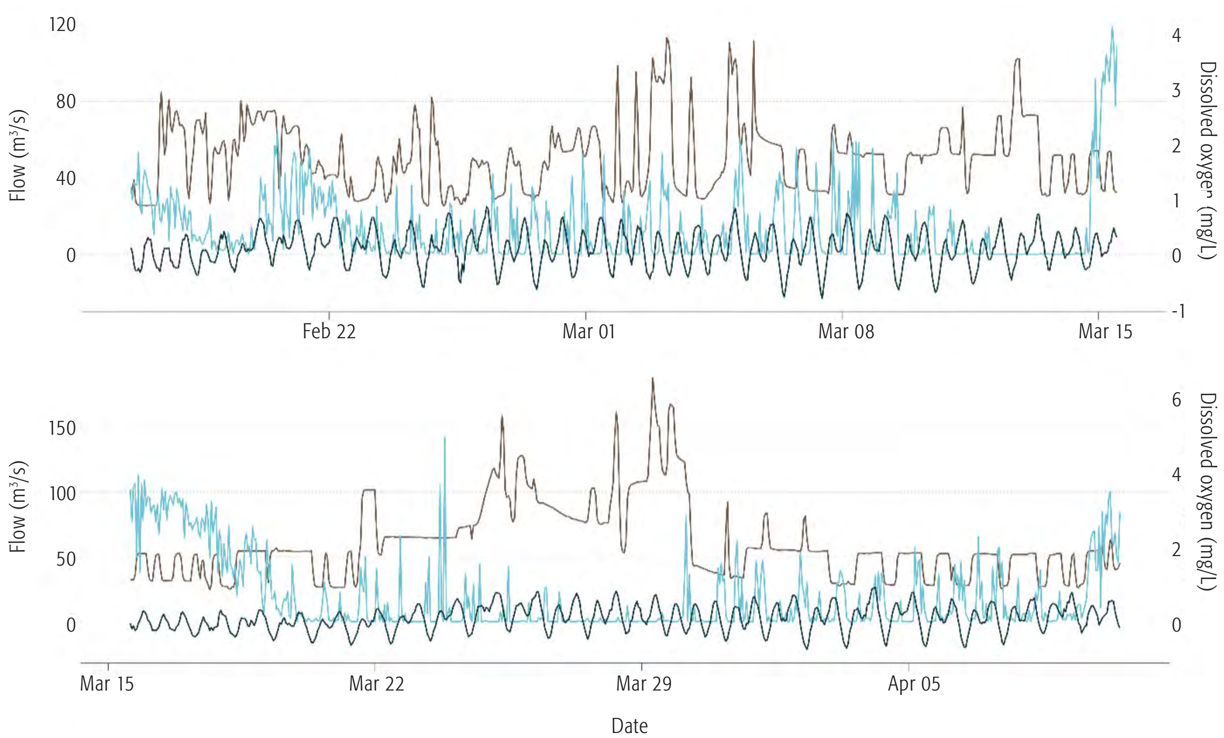
Figure 6.25 Probability exceedance curve of River Derwent flow at Bryn Estyn during spring 2020, 2021 and 2024. Probability exceedance curve ranks observed hourly flow observations during spring 2020, 2021 and 2024, then converts these ranks to percentages, visualising the distribution of observations.



Tidal influence on recharge events

It is known that during low-flow periods, spring high tides can 'recharge' upper estuary bottom water by introducing high-DO marine waters to the region. Inspection of time series plots for the Logger_2020/21 season during low flow periods shows DO increases associated with high tide events were on average 0.74 (+/- 0.23) mg/L, sustained for a period of 2–8 hours, after which DO concentrations generally decrease back to levels which were previously recorded (Figure 6.26).

Figure 6.26 2021 time-series plots of River Derwent flow (m^3/s) (brown line), HOBO logger dissolved oxygen (mg/L) (blue line) and water level (m AHD) at New Norfolk (black line).

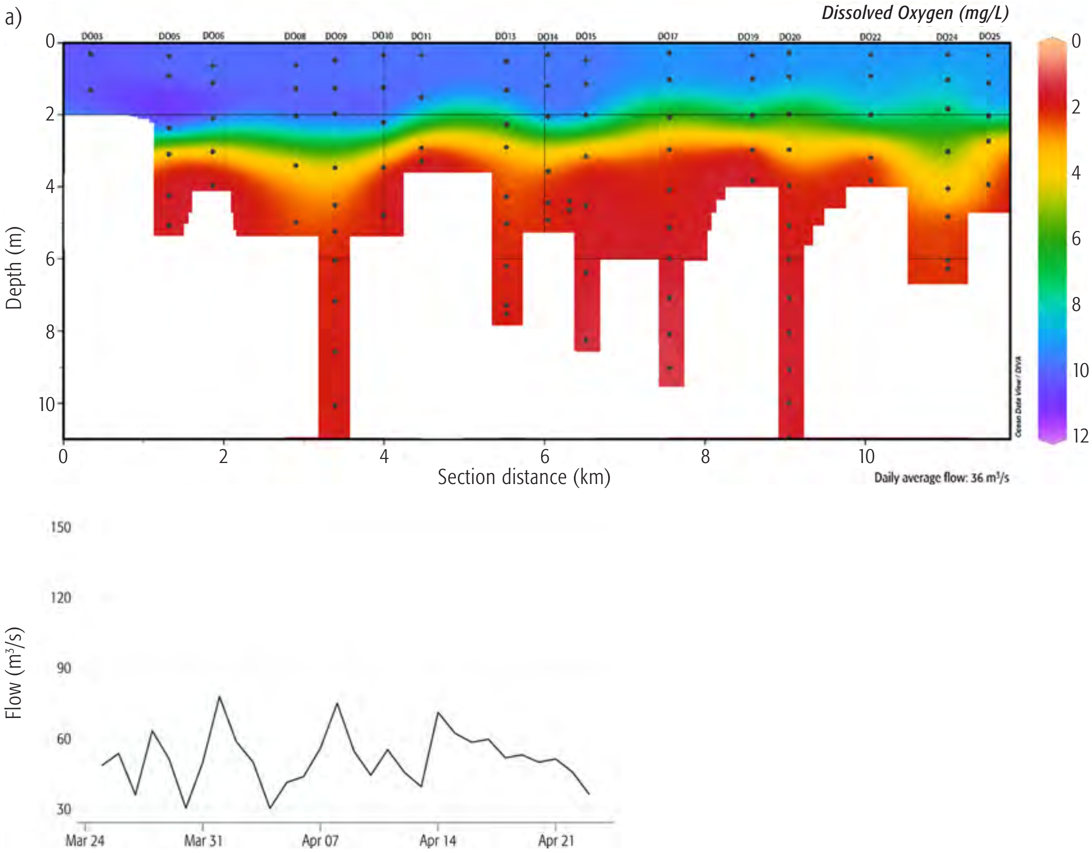


Profile observations

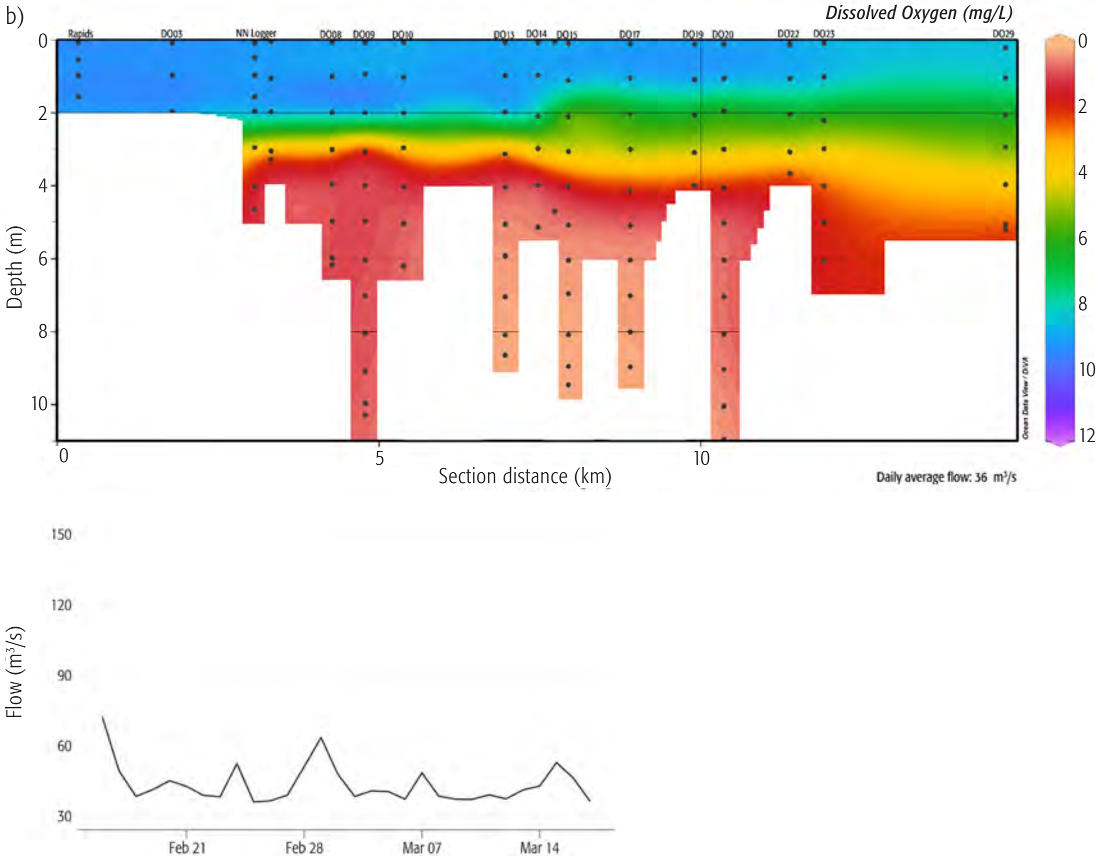
Surface freshwaters (S <5) have high DO, with 95% of observations recording DO conditions >8 mg/L. Subhalocline hypoxia was detected in all upper estuary profile events except for one, which occurred in February 2020 (Figure 6.27). The higher DO concentrations recorded during this profile were likely due to flood conditions in the River Derwent that occurred in December 2019 (daily average flows of 565 m^3/s), that flushed the entire water column of the upper estuary with oxygenated freshwater (>10 mg/L).

In low-flow conditions, the halocline occurs at shallow depths and low DO was recorded in water depths of 3–4 m. In November 2019 and March 2020, moderate freshwater flows can be observed to intersect the shallow sills, trapping marine bottom waters in the deep holes, with hypoxic conditions recorded below the halocline. There is evidence of vertical mixing in the water column, particularly during high flows in the upper estuary. This mixed layer can penetrate to depths of approximately 6 m in the lower reaches of the upper estuary.

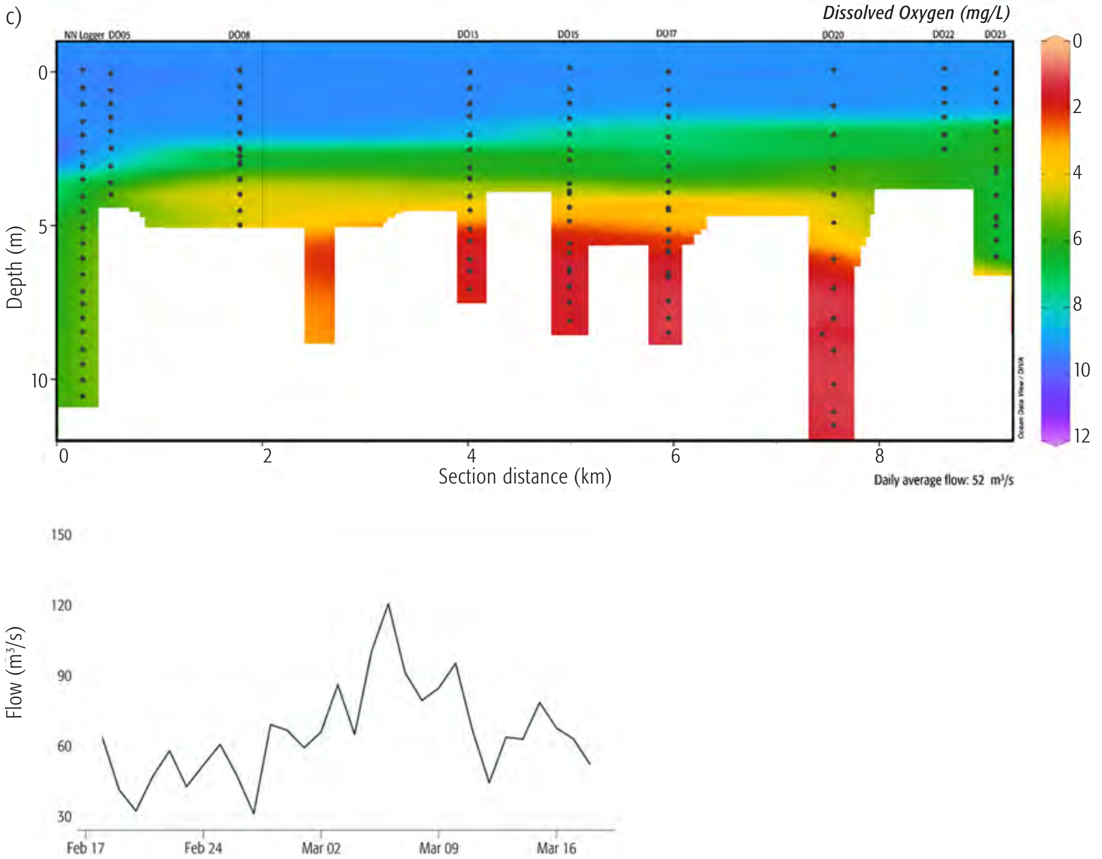
Figure 6.27 Ocean Data View plots visualising transect depth profile dissolved oxygen (DO) (mg/L) data collected in the upper estuary from 2019 to 2025, and approximate bathymetry of the upper estuary. Black dots denote *in situ* measured DO profile locations, DO throughout the remainder of the plot is interpolated between measured points (Schlitzer, 2002). Modelled daily average flow presented bottom right, and flow for the 30 days prior to profile sampling event for the River Derwent is presented below the Ocean Data View plots. Plots are ordered from lowest to highest daily flow for the River Derwent during a profiling event.

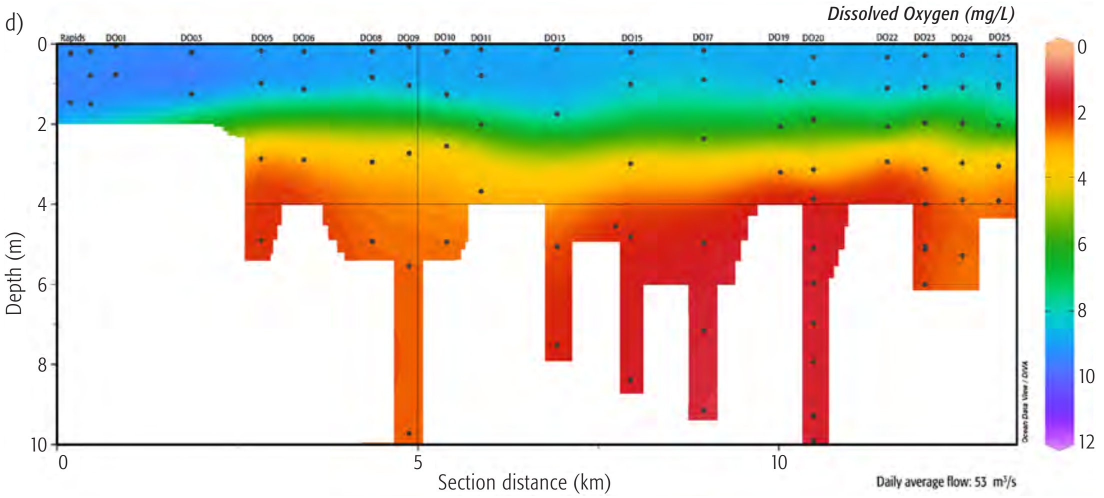


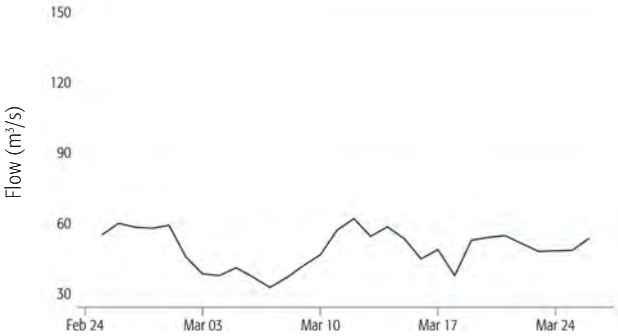
b)

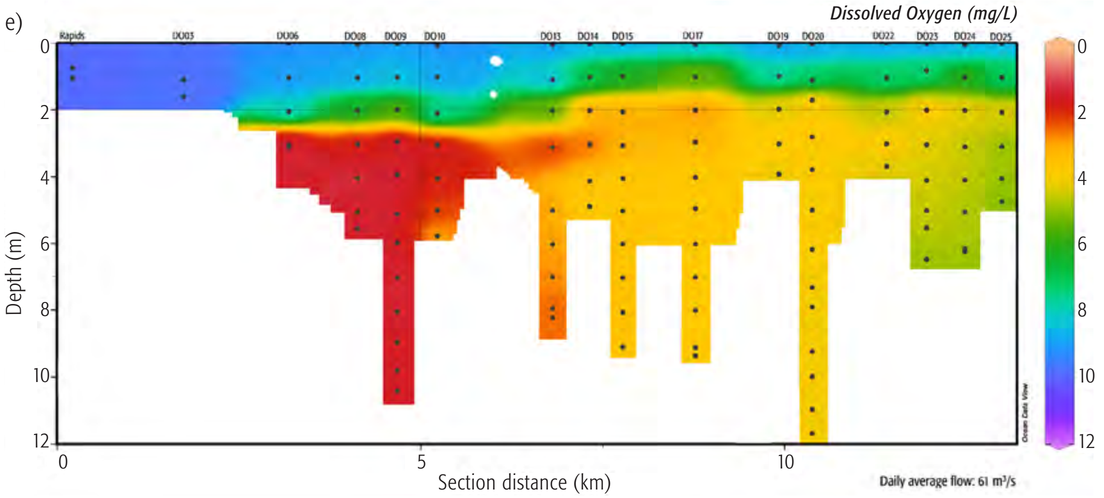

Dissolved Oxygen (mg/L)
Rapids
DO03
NN Logger
DO08
DO09
DO10
DO13
DO14
DO15
DO17
DO19
DO20
DO22
DO23
DO29
Depth (m)
Section distance (km)
Daily average flow: 36 m³/s
Flow (m³/s)
Feb 21
Feb 28
Mar 07
Mar 14

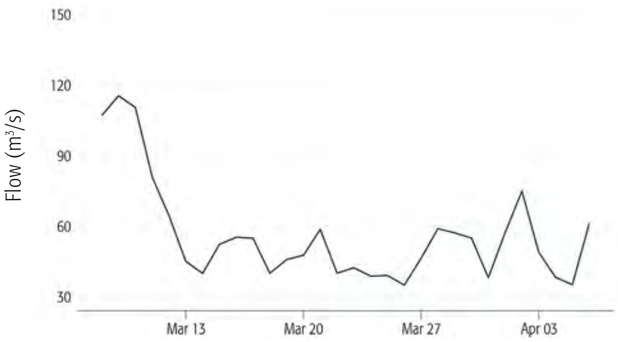
c)


Dissolved Oxygen (mg/L)
NN Logger
DO05
DO08
DO13
DO15
DO17
DO20
DO22
DO23
Depth (m)
Section distance (km)
Daily average flow: 52 m³/s
Flow (m³/s)
Feb 17
Feb 24
Mar 02
Mar 09
Mar 16

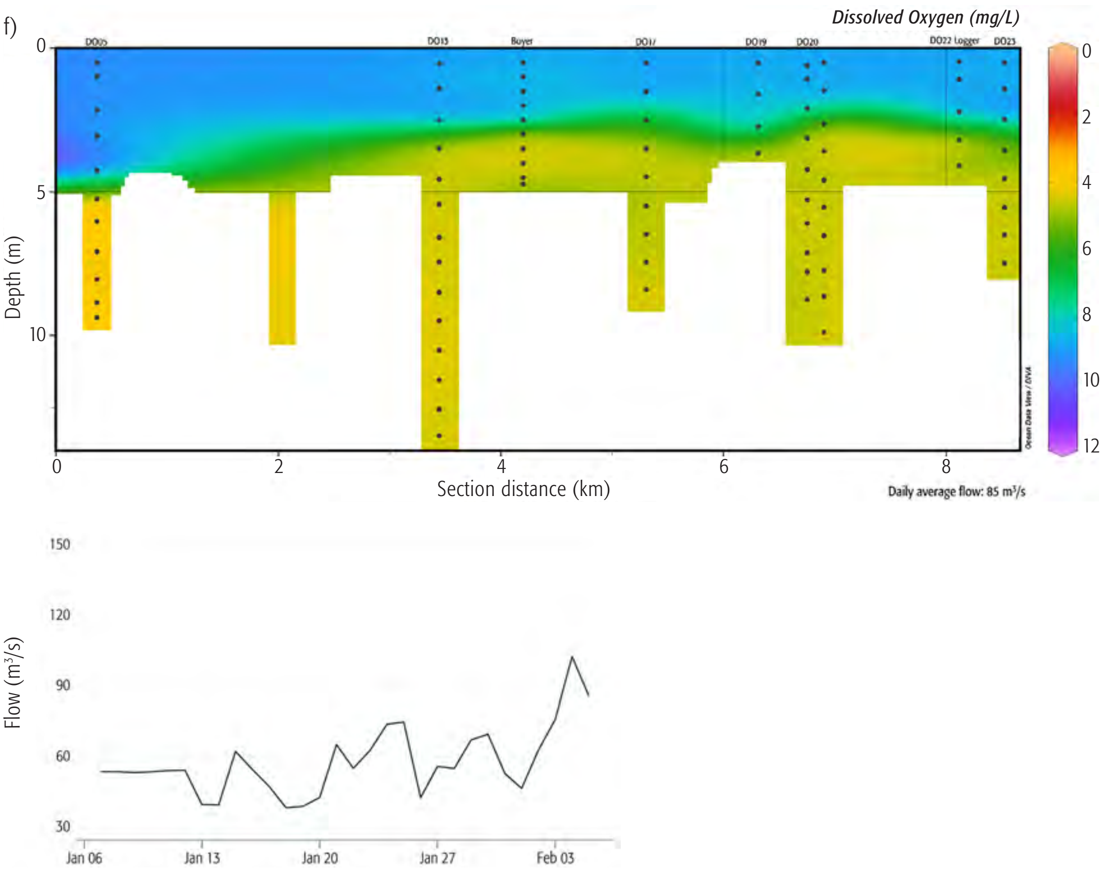

d)
Dissolved Oxygen (mg/L)
Rapids
DO01
DO03
DO05
DO06
DO08
DO09
DO10
DO11
DO13
DO15
DO17
DO19
DO20
DO22
DO23
DO24
DO25
Depth (m)
0
2
4
6
8
10
0
5
10
Section distance (km)
Daily average flow: 53 m³/s
0
2
4
6
8
10
12


Flow (m³/s)
150
120
90
60
30
Feb 24
Mar 03
Mar 10
Mar 17
Mar 24

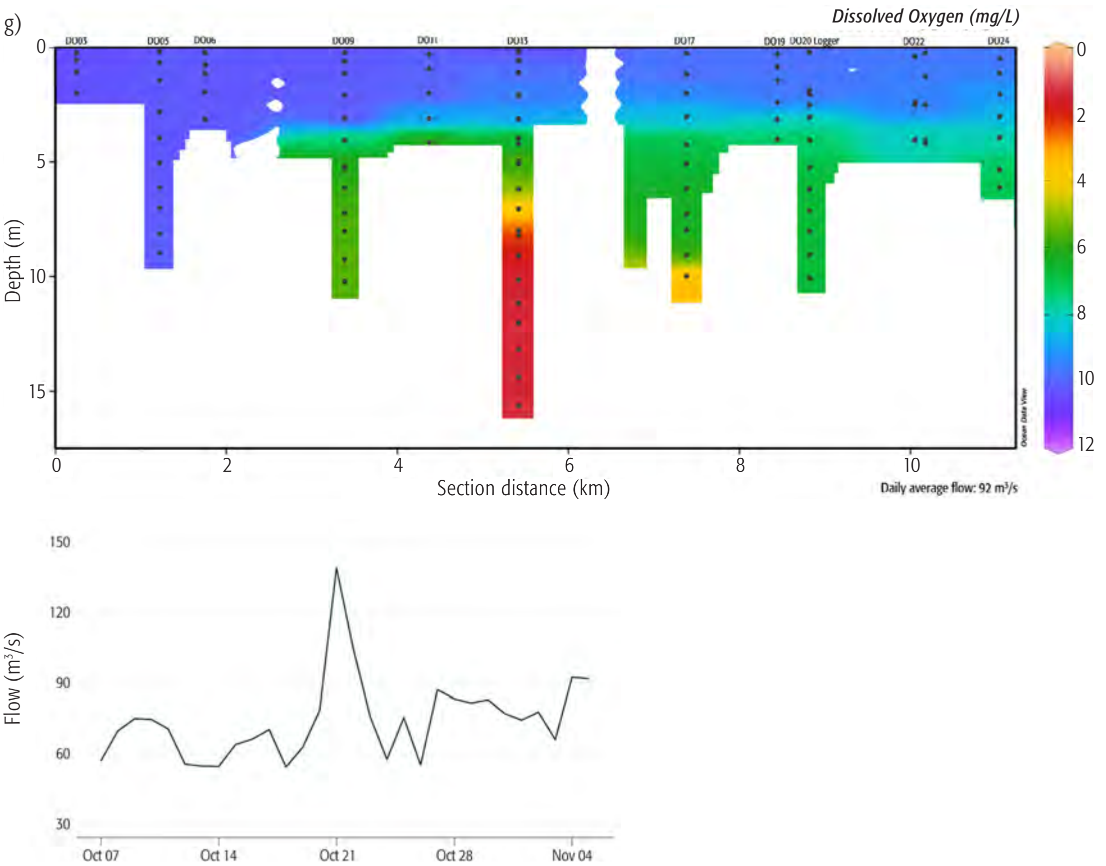

e)
Dissolved Oxygen (mg/L)
Rapids
DO03
DO06
DO08
DO09
DO10
DO13
DO14
DO15
DO17
DO19
DO20
DO22
DO23
DO24
DO25
Depth (m)
0
2
4
6
8
10
12
0
5
10
Section distance (km)
Daily average flow: 61 m³/s
0
2
4
6
8
10
12


Flow (m³/s)
150
120
90
60
30
Mar 13
Mar 20
Mar 27
Apr 03

f)


Dissolved Oxygen (mg/L)
DO05
DO13
Buoy
DO17
DO19
DO20
DO22 Logger
DO23
Depth (m)
Section distance (km)
Daily average flow: 85 m³/s
Flow (m³/s)
Jan 06
Jan 13
Jan 20
Jan 27
Feb 03

g)


Dissolved Oxygen (mg/L)
DO03
DO05
DO06
DO09
DO11
DO13
DO17
DO19
DO20 Logger
DO22
DO24
Depth (m)
Section distance (km)
Daily average flow: 92 m³/s
Flow (m³/s)
Oct 07
Oct 14
Oct 21
Oct 28
Nov 04

Discussion

Low-DO conditions in the upper estuary are almost exclusively associated with the saline bottom waters of the estuarine salt wedge. River Derwent flow is the principal regulator of the location of the salt wedge in the upper Derwent Estuary (Herzfeld et al., 2005). When riverine flows are high, waters are fresh and mixed, therefore high DO concentrations are observed throughout the water column (Marine Solutions, 2025). Our findings suggest that flows >200 m^3/s are required to completely flush the salt wedge downstream of the Boyer site. This is greater than previous findings, which found that flows of >150 m^3/s, were sufficient to displace the salt wedge in the upper estuary to downstream of Bridgewater Bridge (Thomson and Godfrey, 1985). This finding is consistent with observations made by Marine Solutions (2025), who recorded two profiles of unstratified freshwater conditions throughout the water column at Turriff Lodge STP, both monitoring events succeeded flows >200 m^3/s. Profiling shows moderate freshwater flows can trap marine bottom waters in the deep holes of the upper estuary, which, if isolated for extended periods of time can become hypoxic. During low-flow conditions, spring tides can provide pulses of oxygenated marine water to bottom waters of the upper estuary.

Logger observations for bottom waters were collected at one site and are not representative of all areas of upper estuary bottom waters. The severity and duration of hypoxia in the upper estuary bottom waters measured at the Boyer wharf logger site during summer and autumn is variable. Extended hypoxia was recorded at the Boyer site in 2020/21 late summer and autumn comparison period; hypoxia was only recorded 28% and 8% of the comparison period during the 2022 and 2025 season, respectively. Higher spring and early summer flows in the Derwent Estuary may have contributed to the higher DO conditions at the Boyer site in summer/autumn 2022 and 2025. Low-flow conditions throughout spring mean that salt wedge intrusion in the upper estuary begins earlier, increasing residence time of bottom waters leading into summer months. Processes that consume DO in the water will deplete DO leading into summer, increasing severity and duration of hypoxia in bottom waters throughout summer and into autumn.

During the 2020/21 season, severe hypoxic conditions were sustained for over one month at the Boyer site. During February and March 2021, rates of DO decline were observed to be much faster compared to declines occurring in late spring and early autumn. DO was generally found to decline at a steady rate (linear), except for February 2021, where the decline was observed to be exponential. Exponential rates of DO decline were defined in the foundational stream water quality model by Streeter and Phelps (1925), which related BOD from organic matter decomposition to DO concentration in a river or stream. Noting the Streeter–Phelps model does not account for several important factors in addition to BOD which can influence the rate of oxygen consumption in a system. During the 2020/21 season, several high flow events occurred that had negligible impact on the DO conditions of bottom waters at the logger site, which were sufficient to introduce sustained high DO to bottom waters in the 2022 and 2025 season. Similarly, spring-tide incursions of marine waters during 2020/21 only increased DO concentrations on average 0.74 (+/- 0.23) mg/L, for a short period of time. Extended hypoxia in bottom waters will impact upon sediment biogeochemistry, nutrient and metal cycling, primary productivity, and benthic microbial community composition, which in turn will influence oxygen demand in these hypoxic regions (Middelburg and Levin, 2009).

Next steps

- Install logger to collect DO bottom water data from spring into summer to further assess DO declines leading into summer and autumn hypoxic conditions.
- Establish relationship between DOC and non-purgeable organic carbon collected in water samples for ambient water quality monitoring.
- Explore options to conduct multivariate analysis and numerical modelling on upper estuary AWQ data that includes DO, water temperature, salinity, nutrients, non-purgeable organic carbon (if suitable proxy for DOC), and daily average flow to investigate relative influence of each of these parameters on upper estuary DO.
- Explore methods to quantify COD and BOD in the upper estuary stratified bottom waters during summer and autumn.

6.3 Dissolved organic matter sources characterisation trial

Novel methods to 'fingerprint' dissolved organic matter sources in upper estuary water

Provided by Dr Aleicia Holland and Dr Ewen Silvester, La Trobe University

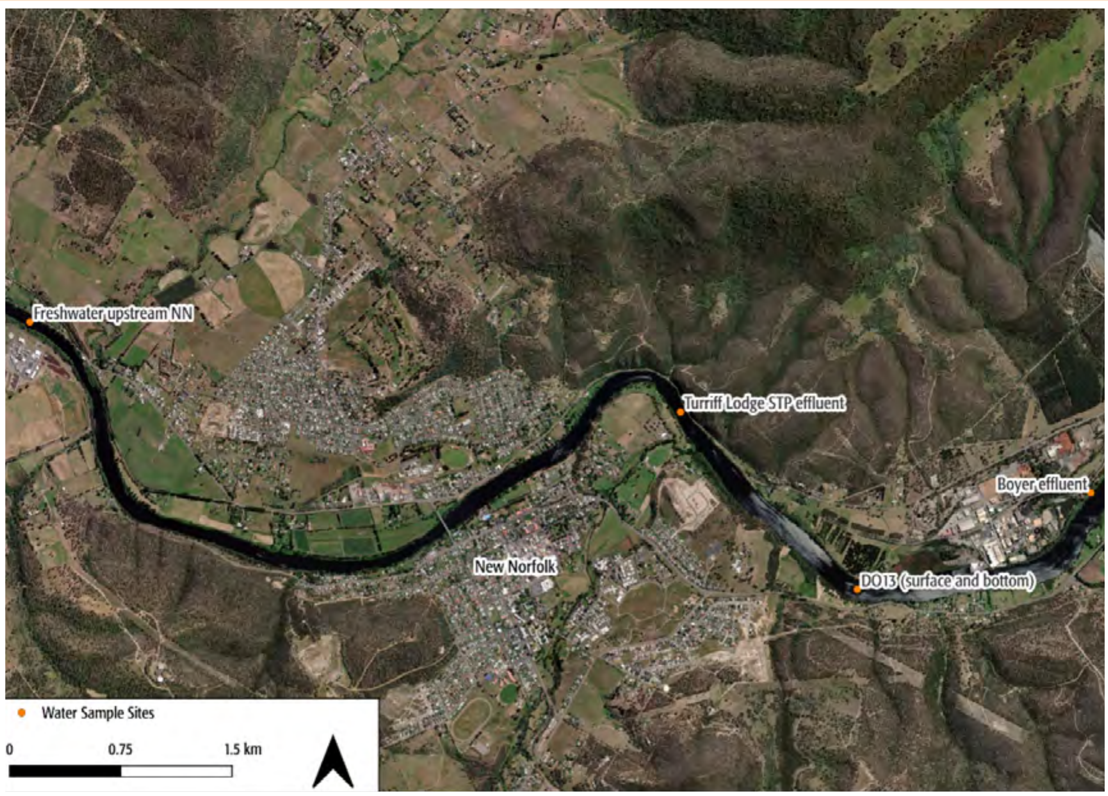
Executive summary

The upper Derwent Estuary experiences seasonally low dissolved oxygen (DO) in bottom waters throughout summer and autumn, with severe and extended hypoxic conditions a known issue (Section 6.2). Elevated dissolved organic matter (DOM) loads in effluent discharge and terrestrially derived catchment sources may be contributing to increased microbial activity and biological oxygen demand (BOD), further exacerbating hypoxic conditions in the upper estuary. DOM characterisation techniques, including novel liquid chromatography characterisation, were tested on water samples collected from riverine, estuarine and effluent outfall sites in the upper Derwent Estuary. These methods were assessed for applicability as a potential tool to trace effluent in estuarine water samples and assess relative contribution of effluent DOM to hypoxic zones in the upper estuary. DOM optical indices (BIX, HIX, SR and SUVA254), protein-like DOM components, organic acid composition (particularly pyruvic acid) and amino acid composition (branch chain amino acids) all show promise to trace effluent sources in collected water samples.

Methods

Three replicate water samples were collected from five sites in the upper estuary in July 2021 and sent to La Trobe University lab for analysis. Freshwater riverine site was located upstream of New Norfolk, surface and bottom water samples were collected from estuarine monitoring site DO13, and effluent samples were collected from Turriff Lodge sewage treatment plant (STP) and Boyer outfall (Figure 6.28).

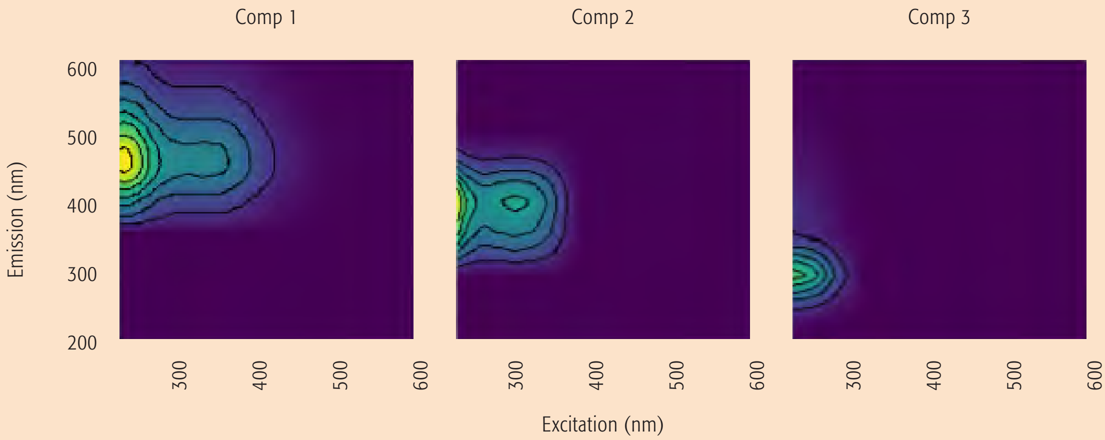
Figure 6.28 Map of water sample sites in the upper Derwent Estuary collected for DOM characterisation analysis.



Optical DOM characterisation method

Optical methods consist of fluorescence and absorbance spectroscopy. These spectroscopic techniques are used to analyse molecules within samples by measuring how light interacts with them. Model analysis determined the presence of three key components within samples. These consisted of humic-like (Comp.1), fulvic-like (Comp.2) and protein-like DOM (Comp.3) (Figure 6.29).

Figure 6.29 Dissolved organic matter (DOM) components determined via excitation emission scans followed by PARAFAC analysis. Comp.1 = humic-like; Comp.2 = fulvic-like; Comp.3 = protein-like DOM.



Each sample was found to be quite unique in terms of its DOM characteristics. DOM found within the freshwater sample was dominated by more terrestrially derived components. The two effluent samples contained significantly higher amounts of dissolved organic carbon (DOC) and smaller molecules of microbial origin (Figure 6.30).

Bottom-water samples contained DOM more similar to that of the freshwater sample, whereas surface samples were more similar to the Turriff Lodge STP in terms of its fluorescence components and several optical indices (BIX, HIX, SR, SUVA254) (Figure 6.31). DOM characterisation methods are described in Acharya *et al*. (2023).

Figure 6.30 Dissolved organic matter (DOM) optical indices and dissolved organic carbon concentration (DOC mg/L) of collected water samples. bix = biological index (high values close to 1 = freshly produced DOM); hix = humification index (high values indicate DOM higher degree of humification); SR = spectral slope ratio (higher values = lower relative molecular weight); FI = fluorescence index (higher values indicate more autochthonous microbially derived DOM); SUVA254 = specific absorbance @ 254nm (higher values = higher relative aromaticity).

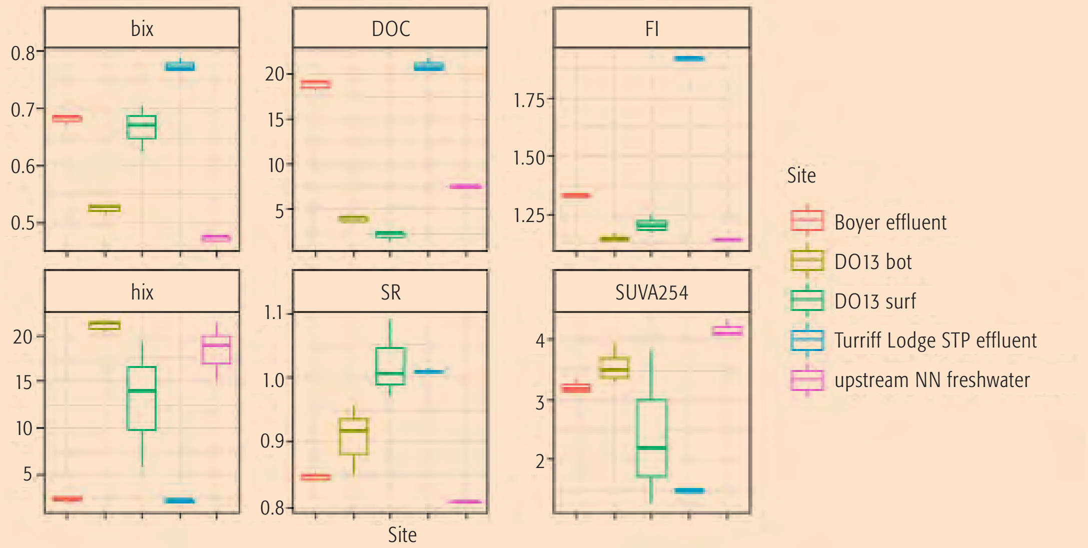
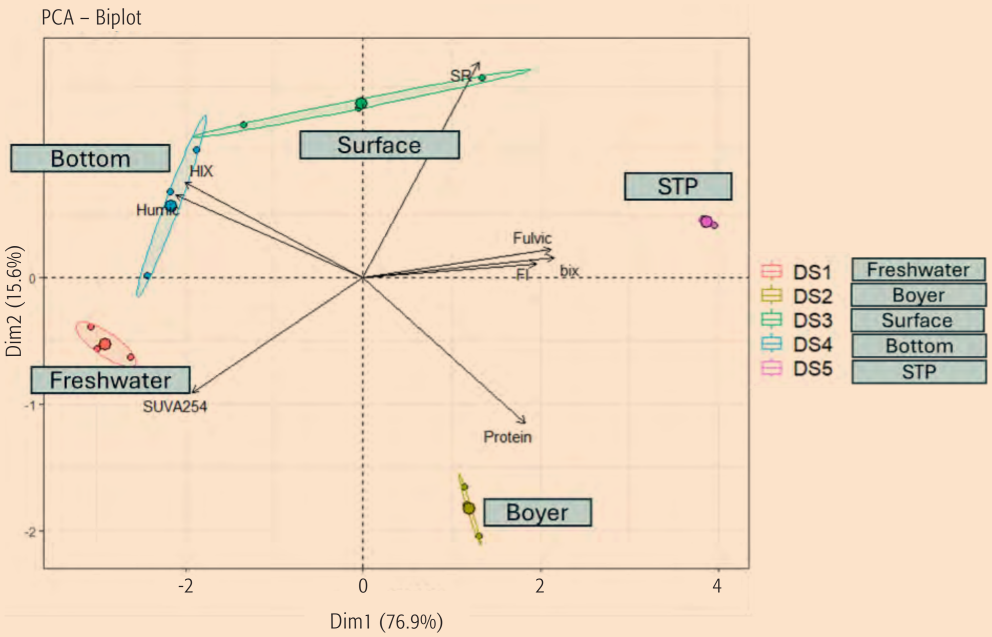


Figure 6.31 Principal component analysis plot showing differences in dissolved organic matter (DOM) composition and optical indices of collected water samples. bix = biological index (high values close to 1 = freshly produced DOM); hix = humification index (high values indicate DOM higher degree of humification); SR = spectral slope ratio (higher values = lower relative molecular weight); FI = fluorescence index (higher values indicate more autochthonous microbially derived DOM); SUVA254 = specific absorbance @ 254nm (higher values = higher relative aromaticity).



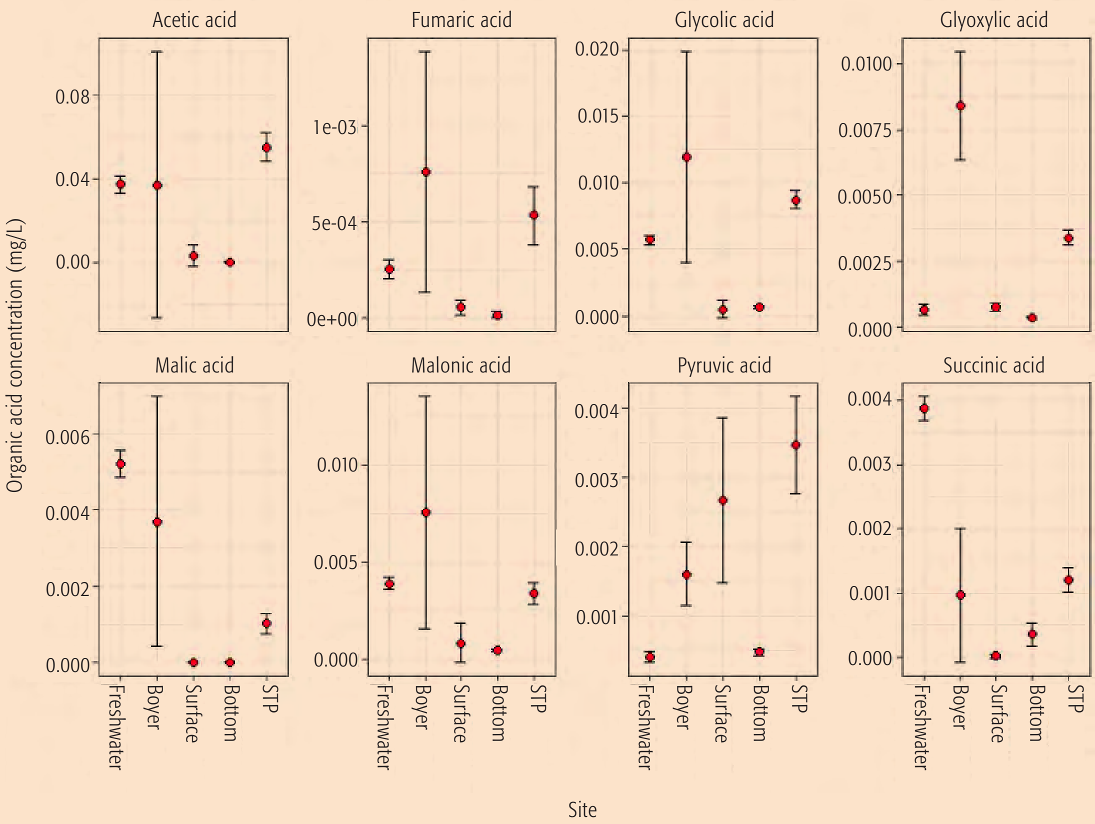
Liquid chromatography (organic and amino acids) characterisation method

Organic acids

Organic-acids were characterised by liquid chromatography-tandem mass spectrometry (LC-MS/MS). Test samples could be separated by organic-acid composition. However, there was not as clear pattern of separation between freshwater and bottom water samples, and effluent and surface water samples, as was found in the optical DOM characterisation.

There was high variability between the Turriff Lodge STP effluent (STP) water samples, suggesting changing effluent characteristics over time. The surface sample was different from both the freshwater and bottom sample in regard to pyruvic acid concentration (Figure 6.32). The surface sample sat in between the STP and Boyer effluent, suggesting that the surface water likely contains pyruvic acid from the STP effluent and this organic acid might be used as a signature molecule for the STP effluent organic material.

Figure 6.32 Concentrations of various organic acids in Derwent water samples.



Amino acids

Liquid chromatography characterisation technique analyses samples for 18 proteinogenic amino acids (Silvester *et al*., 2021). The effluent samples contained elevated levels of all amino acids (i.e., the branched-chain AAs (ILE, LEU & VAL), PRO, PHE) (Figure 6.33), and relatively higher levels of more reduced (i.e, lower oxidation state) AAs compared to the freshwater and bottom water samples (Figure 6.34). Unfortunately, poor recovery of surface water samples meant that no comparisons could be made for this site in terms on its amino acid composition (Figure 6.34).

Figure 6.33 Concentrations of various amino acids (AAs) in Derwent water samples.

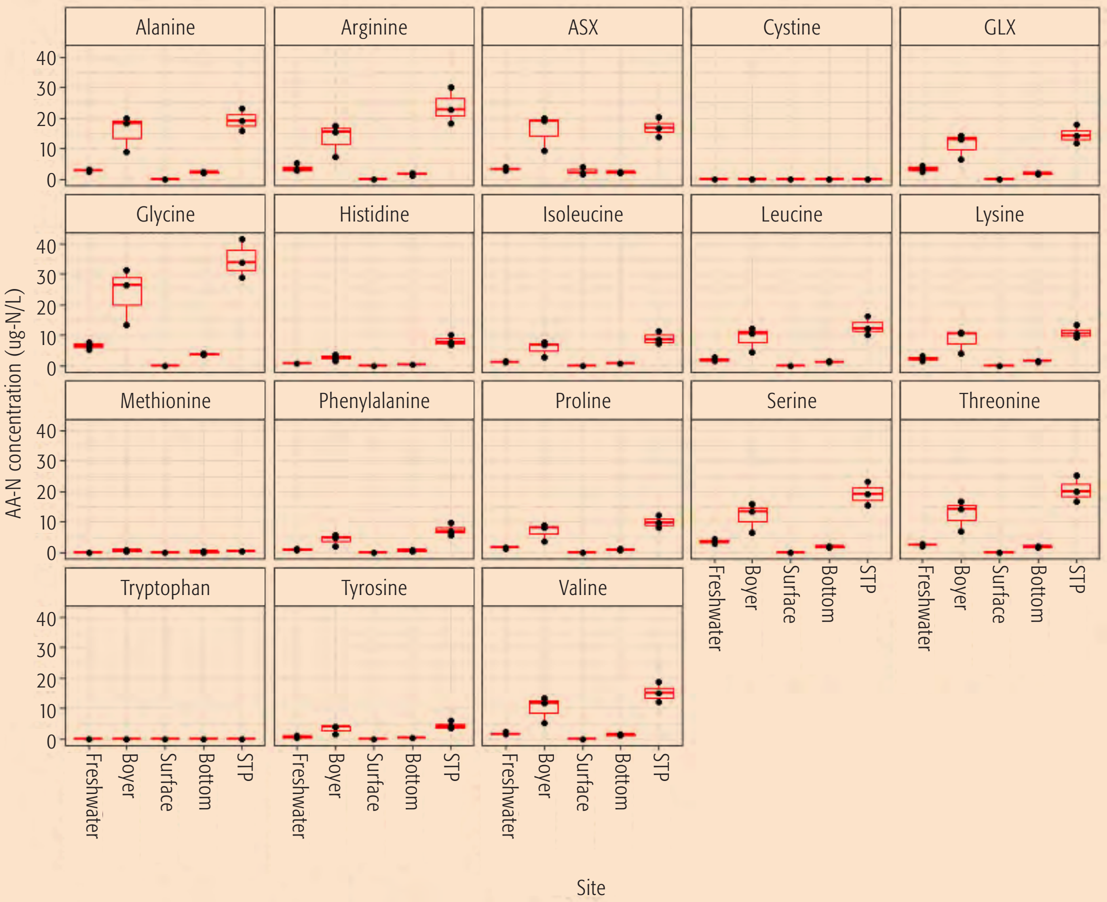
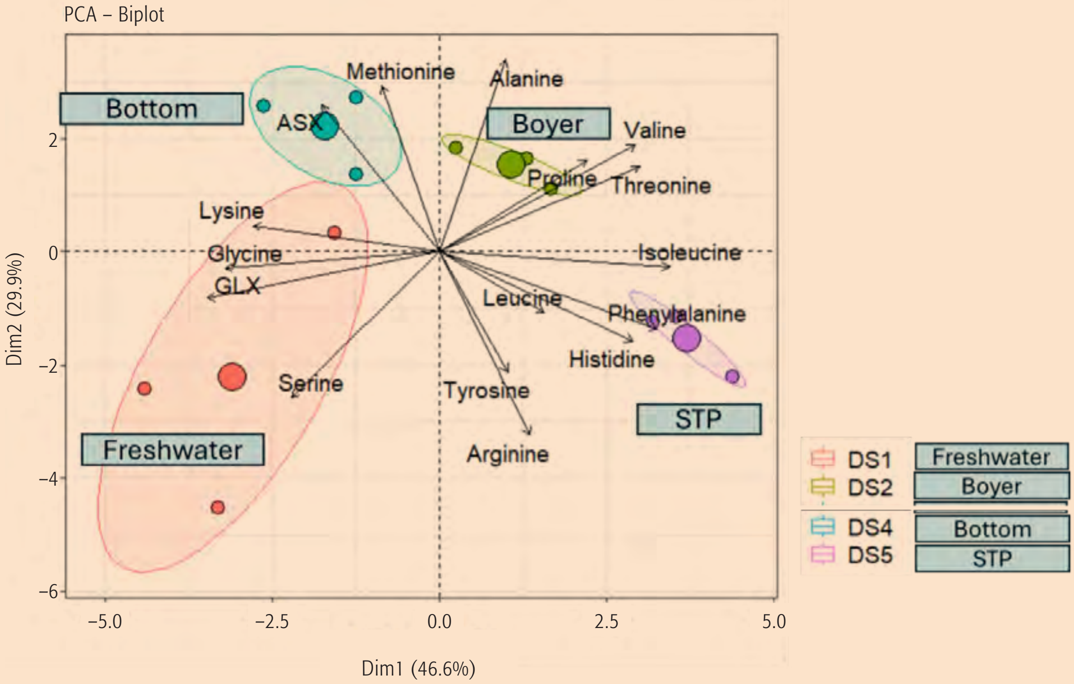


Figure 6.34 Principal component analysis plot showing differences in amino acids between samples. Site DS3 – surface is missing due to poor recoveries.



Discussion

A number of DOM characterisation techniques show promise in their potential as tracers of effluent sources, including DOM optical indices (BIX, HIX, S_R and $SUVA_{254}$), protein-like DOM components, organic acid composition (particularly pyruvic acid) and amino acid composition (branched-chain amino acids).

Effluent samples were clearly different from freshwater and bottom-water estuarine samples in DOM characteristics. The similarity between estuarine surface-water samples and the two effluents suggests the presence of effluent DOM in this layer. Given the strong stratification between surface and bottom waters in this region of the estuary throughout most of the year, perhaps it should not be surprising that effluent discharged to surface waters is not vertically mixed with bottom waters. DOM in the freshwater sample was more similar to bottom waters than surface waters, which is surprising given that physicochemical parameters showed that this site was stratified at the time of sampling and bottom waters were characteristic of salt-wedge marine waters.

The novel method of liquid chromatography characterisation showed promise in its ability to characterise effluent DOM. Liquid chromatography with a built-in organic carbon detector has been proposed as an additional method that could be trialled if work related to DOM characterisation in the upper estuary continues. This method will give a more comprehensive characterisation of DOM components (five components, instead of three), as well as estimates of the molecular weight and aromaticity of humic/fulvic molecules.

It is recommended that these methods are trialled again under a range of different flow regimes and stratification to trace effluent DOM between sites and include an estuarine surface and bottom water sample site downstream of the Boyer effluent outfall. Since this analysis was conducted the Turriff Lodge STP effluent discharge location has changed from surface to bottom water discharge. Therefore, it is of particular interest to investigate if DOM characteristics of bottom waters in the upper estuary now detect presence of STP effluent DOM.

6.4 Metals in porewater

Porewater analysis of three sites in the Derwent Estuary

Provided by Kira Sirois, Professor Zanna Chase, and Noah Menner (University of Tasmania), and Professor Tony Rathburn (California State University)

Porewater is the trapped water between sediment particles, and it is the water in which benthic organisms live. Trace metals in pore waters are considered a highly bioavailable fraction in the sediment as metals in pore water can enter organisms through direct contact with skin or gills (Chapman *et al*., 1998). In contrast, metals associated with sediment particles can only enter an organism, if the particle is ingested (including cell phagocytosis). Not only is porewater a potential exposure route of trace metals for benthic organisms, but it is also a potential source of trace metals to the overlying waters of the estuary (Ouddane *et al*., 2004; Zhang *et al*., 2020; Liu *et al*., 2022; Li *et al*., 2023).

The DEP collaborated with IMAS to carry out a case study to assess porewaters in the historically contaminated estuary. Sediment samples were collected by boat from three bays (Montagu Bay, Newtown Bay, and Cornelian Bay). The cores were then sliced into one-centimetre, disc-shaped increments from the top down. These increments were centrifuged to extract the porewaters, which were filtered to remove particles, and analysed for trace metal content by Analytical Services Tasmania. Results of the analysis are presented in Table 6.6.

The samples were not handled under oxygen-free conditions, which means concentrations may underestimate the true conditions, as some dissolved metals may have been removed due to reactions with oxygen before the samples were filtered. Future porewater analysis would provide a more accurate assessment if samples were collected and prepared under anoxic conditions (Chapman *et al*., 2002).

Generally, metal and metalloid concentrations at the three sites were below Default Guideline Values (DGVs) (Warne *et al*., 2025). Zinc exceeded the DGV of 8 µg/L in all bays, and concentrations varied with depth except for 10-12 cm which was at the DGV. Cornelian Bay had the highest concentrations, with values greatest at the surface. Arsenic concentrations in the top 3 cm at New Town Bay 47 µg/L, though not enough data is currently available to develop a trigger level. Iron and manganese were highest in New Town Bay as well, with a mainly uniform distribution down core. Other metals of potential concern (cobalt, copper, lead) were consistently below the limit of reporting and require more sensitive analysis. Molybdenum and selenium need further data to establish trigger level values. An element-by-element summary is presented in Table 6.7.

Trace metals in porewaters can enter the overlying water through diffusion, if sediment pore waters in contact with the water contain higher metal concentrations than the overlying water (Santos-Echeandia *et al*., 2009). This gradient causes diffusion, the movement of ions (in this case metal ions) from higher concentration to lower. For instance, median zinc in bottom waters at AWQ sites U2 and NTB05 (close proximity to the porewater study area) was 20 and 16.5 µg/L, respectively, during reporting period (**Section 6.1.5**). In this study, zinc concentrations in porewater samples were 84 µg/L in Cornelian Bay, 14 µg/L in Montagu Bay, and 20 µg/L in Newtown Bay. If the zinc concentrations in the waters over the sediment in Cornelian Bay are similar to those at the previously reported AWQ sites, the porewater concentration would be markedly higher, and therefore, zinc in the porewaters would be favoured to diffuse across the sediment-water interface and into the overlying waters.

In addition to diffusion, disturbances can cause a high concentration of heavy metals to be released to overlying waters in a large flux, potentially causing a hot spot for contamination (Butler *et al*., 2005).

Disturbances can be chemical, physical, and biological. DO can have drastic effects on how the sediment particles themselves can hold onto metal, keeping them bound up and not available to organisms living in or on the sediment. Levels of DO often are highly variable geographically and seasonally in the Derwent and can cause sediments to either become sinks or sources of legacy metal contaminants.

Boat wakes in shallow areas can cause a physical disturbance by stirring up sediments and releasing porewaters high in metals into the overlying waters. Smaller physical disturbances can be caused by burrowing organism like worms. When then burrow they agitate the sediment, releasing porewaters to overlying waters, called bio-irrigation. If there is a high concentration of metals in these bio-irrigated porewaters, this can also lead to a flux of metals out of the sediment porewaters to overlying waters.

A conceptual model illustrating metal mobilisation from sediment to porewater under varying oxygen conditions is provided in figure 6.33.

All these processes paint a complex picture of the cycling of metals in sediment. Constant monitoring and analysis of benthic organisms can help assess how metals are affecting the estuary. It can also help us to understand how metals stored in sediments can make their way through the food chain and potentially cause harm to human health, ecosystem health, and fisheries and decrease ecosystem services.

A student project is currently being conducted to investigate baseline ecology of foraminifera in the Derwent Estuary and assess if these single-celled organisms can be used as a bioindicator for heavy metal pollution in the estuary.

Table 6.6 shows concentrations of metals and metalloids (As and Se) in pore waters from three bays in the Derwent Estuary. The ANZG Default Guideline Values are provided for comparison and are based on the assignment of the Derwent to the 95% protection level, concentrations higher than the trigger level are in bold. ID= insufficient data (Warne *et al*., 2025). "<" symbol = below detection limit. Concentrations are in µg/L and depths are in centimetre increments.

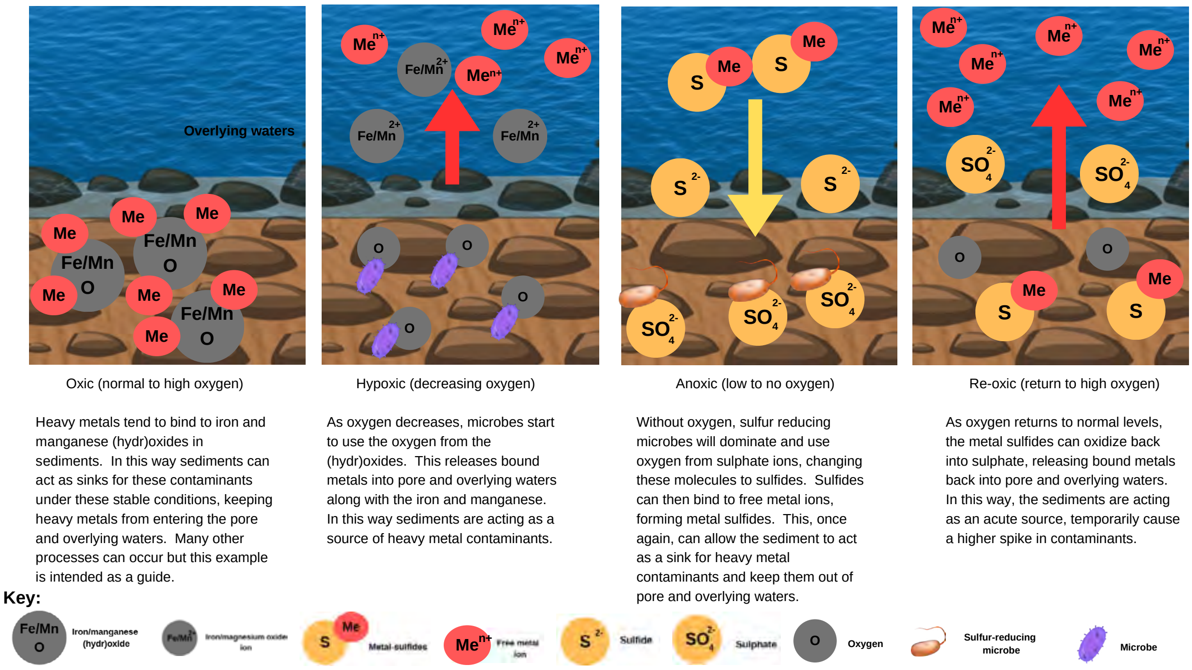
	Hg	Al	As	Cd	Cr	Co	Cu	Fe	Pb	Mn	Mo	Ni	Se	Zn
Montagu Bay														
0-3	<0.05	<20	15	<1	<1	<3	<2	374	<10	**193**	15	<5	<20	**14**
3-5	<0.05	<20	13	<1	<1	<3	<2	140	<10	**166**	32	<5	<20	**11**
5-8	<0.05	<20	16	<1	<1	<3	<2	415	<10	**179**	45	<5	<20	8
8-10	<0.05	<20	19	<1	<1	<3	<2	81	<10	**112**	58	<5	<20	**9**
10-12	<0.05	<20	18	<1	<1	<3	<2	61	<10	71	37	5	<20	**25**
Newtown Bay														
0-3	<0.05	27	47	<1	<1	<3	<2	**6010**	<10	**521**	13	<5	<20	**20**
3-5	<0.05	<20	51	<1	<1	<3	<2	**8270**	<10	**509**	16	<5	<20	**12**
5-7	<0.05	<20	57	<1	<1	<3	<2	**9160**	<10	**569**	18	<5	<20	**17**
7-9	0.06	<20	55	<1	<1	<3	<2	**8280**	<10	**490**	17	<5	<20	**14**
9-11	<0.05	<20	57	<1	<1	<3	<2	**8000**	<10	**419**	20	<5	<20	**11**
11-13	<0.05	<20	60	<1	<1	<3	<2	**8740**	<10	**444**	16	24	<20	**17**
13-15	<0.05	<20	49	<1	<1	<3	<2	**8790**	<10	**515**	21	<5	<20	**12**
Cornelian Bay														
1-3	<0.05	<20	<10	<1	<1	<3	<2	125	<10	**119**	11	<5	<20	**84**
3-4		<20	<10	<1	<1	<3	<2	261	<10	**125**	14	<5	<20	**38**
5-7	<0.05	<20	12	<1	<1	<3	<2	**598**	<10	**138**	22	<5	<20	**16**
7-9	0.1	<20	<10	<1	<1	<3	**3**	236	<10	**97**	61	<5	<20	**12**
9-11	<0.05	<20	<10	<1	<1	<3	<2	454	<10	**97**	47	<5	<20	**11**
11-14	<0.05	<20	<10	<1	<1	<3	<2	207	<10	**85**	43	<5	<20	**11**
Default Guideline Value (95%)	0.4 (in-organic)	37	ID	5.5	4.4	1	0.6	540	4.4	80	ID	70	ID	8

Table 6.7 Background concentrations and Default Guideline Values in Australian estuarine and marine waters provided by ANZECC (2000) and Warne *et al*., (2025).

Mercury (Hg)	Mercury has been known as a serious toxin that can bioaccumulate and biomagnify and negatively impact human and ecosystem health. Most of the pore water samples analysed had mercury levels below the detection limit (0.05 µg/L), which is itself well below the DGV of 0.4 µg/L. The highest observed concentration was 0.1 µg/L in the 7-9 cm increment in Cornelian Bay. Reliability of this DGV is very high.
Aluminium (Al)	Al is a naturally abundant metal in the environment. Most Al concentrations in pore water were below the detection limit of 20 ug/L except the 0-3 cm increment in New Town Bay (27 ug/L), and all measurements were below the DGV of 37 ug/L. There is no information found for background levels in Australian estuarine waters. Reliability of this DGV is very high.
Arsenic (As)	Background As concentrations in Australia have been reported as 1.0-3.3 µg/L in estuarine waters. Concentrations between 47-60 µg/L were found in New Town Bay pore waters and concentrations between 13-19 µg/L were found in Montagu Bay, with higher concentrations at the surface and below 8 cm. There is insufficient data to establish DGVs, though these concentrations are higher than background concentrations. No reliability reported.
Cadmium (Cd)	Cadmium was under the detection limit of 1 µg/L for all samples, lower than the DGV (5.5 µg/L). More sensitive analytical techniques may be useful to monitor changes. There is no information found for background levels in Australian estuarine waters. Reliability of this DGV is very high.
Chromium (Cr)	Chromium was under the detection limit of 1 µg/L for all samples, lower than the DGV (5.5 µg/L). Background concentrations are reported as 0.01-0.1 µg/L. More sensitive analytical techniques may be useful to monitor changes and to establish background levels. Reliability of this DGV is very high.
Cobalt (Co)	Cobalt was under the detection limit of 3 µg/L for all samples. The DGV concentration is 1 µg/L and therefore Co levels in pore waters could in fact be at or past the trigger level. More sensitive analytical techniques are recommended for Co. There is no information found for background levels in Australian estuarine waters. Reliability of this DGV is high.
Copper (Cu)	Copper was under the detection limit of 2 µg/L for all samples except for the 7-9 cm increment in Cornelian Bay (3 µg/L). The DGV concentration is 0.6 µg/L and therefore Cu levels in pore waters could in fact be at or past the trigger level. More sensitive analytical techniques are recommended for Cu. Reliability for this DGV is very high.
Iron (Fe)	Fe is naturally occurring in the environment, with a trigger level of 540 ug/L. Background levels for estuarine waters of Australia are reported as 0.76-67 µg/L. Levels of Fe range from 125-598 µg/L in Cornelian Bay (with highest concentrations in the 5-7 cm increment), from 61-415 µg/L in Montagu Bay (with higher levels at the surface to 8 cm), and from 6010–9160 µg/L in New Town Bay–over ten times the DGV. The high iron concentrations may be related to redox processes but require further study. Reliability for this DGV is moderate.
Lead (Pb)	Background concentrations for Pb are between 0.02-0.13 µg/L but were under the detection limit (10 µg/L) for all samples. More sensitive analytical techniques are needed to assess if concentrations are under the DGV of 4.4 µg/L. Reliability for this DGV is low.
Manganese (Mn)	Background concentrations are between 0.55-3.1 µg/L with a DGV of 80 µg/L. All samples are higher than background levels and the DGV (except increment 10-12 cm in Montagu Bay, 71 µg/L). Concentrations range between 71-193 µg/L in Montagu Bay with highest levels in the surface, 419-569 µg/L in New Town Bay with higher levels in the top 7 cm, and 85-138 µg/L in Cornelian Bay with highest levels in the top 7 cm as well. Further exploration of these high levels of Mn is needed. Reliability of this DGV is unknown.

Molybdenum (Mo)	Levels of Mo range between 15-58 µg/L in Montagu Bay with highest levels in the 8-10 cm increments, 13-21 µg/L in New Town Bay with highest levels below 13 cm, and 11-61 µg/L in Cornelian Bay with highest found in the 7-9 cm increment. There is insufficient data to establish a DGV and no information found for background concentrations. No reliability reported.
Nickel (Ni)	All samples were below the detection limit of 5 µg/L except for the 10-12 cm increment in Montagu Bay (5 µg/L), all of which are well below the DGV of 70 µg/L. Background concentrations are reported as 0.14-1.10 µg/L so more sensitive analytical techniques may help assess if Ni levels are higher than background concentrations. Reliability of this DGV is very high.
Selenium (Se)	All samples are under the detection level of 20 µg/L and there is insufficient data to establish a DGV and no information found for background levels. More data is needed for Se. No reliability reported.
Zinc (Zn)	Background levels have been reported to range from 0.39-3.8 µg/L and 0.4-1.8 µg/L. All samples were well above these background levels ranging from 8-25 µg/L in Montagu Bay with highest levels between 10-12 cm, 12-20 µg/L in New Town Bay with the highest level (20 µg/L) in the top 3 cm, and 11-84 µg/L in Cornelian Bay with highest level (84 µg/L) in the 1-3 cm increment. Highest levels in all samples were well above the DGV of 8 µg/L, especially in Cornelian Bay where the highest level was in the 1-3 cm increment at 83 ug/L-ten times the DGV. Ongoing monitoring of Zn is needed. Reliability of this DGV is very high.

Figure 6.35 Conceptual model illustrating the mobilisation of metals between sediments and porewaters. This schematic is intended as a general guide and does not represent a universal process.



6.5 Environmental DNA for biomonitoring

Provided by Andrew Bissett and Jodie Van De Kamp (CSIRO)

The following content has been provided by CSIRO and the Australian Microbiome to provide awareness of the data we collect and make publicly available and of the potential applications that can be explored. eDNA data is available at **Australian Microbiome – Bioplatforms Australia**.

Environmental DNA (eDNA) is DNA purified from environmental samples, such as sediment and water, from which we can use DNA sequencing of marker genes to identify organisms across the tree of life – from microbes to fish, as well as functional genes. It is a non-invasive, standardisable sampling approach which coupled with rapid advances in DNA sequencing technology is revolutionising biodiversity science. Researchers, industry and governments are increasingly incorporating eDNA surveys into their toolkits for biomonitoring because of its high accuracy, taxonomically holistic lens and ease of deployment. Biomonitoring via eDNA can be applied to studies of biodiversity, including identifying and monitoring rare and elusive species. It is useful for a variety of applications such as conservation, biosecurity, assessing ecosystem health and tracking functional genes such as those involved in antimicrobial resistance, metal degradation or biogeochemical cycling.

The Derwent Estuary is affected by several environmental issues of significant concern: historical contamination by heavy metals, loss of habitats and species, introduced marine pests, weeds and other exotic species, altered river flow regimes and blocked migratory paths for fish and eels, and increasingly, elevated nutrient loading in the estuary due to inputs from sewage treatment plants, industry, and the catchment, algal growth, and hypoxia in localised areas (Green and Coughanowr, 2004; DEP, 2010b, 2015b; Macleod and Coughanowr, 2019). Beginning in 2019, as part of the **Australian Microbiome project**, staff at the CSIRO in Hobart, in collaboration with the Derwent Estuary Program (DEP), have been analysing eDNA in water samples collected as part of the AWQ monitoring program from several sites in the lower and middle estuary and producing a timeseries dataset of microbial, planktonic and eukaryote biodiversity.

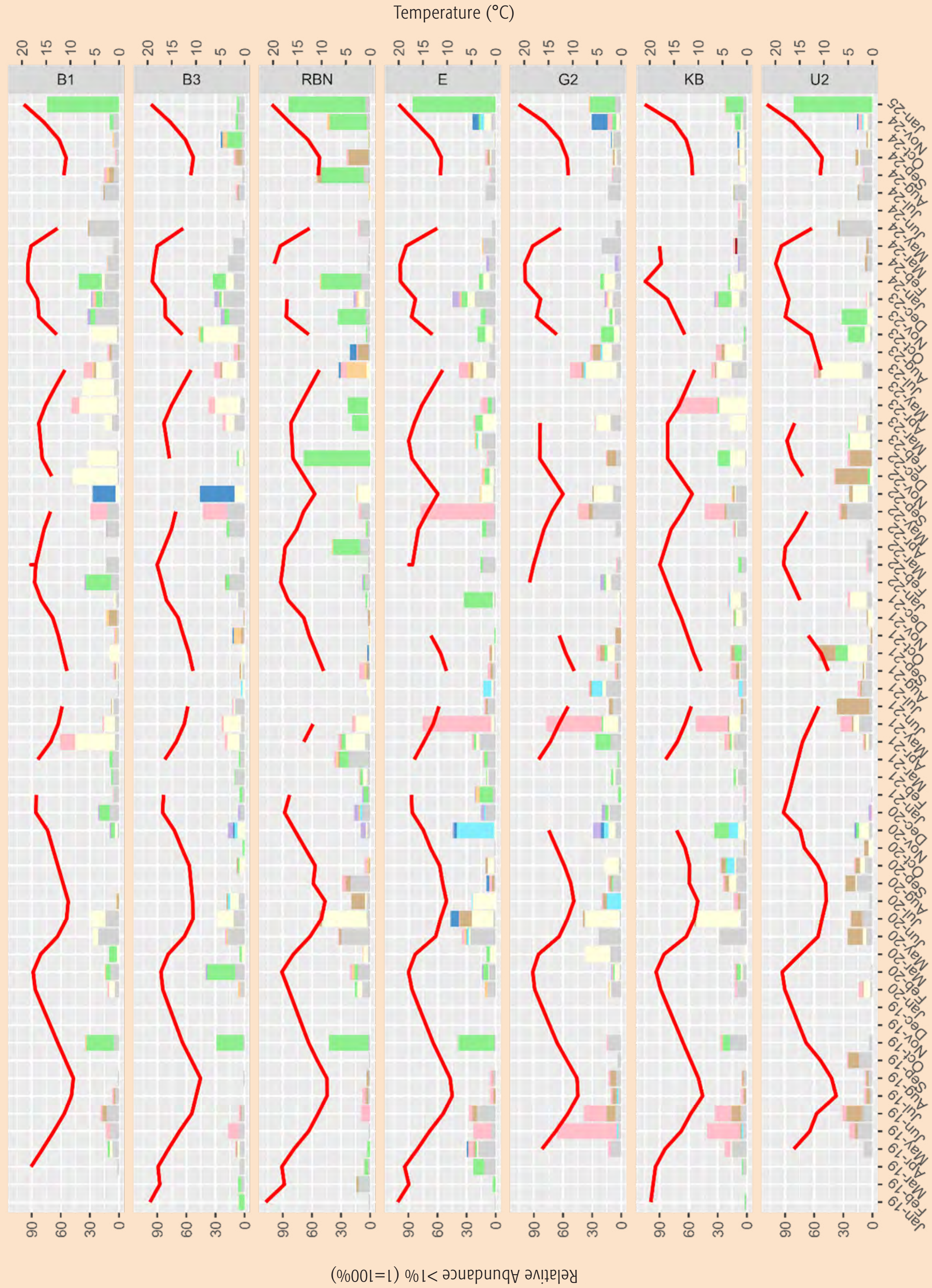
6.5.1 Dinoflagellates in the Derwent Estuary

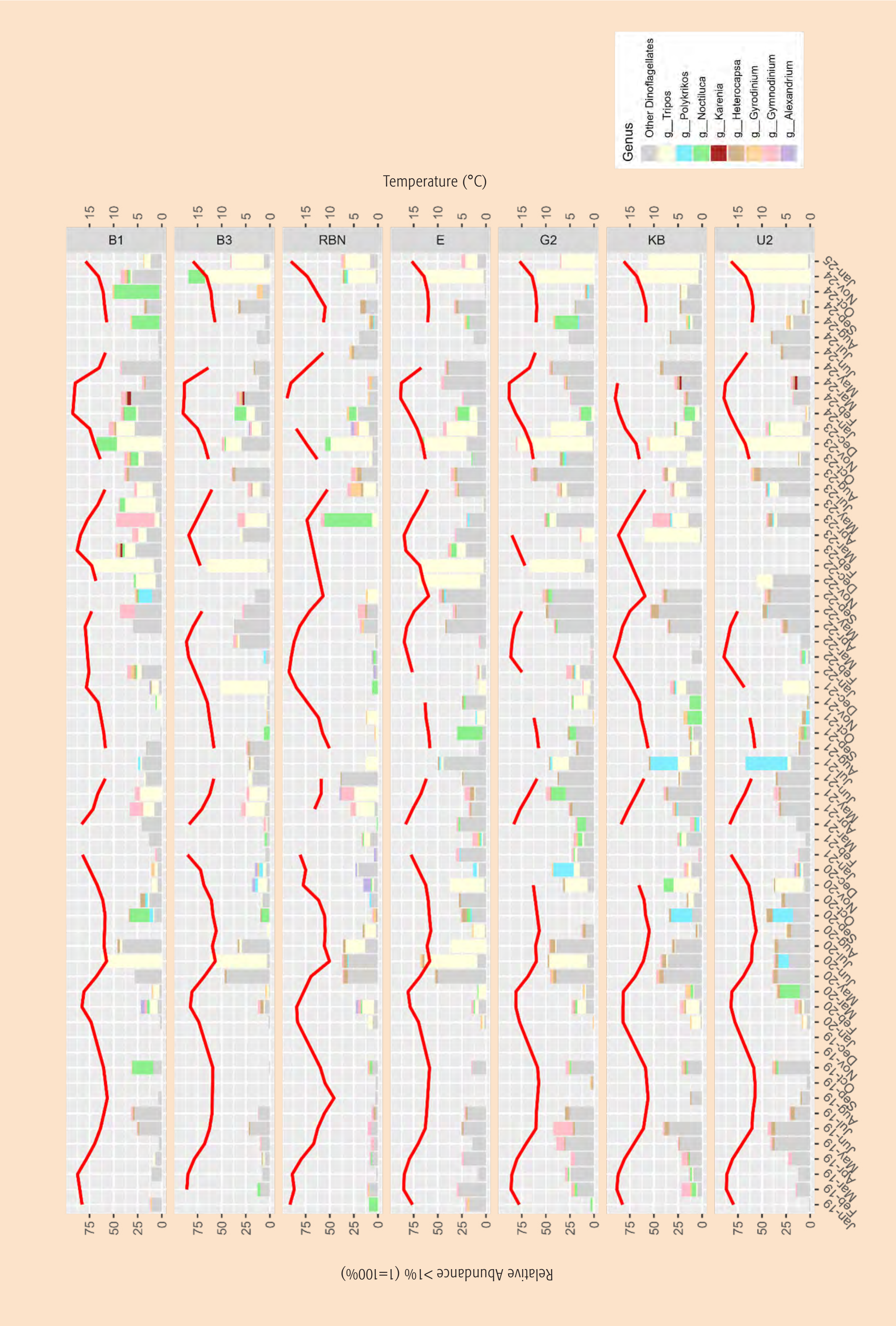
The economic impact of harmful algal blooms (HABs) on Australia's aquaculture and tourism industries can be extensive with severe blooms of toxic dinoflagellates and cyanobacteria impacting hundreds of kilometres of coastline and estuaries. The toxic dinoflagellate *Gymnodinium catenatum* was discovered in the Derwent Estuary in the mid-1980s (Hallegraeff and Sumner, 1986; Hallegraeff *et al*., 1989) and can be distributed throughout the water column during blooms or in sediments as resting stage cysts. Toxic dinoflagellates are considered a major ecological threat in the Derwent Estuary though there is no routine algal monitoring or species identifications undertaken (Whitehead, 2008). We used metabarcoding of a marker gene for small eukaryotes to investigate the distribution and relative abundance of dinoflagellate genera in water samples collected as part of the DEP Ambient Water Quality Monitoring Program from January 2019 to January 2025.

Our data demonstrate increased relative abundance of the genus *Gymnodinium* in surface waters at mid-estuary sites in autumn and early winter most years (Figure 6.36A); particularly during May and June in the middle estuary (sites E, G2, KB) where it has sometimes reached up to ~70% relative abundance of the total community. *Gymnodinium* is detected in the bottom waters, but in lower relative abundance (Figure 6.36B). We also detect sharp increases in relative abundance of another dinoflagellate, *Noctiluca* (commonly known as red tide or sea sparkle), more often in surface waters in the lower estuary, and generally in spring and summer. January 2025 saw a significant increase in relative abundance of *Noctiluca* at almost all sites, peaking at 74-85% at B1, RBN, U2 and E. This coincides with widespread media reports of a bloom of uncharacteristic magnitude in southeast Tasmania. *Noctiluca* is sometimes present in the bottom waters during a bloom, though not as prevalent. The dinoflagellate *Tripos* regularly blooms in the Derwent and was seen in samples from the summer of 2020 onwards, particularly in bottom waters where there are sharp summertime increases in relative abundance at most sites. *Tripos* has been suggested as a water mass indicator to detect environmental change (Hallegraeff *et al*., 2020).

By increasing the temporal and spatial resolution of ongoing data collection we have the potential to enhance our understanding of algal bloom dynamics through coupling eDNA observations of dinoflagellates with the physico-chemical observations collected through the DEP AWQ monitoring program.

Figure 6.36 Dinoflagellates in the Derwent Estuary – Relative abundance of Dinoflagellates at DEP Ambient Water Quality Monitoring sites, overlayed with water temperature. A) Surface waters. B) Bottom waters. eDNA was isolated from water samples collected monthly between January 2019 and January 2025. Data displayed was derived from metabarcoding of the eukaryotic 18S (v4) marker gene and aggregated at the Genus taxonomic level.




Temperature (°C)
Relative Abundance >1% (1=100%)
B1
B3
RBN
E
G2
KB
U2
Genus
Other Dinoflagellates
g__Tripos
g__Polykrikos
g__Noctiluca
g__Karenia
g__Heterocapsa
g__Gyrodinium
g__Gymnodinium
g__Alexandrium

6.5.2 Derwent Estuary Resistome

Impacts of anthropogenic pollution are widely recognised public and environmental health concerns, but there are also indirect impacts that are lesser known, for example, affecting the environmental microbiome. As a survival strategy, some microbes can acquire genes to "resist" environmental stressors, for example, the presence of heavy metals in the environment selects for heavy metal resistance genes. The Derwent Estuary is recognised as one of the most highly metal polluted estuaries in the world with levels of heavy metals (zinc, mercury, lead, cadmium, copper and arsenic) in sediments exceeding national guidelines (Australian and New Zealand Guidelines for Fresh and Marine Water Quality ANZECC Sediment Quality Guidelines SQGs; Simpson *et al*., 2013) throughout the mid-estuary. Recent studies have demonstrated a link between the presence of heavy metal resistance genes and antimicrobial resistance (AMR) genes in the environment. This may be due to several factors including the overlap of heavy metal and antibiotic waste pathways, the co-location of metal and antibiotic resistance genes (and thus simultaneous inheritance), and cross-resistance where a single mechanism confers resistance to both antibiotics and metals (reviewed in Gillieatt and Coleman, 2024). The development of AMR is recognised as a major global health threat to the public and environment. Co-selection has heightened the rate of spread and dissemination of AMR genes in the environment which, in turn has increased the emergence of multidrug resistance clinical pathogens (Salam, 2020).

We used another genomics technique, shotgun metagenomics, to determine the "resistome" of Derwent Estuary surface and bottom-water samples collected between August 2021 and May 2025 from lower and mid estuary sites (B1, E, U2). Metal resistance genes were detected in the water column in the lower estuary to middle estuary, but in generally very low relative abundance (<0.005%; Figure 6.37A). Detection was most consistent at site U2 which has higher levels of Zn in the water than either B1 or E (**Section 6.1.5**). Antimicrobial resistance genes were detected in the water column more consistently and in higher abundance across all sites and timepoints (Figure 6.37B). Abundance often peaks during late winter (August). The data offer a potential window into investigating trends in the distribution and abundance of antimicrobial resistance genes in the estuary.

6.5.3 Future Looking

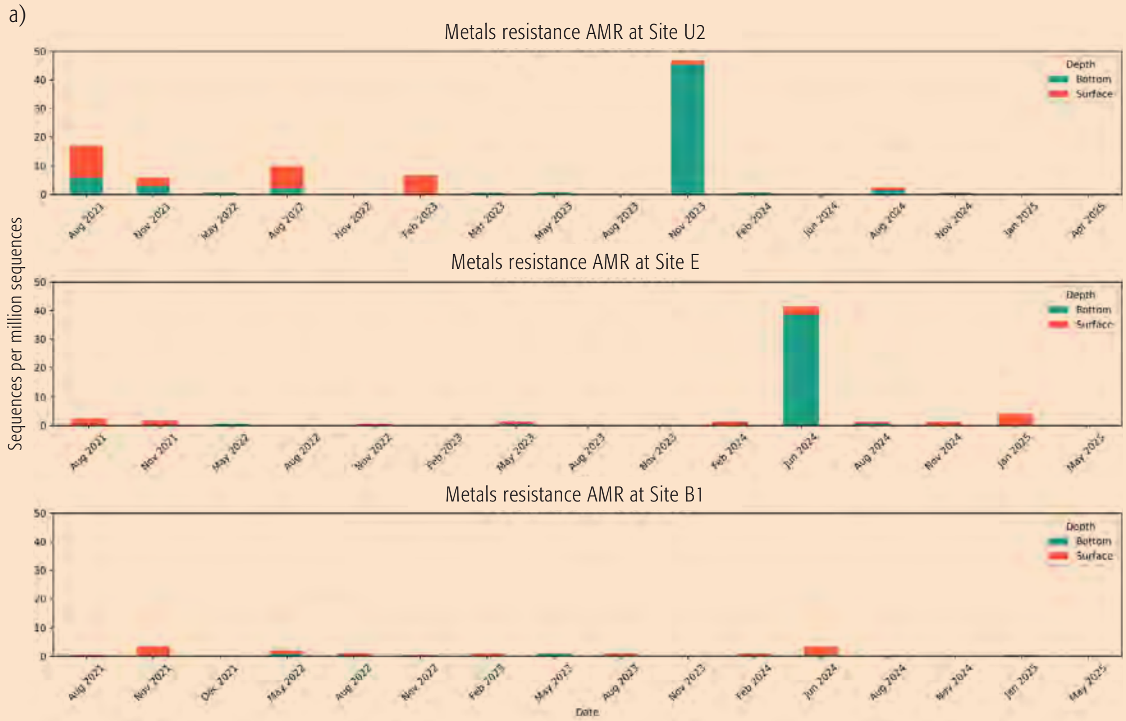
Integration of eDNA analysis into current regular monitoring programs can provide uplift to biodiversity and biosecurity monitoring given the non-invasive sampling, taxonomically holistic lens and ease of deployment. The collaboration between CSIRO, Australian Microbiome and DEP to continue building a comprehensive timeseries dataset in the Derwent Estuary represents an important resource for stakeholders to monitor biodiversity and track change.

Acknowledgment

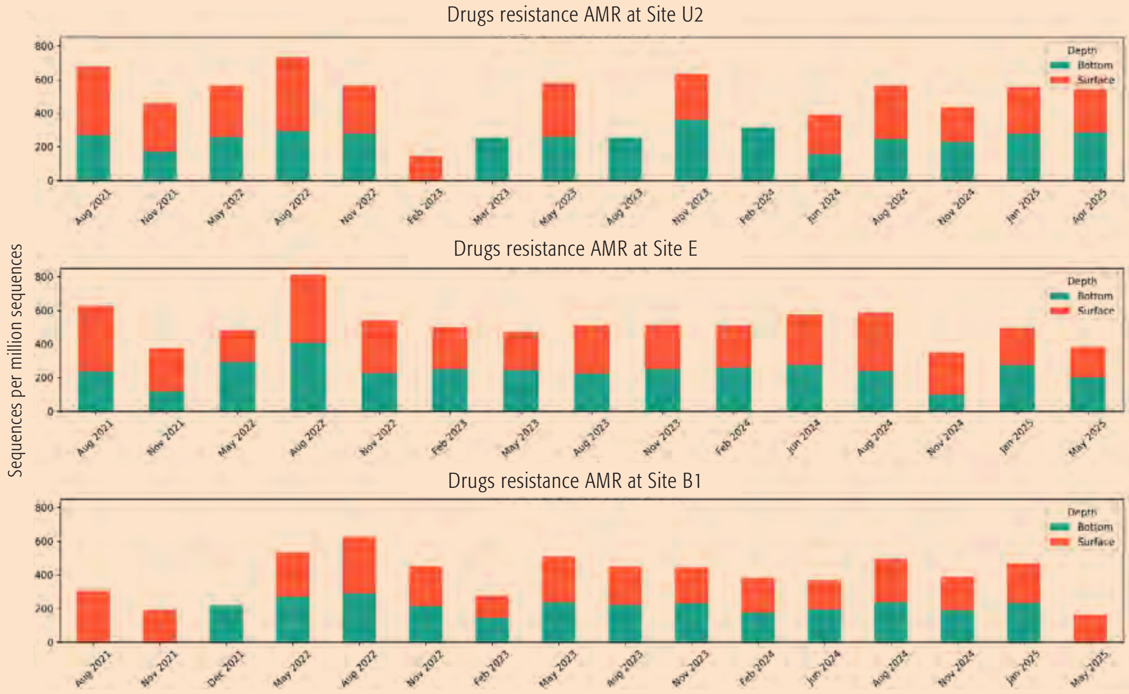
We would like to acknowledge the contribution of the Australian Microbiome consortium in the generation of data used in this publication. The Australian Microbiome initiative is supported by funding from Bioplatforms Australia and the Integrated Marine Observing System through the Australian Government's National Collaborative Research Infrastructure Strategy, Parks Australia through the Bush Blitz program funded by the Australian Government and BHP, and CSIRO.

Figure 6.37 Derwent Estuary Resistome – Relative abundance of microbial resistance genes in surface and bottom waters at DEP ambient water quality monitoring sites. a) Relative abundance of metal resistance genes. b) Relative abundance of antimicrobial resistance genes. eDNA was isolated from water samples collected between August 2021 and May 2025. Samples were selected for metagenomic sequencing representing the lower and mid estuary (B1, E, U2) and seasonal change August, November, February, May) where available. Data displayed were derived from metagenomic sequencing.

a)



b)



7

Recreational water quality

7 Recreational water quality

7.1 Pathogens and health risks

Water contaminated by sewage and animal faeces may contain pathogenic micro-organisms (bacteria, viruses, protozoa), which pose a health hazard when the water is used for primary contact recreation, such as swimming. Infection may occur by swallowing, inhaling or by direct contact of contaminated water with ears, nasal passages, mucous membranes, and cuts in the skin, which allow the pathogens to enter the body (Ministry for the Environment NZ, 2002). The most common health conditions associated with primary contact recreation in contaminated water are gastrointestinal disorders, respiratory illnesses, eye, nose and throat infections and skin disorders.

Direct detection of pathogens is not a feasible option for routine assessments since they occur intermittently and are difficult to recover from water. Thus, water samples are analysed for the concentration of more easily detected microorganisms, which may indicate the presence of pathogens, referred to as faecal indicator bacteria (DEP, 2015). In the Derwent Estuary, enterococci are sampled as the key faecal indicator bacteria, as required by the Tasmanian Recreational Water Quality Guidelines 2007 (DoH, 2007).

Enterococci results are reported as the most probable number of bacterial colonies in 100ml of water (MPN/100ml). Results are used by councils and the Department of Health to classify each site according to long term results, and to manage human exposure to a short-term pulse in poor water quality.

7.2 Sources of contamination

Key sources of faecal contamination in coastal waters include untreated sewage, faecal matter from catchments transported via stormwater systems, animal faeces, and the resuspension of contaminated sediments.

In urban areas, stormwater systems can become contaminated with sewage due to failures in the wastewater system. These failures can result from infrastructure damage, blockages caused by tree roots or wet wipes, overflows during high rainfall events, or direct cross-connections and leakages. Additionally, animal faeces, particularly from high-density aggregations of birds or dogs on beaches, can contribute significantly to contamination. Wind or wave action can also resuspend contaminated sediments, further exacerbating the issue.

Differentiating between these sources can be challenging. Regular sanitary surveys and specialist laboratory techniques, such as sterol analysis, can help identify the specific sources of contamination (DoH Tas, 2007). Systematic investigation is crucial to locate pollution sources effectively. For more information, refer to the DEP Source Tracking Framework and Toolkit (DEP, 2020b).

In the urbanised Derwent Estuary catchment, stormwater and urban rivulets are often contaminated after rainfall. This makes primary contact (swimming) near stormwater outfalls or within the estuary inadvisable for two days after heavy rain. Therefore, interpreting recreational water quality results should always consider the volume of preceding rainfall to ensure accurate assessments (DEP, 2015c).

7.3 Recreational water quality guidelines

Swimming and environmental sites in the Derwent Estuary are graded as Good, Fair or Poor. This is in accordance with the Recreational Water Quality Guidelines for Tasmania (DoH, 2007), which are largely based on the national Guidelines for Managing Risks in Recreational Water (NHMRC, 2008). The guidelines are based on aseptic grab sample analysis for the faecal indicator microbial group enterococci, and the Tasmanian guidelines adopt a three-tiered approach to classifying the long-term quality of a site based on five years of data. The tiers are:

Good: rolling 5-year 95th Hazen percentile value of < 200 enterococci MPN (Most Probable Number) 100 mL^{-1}.

Fair: rolling 5-year 95th Hazen percentile value of 200 - 500 enterococci MPN 100 mL^{-1}.

Poor: rolling 5-year 95th Hazen percentile value of > 500 enterococci MPN 100 mL^{-1}. In this case, water at these sites is considered a threat to public health in the event of primary contact recreation and local councils are required to advise the public and to erect warning signs.

In addition to long-term site classification, trigger levels have been set to manage public exposure to episodic or emerging water quality issues. If a sample exceeds 140 enterococci MPN 100 mL-1, the council is required to resample as soon as possible, and if two consecutive samples return enterococci results above 280 MPN 100 mL-1, the public must be advised directly via signage on the beach in question. This signage can only be removed by Council's Authorised Officer in consultation with DoH.

7.4 Recreational water quality monitoring program

Water quality monitoring of beaches and bays in the Derwent Estuary is coordinated by the DEP in collaboration with Department of Health (DoH), EPA Tasmania and the six councils that border the estuary (Brighton, Clarence, Derwent Valley, Glenorchy, Hobart and Kingborough). The primary objectives of the program are to coordinate monitoring, support investigations and assist councils and the DoH in managing human health risks associated with poor water quality. The DEP's role in the program is to:

- Coordinate recreational water quality (RWQ) monitoring in the Derwent Estuary.
- Compile and analyse data, including classification of beaches and bays, annual reporting and analysis of long-term trends (using methods outlined Tasmanian Recreational Water Quality Guidelines 2007 (DoH, 2007).
- Share water quality data, ratings and forecasts publicly on the DEP website.
- Support and encourage site specific investigations into poor or deteriorating water quality at targeted sites.

The water quality data is made publicly available via the DEP website and Facebook page on a weekly basis throughout the summer (December-March), to allow the community to make informed decisions as to where and when to swim. This data is also used to inform decision-making processes, by identifying stormwater and wastewater assets that require investigating.

Grab samples are collected each Tuesday by Council and the EPA/DEP throughout the Derwent Estuary, during summer and early autumn each year (from 1 December to 31 March) (Figure 7.1). Sites are categorised as either swimming sites or environmental sites, as described below, and locations are shown in Figure 7.2 .

The 19 swimming sites monitored this reporting period (2020 – 2025) are in locations where a significant number of people swim or conduct other primary contact recreation. Primary contact refers to where recreational water is used for whole-body contact, i.e., where there is a risk of swallowing water (NHMRC, 2008). These sites are sampled by councils.

The 19 environmental sites monitored this reporting period (2020–2025), sampled by either councils or EPA/DEP were selected using the following rationale:

- Bays and coves that are frequently used for secondary contact recreation and/or have foreshore parks. Secondary contact refers to incidental contact, i.e., activities where only the limbs are regularly wet and in which greater contact (including swallowing water) is unusual, such as boating and fishing (NHMRC, 2008).
- Areas with potential sources of faecal contamination.
- Sites with relatively low risk of contamination, sampled to contextualise swimming site results.
- Sites associated with major swimming events, such as the Trans-Derwent Swim.

Figure 7.1 Council staff collecting water quality samples at a local Hobart beach.

Since the previous state of the Derwent report, the following sites have been removed or added to the program.

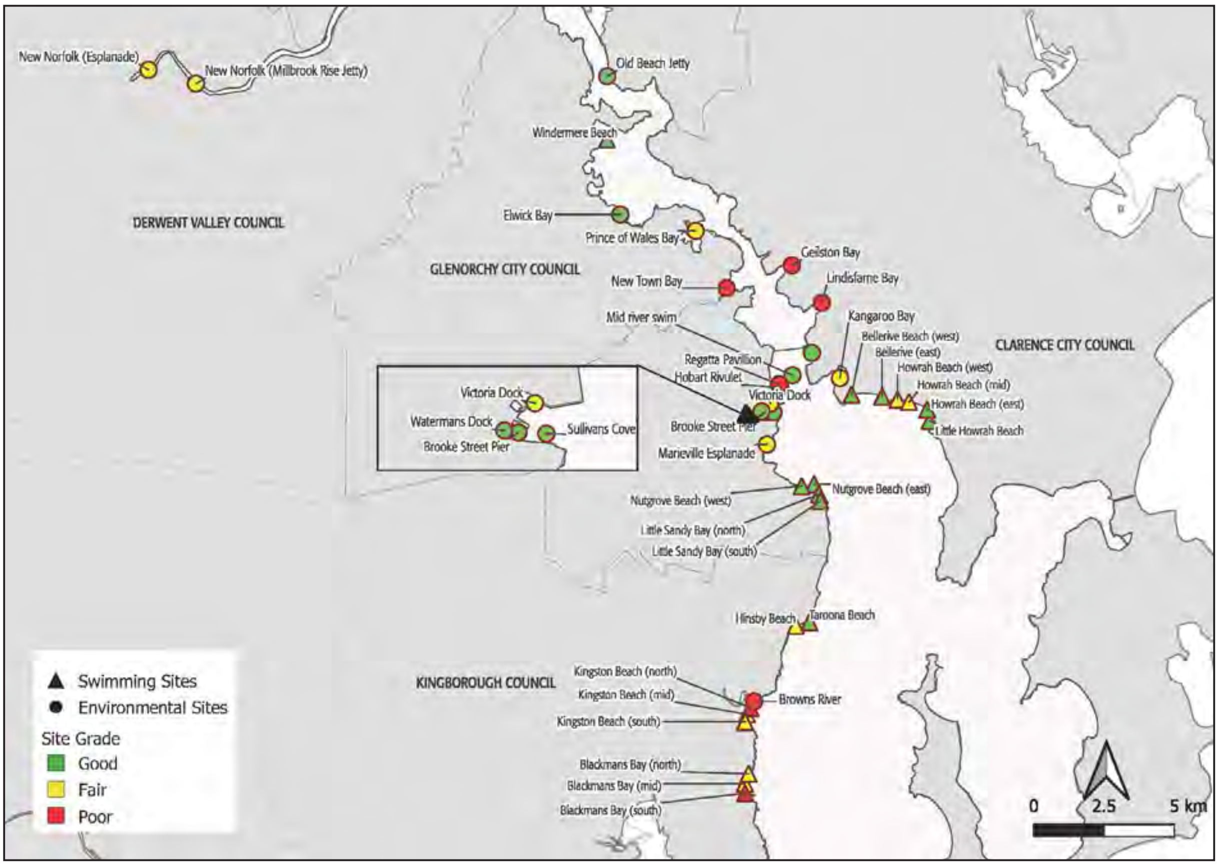
Added sites:

- Bellerive Beach East (Swimming)
- Blackmans Bay North (Swimming)
- Blackmans Bay South (Swimming)
- Kingston Beach South (Swimming)

Removed Sites:

- Mona Jetty (Environmental)
- Mona Berriedale Bay (Environmental)
- Mona Cameron Bay (Environmental)
- Cornelian Bay (Environmental)

Figure 7.2 Location of Recreational Water Quality sampling sites (swimming and environmental sites) with current water quality classification based on data collected in the summer months over the reporting period (December 2020 – March 2025).



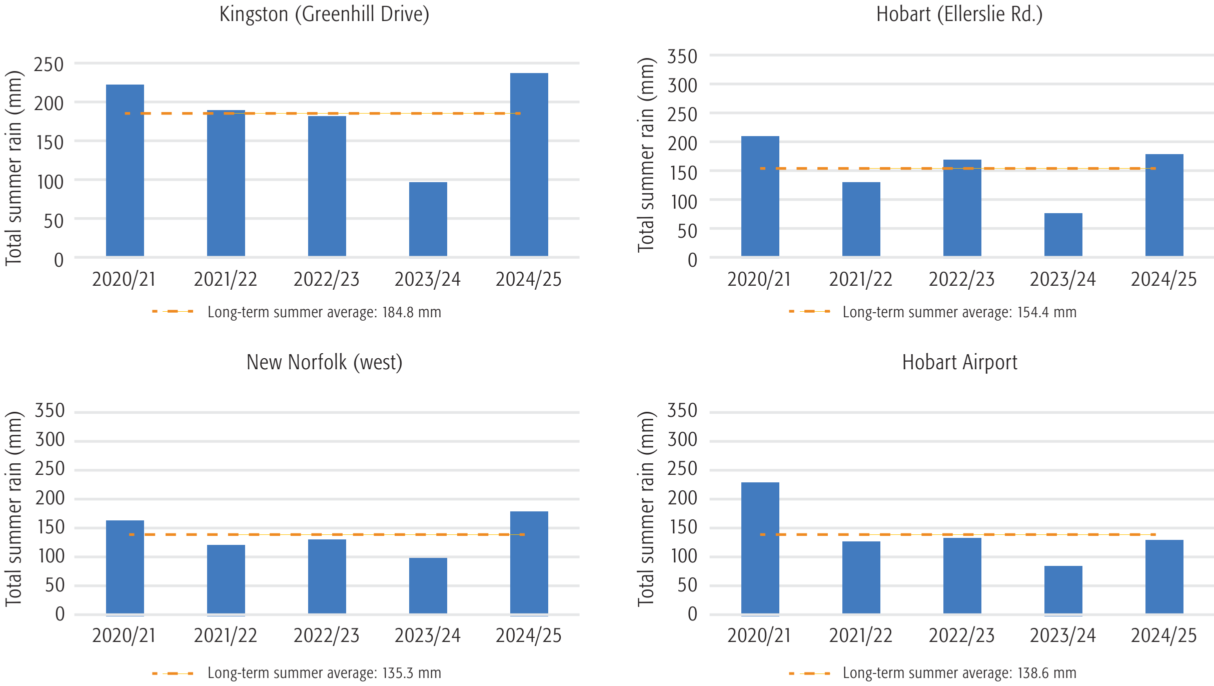
7.4.1 Status and trends

Seasonal rainfall

Given the significant impact of rainfall on recreational water quality, it is important to consider these results in context of summer rainfall across the estuary. Summer rainfall records for the past five years, at four representative rainfall station across the estuary are presented in Figure 7.3 . A five-year, RWQ-season average for each station has also been generated.

Rainfall across the estuary varies, with averages over the last five years ranging from 135 mm in New Norfolk (West) to 184 mm Kingston (Greenhill Drive). Whilst there is variation in the amount of rain recorded at each Bureau of Metrology station, general trends are consistent at all sites across the reporting period. Above-average rainfall in both 2020/21 and 2024/25 was mainly due to wetter Decembers, typically marked by a few large events, while the other months saw average rainfall. In contrast, below-average rainfall during 2023/24 was driven by drier-than-average conditions in February and March.

Figure 7.3 Total Rainfall (mm) at four weather stations in the Derwent Estuary Catchment during the last five RWQ-program seasons (between December & March), as recorded by the Bureau of Meteorology (BoM, 2025b). The RWQ rainfall average for the reporting period is indicated in red text and by a dotted line.



Swimming Sites

As shown in Figure 7.4 , recreational water quality at swimming sites has varied over the reporting period. Recreational water quality variability has seen a consistent change in both the number of Poor and Good sites year to year over the reporting period. 2023/24 had the best water quality in terms of number of exceedances (Table 7.1) coupled with an increase in the number of good sites.

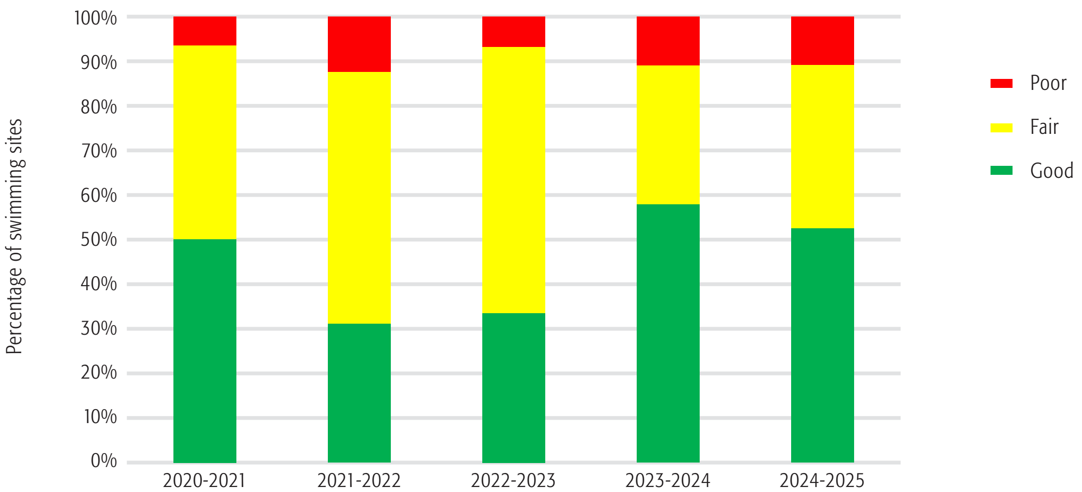
Table 7.1 Number of exceedances (values >140 MPN) for all swimming sites during each RWQ season for this State of the Derwent period.

RWQ season	Number of exceedances (>140 MPN)
2024/25	30
2023/24	16
2022/23	22
2021/22	49
2020/21	28

Compared to other seasons, the 2021/22 reporting period experienced the most degraded water quality, marked by both a high number of exceedances and a record number of sites rated as having poor water quality.

When looking at swimming site ratings over the last 5 years we see that all seasons finished with at least one site with a poor rating, suggesting the need to prioritise investigations in problematic catchments to help ascertain the cause of contamination. An example of these investigations is discussed in Section 7.5 where Clarence City Council conducted a targeted investigations program in Howrah. Good and fair ratings fluctuate the most during this period however it is worth highlighting that at the end of the current RWQ season (2024/25) there were a greater number of sites with a good rating compared to fair sites. We strongly recommend councils to use fair ratings as the catalyst to commence water quality investigations to help identify problems to prevent further decline in water quality that may lead to a poor water quality rating.

Figure 7.4 Proportion of swimming sites graded as 'Good', 'Fair', and 'Poor' in the last five RWQ seasons. Proportions are based only on those sites with five years of data.



Over the reporting period significant changes include:

All City of Hobart sites rated "Good" for the last two seasons.

Improvement in water quality at Howrah Beach Middle following infrastructure rectification (discussed in Section 7.5).

Windermere Beach continues to be one of the most consistent good, rated swimming sites within the Derwent.

Rainfall at swimming sites

An assessment of the relationship between enterococci results and recorded rainfall data has been conducted for the entire reporting period. This is an extension on the analysis that is included in the annual RWQ reports. This assessment includes all enterococci samples collected across the swimming sites over the reporting period (2020–2025), a total of 1616 samples. Results are separated into two groups:

- Group 1. Enterococci results < 140 MPN 100 ml $^{-1}$: 1471 samples.
- Group 2. Enterococci results > 140 MPN 100 ml $^{-1}$: 145 samples.

These two groups were separately assessed for a possible response to rainfall (Figure 7.5). Rainfall data was used from the two local BoM stations covering the swimming sites, with records for the 24 hours prior to 9 am on the day of sampling. Rainfall occurring after 9 am on the day samples were collected, as well as rainfall from previous days, was not included in this assessment. However, both could have significantly influenced beach water quality by contributing to runoff or other environmental changes.

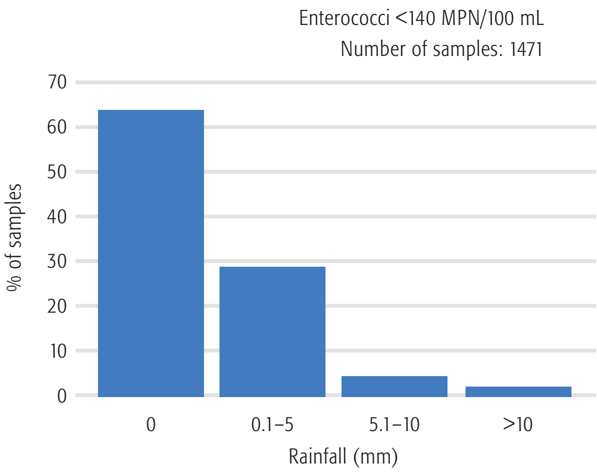
Group 1 (enterococci < 140 MPN):

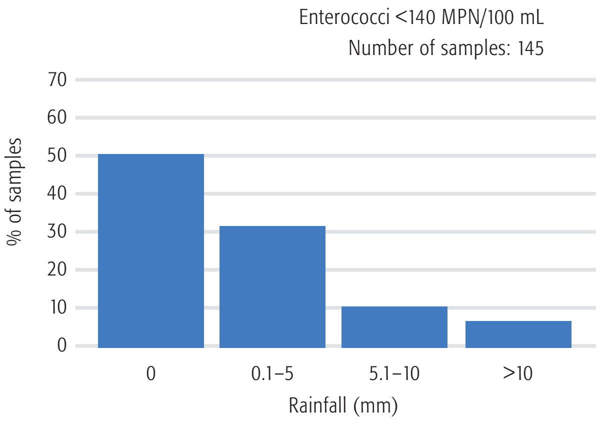
- 1471 samples.
- 64 % of the enterococci results (< 140 MPN 100 ml $^{-1}$) occurred when no rain fell in the preceding 24 hours.
- 29 % of results occurred on days when the total rainfall in the preceding 24 hours was > 0 and < 5 mm.
- 5 % of results occurred on days when the total rainfall in the preceding 24 hours was between 5.1 and 10 mm.
- 2 % of results occurred on days when the total rainfall in the preceding 24 hours was greater than 10 mm.

Group 2 (enterococci > 140 MPN):

- 145 samples.
- 51 % of high enterococci values (> 140 MPN 100 ml $^{-1}$) occurred when no rain fell in the preceding 24 hours.
- 32 % of high enterococci values occurred on days when the total rainfall in the preceding 24 hours was > 0 and < 5 mm.
- 10 % of high enterococci values occurred on days when the total rainfall in the preceding 24 hours was between 5.1 and 10 mm.
- 7 % of results occurred on days when the total rainfall in the preceding 24 hours was greater than 10mm.

Figure 7.5 Proportion of enterococci sample results <140 MPN 100 ml (upper) and > 140 MPN 100 ml (lower), matched with rainfall data recorded on sampling day, from two BoM stations across the estuary. Graphs include all samples collected at swimming sites during the reporting period (2020 – 2025).





As Figure 7.5 shows of the 1616 swimming site samples collected during this reporting period (2020–2025), 91% of enterococci results were < 140 MPN 100 ml $^{-1}$¬ (1471 samples). Analysis of the enterococci results suggests that lower rainfall can lead to better results. However, it should be noted that with dry weather exceedances are likely to come from another external source. There can be numerous reasons for dry weather fails, including sewage cross-connection, sewage spill, sewer leak, residential or business discharge, as well as swell and high winds resuspending sediments. The higher prevalence of external pressures such as dry weather exceedances suggests more targeted investigation is required to address infrastructure issues.

Environmental Sites

Water quality is also monitored weekly during the RWQ season at several environmental sites throughout the Derwent Estuary. These sites whilst not classified as swimming sites are regularly visited by the community for other activities including boating, kayaking and walking, thus, understanding the water quality is important.

A number of these sites are also home to events which take place in the estuary including the Regatta Day swim.

Table 7.2 shows that the number of exceedances (samples >140 MPN) whilst high has been consistent over the current reporting period. The exception to this is season 2023/24 which was dryer on average compared to the previous seasons as discussed in **Section 7.4.1**.

Table 7.2 Number of exceedances (values >140 MPN) for all environmental during each RWQ season for this State of the Derwent period.

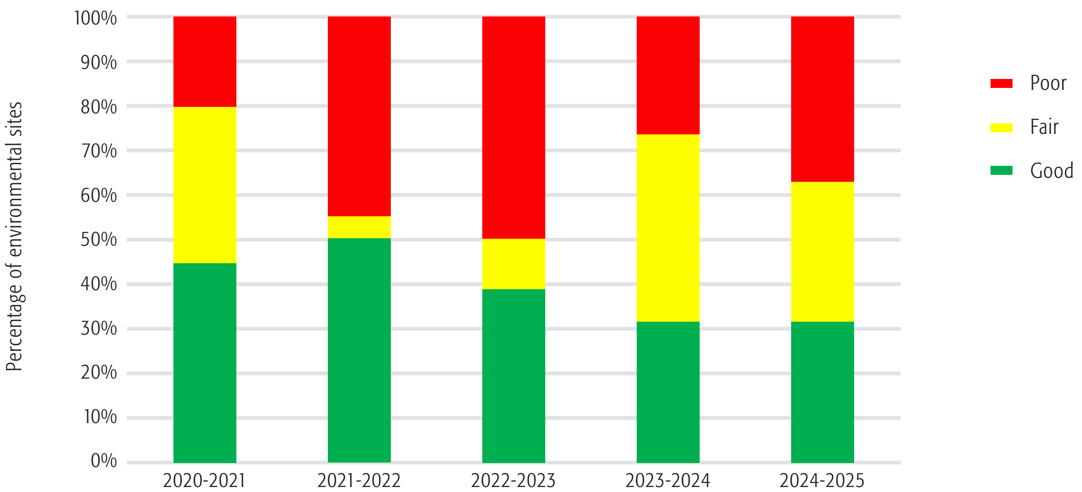
RWQ season	Number of exceedances (>140 MPN)
2024 – 2025	43
2023 – 2024	24
2022 – 2023	56
2021 – 2022	51
2020 – 2021	40

When looking at environmental site ratings over the last 5 years we see all seasons finished with a significant number of poor sites. These ratings may be influenced by the fact that many of the sites are situated in already degraded bays across the Derwent, where ongoing urbanisation contributes to continued contamination. Good and fair ratings fluctuate season to season but at the end of the 2024/25 program, good and fair sites were equally represented (Figure 7.6). It is proposed that a greater emphasis be placed on improving environmental water quality in the Derwent Estuary.

Over the reporting period significant changes include:

- New Norfolk Esplanade was reclassified as an environmental site as council located a stormwater outfall within the swimming site.
- Sampling of Environmental Sites (Jetty, Berridale and Cameron Bay sampled by Mona) and Cornelian Bay (sampled by the City of Hobart) ceased during this reporting period.
- Mid River Swim remains the environmental site with the best water quality in the RWQ program.

Figure 7.6 Proportion of environmental sites graded as 'Good', 'Fair', and 'Poor' in the last five RWQ seasons. Proportions are based only on those sites with five years of data.



7.4.2 Summary and recommendations

Recreational water quality increased in prominence over the last five years with significant interest from the community and local councils about access to sites and improving water quality further highlighting the growing demand for a safe and accessible recreational environment. With this increased exposure, we propose several recommendations:

- Actively investigate sites with a fair water quality rating to identify causes of degraded water quality to prevent a rating downgrade.
- Water quality at environmental sites not typically used for swimming should be flagged as a concern if it deteriorates, with appropriate time and resources allocated to investigate issues.
- Investigate options to provide beach water quality information for winter swimming, which has noticeably increased with the rise of cold-water swimming groups in Hobart; it is also promoted as part of Discover Tasmania's Off-Season tourism campaign.

7.5 Case studies

7.5.1 Water quality forecasting in the Derwent

Forecasting pollution levels at popular swimming sites is a new initiative in Tasmania, though it's widely used elsewhere and supported by the World Health Organisation. It complements regular enterococci sampling by offering timely, up-to-date advice.

Since the 2022/23 RWQ season, forecasting has been trialled and is now part of the Beach Watch program. Key drivers include reducing the delay between sampling and public use, improving communication for beaches with poor long-term ratings, enabling rapid alerts for sewage spills, empowering public decision-making, and supporting large-scale events with current water quality data.

Forecasting Methodology

To develop a water quality forecasting program tailored to Hobart and the Derwent Estuary, we drew on successful models from Western Australia, New South Wales, and Victoria. These programs assess pollution likelihood based on rainfall. Central to our approach was the use of the Enterotester model (DoH WA, 2011), which helped establish site-specific pollution thresholds for each swimming location in the Derwent. This allows for tailored assessments and reflects improvements in individual catchments.

We compiled five years of enterococci and rainfall data for each Beach Watch site, aligning with long-term rating calculations. This data was uploaded into the Enterotester template, which calculated high and low pollution thresholds and recovery times. The model uses 95th percentile calculations to estimate gastroenteritis risk (GI risk), based on national and international guidelines (NHMRC, 2008; WHO, 2021). In Tasmania, we adapted these guidelines to define water quality categories: Good (<5% GI risk), Fair (5–10%), and Poor (>10%), consistent with the Hazen percentile method used in long-term grading.

Figure 7.7 Table 5.7 from the NHMRC Guidelines – Basis of derivation of percentile values for determining microbial water-quality assessment categories. GI – Gastrointestinal, AFRI – Acute febrile respiratory illness (not used for forecasting)

Category	95th percentile value for intestinal enterococci/ 100 mL (rounded values)	Basis of derivation	Estimation of probability
A	<40	This value is below the NOAEL in most epidemiological studies.	GI illness risk:< 1% AFRI risk:< 0.3% The upper 95th percentile value of -40/ 100 mL relates to an average probability of less than one case of gastroenteritis in every 100 exposures. The AFRI burden would be negligible.
B	41-200	The 200/100 ml value is above the threshold of illness transmission reported in most epidemiological studies that have attempted to define a NOAEL or LOAEL for GI illness and AFRI.	GI illness risk: 1-5% AFRI risk: 0.3-1.9% The upper 95th percentile value of 200/ 100 mL relates to an average probability of one case of gastroenteritis in 20 exposures. The AFRI illness rate would be 19 per 1000 exposures or approximately 1 in 50 exposures.
C	201-500	This represents a substantial elevation in the probability of all adverse health outcomes for which dose-response data are available.	GI illness risk: 5-10% AFRI risk: 1.9-3.9% This range of 95th percentile values represents a probability of 1 in 20 to 1 in 10 risk of gastroenteritis for a single exposure. Exposures in this category also suggest a risk of AFRI in the range of 19-39 per 1000 exposures or a range of approximately 1 in 50 to 1 in 25 exposures.
D	>501	Above this level there may be a significant risk of high levels of illness transmission.	GI illness risk: > 10% AFRI risk: > 3.9% There is a greater than 10% chance of illness per single exposure. The AFRI illness rate at the guideline value of 500 enterococci per 100 mL would be 39 per 1000 exposures or approximately 1 in 25 exposures.

Thresholds from Enterotester were then integrated into our in-house Forecaster tool, which generates daily forecasts using a simple messaging system: Unlikely (Good), Possible (Fair), and Likely (Poor). This approach mirrors NSW's communication style, offering clarity and ease of understanding for the public.

Additionally, TasWater provides daily updates on potential sewage impacts, allowing manual overrides of forecasts when necessary. This ensures timely and accurate communication, empowering the public to make informed decisions about swimming site suitability.

Forecasting Results

Throughout both the initial trial and ongoing implementation, we evaluated our results by comparing the forecasting advice provided with the actual enterococci results collected. Central to this evaluation are the criteria that define when advice is considered Appropriate, or when False Alarms and Missed Alarms occur. These criteria, outlined below, are based on methodologies used in both Victoria and New South Wales.

Table 7.3 Metrics of how the forecasts are assessed, based on the same methods as used in New South Wales (NSW) and Victoria (VIC).

Metric	Result
Appropriate advice	If microbial water quality is good (< 140 MPN), and our report forecasted Unlikely or Possible, OR If microbial water quality was elevated (>140 MPN) and our report forecasted Possible or Likely.
False alarm (type 1 error)	When we forecast Likely, and water quality is good (<140 MPN).
Missed alarm (type 2 error)	When we forecast Unlikely, and the pollution level is elevated (>140 MPN).

This approach has allowed us to assess the accuracy of the forecasts each year to determine the appropriateness of this program, with the aim of providing appropriate advice at least 85% of the time as is seen in both New South Wales and Victoria. To date, we have seen appropriate advice given in excess of 85% for all years besides the limited scope initial trial Table 7.4 . The success of this trial and subsequent roll out has enabled forecasting to become an integral part of our RWQ program as we lead the way in this space across Tasmania.

Table 7.4 Accuracy of advice summary based upon confirmed enterococci samples during 2 Derwent trial (2022/23 & 2023/24) and following roll out 2024 –2025. *Initial trial year used reduced swimming sites (12 in total) hence the skewed accuracy.

Accuracy of Advice	Percent %		
	2022/23*	2023/24	2024/25
Appropriate advice	79.8	92.8	86.5
False Alarm	16.25	4.7	7.6
Missed Alarm	3.94	2.8	5.7

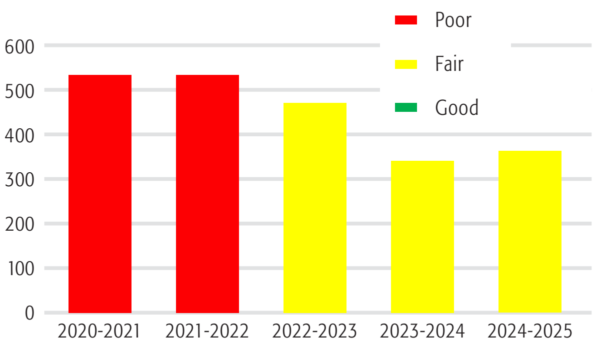
7.5.2 Improvements in water quality at beaches

Poor-rated swimming sites have long existed throughout the Derwent Estuary, with only sporadic efforts made in the past to address water quality issues. However, over the current five-year period, improving beach water quality has become a higher priority. Councils are now actively investigating pollution sources and collaborating to find solutions. Despite this progress, identifying the causes of degraded water quality can be complex and confusing for stakeholders.

Two key examples illustrate the varied outcomes of these efforts. At Howrah Beach Middle, water quality declined to a Poor rating in 2020/21, prompting Clarence City Council to launch extensive investigations across the Howrah catchment. The council secured additional resources to identify and fix over 50 infrastructure defects, mainly involving cross-contamination between sewer and stormwater systems. Supporting initiatives included a beach sediment survey, an educational video, and a gross-pollutant-trap audit. Most of this work was concluded in 2023, with ongoing investigations continuing as needed.

Improvements at Howrah Beach Middle took time to show results. In 2022/23, the beach returned to a Fair rating, with all five Poor results occurring before the rectification works. Since then, the site has maintained a Fair rating and is trending positively (Figure 7.8).

Figure 7.8 Long term rating for Howrah Beach Middle over the current State of the Derwent reporting period (2020 – 2025) showing an improvement in water quality at the site.



In contrast, Blackmans Bay South in Kingborough has faced persistent challenges. Despite being the first council in the Derwent to invest heavily in understanding poor water quality, the site's rating has consistently declined from 2020 to 2025 (Figure 7.9).

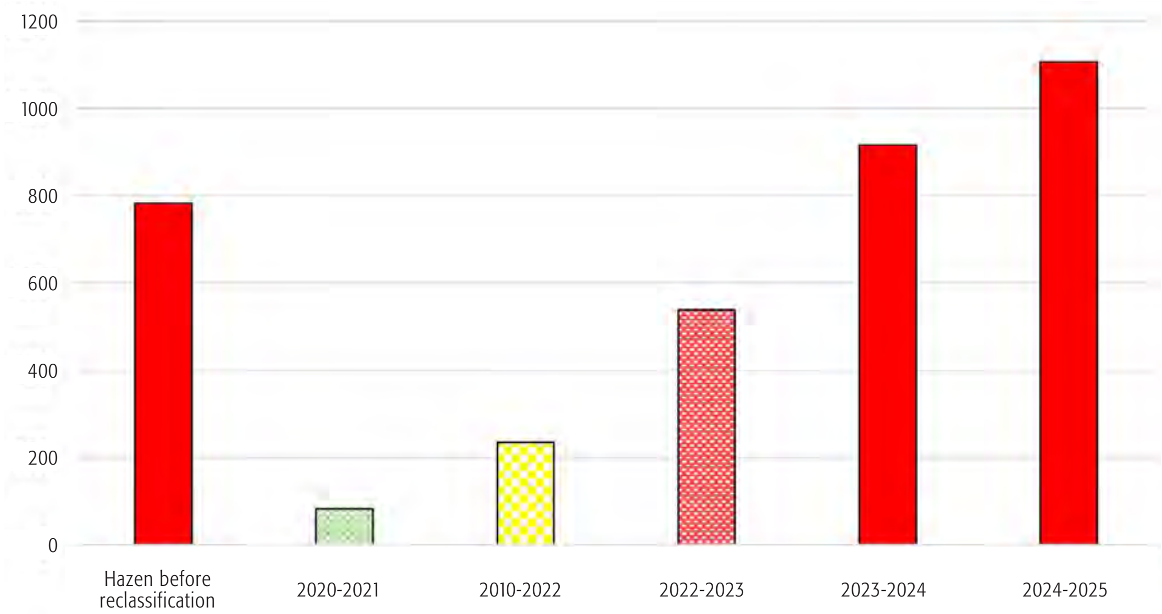
Kingborough Council allocated resources to conduct detailed investigations, including:

- Engaging a Stormwater Investigation Officer
- Visual inspections of stormwater infrastructure
- Installation of stormwater diversions
- Fluorescein dye testing
- Infrastructure upgrades
- Collaboration with TasWater by prioritising rectification and interventions of the sewer network
- Undertaken additional sampling across the year to understand trends
- Engagement of external consultants to undertake specialised investigations

Despite these efforts, the site recorded 10 Poor results over the reporting period. During the period from 2020 to 2023, there were insufficient data points to assign a formal long-term rating. This was due to the site being reclassified and having fewer results than required for a five-year rating. Despite this, Council continued to actively investigate the site, with regular monitoring taking place throughout the period.

Kingborough Council continues to invest in understanding these complex sources, helping to fill knowledge gaps and challenge traditional assumptions about water quality drivers. These examples highlight both the progress and ongoing challenges in improving swimming conditions across the Derwent Estuary.

Figure 7.9 Long term ratings for Blackmans Bay South during the reporting period for this State of the Derwent report (2020 – 2025). Ratings from 2020/21, 2021/22 and 2022/23 are highlighted as these sites had insufficient data points for a formal long-term rating. Hazen before reclassification utilises the long-term rating from 2018/19 – the final season prior to reclassification.



7.5.3 Beach Watch field assessment of *Bacteroides dorei* test

This summer, we partnered with ZiP Diagnostics to assess their field-deployable *Bacteroides dorei* test – a highly specific human faecal indicator bacterium (FIB) – in parallel with routine Enterococci testing. Its key strength lies in its ability to distinguish human faecal contamination from other sources, such as livestock, birds, native marsupials or domestic animals.

The assessment was conducted in two distinct environments:

- Twelve Beach Watch sites within Kingborough Council; and
- Three stormwater outfall sites discharging into the Derwent Estuary.

Over an 8-week sampling period (4 February – 25 March), 48 water samples were collected from the beach sites. Of these, only four tested positive for *Bacteroides dorei*, all corresponding to Enterococci levels below the Tasmanian recreational water retest threshold (>140 MPN/100 mL). Conversely, there were 12 samples where Enterococci exceeded the threshold–some above 500 MPN/100 mL–but the Bacteroides test returned negative results. This suggests that the elevated Enterococci at beach sites could be due to non-human faecal sources such as wildlife or possible dilution within the estuary.

In contrast, the stormwater samples told a different story: all three stormwater sites that tested positive for Bacteroides dorei also showed elevated Enterococci levels. This correlation indicates that human faecal contamination is possibly the primary contributor to microbial pollution in those systems.

Taken together, the results demonstrate that the *Bacteroides dorei* test is a promising tool for rapidly identifying human faecal contamination in stormwater systems, enabling local councils and regulators to trace sources more efficiently and prioritise remediation (Figure 7.10). While best used alongside other methods, this approach provides valuable insights for beach monitoring, particularly in identifying human sources of faecal contamination. By incorporating complementary tools to detect non-human sources, this approach enables a more comprehensive assessment of recreational water quality–particularly useful for targeted investigations of human pollution in stormwater and catchment areas.

Figure 7.10 Water quality testing with council staff.

8

Contamination in fish and shellfish

8 Contaminants in fish and shellfish

8.1 Southern Sand Flathead, deployed Pacific Oyster and wild shellfish monitoring

8.1.1 Background

Timtumili Minanya/River Derwent, including the Derwent Estuary, contains some of the highest recorded concentrations of zinc, lead, cadmium and mercury in sediments compared to other estuaries globally (Macleod and Coughanowr, 2019). While historical industrial activity remains the primary source, ongoing contributions from fugitive dust, diffuse groundwater inflows, and point-source discharges from current operations may continue to influence both bound and soluble metal levels in the estuary. Historic and current metal sources and water-column concentrations are discussed in sections 4.3 and 6.1.5, respectively.

Aquatic organisms have the capacity to bioaccumulate metals, creating potential ecological and health risks, if they are consumed by humans. Consequently, biomonitoring programs often employ these organisms as bioindicators to evaluate human health hazards and track shifts in metal bioavailability over time (Stankovic et al., 2014). Shellfish and finfish collected from the Derwent Estuary have been assessed since the 1970s when oysters from a shellfish farm in Ralphs Bay caused severe vomiting in consumers due to high metal concentrations. Seminal studies by Thrower and Eustace (1973), Ratkowsky et al., (1974) and Bloom (1975) documented elevated levels of zinc, cadmium, lead and mercury in farmed oysters and later in fish (Ratkowsky *et al.*, 1975). More recent studies have looked at the effects of contamination on macroalgae (Farias *et al.*, 2018; Fowles *et al.*, 2018), sessile invertebrates (Fowles *et al.*, 2018), fish (Jones *et al.*, 2014) and birds (Einoder *et al.*, 2018).

Comprehensive and continuous biomonitoring has been undertaken by the current and former operators of the Hobart zinc smelter. This program has included mercury monitoring in flathead since 1984, wild oyster and mussel monitoring since 1994, and deployed oyster cage experiments since 2004. Southern Sand Flathead (*Platycephalus bassensis*) was selected as a preferred species for biomonitoring as they can provide an understanding of both ecological and human health risk throughout the estuary i.e. they reside permanently in the estuary, have strong site fidelity, are relatively sedentary bottom feeders (Tracey *et al.*, 2011; Verdouw *et al.*, 2011), are ambush predators that consume smaller fish and other benthic biota and are highly sought after by recreational fishers.

In contrast, while the Pacific Oysters (*Crassostrea gigas*) and Blue Mussels (*Mytilus edulis*) also provide information on both ecological and human health risk, they are sessile filter feeders and as such reflect site-specific bioavailability in the water column, providing an opportunity to assess changes in bioavailability at a given location over time with deployed shellfish and health risks with wild harvested shellfish.

This study aims to evaluate current metal concentrations in selected biota, compare concentrations to The Food Standards Australia New Zealand (FSANZ) trigger levels to inform public health recommendations, and assess spatial and temporal trends in mercury and other environmental data. The following sections present methods and findings for the following biomonitoring programs: deployed Pacific Oysters, wild Pacific Oysters and Blue Mussels, and Southern Sand Flathead.

Methods

Sample collection and lab analysis

All monitoring was conducted by Nyrstar Hobart (DEP, 2020a) and all samples were analysed at Analytical Services Tasmania (AST) (Analytical Services Tasmania, 2025).

Deployed Pacific Oysters

Farmed Pacific Oysters of the same known age were sourced from uncontaminated and disease free shellfish growing areas around Tasmania to ensure baseline conditions below safety standards. In 2004, oysters were sourced from Barilla Bay, then Little Swanport from 2005–2015. From 2016–2024, oysters were sourced from Circular Head near Smithton, a site free of Pacific Oyster Mortality Syndrome (POMS), to reduce the risk of spreading POMS to local aquaculture operations in Mickeys Bay, D'Entrecasteaux Channel, South Bruny Island.

A minimum of 20 individuals were analysed prior to estuarine deployment to give a baseline for metal accumulation at source locations. Thirty-two oysters were placed into oyster baskets and deployed into surface waters at nine sites in the middle estuary and one control site (Mickeys Bay) (Figure 8.1). One basket was deployed into mid-water and one just off the seabed at each of the following sites: Elwick Bay Pavilion, the Nyrstar Hobart wharf and Beltana Beacon. Baskets were deployed in December and retrieved in January, as per previous surveys. The oysters were secured sub-tidally to existing structures.

Upon retrieval, oysters were shucked and placed on ice in the field and transported directly to a freezer. Oysters from each basket were combined, homogenised and analysed as a single sample at AST.

Baseline-corrected data was calculated by subtracting concentrations measured in non-deployed oysters from post-deployed oyster concentrations.

Wild Pacific Oysters and Blue Mussels

Wild Pacific Oysters and Blue Mussels were collected triennially from the Derwent Estuary and D'Entrecasteaux Channel. Wild oysters were sampled from 26 locations and mussels from 30 locations. Sites are categorised into four Derwent Estuary regions (middle estuary, eastern shore, western shore, and Ralphs Bay), a reference region (D'Entrecasteaux Channel) and a control site (Mickeys Bay) (Figure 8.1).

Twenty oysters and twenty mussels were randomly collected from each site and composited to give a single sample per location. Samples were shucked, rinsed in deionised water then placed in a labelled sampling bag and frozen prior to submission.

Southern Sand Flathead

Wild Southern Sand Flathead were sampled in spring, targeting up to 10 fish above legal size (>350 mm) from seven locations in the Derwent Estuary, 20 fish were collected from a control site at Mickeys Bay in the D'Entrecasteaux Channel (Figure 8.1). In November 2023, the legal catch size of Southern Sand Flathead was increased from 320mm to 350mm. Southern Sand Flathead were sampled annually from 2002 to 2015 and biennially from 2016 to 2024.

Fish were measured, weighed, gutted, skin removed, rinsed in deionized water, and submitted to AST. Skins were removed because consumers typically discard them before consumption; including skin in the analysis would therefore overestimate exposure and artificially elevate the perceived risk to consumers. In 2018, fish skins were not removed prior to analysis resulting in elevated estimates of metal concentrations, consequently, 2018 data were excluded from analysis.

Laboratory Analysis

Shellfish were blended and homogenised prior to subsampling for analysis. A random ~10–20 g portion of skinless flesh was taken from each flathead fillet for metals and mercury analysis.

For mercury analysis, ~1 g of wet sample was digested in concentrated nitric and sulfuric acid for 3 hours at 97 °C. The digest was then analysed using cold vapour atomic fluorescence spectrometry (CV-AFS). Prior to metals analysis a portion of sample was dried at 104 °C. The dried sample was ground and a representative ~1 g subsample digested in aqua regia for 2 hours at 95±4 °C. Analysis for cadmium, copper, lead, zinc (and arsenic for fish only) was performed by inductively couple plasma atomic emission spectrometry (ICP-AES).

Results for both metals and mercury were reported in mg/kg on a wet matter basis.

Data handling and analysis

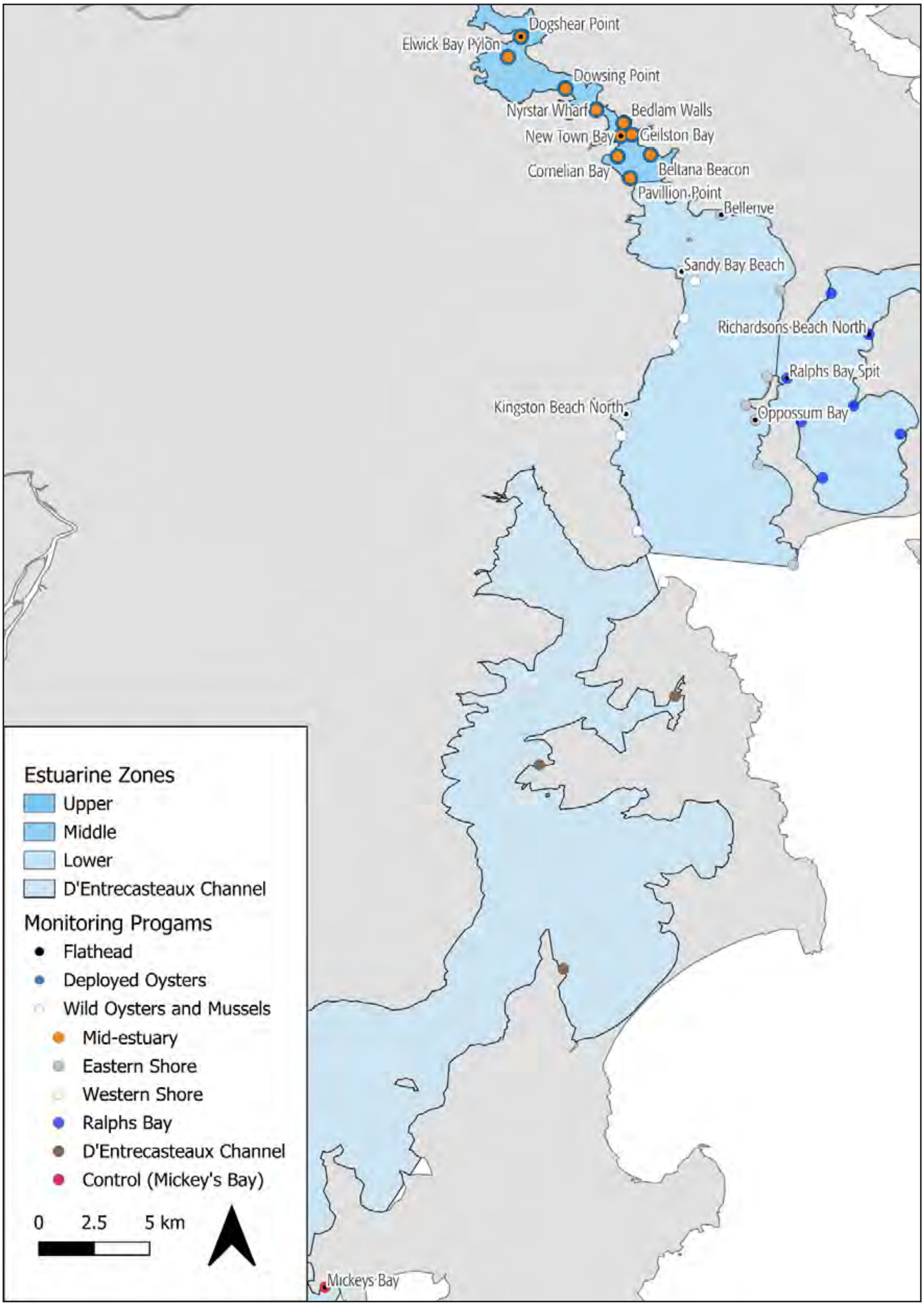
For assessment against FSANZ trigger levels and spatial variation, data for the reporting period (2020–2025) was analysed. For temporal trend analysis, data for the monitoring period was utilised (2002–2024).

Metal concentrations in Pacific Oysters, Blue Mussels and Southern Sand Flathead were displayed using box plots which show the minimum, 20th percentile, median, 80th percentile and maximum values across control, reference and estuary sites for the monitoring period. These visualisations support spatial comparisons and assessment against FSANZ guidelines (2024), including maximum permitted levels and generally expected levels, to inform public health recommendations.

Statistical time-trend analysis was performed in R version 4.5.1 using the HARSAT (Harmonised Regional Trend Analysis Tool) package version 1.0.3 (https://harsat.amap.no/). HARSAT is applied for data analysis by the Arctic Monitoring and Assessment Programme (AMAP), the Baltic Marine Environment Protection Commission (HELCOM) and the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) for assessment of data concerning contaminants and their effects in the marine environment. A detailed description of the statistical methodology can be found in the AMAP Human Health Assessment (AMAP, 2022). The use of this tool came about from the DEP's contribution to the Open-Ended Scientific Groups (OESG) evaluation of the Minamata Convention (see 8.2). Presented below are preliminary trend analysis results. The wild Pacific Oyster and Blue Mussel datasets were excluded from trend assessment as they are monitored solely for human health risk evaluation.

Metal concentrations were log-transformed and changes in the log concentrations over time were modelled using linear mixed models. When there were more than 15 years of data, both a linear model and smoothers (2,3 and 4 degrees of freedom) were fitted. Model selection was based on the lowest Akaike Information Criterion. The primary metric used to summarise changes over time was the change in total metal concentrations from 2002 to 2024, with significance assessed at the 5% level. The significance of each fitted trend–linear or non-linear–was assessed using likelihood ratio tests. Further methodological details for HARSAT are available in AMAP (2022) and Morris et al. (2022).

Figure 8.1 Monitoring sites map showing flathead, deployed oyster and wild mussel and oyster sites. Deployed oyster (●) and flathead (●) sites are labelled with site names. Wild oyster and mussel sites are colour coded by sampling region: Middle estuary (●), eastern shore (●), western shore (○), Ralphs Bay (●) and D'Entrecasteaux Channel (●). The control site Mickeys Bay is the control site for all programs (●).



Contaminant guidelines and assessment approach

To assess human health risks associated with metal contaminants in fish and shellfish, we applied guidelines established by FSANZ. FSANZ (2024) sets Maximum Levels (MLs) for mercury, lead, cadmium, and arsenic in seafood under Standard 1.4.1, Schedule 19. These MLs are legally enforceable and are based on dietary exposure and human health risk calculations. MLs are applied to foods that significantly contribute to total contaminant intake.

In addition, Generally Expected Levels (GELs) for zinc and copper were referenced from FSANZ (2001). GELs are not legally enforceable but provide indicative concentrations for contaminants with low-level risk to consumers. These values were used to contextualise environmental concentrations where MLs were not applicable.

The metals and metalloid (arsenic) assessed in this study include:

- Mercury, lead, cadmium, inorganic arsenic (Southern Sand Flathead only) – due to their particular toxicity and known association with industrial processes.
- Zinc and copper – due to elevated environmental concentrations throughout the estuary and relevance to ecosystem health.

Contaminant concentrations in seafood samples were compared against MLs and GELs to evaluate potential human health risks (Table 8.1) (FSANZ, 2024).

Table 8.1 Maximum levels (MLs) and generally expected levels (GELs) for metal toxicants in fish and shellfish (FSANZ, 2024).

	Maximum levels (mg/kg)				Generally Expected Levels (median/90 percentile) (mg/kg)	
	As*	Cd	Hg	Pb	Cu	Zn
Fish	2	no set limit	0.5**	0.5	0.5/2	5/15
Molluscs	1	2	0.5***	2	3/30	130/290

* Arsenic (inorganic)
** for most fish (1.0 for sharks and other specified fish)
*** specifies mean for minimum number of fish required

8.1.2 Results

Deployed Pacific Oysters

FSANZ guidelines assessment

For surface deployed oysters (n=47), cadmium exceeded the ML in 9% of samples, while lead exceeded the ML in 73% of samples. Mercury concentrations remained below the ML in all samples. Additionally, GELs were exceeded in 83% of samples for zinc and 75% of samples for copper (Figure 8.2). Similar patterns were observed in oysters deployed to middle and bottom waters, where median concentrations in middle- and bottom-deployed oysters at three sites exceeded trigger levels for lead, copper and zinc (Figure 8.3).

Spatial

Spatial variation in metal concentrations was observed in oysters deployed to surface waters of the Derwent Estuary and to the control site (Mickeys Bay). Median metal concentrations during the reporting period were higher at all estuary sites when compared to the control site at Mickeys Bay.

Within the estuary, median metal concentrations were highest at Nyrstar Wharf site, reflecting close proximity to the source. Concentrations declined upstream of the Nyrstar Wharf site (Dowsing Point and Elwick Bay Red Pylon) and increased from New Town Bay to Pavilion Point. This pattern was consistent for all analytes (Figure 8.2).

Similar variation was observed in oysters deployed to middle and bottom waters, where concentrations were greatest at Nyrstar Wharf and declined in up and downstream directions (Figure 8.3).

Water depth

Significant variation was observed across deployment depths. Copper, lead and mercury concentrations were greatest in oysters deployed to bottom waters, and lowest in surface deployed oysters. This trend was particularly evident in lead and copper at the Nyrstar Wharf site. The opposite was true for cadmium and zinc where concentrations were greatest in surface oysters and lowest in bottom oysters (Figure 8.3).

Temporal

Across the monitoring period (2002–2024), statistically significant ($p<0.05$) linear and non-linear trends were observed at seven of the nine sites for three of the metals measured. Mercury concentrations in surface-deployed oysters increased over time at four estuary sites (Dowsings Point, Elwick Bay Red Pylon, New Town Bay and Pavillion Point), copper increased at one site (Bedlam Walls), zinc increased at one site (Nyrstar Wharf) and cadmium declined at two sites (Dowsings Point and Cornelian Bay) (Figure 8.4; Table 8.2)

In 2024, surface-deployed oysters at Nyrstar Wharf showed higher concentrations of all measured metals compared to all previous samples from this location, except for mercury, where the outlier was recorded in 2022 (Figure 8.2).

Figure 8.2 Metal concentrations in soft tissue of surface-deployed Pacific Oysters (2020–2024). Box plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at eight sampling sites in the middle estuary and a control site in Mickeys Bay, the D'Entrecasteaux Channel. Grey dots represent baseline corrected data. Dashed red line represents Food Standards Australia and New Zealand national Maximum Levels (cadmium, lead, mercury) and Generally Expected Levels (copper and zinc). Metal concentrations are expressed in mg/kg wet matter basis (WMB).

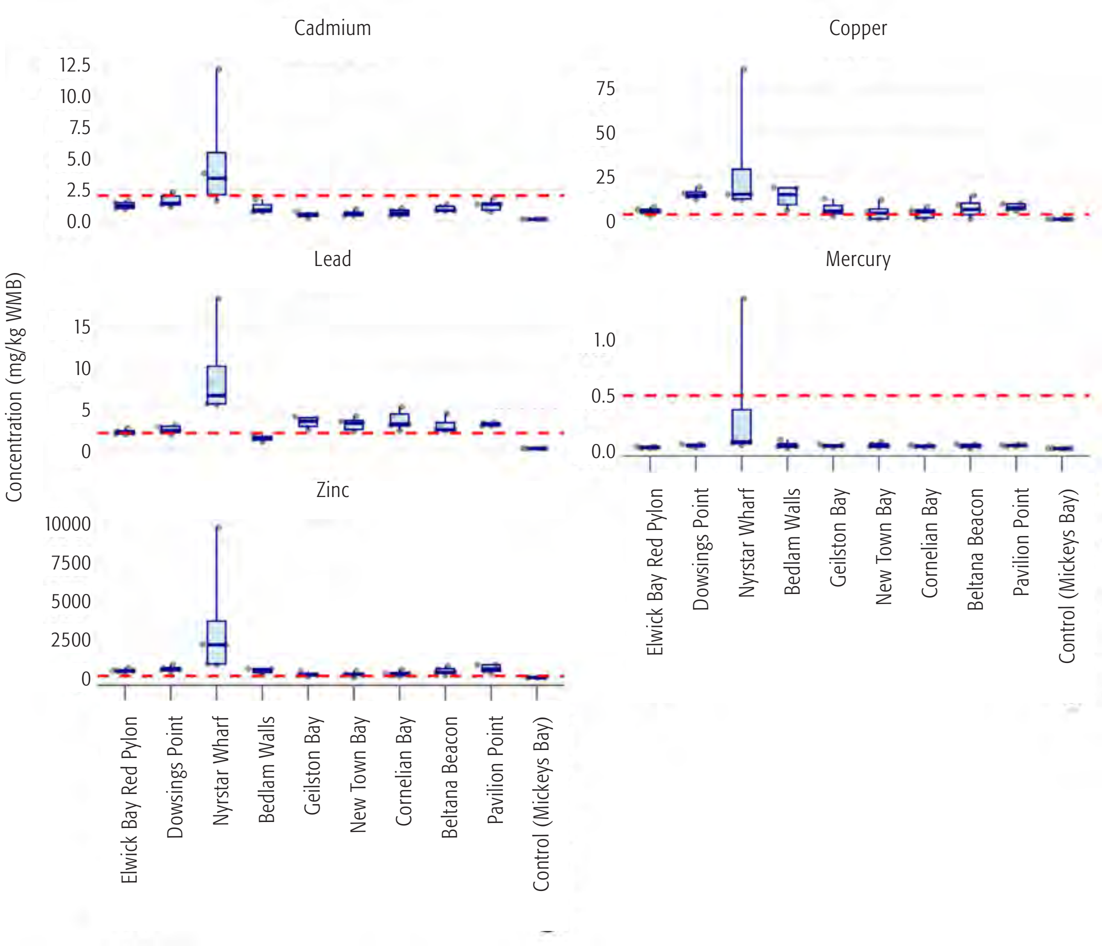


Figure 8.3 Metal concentrations in soft tissue of deployed Pacific Oysters deployed to surface, middle and bottom waters (2020–2024). Plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at three sampling sites in the middle estuary (Elwick Bay Red Pylon, Nyrstar Wharf, and Beltana Beacon). Dashed red line represents Food Standards Australia and New Zealand national Maximum Levels (cadmium, lead, mercury) and Generally Expected Levels (copper and zinc). Metal concentrations are expressed in mg/kg wet matter basis (WMB).

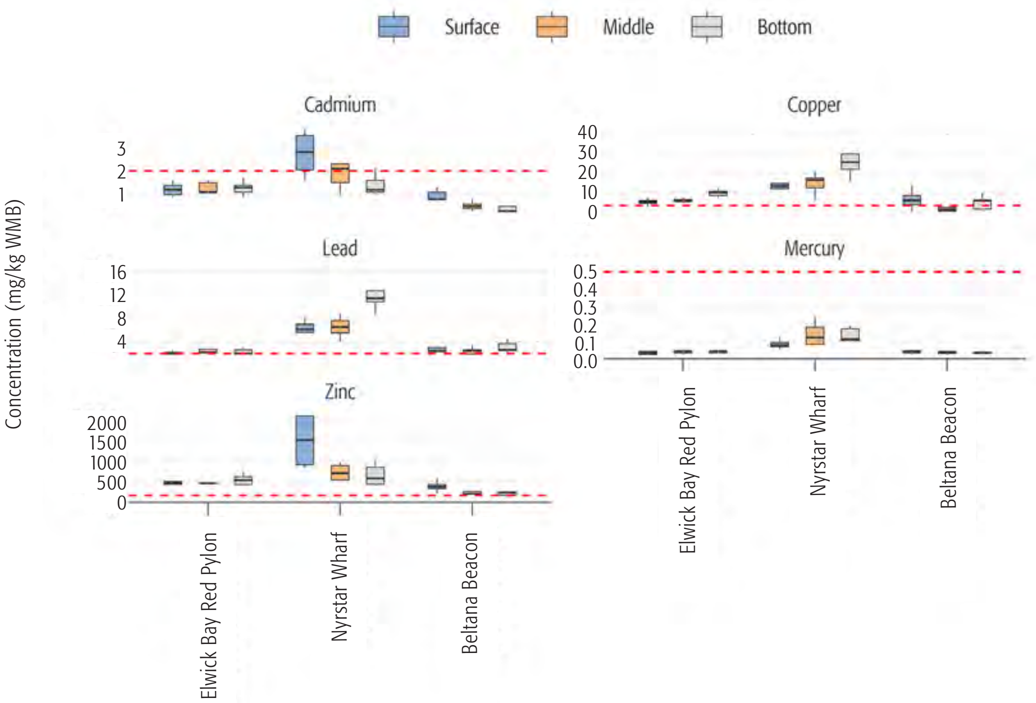


Figure 8.4 Significant linear and non-linear temporal trends in cadmium (a–b), mercury (c–f), zinc (g), and copper (h) in surface-deployed oysters at middle Derwent Estuary sites. Trends were assessed using HARSAT (**https://harsat.amap.no**). Metal concentrations are expressed in µg/kg- wet matter basis (WMB). Limit of detection for mercury is 20 µg/kg.

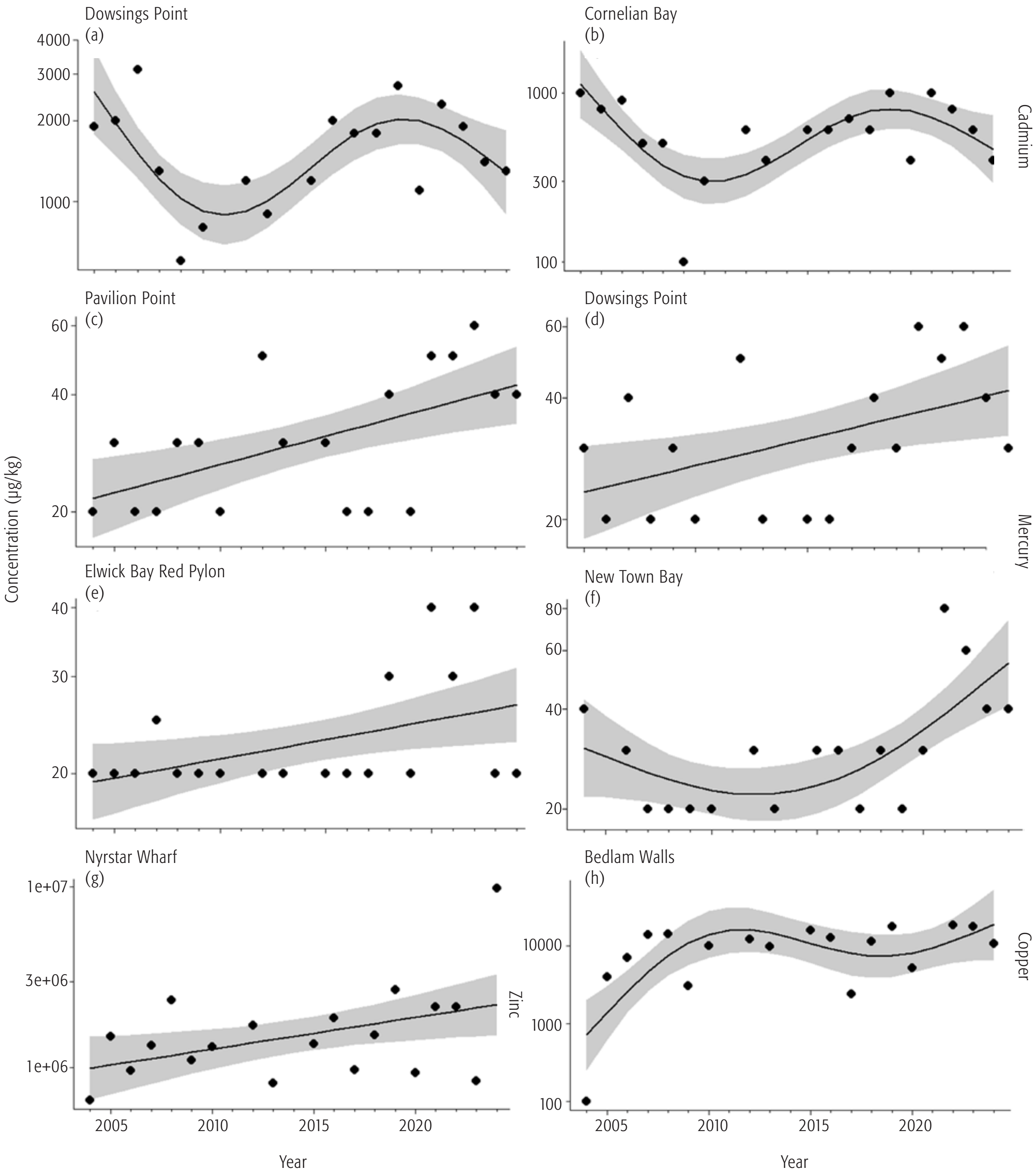


Table 8.2 Significant trends detected in metal concentrations in surface deployed oysters at sampling sites in the middle estuary. Percent change is an estimate of the shift in concentration per year over the monitoring period. P-values indicate the level of significance, and trend types are classified as linear or non-linear. Non-significant trends are not reported.

Site	Metal	Years	% change	p-value	Trend
Bedlam Walls	Copper	18	16.3	0.002 **	Non-linear
Cornelian Bay	Cadmium	19	-4.4	0.041 *	Non-linear
Dowsings Point	Cadmium	18	-3.7	0.041 *	Non-linear
Dowsings Point	Mercury	18	3	0.031 *	Linear
Elwick Bay Red Pylon	Mercury	19	1.6	0.045 *	Linear
New Town Bay	Mercury	18	2.9	0.019 *	Non-linear
Pavilion Point	Mercury	19	3.3	0.007 **	Linear
Nyrstar Wharf	Zinc	19	4	0.046 *	Linear

Wild oysters and mussels

FSANZ guideline assessment

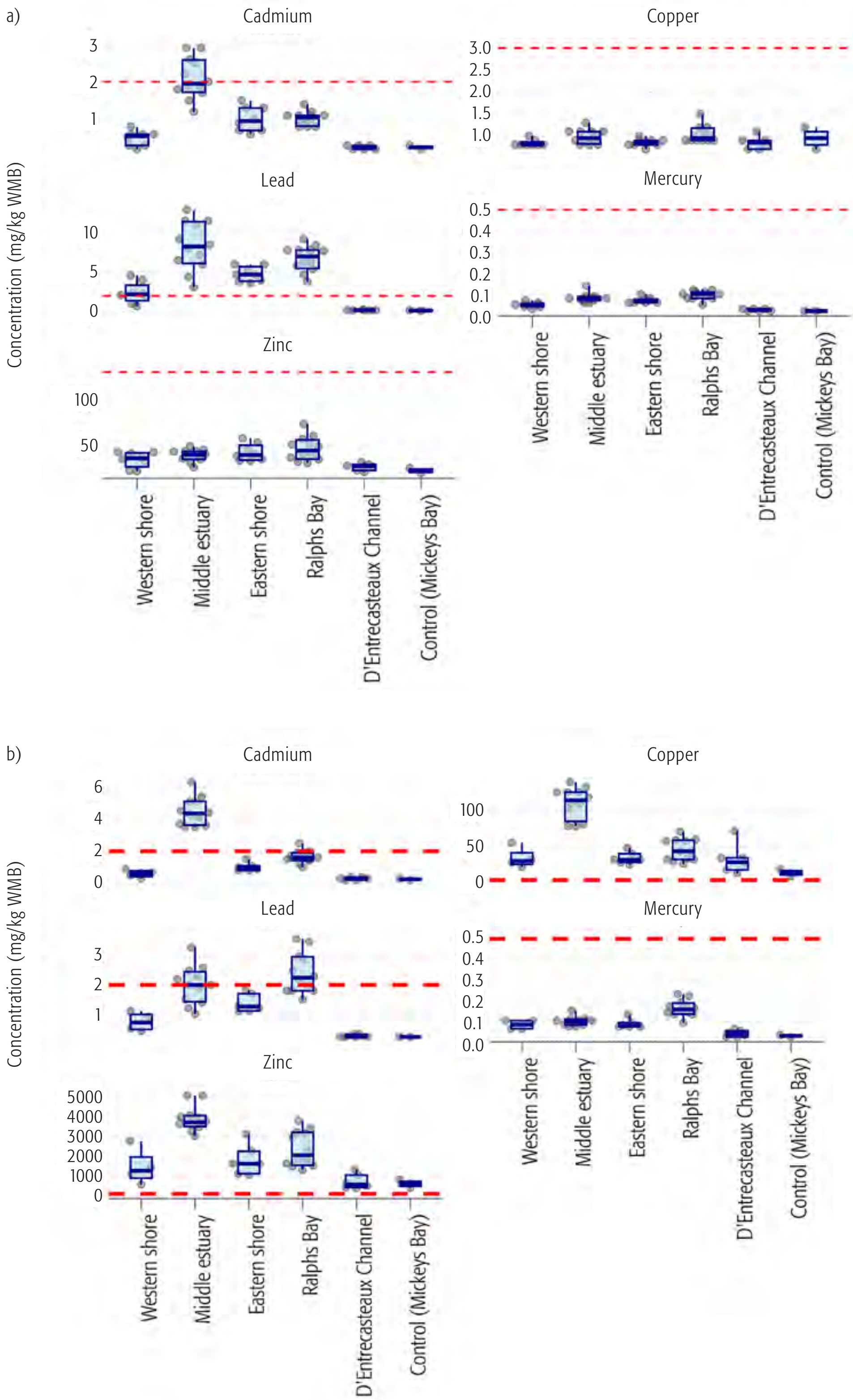
For wild mussels (n=48), cadmium exceeded the ML in 10% of samples and lead exceeded the ML in 77% of samples. Mercury concentrations remained below the ML in all cases. GELs for copper and zinc were not exceeded (Figure 8.5a).

For wild oysters (n=42), cadmium exceeded the ML in 31% of samples, and lead exceeded the ML in 29% of samples. Mercury concentrations remained below the ML in all cases. GELs for copper and zinc were exceeded in 100% of samples. Median zinc and cadmium concentrations also exceeded the GELs at channel and control sites (Figure 8.5b).

Spatial patterns

Regional variation in metal concentrations was observed in wild oysters and mussels across the Derwent Estuary and D'Entrecasteaux Channel. Median concentrations during the reporting period were higher in all estuary regions (western shore, middle estuary, eastern shore, and Ralphs Bay) compared to sites in the D'Entrecasteaux Channel and at the control site. Within the estuary, the highest concentrations were typically recorded in the middle estuary and Ralphs Bay, whilst the western shore exhibited the lowest levels. These spatial patterns likely reflect the influence of estuarine circulation, ambient water metal concentrations, proximity to contaminant sources, and estuarine processes. Sites in the middle estuary are closest to the zinc smelter, whereas marine-influenced regions, such as the western shore, are exposed to lower concentrations in the water column (Figure 8.5).

Figure 8.5 Metal concentrations in soft tissue of a) wild Blue Mussels, and b) wild Pacific Oysters (2020–2024). Plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at five regions (western shore, eastern shore, middle estuary, Ralphs Bay, and the D'entrecasteuax Channel) and a control site (Mickeys Bay). Grey dots show the raw data. Red line represents Food Standards Australia and New Zealand national Maximum Levels (cadmium, lead, mercury) and Generally Expected Levels (copper and zinc). Metal concentrations are expressed in mg/kg wet matter basis (WMB).



Southern Sand Flathead

FSANZ guidelines assessment

Only twenty three of the 300 Southern Sand Flathead caught during the monitoring period (2020–2024) were above the legal size. Lead did not exceed MLs in any Southern Sand Flathead samples. However, mercury concentrations in flathead (n=23) from middle and lower estuary sites exceeded the ML of 0.5 mg/kg WMB in 61% of samples. Total arsenic exhibits higher values (>2 mg/kg) in a significant number of samples. Whilst copper did not exceed GELs in any samples, zinc exceeded the GEL in 80% of samples (Figure 8.6).

Spatial patterns

Regional variation was observed in mercury, zinc and arsenic concentrations when comparing sites in the Derwent Estuary to the control site (Mickeys Bay). In contrast, cadmium, copper and lead concentrations showed no spatial variation, with levels typically at or below the limit of reporting (LoR) (Figure 8.6). For these metals, any potential regional differences are likely obscured by results being consistently below the LoR.

Mercury exhibited the most pronounced variation, with significantly elevated concentrations at estuary sites relative to the control site, where levels were typically below the ML. Within the estuary, Ralphs Bay (Richardson Beach and Ralphs Bay Spit) consistently recorded higher mercury concentrations, likely reflecting presence of older and larger fish, or greater mercury bioavailability at these sites.

Zinc and arsenic concentrations were typically higher at estuary sites, however, fish from the control site (Mickeys Bay) also showed levels above the ML, indicating that these elements may be elevated more broadly across the region.

Temporal

Across the monitoring period (2004–2024), no significant trends were observed at four of the nine sites, with five sites exhibiting statistically significant linear and non-linear trends. Copper concentrations in Southern Sand Flathead decreased significantly at three sites (Bellerive Bluff, Ralphs Bay Spit and the control site), zinc decreased significantly at one site (control), and cadmium increased at two sites (Kingston Beach North and Ralphs Bay Spit). No change was detected for lead, arsenic or mercury (Figure 8.7; Table 8.3). Zinc and mercury analysis for all sites was conducted with plots available in Appendix 1.

Figure 8.6 Metal concentrations in Southern Sand Flathead (2020–2024). Plots show the minimum, 20th percentile, median, 80th percentile and maximum concentrations at eight Derwent Estuary sites and a control site (Mickeys Bay). Grey dots show the raw data. Red line represents Food Standards Australia and New Zealand national Maximum Levels (cadmium, lead, mercury and inorganic arsenic) and Generally Expected Levels (copper and zinc). Metal concentrations are expressed in mg/kg wet matter basis (WMB).

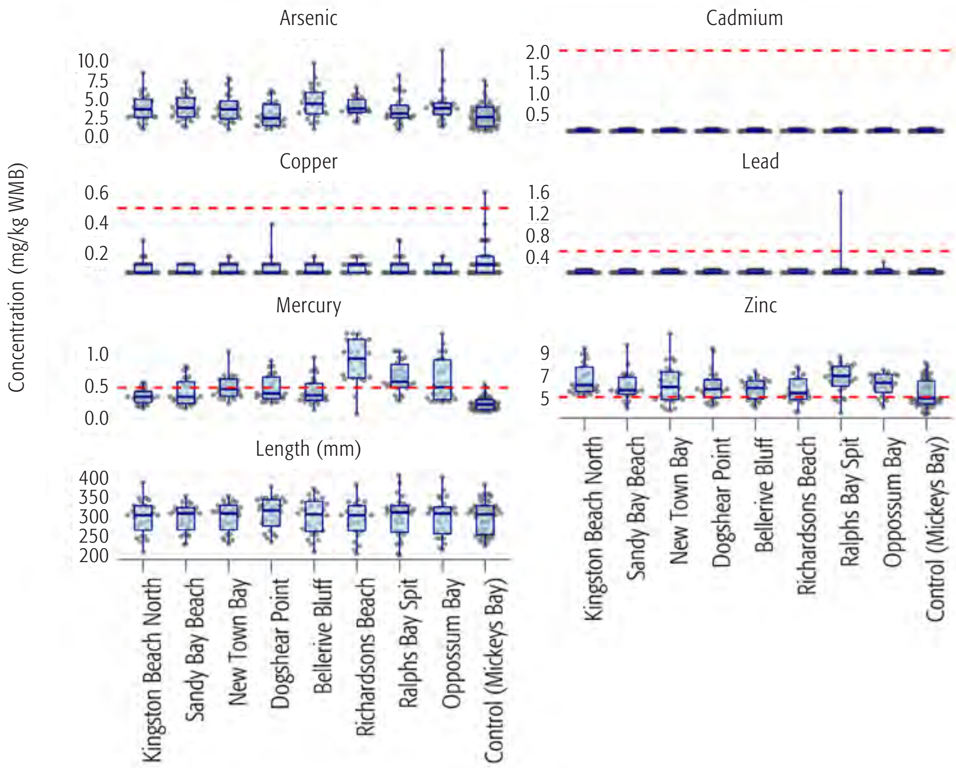


Figure 8.7 Linear and non-linear temporal trends in mean cadmium (a–b), copper (c-e) and zinc (f) concentrations in Southern Sand Flathead in the Derwent Estuary. Trends were assessed using HARSAT (**https://harsat.amap.no**). Metal concentrations are expressed in µg/kg- wet matter basis (WMB). All trend lines were statistically significant (p<0.05).

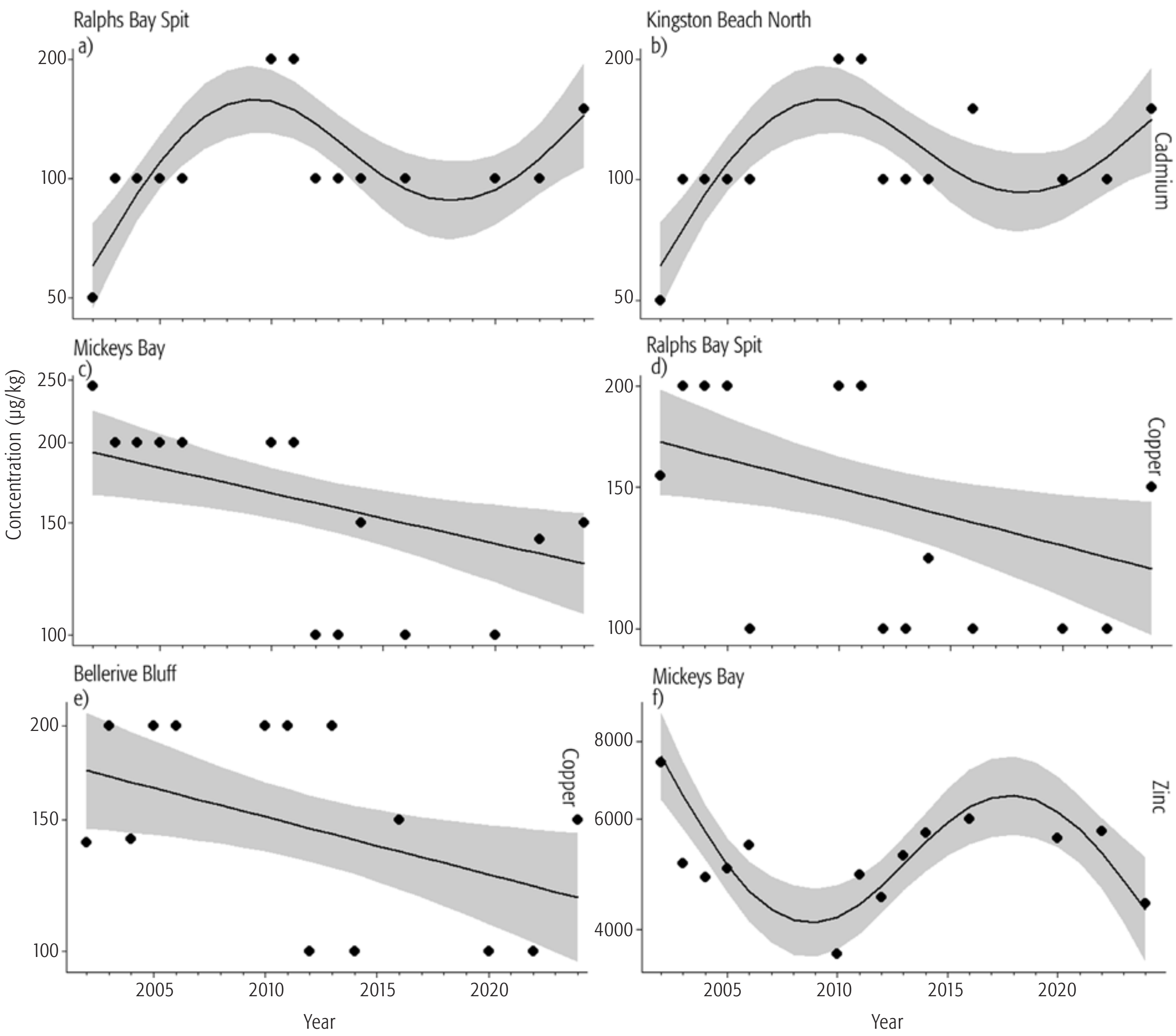


Table 8.3 Significant trends detected in metal concentrations in Southern Sand Flathead from the Derwent Estuary and at the control site (Mickeys Bay, D'Entrecasteaux channel). Percent change is an estimate of the shift in median concentration per year over the monitoring period. P-values indicate the level of significance, and trend types are classified as linear or non-linear. Non-significant tends are not reported.

Site	Metal	Years	% change	p-value	Trend
Kingston Beach North	Cadmium	14	3.8	0.005 **	Non-linear
Ralphs Bay Spit	Cadmium	14	4	0.004 **	Non-linear
Bellerive Bluff	Copper	14	-1.8	0.043 *	Linear
Mickeys Bay	Copper	14	-1.8	0.022 *	Linear
Ralphs Bay Spit	Copper	14	-1.6	0.040 *	Linear
Mickeys Bay	Zinc	19	-2.6	0.004 **	Non-linear

8.1.3 Discussion

Public health advice

Ongoing monitoring of Southern Sand Flathead, Pacific Oysters and Blue Mussels in the Derwent Estuary would suggest that the current public health advisories remain relevant and highlights the importance of these in reducing community exposure to metal toxicants in seafood. Mussels and oysters accumulated high levels of lead over the reporting period, whilst oysters also exhibited high levels of zinc and copper. These findings, combined with the rapid accumulation observed in oysters deployed over six weeks, highlights the continued need to avoid consumption of shellfish from the Derwent.

Southern Sand Flathead contained high levels of mercury, with concentrations exceeding the FSANZ guidelines. Mercury accumulation in Southern Sand Flathead in the Derwent has been well documented (Verdouw, 2008; Jones *et al.*, 2014; DEP, 2020b).

Although total arsenic concentrations in flathead were often greater than 2 mg/kg, it is important to remember that this trigger value applies specifically to inorganic arsenic. Therefore, a direct comparison cannot be made, and it is possible that the inorganic fraction is below the ML. The DEP recommends including all species of arsenic in the analyte suite to definitively characterise the organic arsenic species present in flathead to enable an accurate assessment against MLs. This may also identify the pathway for arsenic bioaccumulation in flathead.

Health risk from exposure to flathead may be partially reduced as a result of reduced numbers of legal-size flathead in the system. Only 8% of flathead caught during the current reporting period were above the legal catch size. The catch size for flathead increased to 35 cm in November 2023 in response to research showing a marked decline in the overall flathead population around Tasmania more broadly (Tracey S. and Stark K.D., 2024).

Previous DEP monitoring of recreationally targeted wild-caught fish found exceedances for at least one metal or metalloid in all species surveyed, particularly in Black Bream (DEP, 2020a; b). These findings reinforce the need for continued monitoring and ongoing communication in the public health space to avoid complacency and ensure the community remain aware of the risks.

Based on current findings, the public advisories from the Director of Public Health for the Derwent remain as follows:

- Do not consume any shellfish or bream from the Derwent, including Ralphs Bay
- Do not eat fish from the Derwent more than twice a week and the following people should further limit their consumption to once a week:
    - Pregnant and breastfeeding women
    - Women who are planning to become pregnant
    - Children aged six years and younger
- When eating fish from the Derwent, it is best to avoid eating fish from other sources in the same week.

This advice is based on long-term monitoring of flathead and studies on the mercury levels in a variety of legally sized fish caught in the Derwent. There is always a risk to human health from eating wild shellfish; thus, the Department of Health recommends that all shellfish is bought from retail outlets, because shellfish for sale is subject to quality assurance programs that specifically test for, and manage, human risks.

Spatial variation

Metal concentrations in all biota were consistently lower at the control site (Mickeys Bay) than in the Derwent Estuary, except for arsenic in flathead, which exceeded guideline values at the control site. This suggests arsenic bioavailability is not confined to the Derwent. Within the estuary, oysters and mussels near the Nyrstar Wharf exhibited the highest concentrations, which is consistent with previous monitoring and reflects proximity to historical and ongoing metal sources. Mercury, zinc and lead were also elevated in wild shellfish at Ralphs Bay. This is consistent with observations of elevated zinc in ambient waters and longer water and sediment retention times in this region.

Similarly, elevated metal levels were also observed in flathead in this region. Previous studies of Southern Sand Flathead in the Derwent suggested that fish from Ralphs Bay (Ralphs Bay Spit and Richardson Beach) had higher concentrations of mercury than fish caught from other regions in the Derwent (Langlois *et al.*, 1987; Verdouw *et al.*, 2011; DEP, 2020b). In contrast, Jones et al., (2013) did not observe significant variation in mercury concentrations between regions once the models were corrected for fish length, however, growth rate variation accounted for a large proportion of observed spatial differences. A follow-up study by Jones et al. (2014) suggested that spatial variation in mercury bioavailability may also be influenced by environmental factors, with the co-occurrence of some contaminants confounding metal contamination data. In this study it was found that there may be an antagonistic relationship with selenium that affects mercury bioavailability. Sites with low selenium were found to have higher proportions of the more toxic mercury species, methylmercury, whereas sites closer to the smelter with high selenium levels had lower mercury bioavailability. The current monitoring data has confirmed the previously observed elevations in mercury and zinc in Sand Flathead in Ralphs Bay.

Median fish length in fish sampled from Ralphs Bay did not differ from that of other Derwent sites during the current reporting period. This may be a consequence of a reported decline in legal-size fish associated with increased fishing pressure and reduced recruitment success (Krueck *et al.*, 2025). More recent research also suggests that the movements of flathead within the Derwent are complex. Tracey *et al. (*2020) showed that flathead exhibit lower home-site fidelity than previously thought and as such has the potential to confound spatial patterns in the estuary.

Temporal variation

Over the 22-year monitoring period, metal concentrations in biota remained relatively stable at most sites; however, a decline in cadmium was observed at several locations. In contrast, mercury levels in oysters showed a measurable increase. HARSAT analysis indicated a statistically significant increase (approximately 1-3% per year, varying by site) in mercury concentrations in deployed Pacific Oysters at western shore sites closest to Nyrstar. Although these concentrations remain well below thresholds of concern for human health, the increase is interesting and further investigation and predictive modelling are required to better understand the processes driving these observed changes. An assessment of the mercury discharge from Nyrstar's combined effluent stream (see Section 4.3), environmental and biogeochemical drivers–such as climatic conditions (wind and rainfall), sediment geochemistry, co-contaminants, mercury methylation, and food-web dynamics is suggested as all these processes and inputs may influence mercury bioavailability in surface deployed oysters.

As noted above, assessing temporal changes in mercury bioaccumulation in fish can be complicated by shifts in habitat, fish length, growth rates, prey preferences and the potential for seasonal movements or migration. Jones et al., (2013) used linear and non-linear models to examine spatial and temporal variations between Derwent Estuary regions and concluded that calculating length-standardised mercury concentrations is important to accurately evaluate spatial and temporal trends. Both linear (data from 1974-2011) and non-linear models (data from 1991-2011), which accounted for fish biometrics, showed no significant change despite reductions of mercury inputs.

In the current study, length standardisation was not applied due to lack of log-linear relationship between mercury concentration and fork length. Consistent with the findings of Jones et al., (2013), there was no evidence of any significant temporal changes in flathead contamination levels at any of the sampling sites since sampling commenced (2002–2024). This would suggest that significant reductions in anthropogenic mercury inputs achieved to date do not seem to be transferring to reductions in fish concentrations and highlights the complexity of the mercury cycling processes and interactions in the Derwent Estuary. Similar findings have been observed in other fish species, including Striped Bass (Greenfield *et al.*, 2005) and tuna (Médieu *et al.*, 2024).

8.1.4 Conclusion

Monitoring of deployed oysters, wild mussels, and flathead confirms that metal contamination in the Derwent Estuary remains a health risk and that current seafood safety warnings should be maintained. Spatial patterns of metal contamination in shellfish (oyster and mussels) within the main estuary were largely consistent with previous studies, showing that concentrations increased with proximity to historical point sources such as Nyrstar. Elevated mercury concentrations in biota from Ralphs Bay suggest additional environmental factors influencing bioavailability in this region. Despite substantial investment in remediation and reductions in metal inputs over recent decades, temporal trends indicate limited improvement. These findings underscore that historical contamination continues to have a lasting influence on metals and metalloids in the food web.

8.1.5 Recommendations

To protect public health, long-term monitoring and management strategies remain essential. To improve understanding of bioavailability of metals in biota, the following gaps and limitations are recommended:

- Include all species of arsenic in the analyte suite to definitively characterise the organic arsenic species present in flathead to enable an accurate assessment against MLs. This may also identify the pathway for arsenic bioaccumulation in flathead.
- Apply the length-standardisation method outlined in Jones et al. (2013) to adjust mercury concentrations for fish size before conducting trend analysis.
- Investigate selenium dynamics in sediments and fish. Strong correlations between total mercury and

selenium suggest selenium may play a key role in modulating mercury's ecological impacts (Jones *et al.*, 2014). Future assessments should incorporate selenium when evaluating mercury methylation and bioaccumulation.

- Develop predictive models to link changes in anthropogenic inputs with metal accumulation in the food web and associated spatial variability.
- Maintain and develop targeted research collaborations focused on better understanding ecological and geochemical interactions and ensuring appropriate and risk relevant monitoring approaches.

8.2 Minamata Convention on Mercury

The DEP has been actively collaborating with the Open-Ended Scientific Group (OESG), a technical subgroup under the Minamata Convention responsible for evaluating the Convention's effectiveness in reducing mercury in the environment. This section provides background on the Minamata Convention, the OESG's work and DEP's contributions to date.

Background on the Minamata Convention

Australia is a Party to the Minamata Convention on Mercury – an international treaty aimed at protecting human health and the environment from mercury pollution. The Convention sets measures to control mercury throughout its lifecycle: from entry into the economy, use in products, to emissions from industrial processes, and through to waste management and storage.

OESG contributions to the global scientific report

The Open-Ended Scientific Group (OESG) is a technical subgroup under the Minamata Convention. The OESG is tasked with compiling and analysing data to ensure sound science informs global policy implementation and evaluation.

Reports being prepared by the OESG will provide a scientific basis for determining whether international actions taken to implement the Minamata Convention have reduced emissions and contaminant levels in environmental media (such as air, soils and water), biota (such as fish, birds and mammals), and humans globally. The draft reports will undergo a review process by countries who are Parties to the Convention, this includes Australia. This process will take place in 2026-2027.

DEP contributions to the global scientific report

Australian data holders, including the DEP, submitted their data directly to the OESG through a data authorisation process, giving permission to the OESG and participating experts to use the submitted data for the purposes of the Minamata Convention's evaluation. Data is stored in the OESG's temporary data repository, hosted by the Biodiversity Research Institute.

The DEP datasets are among the few long-term series from the Southern Hemisphere - a region often underrepresented in global monitoring. These datasets offer valuable insights into mercury trends over time, particularly in estuarine environments near industrial activity, reflecting a strong commitment to collaboration and knowledge generation.

In addition, in 2025, the DEP scientists assessed mercury temporal trends in oysters and fish using the Harmonised Regional Seas Assessment Tool (HARSAT) being developed as a collaboration between AMAP (Arctic Monitoring and Assessment Programme), HELCOM (Baltic Sea Regional Seas Convention) and OSPAR (NE Atlantic Regional Seas Convention). The DEP also contributed to the integrated case study *"Analysis of Mercury in the Derwent Estuary, Australia"*.

8.3 Harmful Algal Blooms

Phytoplankton are photosynthetic, microscopic algae that live in the water column of freshwater, estuarine and marine environments. They are primary producers in aquatic food webs, being important sources of food for higher taxa, oxygenate the water through photosynthesis and play an important role in absorbing atmospheric carbon dioxide. Their abundance in the water column is highly variable, being dependent on temperature, light, salinity, wind and nutrient availability. When environmental conditions are optimal there can be rapid, widespread and persistent increases in phytoplankton abundance, termed phytoplankton/algal blooms, and can be seasonally recurrent, annual, or event based (Cloern and Jassby, 2010).

Algal blooms are a natural phenomenon and are mostly harmless to ecosystems and humans. However, sometimes these blooms can be detrimental to environments, by reducing oxygen in water, damaging fish gills and resulting in excessive mucus production, which can lead to fish mortality (Hallegraeff *et al.*, 2017). Certain algal species produce harmful toxins, which if consumed can be harmful to fish (ichthytoxins) and humans (neurotoxins). These types of blooms are termed harmful algal blooms (HABs).

In coastal and estuarine regions of southeast Tasmania HABs have been linked to environmental conditions following high rainfall and land run-off, relatively light winds, warmer water temperatures, lower air temperatures, and strong stratification of coastal waters, which inhibits mixing in the water column (Hallegraeff *et al.*, 1995; Condie *et al.*, 2019). These blooms can result in mass mortality of fish and other organisms (Roberts *et al.*, 2019), impact on human health (Edwards *et al.*, 2018), and result in loss of income for wild fisheries and aquaculture industries (Hallegraeff, 2014).

There is growing interest in better understanding and responding to HABs in coastal and estuarine areas, due to the 2025 unprecedented large and extended bloom of toxic microalgae *Karenia mikimotoi* in South Australia. This bloom began in mid-March 2025 and coincided with a prolonged marine heatwave in the area. This algae is harmful to marine wildlife, particularly fish, impacting their gills. This bloom has significantly impacted marine-dependent industries, with the South Australian and Commonwealth governments providing significant funding to support impacted industries, research and communications. *Karenia mikimotoi* does not produce a toxin that is harmful to humans, however exposure could cause slight irritations to skin and eyes, and the public are advised to avoid swimming in areas impacted by the bloom.

Currently in Tasmania there is routine weekly monitoring for HABs in shellfish growing areas and some wild fish catch monitoring sites, with fish aquaculture sites monitoring daily. NRE Tas aims to set up a HAB working group to increase state capacity and preparedness for HABs in coastal Tasmanian waters. This includes better understanding of conditions that cause HABs, how to efficiently and effectively monitor for HABs and how to coordinate and cooperate across multiple stakeholders with competing priorities and interests.

8.4 Additional HARSAT Analysis of Mercury and Zinc

Figure 8.8 Temporal trends in median mercury concentrations in deployed oysters at eight Derwent Estuary sites and one control site (Mickeys Bay) for the monitoring period (2004–2024).

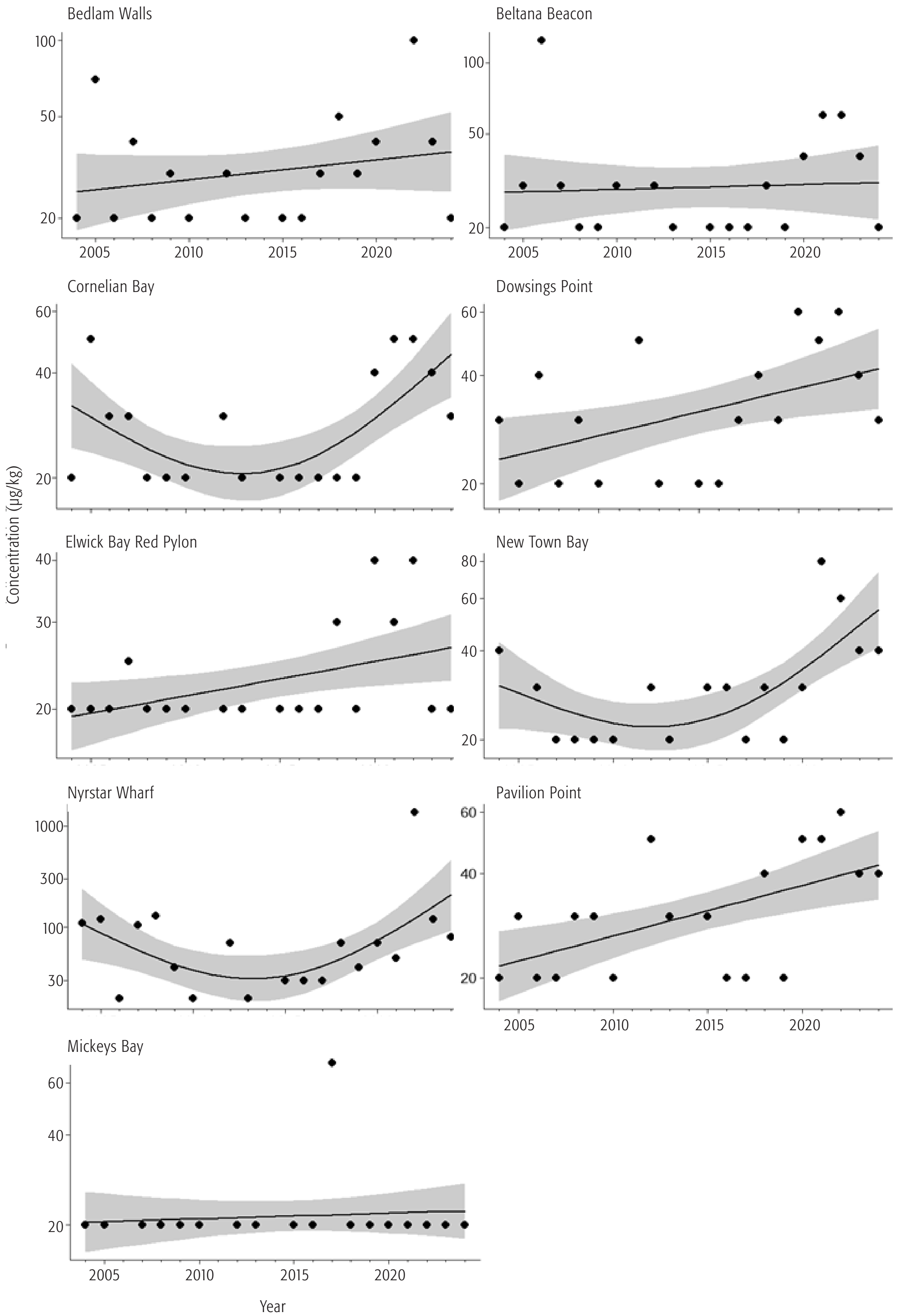


Figure 8.9 Temporal trends in median zinc concentrations in deployed oysters at eight Derwent Estuary sites and one control site (Mickeys Bay) for the monitoring period (2004–2024).

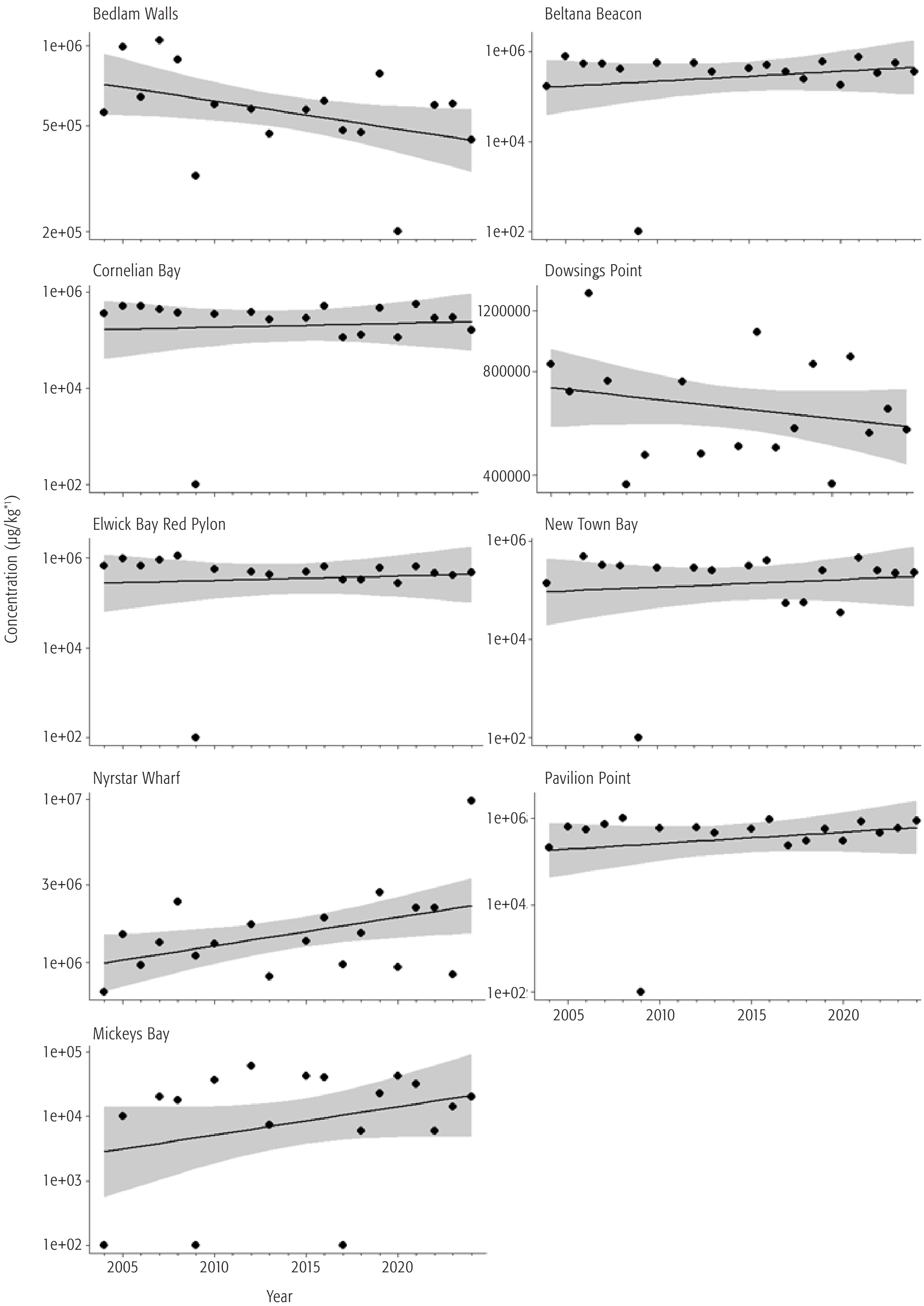


Figure 8.10 Temporal trends in median mercury concentrations in flathead at eight Derwent Estuary sites and one control site (Mickeys Bay) for the monitoring period (2004–2024).

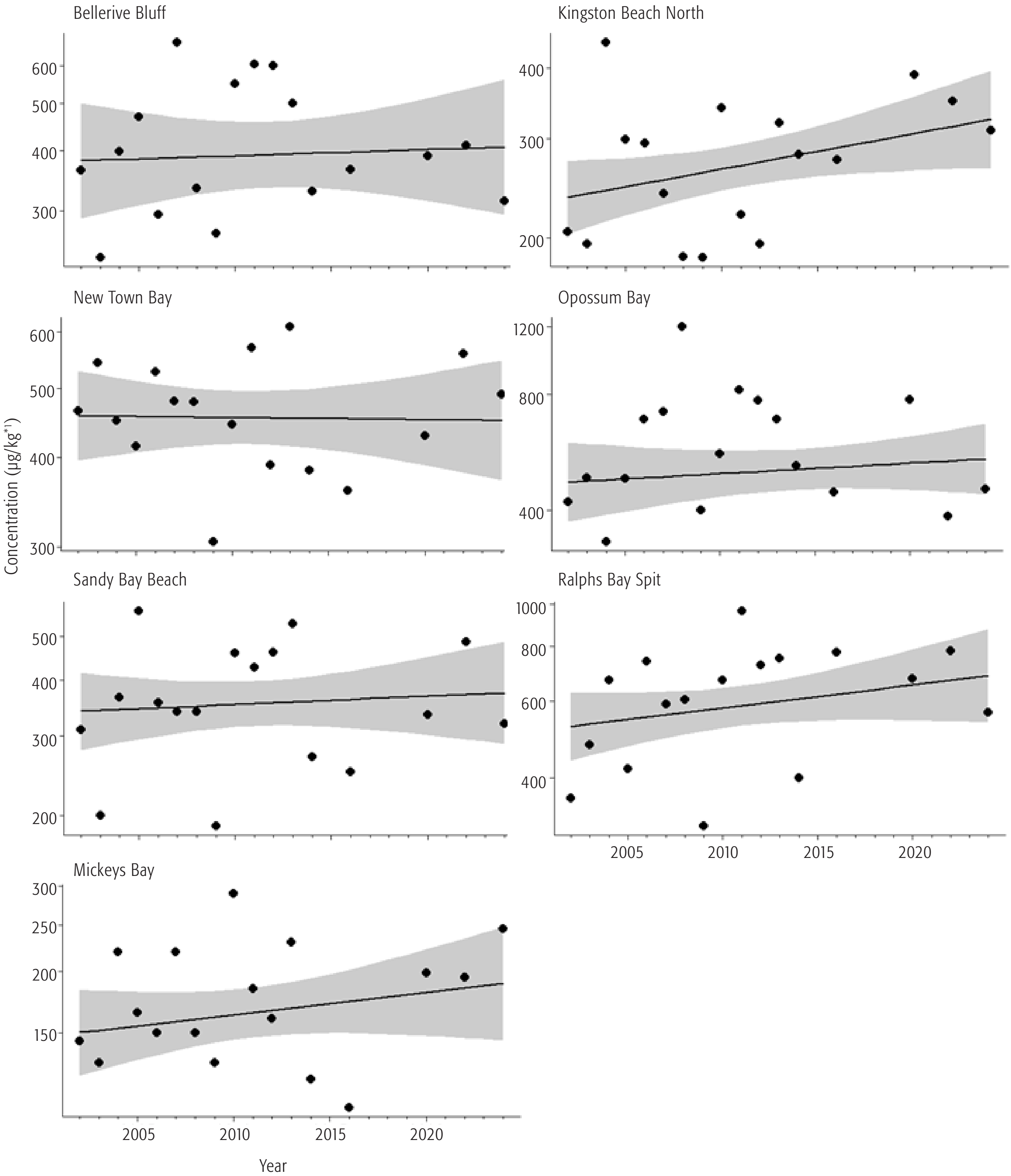
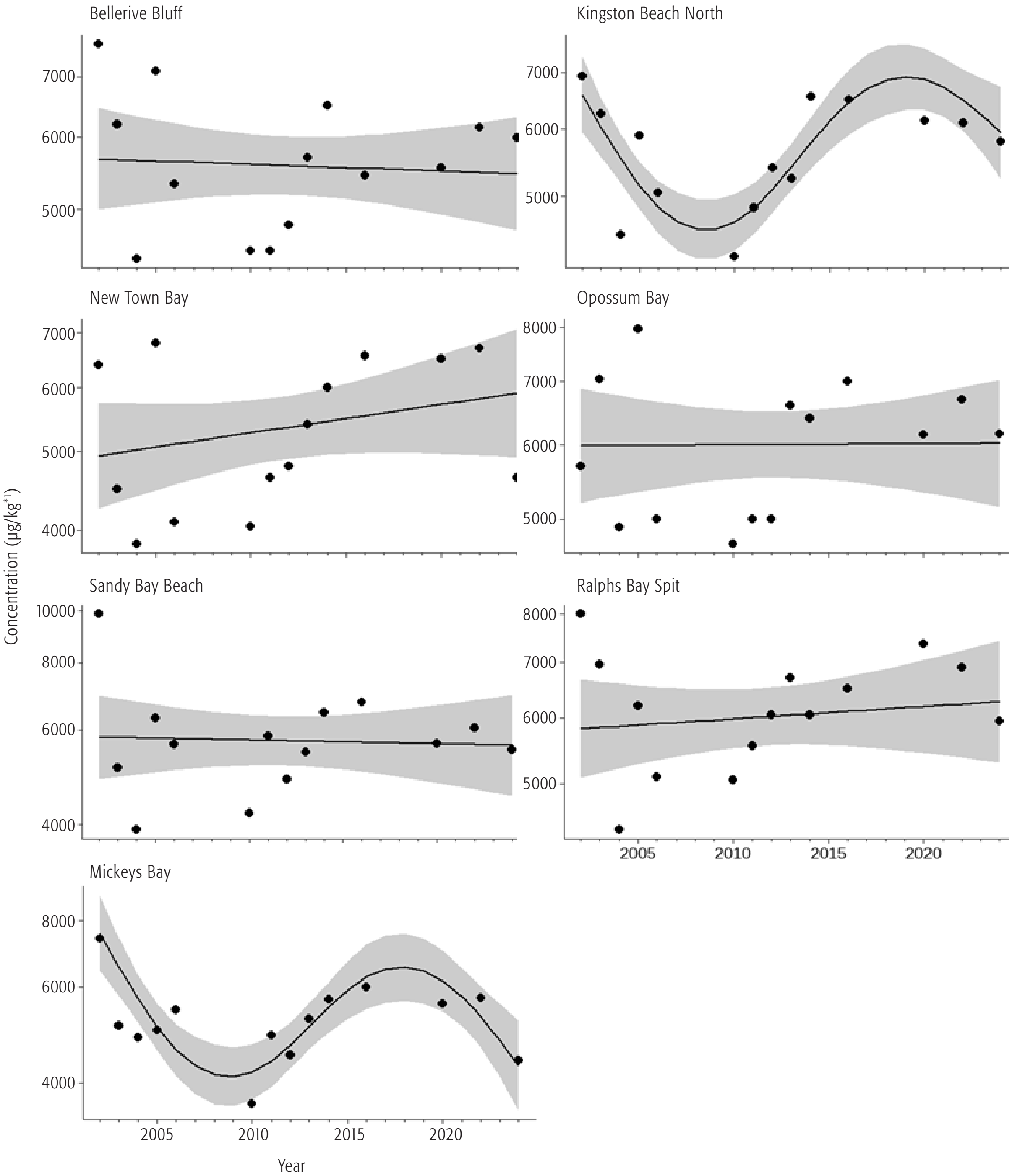


Figure 8.11 Temporal trends in median zinc concentrations in deployed oysters at eight Derwent Estuary sites and one control site (Mickeys Bay) for the monitoring period (2004–2024).



9

Estuarine habitats

9 Estuarine habitats

Timtumili Minanya/River Derwent, including the Derwent Estuary's climatic region, geomorphic form, geology, and environmental condition has formed a variety of habitats throughout the estuary. The upper estuary is narrow and relatively shallow, dominated by macrophyte beds and freshwater wetlands. The middle estuary widens and deepens in its lower reaches, the mid-channel is dominated by silt, with ephemeral seagrass meadows, and areas of intertidal saltmarsh found within protected bays. The lower estuary is highly marine influenced with sandy beaches, rocky reefs, patches of seagrass, intertidal sand flats and saltmarsh community (Table 9.1).

These habitats support a broad range of flora and fauna and are also highly valued for human recreation and enjoyment. They also face pressures from legacy contamination, industry, urban and agricultural development, nutrient loading, human disturbance and climate change. There is growing interest in ecosystem restoration to restore degraded or lost habitats, with the aim of restoring associated ecosystem services, biodiversity and socio-economic outcomes. Several exciting habitat restoration projects have been conducted in the Derwent Estuary and are summarised in this section.

Table 9.1 Percentage of sub-tidal habitat cover across the Derwent Estuary, split into upper, middle and lower sections. From (Lucieer, Lawler, and Pender, 2007).

Section	Macrophytes (%)	Cobble (%)	Reef (%)	Sand (%)	Seagrass (%)	Silt (%)	Vegetated (%)	Total (%)
Upper	33.3	3.48	3.54	0	0	34.4	25.32	100
Middle	15.74	0	1.84	0.066	0.36	82	0	100
Lower	0	0	2.06	64.38	0.054	33.5	0	100
Whole Estuary	3.39	0.14	2.1	53.37	0.09	39.9	1.01	100

9.1 Sub tidal soft sediments

Sub-tidal soft sediments constitute the majority of benthic habitat across the Derwent Estuary (Lucieer, Lawler, and Pender, 2007). Silt is the dominant benthic habitat in the upper and middle estuary, in the lower estuary silt sediment is only found in deeper, mid-channel regions, with high energy areas of nearshore bays and the estuary's mouth constituting sand. These sub-tidal soft sediments are important habitats for macrophytes and seagrass (**Section 9.5**) in shallow areas, as well as benthic invertebrate infauna and epifauna communities (**Section 10.2**). These habitats also perform important ecosystem services including nutrient cycling, organic matter decomposition, metal regulation and sediment stabilisation.

Sub-tidal soft sediments in the upper and middle estuary are highly contaminated from legacy industrial metal pollution (**Section 4.3**). Sediments in the upper estuary are also found to contain high organic carbon content, due to effluent from Boyer paper mill (**Section 4.2**), which has influenced the structure of benthic infauna in this region of the estuary (Macleod and Helidoniotis, 2005). Sediments surrounding these industrial sites have higher levels of contamination, in the upper and middle estuary, however sediment particle composition is likely also influencing contamination levels. Finer silt sediments are known to accumulate metals better than coarser-grained particles, due to their surface-to-volume ratio and organic matter content (Jones *et al*., 2003; Mucha *et al*., 2004; Buyang *et al*., 2019).

9.2 Native Oyster Reef Restoration in the Derwent Estuary

Provided by Jennifer Hemer, NRM South

Native or Flat Oysters (*Ostrea angasi*) are bivalve molluscs, which exist as individuals, clumps or reef forming communities in subtidal coastal marine systems. Native Oyster reefs were once an important part of the marine landscape in southeast Tasmania. These reefs were responsible for providing a range of ecosystem services and were an important food source for Tasmanian Aboriginal people. Following European colonisation, Native Oyster reefs were the focus of a dredge fishery for food and construction materials – colonists mined the reefs for building material, burning the lime-rich shells to create cement. Unfortunately, by the turn of the nineteenth century, the impact of a destructive harvesting approach and localised declines in environmental condition (e.g. water and sediment quality) meant that this biodiverse and productive habitat was effectively lost from southern Australia, including the Derwent Estuary (Edgar and Samson, 2004). The only known remaining intact Native Oyster reef-habitat is found in Georges Bay, St Helens. The Reef Builder South-East Tasmania project, which was delivered by NRM South in partnership with The Nature Conservancy is part of the national Reef Builder initiative. This project was the first marine restoration project to restore Native Oyster reefs in Tasmania.

The project was delivered under permit between 2021 and 2023. In May 2023, an array of six reefs were constructed at the Dixons Beach restoration site in Taroona. These included four 30 x 10 m reefs and two 15 x 10 m reefs. The combined area of these reefs is approximately 1,400 m^2 and sits between 300 mm and 500 mm above the seabed at a depth of 7.5–8.9 m.

Reefs were constructed by deploying a locally sourced limestone rock reef base to the sea floor. The rock ensures the reef has structural integrity over time and lime rock encourages recruitment and growth of Native Oysters. Once constructed, reefs were seeded by divers with approximately 500,000 Native Oyster spat 5–10 mm in size, grown in a local hatchery. An additional approximately 30,000 wild spats around 40 mm in size were collected on a nearby oyster farm and seeded over the reefs. The oyster spat was settled on clean, waste scallop shell.

Figure 9.1 Native Oyster (*Ostrea angasi*) spat seeded onto scallop cultch. Image: The Nature Conservancy.

A restoration suitability model, developed by The Nature Conservancy in collaboration with NRM South, was used as a decision support tool to identify appropriate sites. This was further refined through stakeholder consultation and ground-truthing (surveys). Initially, 16 sites were identified through the restoration suitability model for further investigation. Potential sites were examined further through surveys, analysis of sediment samples, and video transects of the seabed (using a drop-video-camera technique). From the initial 16, six priority sites were identified for further investigation. Detailed investigation included dive-transects and collection of salinity and temperature data throughout the water column. All six sites showed potential for restoration. One of two sites considered most suitable was Dixons Beach, Taroona. Factors that influenced decision making included: construction logistics, biological suitability, oceanographic influences and site exposure, and community accessibility.

Prior to constructing the reefs, NRM South commissioned baseline surveys of the area. The Taroona reefs were built on an open area of fine sand and shell fragments with sparse flora and fauna, approximately 250 m from the shore. Monitoring surveys were completed at approximately six months and 18 months after construction and seeding with oyster spat to measure the growth and survival of Native Oysters and the development of reef ecology, particularly marine flora and fauna species colonising the constructed reef surface.

Monitoring was undertaken in October 2023, six months after the reefs were constructed and seeded. At that time, it was observed that there was both oyster mortality and survival, and that reef growth was low. However, recruitment of other sessile and mobile animals and macroalgae improved habitat quality. Further monitoring by The Nature Conservancy in December 2024 aimed to provide a snapshot of the state of the reefs, considering oyster size and density, and to provide a high-level overview of reef diversity and habitat quality. At the time of monitoring, oyster density on the reef was still low, however, there is evidence of natural recruitment indicating there is larval supply in the system. Monitoring also found that surviving oysters, made up of the original hatchery-reared and wild collected spat, had grown.

Based on field observations and reviewing video footage, the constructed reefs at Taroona exhibited high biodiversity, an abundance of several species, and generally good habitat quality. There was evidence of rapid colonisation of the hard substrates by early-to-mid successional stage species of sessile flora and fauna. Algal turfs, green filamentous algae, colonial and stalked ascidians, sponges, and large red and brown macroalgal species were observed in abundance. Most conspicuous was widespread presence of the large macroalgae Golden Kelp (*Ecklonia radiata*). The individuals observed were generally <50 cm, and these can be expected to grow and mature with time, potentially to over one meter in length, which will increase the physical complexity and overall habitat quality of the reefs.

Large mobile invertebrates, including sea stars, gastropods, and decapod crustaceans were observed. Several large, conspicuous fish species were observed, primarily invertivores, such as Bluethroat Wrasse (*Notolabrus tetricus*) and Toothbrush Leatherjackets (*Acanthaluteres vittiger*). Several Banded Stingaree (*Urolophus cruciatus*) were observed on the soft sediments just outside the edges of the reefs and attempting to shelter in the rock crevasses along the reef edges. All observed and recorded fish species are reef associated and non-migratory and can be expected to be present year-round.

NRM South and The Nature Conservancy will continue to monitor the reefs on an annual basis to track reef health and performance. Based on experience of other restored Native Oyster reefs around Australia, it is expected to take between seven and ten years for shellfish reefs to recover into functioning ecosystems. NRM South welcomes information about the reefs from recreational divers. Recreational divers should be aware that no shellfish collected from the Derwent Estuary should be consumed due to high levels of metal detected in shellfish tissue (**Section 8**).

Figure 9.2 Macroalgae on Taroona reefs December 2024. Image: The Nature Conservancy.

Figure 9.3 Banded Stingaree (*Urolophus cruciatus*) among the reef at Taroona December 2024. Image: The Nature Conservancy.

Strain *et al*. (2024) investigated community respiration rates, inorganic nitrogen fluxes, net denitrification, filtration rates and associated biological diversity in remanent Native Oyster (*Ostrea angasi*) reefs in southeast Tasmania. Oyster biomass was found to be positively associated with community respiration, filtration rates, and inorganic nitrogen fluxes, denitrification rates and biodiversity associated taxa. Supporting the hypothesis that oyster reefs enhance water quality and enhance diversity and abundance of associated taxa compared to bare sediments.

This research shows that Native Oyster reef restoration is a potential tool to enhance biodiversity and improve local water quality in coastal embayment's in southeast Tasmania. The remnant oyster reef sampled at Ralphs Bay was found to contain the highest density of oysters of all surveyed oyster reefs.

Many Native Oysters were observed throughout the estuary's middle and lower rocky reefs during recent rocky reef survey (**Section 9.3.1**), and natural recruitment was observed on the deployed reefs at Dixons Beach, Taroona. This shows that there is larval supply of Native Oysters in the Derwent Estuary, and any further restoration efforts would increase habitat for oysters to colonise, enhance biodiversity and improve local water quality.

9.3 Rocky Reef

Rocky reefs and their rocky intertidal shorelines are important habitats for a range of species, often with markedly higher diversity that adjacent soft sediment habitats. Within estuaries the distribution and abundance of species often varies considerably between habitats due to the differences in environmental conditions along the estuarine gradient from the open ocean to the river that creates it (e.g. Barrett *et al*., 2012, 2025). In the Derwent Estuary, reef habitat represents only approximately 2% of the total benthic area (Lucieer, Lawler, and Pender, 2007), yet it supports a distinct assemblage of species compared to surrounding habitats (Barrett *et al*., 2012b).

9.3.1 Rocky Reef Project

Provided by Associate Professor Dr Neville Barrett and Charlotte McAneney, IMAS

In 2010, a survey was initiated by the Derwent Estuary Program (DEP) to quantify the distribution and abundance of reef-associated biota on subtidal and intertidal reef habitats in the Derwent Estuary (Barrett *et al*., 2012b), including a thorough survey for rare species, such as Derwent River Seastar (*Marginaster littoralis*) – an endemic intertidal seastar recorded only from the Derwent, and quantitatively describing the abundance and distribution of introduced species. In 2024, this survey of 27 sites along the Derwent Estuary was repeated to assess decade-scale changes in the abundance and distribution of fish, invertebrates, and algal communities on intertidal to subtidal reefs and adjacent sediment habitats.

Aims and objectives

The 2024 study replicates biological surveys completed in 2010, establishing a framework for long-term decadal monitoring in the Derwent Estuary. This monitoring enhances understanding of native species responses to changing environmental conditions, while also tracking the distribution and abundance of introduced pest species.

This study aimed to:

1. Provide further data on the spatial distribution and abundance of species and benthic communities,
2. Quantitatively describe changes in historically abundant and newly dominant species, and,
3. Assess population changes through time for key introduced species.

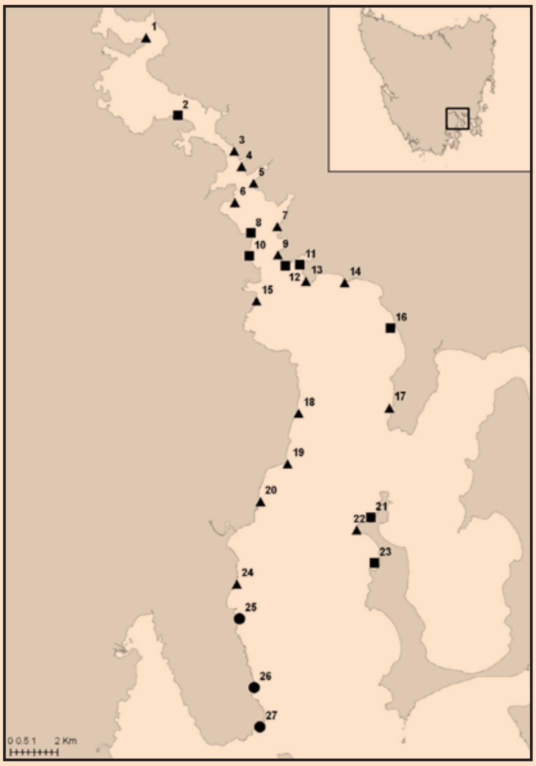
Findings from this survey are critical for detecting shifts in reef community dynamics, evaluating management effectiveness, informing targeted restoration, and guiding future monitoring.

Methods

This study utilises a mix of methods to comprehensively examine patterns of biodiversity in a cost-effective but robust way. The first method is fixed-depth diver transects along transect lines, counting fish and mobile invertebrates and estimating the cover of algae and attached invertebrates. This is a nationally adopted standard measure as part of the Australian Temperate Reef Collaboration (**https://atrc.au/**). These surveys were conducted at 19 sites along the Derwent

Estuary (Figure 9.4). To further assess diversity and rank abundance of key species across a wider range of habitats from intertidal rocky shores to deeper reef margins and associated sediments, replicated timed intertidal and subtidal surveys were undertaken across various depth ranges at 24 sites. These surveys were typically conducted along reef systems, but where possible, soft sediment and ecotone search (comprising of patch reef and open sediments) were completed to capture the habitat of known rare species.

Figure 9.4 General map showing locations of sites surveyed in the Derwent Estuary in February – April in 2010, and March – June 2024. All sites were surveyed using the same methods in both survey periods. ● standard surveys; ■ timed surveys; ▲ standard and timed species surveys.

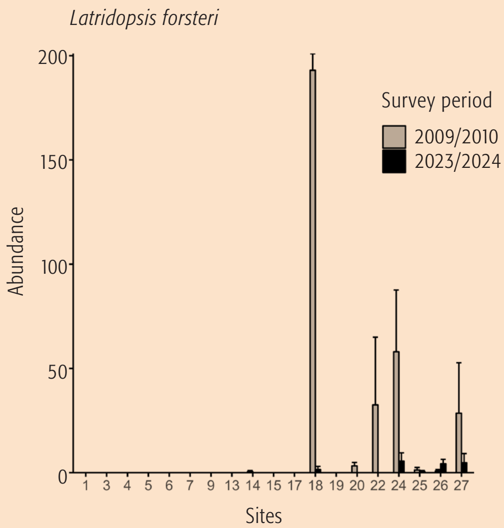


Results and discussion

Fish

The fish community composition remained broadly consistent with observations from 2010, with seven species overlapping amongst the most abundant in both surveys: Longfin Pike (*Dinolestes lewini*), Variable Threefin (*Forsterygion varium*), Bastard Trumpeter (*Latridopsis forsteri*), Bluethroat Wrasse (*Notolabrus tetricus*), Bigscale Bullseye (*Pempheris multiradiata*), Senator Wrasse (*Pictilabrus laticlavius*), and Southern Hulafish (*Trachinops caudimaculatus*). While site-level variability was present, changes in richness between years were minimal, across both survey methods. Of concern is the marked decline of Bastard Trumpeter from 883 individuals in 2010 to 63 in 2024 on quantitative transects (Figure 9.5). To examine whether this decline reflects natural variation in recruitment or human influence, we contrasted these results with unpublished monitoring data from nearby protected sites within the Tinderbox Marine Reserve. There, numbers declined from a total of 52 to 38 individuals sighted on transects between 2010 and 2024, a much smaller decline than observed in the Derwent Estuary sites. Anecdotally, recreational spearfishing in the Derwent has markedly increased over this period, and when combined with observed sequential recruitment failure over multiple years, this could be driving the sharp decline in Trumpeter numbers observed in 2024.

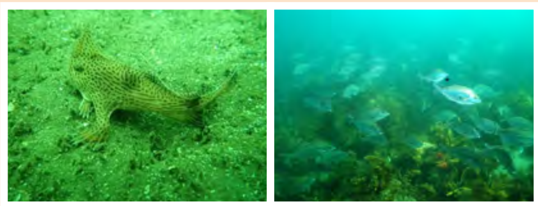
Figure 9.5 Abundance of Bastard Trumpeter (*Latridopsis forsteri*) at Derwent Estuary survey sites in 2010 and 2024. Count on Y axis is number per 2,000 m^2. Sites from Figure 9.1 are shown on X axis.



New range-extending species were observed in 2024, including Mado Sweep (*Atypichthys strigatus*) at multiple sites and Snapper (*Chrysophrys auratus*) at Battery Point, reflecting the growing influence of coastal warming. While another range-extending species, the Yellowtail Kingfish (*Seriola lalandi*), was not recorded in surveys, the species was present in high numbers, with shore and boat fishing activity limiting access to several sites. Collectively, these observations of warmer-water species provide an important indication of climate-driven range expansion within the Derwent Estuary.

Increased observations of Spotted Handfish (*Brachionichthys hirsutus*) on timed swims over reef-adjacent sediments are a promising indication of potential habitat and population recovery and a reflection of the ongoing conservation efforts targeting this highly range-restricted species.

Figure 9.6 Spotted Handfish (*Brachionichthys hirsutus*) (left image) observed at Bellerive Bluff (site 13) in 2024. Bastard Trumpeter (*Latridopsis forsteri*) schooling at The Grange (site 18) in 2010.



Cryptic fish

Cryptic fish species richness and overall abundance both increased in 2024, potentially responding to less extreme environmental conditions in the lead up to that survey. Estuaries are often highly dynamic, with irregular major flood events leading to radical redistribution of many marine species, that gradually recover their range in subsequent years. The introduced Variable Threefin tripled in abundance between surveys, including extending its upstream range. Likewise, another introduced fish, Estuarine Threefin (*Forsterygion gymnotum*), found only in the Derwent, also increased in abundance and expanded its range, now occurring at nine sites in 2024, compared to only two in 2010.

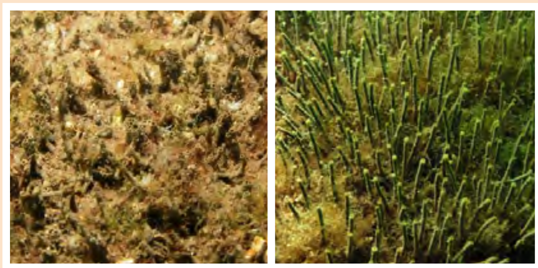
Mobile macro-invertebrates

Species richness of mobile macro-invertebrates was comparable to 2010, with no clear increases or spatial trends observed across the estuary. The introduced New Zealand Common Cushion Star (*Patiriella regularis*) remained the most abundant macro-invertebrate species, with considerably higher numbers recorded in 2024 – more than double the 2010 counts at many sites. The introduced North Pacific Seastar (*Asterias amurensis*) also showed a marked twentyfold increase in abundance over this time on the reef-based transects. As per the standard surveys, on timed surveys macro-invertebrate species richness increased seaward from the upper estuary sites, particularly in intertidal habitats. In addition to showing similar species-related patterns to the transect surveys, the wider variety of habitats and depths covered allowed further insights. One example is that the Native Oyster (*Ostrea angasi*) appears to have become more prevalent at many sites, including sites north of site 15 (particularly sites 3, 5, and 7). Potentially indicating some population recovery of this species. However, further taxonomic examination is planned and necessary to confirm that this result is not a misidentification with introduced Pacific Oyster (*Crassostrea gigas*) that can appear similar in some environments.

Algae

Algal species richness and spatial distribution patterns were broadly similar between surveys, with higher richness consistently observed in the lower estuary in both 2010 and 2024. However, percent cover of algae increased dramatically in 2024 for several sites, spanning from Bedlam Walls North (site 3) to Battery Point (site 15) – locations that were largely dominated by tubeworm matting in 2010, representing an alternate stable state to Kelp beds (Figure 9.7).

Figure 9.7 Tubeworm matting present in 2010, dominating several sites (from site 3 – 15: Bedlam Walls North to Battery Point).

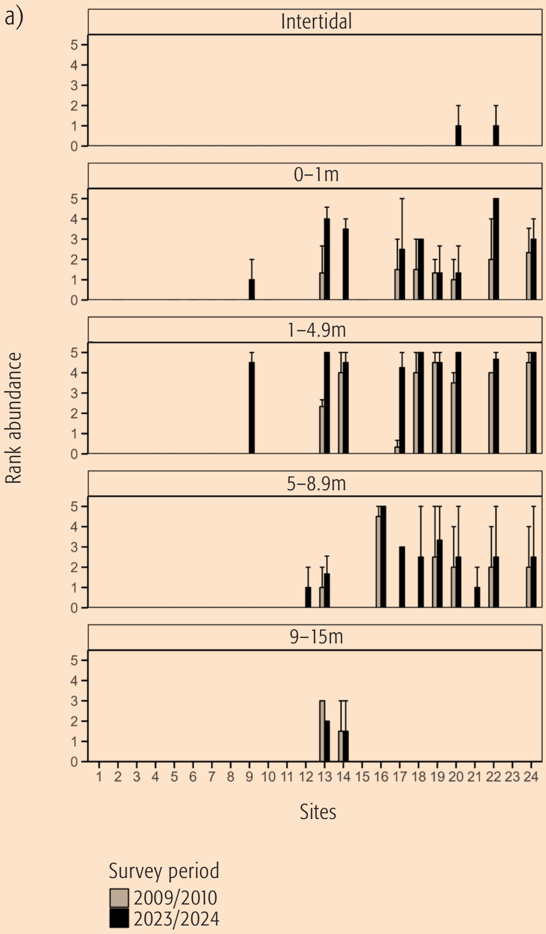


One of the most notable observations was that the common Golden Kelp (*Ecklonia radiata*) expanded its distribution northwards into areas where it had not been previously recorded. This shift–alongside the complete disappearance of an extensive tubeworm matting that covered/smothered many shallow reefs between Bedlam Walls North and Battery Point–indicates a substantial ecological transition. Golden Kelp can be used as an indicator for water clarity and quality, and as such, improved environment conditions. We found this species to have increased in both rank abundance and an expanded depth range in 2024, particularly within 0–1 m, 1–4.9 m and 5–8.9 m subtidal zones surveyed on timed swims (Figure 9.8). Notably, Bellerive Bluff and Bellerive/Howrah Bluff increased from a rank of 1–2 (≤10 individuals) in 2010, to near-complete coverage (rank 4–5; 101–1000+ individuals) in 2024.

This expansion in kelp coverage at deeper depths, potentially due to improved water quality, and therefore, greater light penetration is strongly coupled with the disappearance of widespread tube-matting previously recorded between Bedlam Walls and Battery Point. That matting excluded other species from occupying that space, with such matting known to be typically found in the middle sections of highly organically enriched estuaries. Its overall disappearance suggests a significantly positive ecological shift has occurred, allowing signs of kelp recovery in the estuary.

Figure 9.8 a) Rank abundance (mean ± SE) of Golden Kelp (*Ecklonia radiata*) recorded during timed swims in the Derwent in 2010 (grey) and 2024 (black) at different depths. Abundance score; 1 = only 1 individual, 2 = 2–0, 3 = 11–100, 4 = 101–1000, 5 = 1000+. b) Image of healthy, abundant Golden Kelp, taken during the 2024 survey of site 14 (Bellerive/Howrah Bluff). This site showed considerable increases in Kelp in 2024, particularly at the 0–1 m depth (rank = 0 in 2010, versus rank = 4 in 2024).

a)



b)

Introduced species

As per the 2010 survey, in 2024, introduced species richness were a dominant part of the biota observed, most notable in the upper sites to middle sites extending to Battery Point and Rosny Point peaking in abundance around the Hobart port region. Native species were more diverse in the lower estuary, closer to the marine extreme. The significant numbers recorded for both introduced seastars, New Zealand Common Cushion Star and Northern Pacific Seastar, are of concern with observations markedly increasing for both species over the decade period. This significant increase, particularly across sites 1–15 (Cadbury Point to Battery Point), reinforces the dominance of introduced species in degraded areas and highlights the need for continued monitoring and management of invasive species. While the seastars were the most conspicuous component of the introduced biota, other elements were also recorded. This includes the red algae *Grateloupia turuturu* that notably increased in percent cover and spatial distribution by 2024, occurring on intertidal to immediate subtidal reefs at six new sites across the estuary. Importantly, this rise, and that of other introduced species, including the fishes Variable Threefin and Estuarine Threefin, has not been matched by a corresponding decline in native species abundance or cover, suggesting that coexistence, rather than replacement, may be occurring at some sites.

In contrast, other introduced species showed no significant increases in abundance and exhibited minimal range shifts. For example, the large introduced brown seaweed Wakame (*Undaria pinnatifida*) remained at low levels and in the outer estuary sites, as does the red algae *Aeodes nitidissima*, located in a restricted band near the Tasman Bridge. While the European Shore Crab (*Carcinus maenas*) was a recent arrival to the Derwent Estuary at the time of the 2010 study and was anticipated to both spread more widely and increase its abundance in following years, there has been no evidence of this during the recent survey. The New Zealand Half Crab, (*Petrolisthes elongatus*) remains the most abundant and well-established introduced crab in the system; however, this is primarily an intertidal species, so the threat is mostly constrained to that habitat. Interestingly, the Pacific Oyster appears to have declined by more than ten-fold at nearly all sites and was absent from all ecotone habitats in 2024. However, expert validation is planned in upcoming months to determine if this was a real effect or a result of misidentification as Native Oysters in habitats where they can look similar.

Recommendations and management considerations

Key observation from the 2024 survey of decadal changes in the Derwent Estuary included: 1) a significant recovery of Golden Kelp and associated algal species resulting from polychaete matting loss in the middle estuary; 2) a marked decline in Bastard Trumpeter numbers throughout the estuary; and 3) a significant increase in abundance of the introduced seastars.

Changes to kelp cover and polychaete matting are likely in response to decreasing organic pollution and associated increasing light levels in the estuary in response to improved management of water quality. Such improvements need to be maintained and increased where possible. For fish declines, most notably around Trumpeter, some changes are likely environmental, but magnified by human impacts. The biological and recreational (snorkelling) value of Derwent Estuary reefs could be enhanced through increased education and controls around the impacts of recreational spearfishing on Bastard Trumpeter and other fish populations, that would also increase public safety in high use areas. For introduced species, ongoing decadal monitoring of Northern Pacific Seastar and other introduced seastars is essential to understand the magnitude of problems these species cause and inform and support targeted responses, such as volunteer invasive seastar removal in areas where Spotted Handfish are particularly vulnerable.

Further research into the impacts of vessel activity on reef diversity (e.g. wave-driven effects on sediment resuspension) would support management of this busy waterway. This summary provides a snapshot of key findings and considerations from the 2024 rocky reef survey, with a comprehensive analysis and detailed management recommendations to be presented in the forthcoming full report.

9.4 Giant Kelp

Coastal waters of Tasmania represent the most extensive Giant Kelp (*Macrocystis pyrifera*) habitat in Australia. Giant Kelp beds form a closed and semi-closed surface canopy representing a distinctive and important habitat type on shallow Tasmanian subtidal reefs. Review of aerial photographs found that canopy cover of Giant Kelp has declined by approximately 90% along Tasmania's east coast (Johnson *et al*., 2011). Giant Kelp forests of southeast Australia are listed as Endangered under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), the first listing of a marine community in Australia. A 2019 aerial survey of Giant Kelp along the east coast of Tasmania identified areas of Giant Kelp cover in the fringing reef areas of Blackmans Bay, Tinderbox, Cape Deliverance and Cape Direction out to the Iron Pot in the lower reaches of the Derwent Estuary.

Giant Kelp beds face compounding threats from climate change in coastal areas of Tasmania. This species is sensitive to increasing sea surface temperatures, typically not surviving or establishing in waters above 20°C (Cavanaugh *et al*., 2019). Canopy cover of Giant Kelp forests have been linked to El Niño – Southern Oscillation cycles and sea surface temperatures, with all peaks in canopy cover found to occur in strong El Niño years (Butler *et al*., 2020), which in south-east Australia is characterised by cooler than average sea surface temperatures and reduced storm activity (Harris *et al*., 1988). The establishment of Longspined Sea Urchin (*Centrostephanus rodgersii*) in coastal waters of Tasmania, attributed to range expansion of the East Australian Current (EAC), has also been detrimental to Giant Kelp forests. These urchins can overgraze macroalgae to form urchin 'barrens' devoid of Giant Kelp and associated biodiversity (Johnson *et al*., 2005; Ling *et al*., 2008), impacting on commercially important species including abalone and rock lobster (Johnson *et al*., 2011).

In late 2022, the Institute for Marine and Antarctic Studies (IMAS) began a project to review, design, trial and investigate viability of offshore commercial production of Giant Kelp. The site of this trial was Tinderbox, located at the mouth of the Derwent Estuary. Initially the project team designed and trialled offshore engineering systems for kelp production. Three different cultivars (kelp 'families' previously tested in the lab for thermal tolerance, and thus, the potential to cope with ocean warming) were deployed to assess whether certain cultivars performed better than others. A range of measurements were taken to assess growth rates and performance including plant size, biomass and density along grow-lines. A range of environmental variables (including light, temperature and nutrients) impacting growth and plant health were also assessed.

The results of the project were a resounding success. Approximately 3.6 t of Giant Kelp was grown at the Tinderbox site from outplanted hatchery-reared microscopic juveniles. Across the site, the average plant size ranged from 1.6 m to a maximum length of 3.4 m, grown in just six months, highlighting the potential for growing Giant Kelp as an annual crop (Figure 9.9). Key learnings from this work include a high sensitivity in Giant Kelp to light and temperature, which limit its production within the Derwent Estuary to shallow depths and only 7 – 8 months of the year to avoid warm summer periods which could kill crops. Optimal depths for outplanting at offshore sites are recommended to be between 8 and 9 m.

Figure 9.9 Giant Kelp (*Macrocystis pyrifera*) grown from outplanted hatchery-reared juveniles at Tinderbox. Image: Victor Shelamoff, IMAS.

9.5 Aquatic macrophytes and seagrass

9.5.1 Upper estuary macrophyte and seagrass monitoring

Aquatic macrophytes are a diverse group of plants whose lifecycle takes place entirely or periodically in the aquatic environment, they include green macroalgae, charophytes, bryophytes and vascular plants (Lesiv *et al*., 2020). Seagrasses are aquatic angiosperms (flowering plants) that have evolved from terrestrial plants, with roots, stems and leaves (Hemminga and Duarte, 2000). Macrophyte and seagrass beds form the foundation of highly productive regions, and important habitats in coastal marine and estuarine ecosystems worldwide. As primary producers, they provide essential ecosystem services including habitat provisioning, nutrient cycling, water-column filtration, sediment stabilisation, coastal protection and sequestration of atmospheric carbon (Orth *et al*., 2006; Nordlund *et al*., 2018; do Amaral Camara Lima *et al*., 2023).

Macrophyte beds are sensitive to environmental change with many species declining in coastal areas as a result of increased urbanisation and disturbance. Anthropogenic impacts, such as coastal development, poor water quality and climate change have contributed significantly to habitat loss (Waycott *et al*., 2009). Decreased light availability as a result of increased epiphytic algal growth and/or turbidity is widely regarded as a key driver of decline (Burkholder *et al*., 2007).

In the Derwent Estuary, the brackish aquatic macrophyte beds of the upper and middle estuary cover an area of over 600 ha. These communities serve as major benthic primary producers and provide critical habitat for recreationally and ecologically important fish species including Black Bream (*Acanthopagrus butcheri*), Brown Trout (*Salmo trutta*), whitebait, Yelloweye Mullet (*Aldrichetta forsteri*) and the threatened Australian Grayling (*Prototroctes maraena*). These macrophyte beds also an important food resource for large Black Swan (*Cygnus atratus*) populations and other waterbirds. In addition, the beds oxygenate the water column, and settle suspended sediment, likely contributing to the continued burial of metal-contaminated sediments in the region (Wild-Allen *et al*., 2009a, 2011).

However, these macrophyte beds face mounting pressures from multiple sources. Elevated turbidity from catchment runoff, particularly during winter, reduces light availability. Nutrient inputs from the catchment enhance primary productivity and promotes algal overgrowth, further smothering macrophyte beds. The organic-rich sediments in the estuary may exacerbate sulphide intrusion into plant rhizomes, potentially leading to mortality (Frederiksen *et al*., 2007). Observations of increasing algal blooms and smothering events (DEP, 2020a; Marine Solutions, 2021), along with intensifying marine heatwaves, raise concern for the long-term resilience of these systems.

In 2015, following significant algal smothering and macrophyte bed dieback, the DEP established a seagrass condition monitoring program. The key objectives of the program are to:

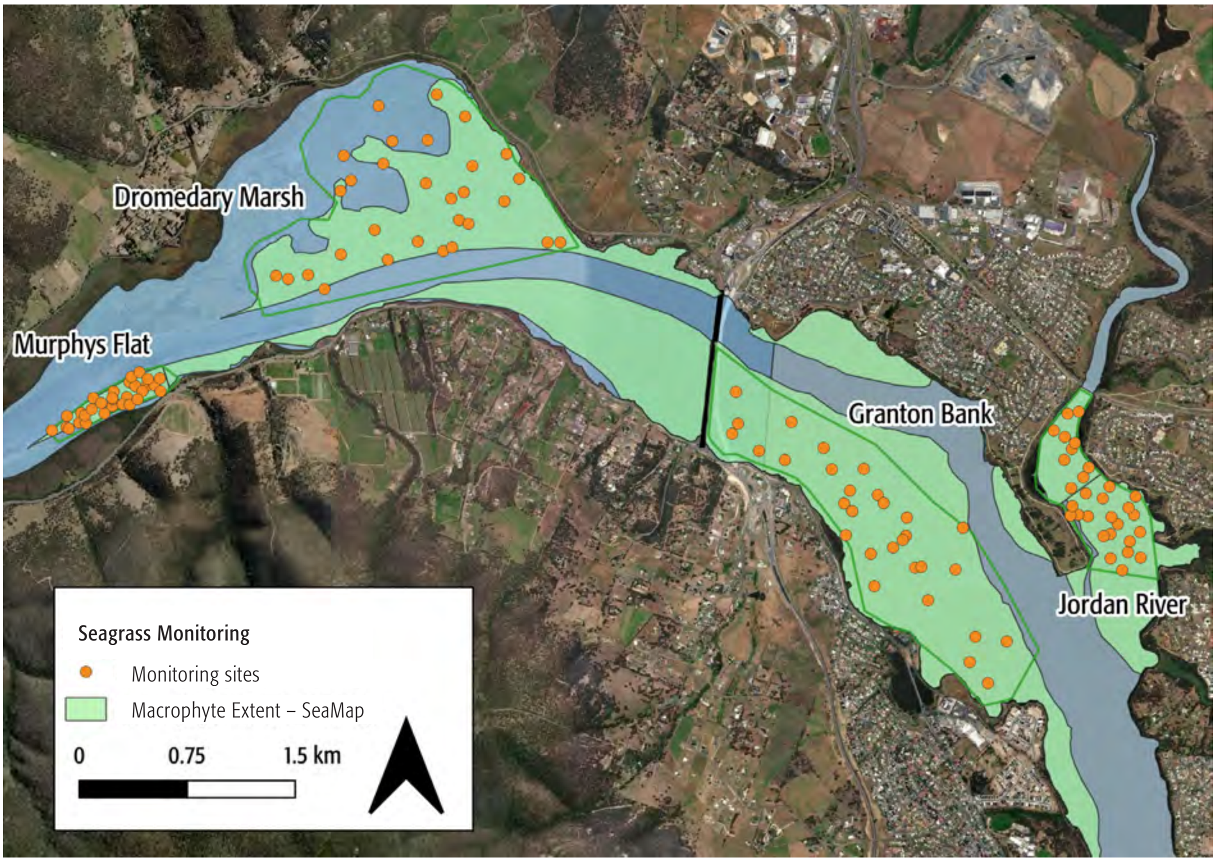
- Examine spatial and seasonal fluctuations in macrophyte meadow condition;
- Understand factors influencing macrophyte condition; and,
- Better understand species composition and distribution.

This section summarises data for the period 2015 – 2024.

Methods

Surveys were conducted at four macrophyte beds in the lower section of the upper estuary (Murphys Flat and Dromedary Marsh) and upper section of the middle estuary (Granton Bank and Jordan River). Thirty sites were randomly selected at each meadow, with sites located and logged on a Garmin GPS 72H in the field (Figure 9.10).

Figure 9.10 Location of macrophyte monitoring sites at the Murphys Flat, Dromedary Marsh, Granton Bank and Jordan River macrophyte beds. Approximate macrophyte bed extent shown in green.



Macrophyte beds were surveyed up to four times a year (once per season) between 2015 and 2024. From December 2019 to December 2020, sites were reduced to 15 per bed and subsequently returned to 30 from December 2021 onwards. Due to operational constraints, boat availability and weather conditions, frequency of monitoring has been closer to two events per year in recent years.

Images were captured using a GoPro Hero 4 (4K UHD Camera, 12mp) mounted to a 0.5 x 0.5 m metal quadrat frame. The frame was deployed upright onto the bed for 30 seconds whilst images were collected at 10s intervals. Photos were taken approximately 0.5 m above the seabed.

The photo quadrat images were analysed using the freeware package, Coral Net. A randomised 30-point scatter was overlayed over each image and each point scored for macrophyte, algae or sediment cover from 2015-2020. From 2020, the method changed to a 49-point grid to score the images in order to standardise with other research groups. Scores for each quadrat were converted to a percentage to calculate percent cover.

Results

Species composition

The four beds are typically dominated by the rare aquatic macrophyte *Ruppia megacarpa*, which is distributed from 0–3 m (Jordan *et al*., 2001; Lucieer *et al*., 2007). This species is most dominant on the shallow banks where salinity is lower. Additional aquatic macrophyte species have been observed, particularly at Dromedary Marsh and Murphys Flat where diversity is greatest. These species include *Ruppia polycarpa*, *Lepilaena cylindrocarpa* and *Myriophyllum* spp.

In 2024, the rare aquatic macrophyte Lepilaena patentifolia and the charophyte Nitella hyalina were observed at Dromedary Marsh (pers comms, R. Schahinger, 2024). These observations were significant as it is only the second documented observation of *Lepilaena patentifolia* in the Derwent (the first being in 1976), and the second observation of *Nitella hyalina* in Tasmania. In the Jordan River mouth bed, the rare aquatic macrophyte *Stuckenia pectinata* is sparsely interspersed amongst *Ruppia megacarpa*. The brackish seagrass, *Zostera muelleri* is restricted to small patches on the edges of the river channel (Marine Solutions, 2021).

At times (Spring/Summer), these beds can be covered by growth of dense epiphytic algae. Lawler (2009) and (Mount, 2011) found the upper Derwent macrophytes were in good condition with less than 10% of macrophytes covered by algae. DEP observed dense algal smothering in summer 2015/16 where up to 90% of beds were covered with epiphytic algae (DEP, 2020a). Algal epiphytes documented in these beds include *Gracilaria* sp., *Cladophora* sp., *Polysiphonia* sp., and *Ceramium* sp. (Lawler, 2009).

In 2018 and 2025, DEP collected epiphytic algae samples present at macrophyte monitoring sites in the middle to upper estuary for identification at the Tasmanian Herbarium (TMAG). The main epiphytic algal species present in both collections were *Chaetomorpha billardierii*, *Ulva intestinalis*, and *Gracilaria chilensis*. Specimens of *Gayralia oxysperma* and *Derbesia tenuissima* were also identified in the 2025 collection, however, they are believed to be less common. The epiphytic algal community abundance, distribution and dynamics in these macrophyte beds still remains relatively understudied.

Spatial and temporal variability

Macrophyte beds of the upper and middle Derwent Estuary cover an area of over 600 ha and are mostly continuous along the shallow bank areas either side of the estuary (Figure 9.10). The cover of macrophytes varies between beds and years, with Granton Bank bed generally having the greatest macrophyte cover, followed by Dromedary Marsh, and Murphys Flat. The Jordan Riverbed has historically had the lowest macrophyte cover; however, in recent years there has been an increase in cover at this site (Figure 9.11).

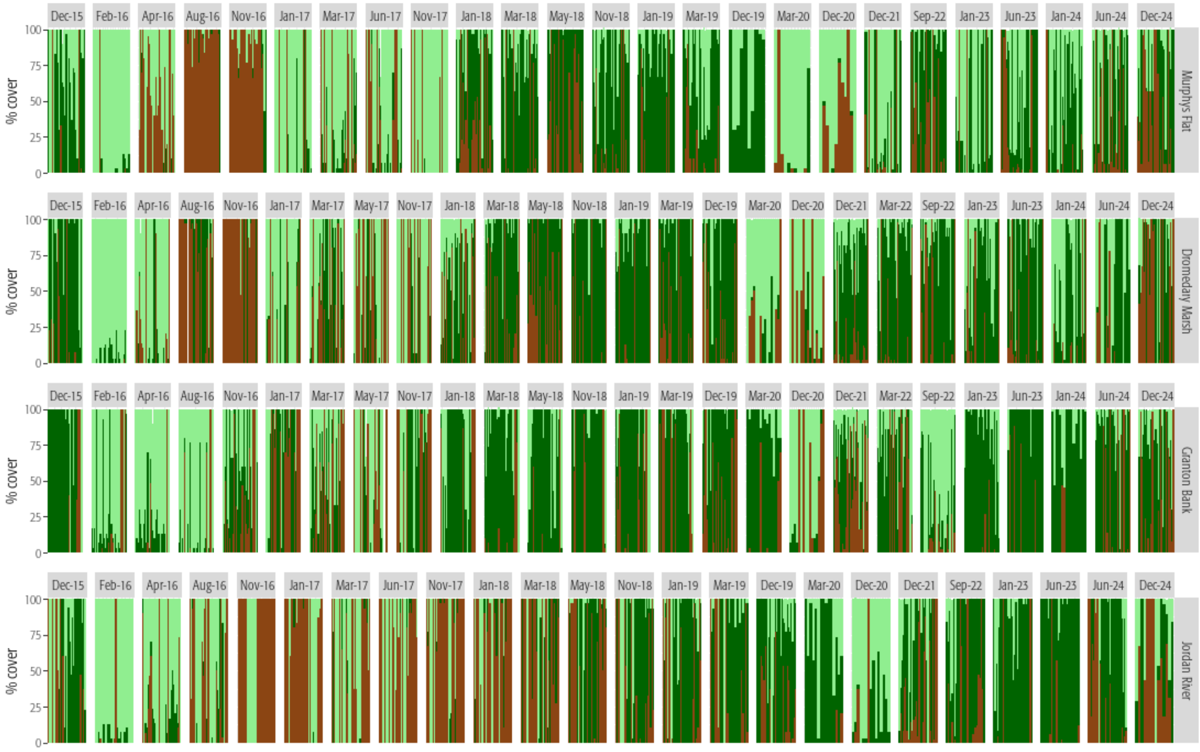
Long-term trends in macrophyte, epiphytic algae and sediment cover across monitoring events were similar across macrophyte beds. An epiphytic algal bloom that occurred in February – April 2016 resulted in widespread macrophyte dieback and increase in sediment cover in August – November 2016 for Murphys Flat and Dromedary Marsh, and November 2016 to January 2017 for Granton Bank and Jordan River. The dieback of macrophytes was not as widespread at Granton Bank compared to the other beds, perhaps reflecting the greater density of macrophyte cover. The cause of this algal bloom and dieback was likely due to a significant warming event that happened in the estuary in 2015, followed by a marine heatwave that occurred in coastal waters of southeast Tasmania in 2015/16 (Roach and Holbrook, unpublished) (**Section 2.4.6**).

Prolonged elevated temperatures can enhance the growth of epiphytic algae, particularly if associated with other optimal environmental conditions including nutrient availability and light conditions (Kim *et al*., 2021). Marine heatwaves can also result in heat stress die-offs of estuarine macrophytes (Shields *et al*., 2019). Re-establishment of macrophytes occurred through 2018, with macrophyte cover peaking in 2019.

High algal cover was recorded across all macrophyte beds in March to December 2020, with the densest algal cover observed at Murphys Flat and Dromedary Marsh, with macrophytes recovering at these beds by September 2022, and January 2023 at Granton Bank and Jordan River. Dense algal growth affecting macrophyte beds is generally first observed in the upstream beds of Murphys Flats and Dromedary Marsh, before being observed downstream in Granton Bank and Jordan River months later. This indicates that environmental influences causing the rapid growth of epiphytic algae in these macrophyte beds are derived upstream of Murphys Flat. Since 2019, there has been an increase in nutrients in the upper estuary, Mann-Kendall trend analysis found there to be a significant positive trend in total nitrogen and dissolved phosphorus in bottom waters, and total nitrogen and total ammoniacal nitrogen in surface waters at U12 Ambient Water Quality Monitoring site located near Murphys Flat and Dromedary Marsh (See **Section 6.1**). Increased nutrient loads in this shallow region of the upper estuary are likely impacting macrophyte and epiphytic algal growth regimes across these beds, potentially favouring fast-growing ephemeral algae that thrive in higher nutrient conditions (Raven and Taylor, 2003).

These results suggest that despite great stressors, and dieback events *Ruppia* spp. and *Zostera* sp. present in the upper and middle estuary can recover and re-establish to dense coverage. This is likely reflective of their growth strategies where plants have multiple mechanisms of re-establishment, including rhizomes, vegetative fragments and seed germination. Good recovery of these species from disturbance has been observed in previous studies (see Macreadie *et al*., 2014; O'Dea, 2023). These studies found that disturbance intensity greatly influenced the recovery time of macrophyte beds, and neither observed recovery by seed germination. Therefore, with increasing pressures from catchment-derived nutrient loads, and large disturbance events, such as marine heatwaves, these upper estuary macrophyte beds could be at risk from compounding events, with limited recovery time until the next event, macrophyte health and coverage could decline.

Figure 9.11 Percent cover of macrophyte (dark green), epiphytic algae (light green) and sediment (brown) for each monitoring site (n=30) at Murphys Flat, Dromedary Marsh, Granton Bank and Jordan River from December 2015 to December 2024. Note that from December 2019 to December 2020 only 15 of 30 sites were sampled at each meadow.



Conclusions

- Granton banks has the greatest macrophyte cover, and macrophyte coverage at Jordan River has improved significantly in recent years.
- An extreme warming event and marine heatwave in 2015/16 was likely the cause of large algal blooms and significant die-back of macrophyte beds in the upper and middle estuary macrophyte beds.
- Murphys Flat tends to be the first site impacted, and a good indicator site for algal blooms, suggesting that conditions causing the rapid growth of algae are derived from upstream of this bed (catchment derived).
- Macrophyte beds of the upper and middle estuary are resilient and able to recover from increased stressors, and even dieback.

Recommendations

- Reinstate seasonal monitoring (four times a year) of macrophyte beds.
- Scope a project to attribute environmental factors influencing rapid and widespread growth of epiphytic algae in upper and middle estuary meadows.
- Full ecological assessment of macrophyte beds to quantify the abundance and distribution of macrophyte bed-associated biota to better understand the ecological significance of these habitats in the upper Derwent Estuary. This assessment will provide a vital baseline against which future environmental changes can be measured.
- Update the seagrass natural capital account based on macrophyte-extent monitoring.

9.5.2 Seagrass restoration in the Derwent: Identifying best practice restoration methods

Provided by Kelsie Fractal, IMAS, Vere Michiels, UTAS, Associate Professor Dr Jeff Wright, IMAS, Dr Elisabeth Strain, IMAS and Centre for Marine Socioecology, UTAS

Globally, seagrasses are in decline, with total losses estimated at 19% (Dunic *et al*., 2021) and approximately 5% in Australia (Statton *et al*., 2018), however, these estimates are likely to underestimate the scale of loss as many areas around Australia and the globe are insufficiently mapped (McKenzie *et al*., 2020). Losses can result from a range of human-induced pressures, including coastal development, dredging, land reclamation, and boat mooring scars. Natural disturbances such as storms also contribute. However, declining water quality from wastewater, stormwater, and agricultural runoff is one of the leading causes of habitat degradation (van Katwijk *et al*., 2016).

In Tasmania, reports suggest seagrass biomass has increased in some areas over the past decade (IMAS, unpublished data). However, other areas have experienced significant declines since 1950 (first suitable aerial imagery available) with no evidence of recent recovery. In Ralphs Bay, located in the lower Derwent Estuary, historical records indicate complete loss of 4.3 km^2 of seagrass before 1990 (Rees, 1993). Historic high nutrient loading in the Derwent Estuary (DEP, 2015c), which included a wastewater treatment outlet into Ralphs Bay may have contributed to seagrass loss but the exact cause is unknown. However, general improvements to nutrient levels in the estuary, and upgrades to the Rokeby wastewater outfall have since occurred and previous modelling work suggests that ambient nutrient levels and turbidity are low enough to support seagrass growth (Wild-Allen *et al*., 2009b).

In 2022, a study investigating the effects of stormwater on seagrass in the Derwent Estuary found that light availability and sediment characteristics (sediment nutrient levels and particle size distribution) in Ralphs Bay were comparable to those in healthy seagrass meadows elsewhere in the estuary (Fractal, 2022). However, only small remnant patches of *Zostera muelleri* and *Heterozostera nigricaulis* were observed (Fractal unpublished data) suggesting a potential bottleneck to natural recovery. Limited connectivity to seed sources may be a contributing factor, indicating that active restoration may be required to support seagrass recovery in the area.

A variety of seagrass restoration techniques have been trialled across Australia and internationally, broadly classified into two categories:

1. Transplant-based approaches, which involve relocating whole plants from donor meadows; and
2. Seed-based approaches, which involve collecting and deploying seeds at restoration sites.

Transplant-based restoration has historically explored factors such as planting unit size and anchoring methods. More-recently, seed-based techniques have gained interest due to their reduced impact on donor meadows and greater scalability. Drawing on collaboration with other Australian seagrass researchers and a review of the available literature, four restoration methods were selected and adapted for local environmental conditions and the target species, *Zostera muelleri*.

Three restoration sites around Ralphs Bay (Figure 9.12) were selected for trial based on environmental suitability, hydrodynamic condition, proximity to nutrient sources, and site accessibility.

Figure 9.12 The Derwent Estuary, showing Ralphs Bay three restoration sites and the donor meadow.

At each site, the pilot project tested four restoration techniques (Figure 9.13) to determine the most effective and scalable method for restoring *Zostera muelleri*:

1. Core transplants: 80 mm diameter cores of seagrass with intact sediments and roots.
2. Sprig transplants: 3–5 seagrass shoots with roots and rhizomes, sediment removed, secured with bamboo stakes.
3. Seeds bags: hessian bags (7 x 13 cm) filled with site sediments and 50 seagrass seeds.
4. Dispenser Injected Seeding: injection of a seed and sediment slurry into the substrate using a caulking gun.

In 2023, seeds and transplant material were collected from a healthy donor meadow at Knights Reserve in Claremont. The trial was deployed along the low tide line in 30 quadrats (0.5 x 0.5 m) at each of the three restoration sites. To serve as a method control and assess site recovery following transplant removal, quadrats at the donor site were cleared and replanted with each of the four methods.

Figure 9.13 Restoration methods used in the Ralphs Bay trial: a) transplanting cores, b) transplanting springs, c) dispenser injected seeding, d) seed bags. Images: Kelsie Fractal.

Sites have been monitored seasonally for seed germination and seedling emergence, growth (leaf length) and percent cover. In addition, data were collected on the cost and time required for each restoration technique. Trial outcomes will be reported in 2026.

This pilot trial forms part of Kelsie Fractal's PhD research and is a collaboration between IMAS and OzFish Unlimited, with funding support from QBE insurance.

9.6 Saltmarshes and wetlands

Coastal saltmarshes and wetlands occur in low-lying, sheltered, low-energy, upper intertidal environments in estuarine and coastal areas (Prahalad and Pearson, 2013). They are exposed to both saltwater influences from tidal inundation, and freshwater from rainfall, groundwater, creeks or rivers. They are dominated by halophytes (salt tolerant plant species) including herbs, grasses, small shrubs, sedges and rushes (Adam, 1990). Saltmarshes provide many important ecosystem services including; flood mitigation and coastal protection (Shepard *et al*., 2011), improving local water quality (Mitsch and Gosselink, 2015), providing habitat for a range of different organisms including fish, macro-invertebrates, insects, birds and mammals (Barbier *et al*., 2011), sequestering atmospheric carbon (Chmura *et al*., 2003), and are valued for human recreation (Adam, 1990; Clarke *et al*., 2021). Saltmarsh communities occupy an area of over 450 ha within the Derwent Estuary (Prahalad and Jones, 2013), and can be found throughout the estuary's upper, middle and lower reaches.

Given that coastal saltmarshes occur in low-lying, protected embayments with surface and ground water sources, they are particularly vulnerable to a range of threats. It is estimated that approximately 50% of historic saltmarsh area in south-east Tasmania has been lost (Prahalad *et al*., 2012). Loss of saltmarshes has occurred through land reclamation activities including infilling, and construction of levees and drains for coastal urban and farming development. They are vulnerable to changes in freshwater flow regimes through altered catchment flows from dams, diversions, and irrigation schemes (Carreño *et al*., 2008), as well as catchment land-use modification impacting water quality, nutrient supply (Boorman, 2019) and sediment (Ganju *et al*., 2013; Weston, 2014). Saltmarsh vegetation can be damaged by livestock grazing, trampling and compaction (Koppenaal *et al*., 2022), and introduced plant species can outcompete native vegetation, reducing available habitat for fish (Harrison-Day *et al*., 2022). Saltmarsh areas can be highly impacted by anthropogenic stressors including vehicle access, pollution and dumping of rubbish (Thomsen *et al*., 2009). They are also vulnerable to associated impacts from climate change, including sea level rise (Saintilan *et al*., 2023, 2024), reduced rainfall and runoff, and rising temperatures (Saintilan *et al*., 2019). Therefore, subtropical and temperate saltmarsh communities are listed as Vulnerable under the EPBC Act.

Saltmarsh communities support a range of threatened and unique biodiversity within the Derwent Estuary. Several species of state-listed *Threatened Species Protection Act 1995* (TSP Act) threatened flora can be found in Derwent Estuary saltmarsh environments, including Golden Dodder (*Cuscuta tasmanica*), Slender Buttons (*Cotula vulgaris var. australasica*) and Narrowleaf Blown Grass (*Lachnagrostis punicea subsp*. *filifolia*) (DEP, 2010c). Other rare flora includes Candle Saltmallow (*Lawrencia spicata*) and Many-stemmed Bluebell (*Wahlenbergia multicaulis*) (North Barker, 2008). Doran's Road saltmarsh, Lauderdale is critical habitat for the endemic Tasmanian Saltmarsh Looper Moth (*Dasybela achroa*), which is listed as Vulnerable under the TSP Act, with only a few specimens of this moth being observed outside of this area (DEP, 2015b).

Saltmarshes are important habitat for many resident and migratory bird species (Spencer *et al*., 2009). They provide a range of important services to birds including foraging, roosting and nesting habitat, and indirect benefits such as improving water quality, and protection from predators and disturbance (Prahalad *et al*., 2015). Saltmarsh environments within the Derwent Estuary are found to support a broad range of bird species including waterbirds, shorebirds, and passerines (Visby and Prahalad, 2020).

Samphire or Beaded Glasswort (*Sarcocornia quinqueflora*) at Windermere Bay. Image: Ellie Green.

A recent study conducted by (Harrison-Day *et al*., 2024) investigated fish use of microtidal saltmarshes in southeast Tasmania across four seasons. The sites sampled were Marion Bay, Barilla Bay, and Ralphs Bay saltmarsh in the Derwent Estuary. Marion Bay and Barilla Bay recorded higher densities of fish than have previously been found in Australian saltmarshes, re-enforcing findings from other Tasmanian saltmarshes (Prahalad *et al*., 2018; Harrison-Day *et al*., 2022).

Ralphs Bay saltmarsh recorded lower fish densities and species richness compared to the other sites. Four common Tasmanian estuarine fish species were captured: Smallmouth Hardyhead (*Atherinosoma microstoma*), Yelloweye Mullet (*Aldrichetta forsteri*), Congolli (*Pseudaphritis urvillii*) and Greenback Flounder (*Rhombosolea tapirina*). The lack of adjacent seagrass cover, and estuarine geomorphology were thought to influence overall fish abundance at this site. Results from Harrison-Day *et al*. (2024) show the importance of Tasmanian saltmarshes for fish habitat. They also emphasise the importance of considering adjacent habitats within a broader seascape context (Saintilan *et al*., 2007) in restoration efforts aimed at benefiting fish populations (see **Section 9.5.2**).

The upper Derwent Estuary contains large areas of tidally influenced freshwater wetland community, covering an area of over 250 ha (North Barker, 2008), this area includes the Murphys Flat Conservation Area and Dromedary Marsh. The vegetation is predominantly brackish wetland community, with large areas of freshwater Southern Reed (*Phragmites australis*), interspersed with saline grassland and rushland community. The upland areas are dominated by dry scrub of Woolly Teatree (*Leptospermum lanigerum*) community, with small patches of threatened riparian *Eucalyptus ovata – Callitris oblonga* woodland community (**Section 9.8**). The upper estuary wetland area has been found to contain higher plant diversity compared to other areas of saltmarsh in the Derwent Estuary, with less distinct vegetation boundaries and zonation (Prahalad and Mount, 2011). This area of wetland community is important habitat for migratory fishes and threatened Australian Grayling (**Section 10.3.5**), waterbirds, including the EPBC Act Endangered Australasian Bittern (*Botaurus poiciloptilus*) (**Section 10.5.1**), passerines and raptors. These wetland areas have been impacted by restricted hydrology and reduced river flows from damming and irrigation in the Derwent catchment, drainage, altered nutrients, sedimentation, weeds and agricultural grazing (Prahalad and Mount, 2011).

Saltmarshes were identified as a key habitat of the Derwent Estuary through Conservation Action Planning (CAP) undertaken in 2010. CAP has led to the DEP initiating and supporting extensive work to better map (Prahalad *et al*., 2009), monitor (Prahalad, 2012; Prahalad and Jones, 2013), and manage (Visby and Prahalad, 2020) saltmarsh communities within the Derwent Estuary. Visby and Prahalad (2020) identified the Windermere Bay saltmarsh area as a priority site for restoration, due to historic infilling and saltmarsh loss. The study concluded the old infill could be excavated, to facilitate passive saltmarsh regeneration and support landward migration with rising sea levels.

9.6.1 Windermere Bay Saltmarsh Restoration

Background

Windermere Bay, Claremont is located on the western banks of the Derwent Estuary, within the middle estuary zone. Fringing estuarine saltmarsh community occupies approximately 9,000 m^2 within the bay. The surrounding foreshore area is a public reserve, with a foreshore track network, including raised boardwalk through sections of the marsh. The area has high community value; the grassed areas and trails are popular with walkers and runners, there is an off-lead dog area, and popular swimming site Windermere Beach located at the southern end of the bay. There is an active local environmental group, Claremont Coast Care, who host regular weeding and planting activities. The bay has historical significance, being the site for army training camps in the First World War. The bay also holds cultural significance for Tasmanian Aboriginal people for resting and gathering, its northern and southern headlands offering protection from prevailing winds, and the saltmarsh and bay would have provided abundant food resources.

The saltmarsh at Windermere Bay was identified as a priority restoration area in the estuary, due to the impacts from historic infilling (Visby and Prahalad, 2020). Upgrading and extending the boardwalk over the saltmarsh was also a key priority for the Glenorchy City Council (GCC) to connect the foreshore trail network in the bay (GCC, 2020). The new boardwalk would traverse the proposed saltmarsh restoration area and lead to a new bridge over Faulkners Rivulet. The DEP and GCC agreed to collaborate on each of these projects in 2019, to achieve both ecological, recreational and community benefits. The saltmarsh restoration project was funded by the Australian Government's Urban Rivers and Catchments Program.

The aim of the restoration project was to:

- Excavate historic infill to promote saltmarsh vegetation re-establishment.
- Conduct re-vegetative planting upland of the saltmarsh to increase terrestrial connectivity.
- Increase community education and engagement on saltmarsh values compared to pre-project levels.

Nature Glenelg Trust (NGT) was engaged to conduct an initial historical and eco-hydrological assessment of the saltmarsh area, formulate a restoration plan and assist with on-ground restoration works for this project.

Historic and Hydrological Assessment

Historic aerial imagery of Windermere Bay was obtained from the Department of Natural Resources and Environment Tasmania (NRE Tas) Aerial Photo Viewer website. Historic images were assessed to pinpoint the exact timing of infilling of the saltmarsh area. The majority of infilling appears to have occurred between March 1973 and 1975, and was likely linked to surrounding residential and road developments underway at the time (Figure 9.14). As a result of infilling approximately 50% of saltmarsh has been lost from Windermere Bay. Despite infilling, areas have settled and appear to maintain a hydraulic connection to tides and groundwater. Saltmarsh re-establishment is occurring in both the northern and southern basins of Windermere Bay, albeit – due to the change in surface elevation – at a slow pace.

Figure 9.14 Aerial imagery of Windermere Bay saltmarsh pre-infilling (1967) and post-infilling (1975).
Images: Department of Natural Resources and Environment Tasmania Aerial Photo Viewer website.

Faulkners Rivulet flows from Mount Faulkner Conservation Area and discharges into Windermere Bay. The upper catchment areas are natural, however, in the lower reaches the rivulet functions as a stormwater outlet. Infrequent but significant flooding occurs within the rivulet, with flows spilling out into adjacent saltmarsh and foreshore areas after heavy rainfall events. Twelve months of water quality data were collected from Faulkners Rivulet, upstream of the saltmarsh as part of the Derwent Estuary stormwater monitoring program 2024 – 2025 (Section 5.2). Water quality samples were also collected from Faulkners Rivulet downstream of the saltmarsh, over the saltmarsh and in Windermere Bay as a baseline to assess water quality improvements associated with the saltmarsh restoration.

Soil Assessment

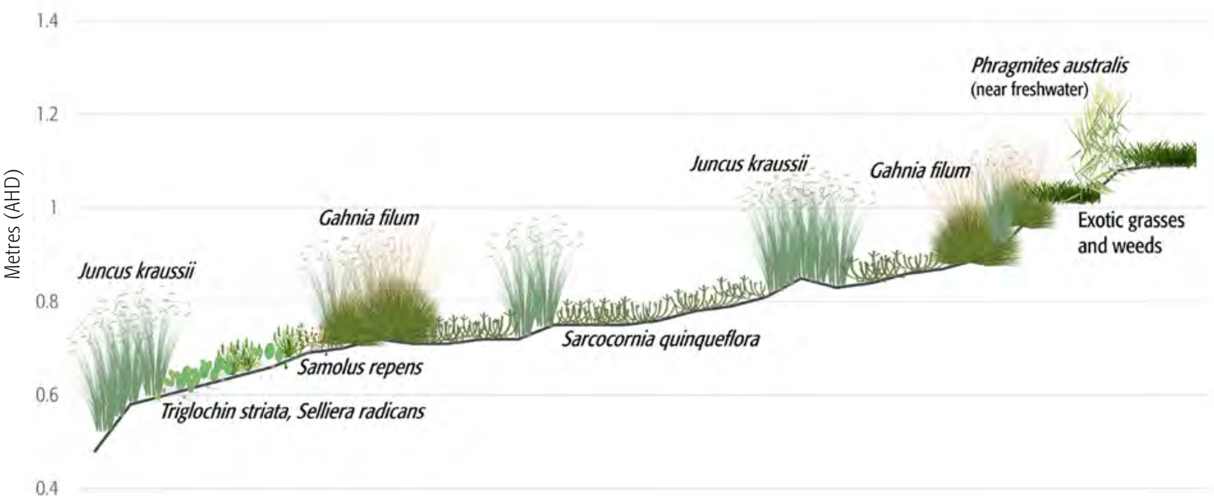
Shallow soil profiles were taken across the proposed restoration area in Windermere Bay. Soil profiling aimed to determine the natural surface level of the saltmarsh, while also inspecting for any forms of contamination within the soil. No potential sources of contamination were identified during soil profiling (e.g. construction waste, household waste, oil slicks, etc.) The area was not included in GCC's contaminated sites register, nor was it considered likely to include contaminated soil. To determine classification of infill material, Environment Protection Authority (EPA) and key regulatory literature were consulted to confirm soil testing requirements. Soil samples were collected from six sites across the proposed restoration area. All analytes tested were well below the limit for classification of contaminated fill material stipulated in *Information Bulletin 105* under the *Environmental Management and Pollution Control Act 1994*, and no asbestos was detected in the collected samples. Therefore, it was deemed sufficient to classify the infill proposed for excavation as 'clean fill'.

Vegetation Assessment

Three vegetation transects were established within the proposed restoration area, and one transect in the more natural northern basin, to map vegetation within the bay. The plant diversity at Windermere Bay is comparable with other saltmarsh areas within the Derwent Estuary (Visby and Prahalad, 2020).

The plant species identified during vegetation surveys are common, with no rare saltmarsh species encountered. There was found to be clear zonation of species across the transects associated with different elevation levels, and therefore, associated hydrological regime (Figure 9.15). It was clear that above elevation levels of 0.95 m AHD exotic grass species dominate the vegetation assemblage. An upper limit of 0.9 m AHD for the saltmarsh restoration level was set to ensure recolonisation of saltmarsh vegetation. These transects will be monitored on an annual basis to assess saltmarsh vegetation recovery over time.

Figure 9.15 Depiction of zonation of plant species, corresponding to ground level elevation and associated hydrological regime, along Windermere Bay saltmarsh transects.



Although not captured in the vegetation survey the invasive species of concern Spiny Rush (*Juncus acutus*) is known to occur in the saltmarsh area at Windermere Bay. This species has been actively managed by GCC for over a decade, however, persists in small numbers. In spring 2024, a brief survey of the area identified several small to medium plants, which were mapped and subsequently treated with herbicide. The DEP in collaboration with GCC will prepare a 5-year weed management plan for Spiny Rush at Windermere Bay, with the aim of eradicating this weed from the bay.

Bird Monitoring

Spring/summer bird monitoring was conducted at Windermere Bay saltmarsh prior to restoration works commencing. Windermere Bay saltmarsh has high bird diversity and numbers compared to other saltmarsh areas within the Derwent Estuary (Visby and Prahalad, 2020). A total of 21 bird species were observed, with bird numbers in the surveys ranging from 64 to 141 individuals.

Black Swan (*Cygnus atratus*), Tasmanian Nativehen (*Tribonyx mortierii*), Masked Lapwing (*Vanellus miles*), Silver Gull (*Chroicocephalus novaehollandiae*) and introduced ducks were recorded during every bird survey at Windermere Bay saltmarsh. Water birds, gulls, hens, and lapwings dominate the bird diversity at the site. The diversity and abundance of passerines was found to be lower at this site compared to other saltmarsh areas. This is not surprising given the limited surrounding riparian and terrestrial vegetation adjacent to the saltmarsh area. We will continue to conduct seasonal spring/summer bird monitoring to assess the impact this restoration project has had on bird species.

Community survey

A community survey at Windermere Bay was conducted to investigate what visitors to the bay enjoy most about the area, how they interact with the bay, their knowledge level of saltmarsh environments and interest in aspects of saltmarsh biodiversity and ecosystem services. A total of 171 participants were surveyed via in-person interrupt surveys and an online survey hosted on the GCC Let's Talk Glenorchy website. Recreating in a natural space was found to be the key reason why people visited Windermere Bay. Walking was the most popular activity, with participants spending the most amount of time on the paths/tracks/

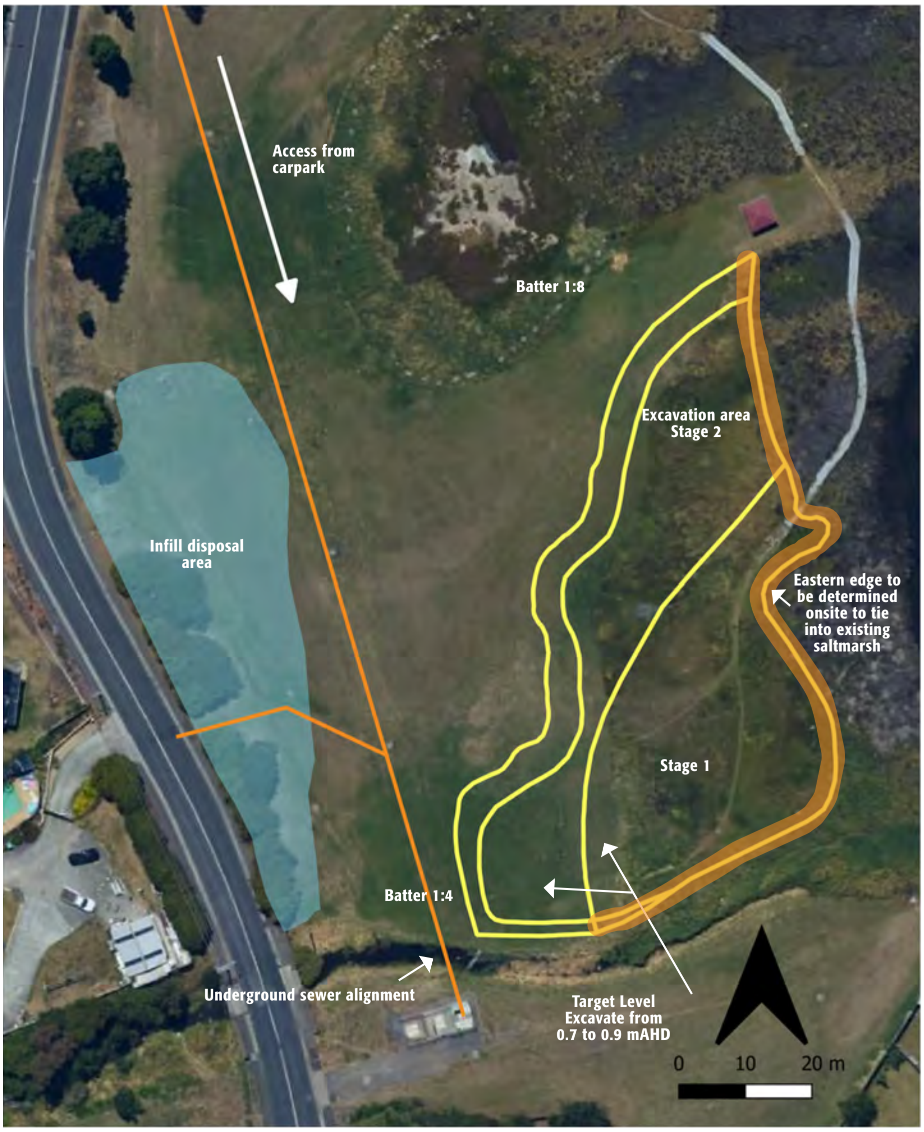
trails. The GCC boardwalk upgrade through the saltmarsh restoration area will increase walker access and connection to the saltmarsh. Strong connections to the natural features of the bay were expressed by participants, who highly value the biodiversity, quiet/peaceful atmosphere, views and access to water, which seem to provide a respite from nearby urban areas. There was found to be good baseline knowledge of and interaction with the saltmarsh at Windermere Bay. The urban setting of this wetland has significantly influenced participants' interest in biodiversity and ecosystem services, particularly when asked what areas they would like to learn more about.

The full report, Community Appreciation of a Local Saltmarsh and Recreational Area, April 2025, is available on the DEP website publication page. This report helped inform community education and engagement throughout the project delivery.

On-ground works

On-ground works to excavate the infill material were split into two stages to accommodate the boardwalk construction, and limit vehicle tracking over newly excavated saltmarsh areas. Stage 1 of works was limited to the areas under and east of the new boardwalk, and Stage 2 focused on areas west of the boardwalk and spreading of infilled material on the bank adjacent to Cadbury Road. This area contained a group of large woody weeds (including African Boxthorn (*Lycium ferocissimum*), *Cotoneaster sp.* and Hawthorn (*Crataegus monogyna*) which were removed prior to spreading the infill material (Figure 9.16).

Figure 9.16 Proposed extent, area and timing for excavation works for Windermere Bay saltmarsh restoration project. Image: Nature Glenelg Trust.



Excavation works were conducted from north to south, to ensure continuity in ground level elevation from the saltmarsh edge (approximately 0.7 m AHD) to the border of the restoration area (approximately 0.9 m AHD) and that newly excavated and saturated areas were not continually tracked over. In total, an area of approximately 3,000 m^2 and volume of approximately 1,200 m3 of infill material was excavated and spread adjacent to the roadside (Figure 9.17).

Figure 9.17 Aerial imagery of Windermere Bay saltmarsh restoration site after Stage 2 excavation works were complete. Image: Mark Bachmann, NGT.

Buffer planting

An area of approximately 860 m^2 upland of the saltmarsh restoration area was designated for buffer vegetation planting. This native vegetation will act as a natural buffer between intertidal and terrestrial areas of the bay, increasing habitat for terrestrial species that also frequent saltmarsh areas, including insects, birds, and mammals. A total of 740 native plant species were selected to be planted at Windermere Bay included riparian species, grasses, herbs, shrubs and overstory trees. Black Gum (*Eucalyptus ovata*) and South Esk Pine (*Callitris oblonga*) were planted as these species are known to occur within nearby wetland communities in the upper reaches of the Derwent Estuary, and constitute the upper canopy of EPBC Act listed Vulnerable ecological community *Eucalyptus ovata – Callitris oblonga* forest.

Interpretation signage

Two interpretation signs are planned, based on feedback from the community survey, and will be installed at Windermere Bay saltmarsh area in late 2025 to complete this restoration project. One sign will include educational information on saltmarsh biodiversity and ecosystem services. The second sign, developed in cooperation with the Tasmanian Aboriginal community, will focus on Palawa cultural content. It will present a story in both palawa kani and English. Engraved rocks featuring palawa kani language and accompanying images will be placed along the new walking track. This trail invites visitors to explore and engage with language and cultural stories as they walk through the saltmarsh–observing natural elements, reflecting on the Old People's connection to Country, and considering their own relationship with the land.

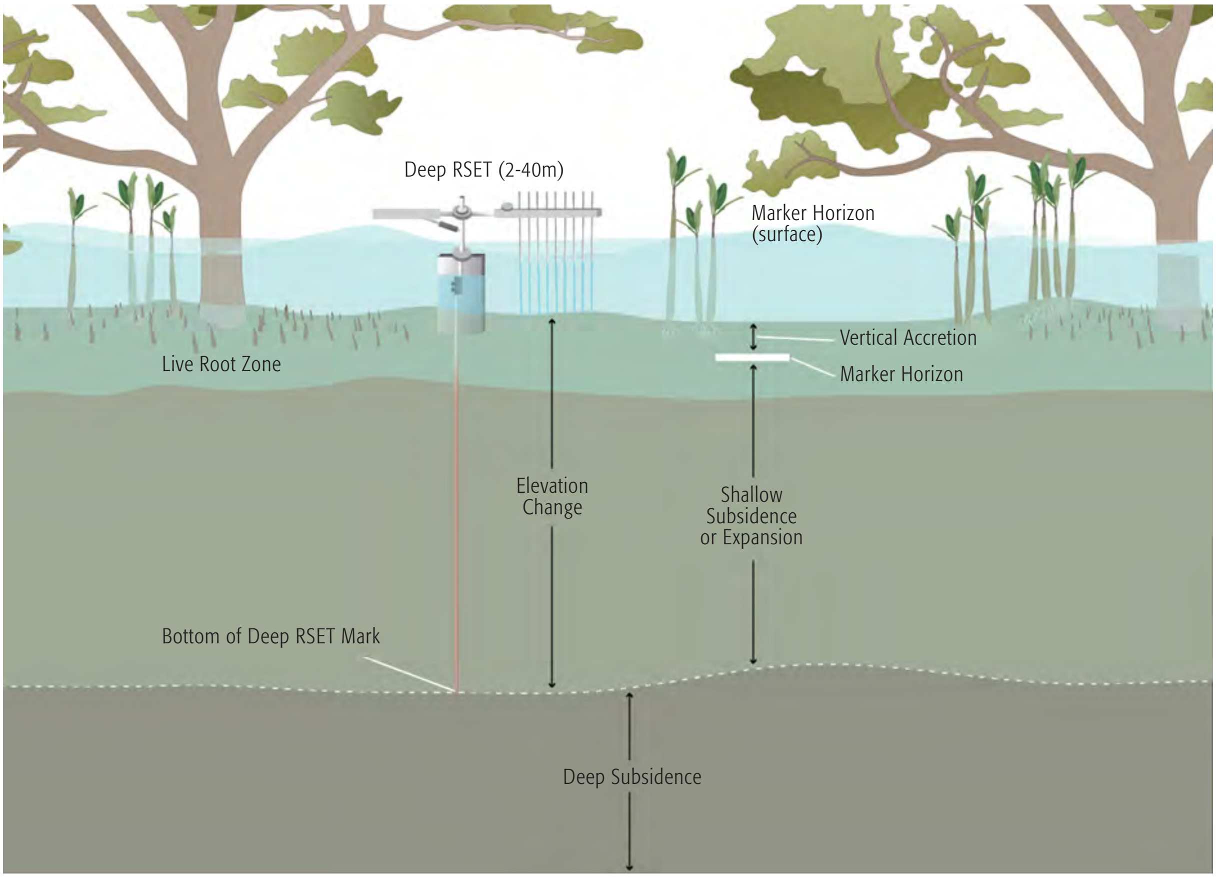
9.6.2 Derwent Estuary saltmarsh vertical accretion monitoring

Saltmarshes occur within the intertidal zone and are particularly vulnerable to changes in sea-level height. These tidal wetlands can adapt to changes in sea level, by accreting sediment from water that passes through them. Vegetation grows over and around accumulated sediment, raising the ground level (Schuerch *et al*., 2018). The rate of vertical accretion of saltmarshes is dependent on multiple factors including catchment sediment budgets (Weston, 2014), vegetation community presence, local sea level rise (Saintilan *et al*., 2024) and tidal range (Balke *et al*., 2016). There is a risk of widespread loss of tidal marsh communities worldwide with projected climate induced sea level rise trends (Saintilan *et al*., 2023). Due to the importance of saltmarsh communities to coastal ecosystems it is important to understand locally how these communities are responding to changes in relative sea level height.

The method for accurately measuring changes in surface height within saltmarsh environments are rod surface elevation tables (RSETs), developed by U.S. Geological Survey (Cahoon *et al*., 2002).

The rod survey benchmark serves as a fixed point from which relative changes in elevation of the wetland surface can be measured. The rod is driven into the wetland substrate until the bedrock has been reached. Measurements are periodically taken by installing a detachable measurement head to the rod, pins are then lowered to the wetland surface layer and measurements taken. As the wetland surface accretes or subsides the height of the pins increases or decreases relative to the measurement level arm (Figure 9.18).

Figure 9.18 Conceptual diagram of rod surface elevation table infrastructure installed in tidal wetlands to measure changes in surface elevation height. From (Lynch *et al*., 2015).



RSETs have been deployed across Australian saltmarshes in a range of geomorphic, tidal and dominant vegetation community types, and routinely monitored since 2001. This long-term dataset provides valuable information for how these communities are responding to sea level rise. A review of all data collected by the Australian RSET network found that most sites recorded increases in elevation over the period of measurement, however, when compared to rates of sea-level rise over the same period found that increases were not enough to keep pace with rising sea level (Saintilan *et al*., 2024). This network of RSETs across Australia is currently in the process of being integrated to Australia's Terrestrial Ecosystem Research Network infrastructure, under the OzSET Network. Tasmania has been identified as a key gap in the available data on response of Australian tidal wetlands to sea-level rise.

Two sites have been proposed to install RSETs in the Derwent Estuary, to provide valuable data on how southern Tasmanian, drowned-river-valley, saltmarsh communities are responding to sea level rise. A rush-dominated upper estuary site and succulent, saline, lower estuary site have been proposed to allow for comparison across an estuarine latitudinal gradient. Pending approvals and funding this infrastructure will be installed *in situ* late 2025 to start collecting data on vertical accretion trends in Derwent Estuary saltmarsh communities.

9.7 Shorelines

Shoreline geomorphology of the Derwent Estuary is quite diverse comprising sandy beaches, pebbled and rocky shorelines, coastal cliffs, intertidal wetlands, coastal lagoons and artificial structures, including seawalls, marinas and wharfs. Shorelines provide important habitat to native species, public amenity for recreation and access to the estuary, transport and industry. The profile and underlying geology of shorelines can make them more susceptible to erosion from sea level rise, storm surge and boat wake waves. Shorelines more susceptible to erosion include sandy beaches, cobbled/pebbled beaches, soil margins overlying hard bedrock surface, clay/silty sediments, and vegetated areas including wetlands. Resilient shoreline types include cliffs and sloping hard rock shores, and well-designed artificial shorelines (Sharples, 2022). Mapping of Tasmania's coastline for erosion susceptibility in 2013 identified 2.5 km^2 of shoreline in the Derwent Estuary to be at high risk of erosion and 2 km^2 at medium risk (Sharples *et al*., 2013).

9.7.1 Ferry wake on shorelines in the Derwent Estuary

Daily ferry transport was established across the Derwent Estuary in 2021, between Bellerive and Brooke Street Pier. Prior to commencing operation, the physical impacts of ferry operation on shorelines in the Derwent Estuary was assessed on behalf of the DEP. The main aim of this work was to investigate the effectiveness of simple, low-cost methods for monitoring and detecting any physical impacts of boat wake waves on the shorelines of the Derwent Estuary. The two methods investigated and trialled were:

1. Ground-based photo point monitoring and
2. Historic aerial imagery assessment.

Photo-point monitoring

Photo-point monitoring was established at Kangaroo Bay (site of Bellerive ferry berth) for a 3-month period to detect shoreline impacts from boat wake as a result of routine ferry traffic in and out of the bay. The shoreline of Kangaroo Bay is mostly comprised of hard surfaces that are not easily erodible. However, in some places the hard rocky shorelines were associated with softer components that showed some evidence of erosion or have the potential to erode.

Thirteen photo-point monitoring sites were established around Kangaroo Bay during the trial. Monitoring sites were selected with a clear view of one or more of the erosion-susceptible shoreline regions. Monitoring photos were taken at the 13 sites around Kangaroo Bay on seven occasions, over a 3-month period. Photos were analysed by simple visual comparison of photos to identify physical changes to shoreline forms and distribution of soils, bedrock, boulders, cobbles and sand deposits.

The photo monitoring provided evidence of minor movements of cobble veneers and beaches in several locations in the bay, although most of these were probably related to natural wind- or swell-wave events. One site (KBPP5) did, however, provide possible evidence of shoreline change with the progressive removal of intertidal cobbles over the monitoring period observed after the commencement of ferry operations (Figure 9.19). This site is relatively sheltered from swell and wind action, thus, is inferred to be a response to the introduction of ferry wake waves.

This trial proved successful in demonstrating that photo-point monitoring is easy to establish and conduct in the Derwent Estuary and can be effective in identifying changes in shoreline components that could be attributable to ferry wake waves.

Figure 9.19 Comparison of KBPP5 monitoring photos captured on 08/08/2021 (prior to ferry operation) (left) and 21/10/2021 (right). Left image shows a cobble-dominant foreshore, and right image shows more exposed underlying dolerite bedrock, particularly towards to lower right-hand area of the photo. Image: Chris Sharples.

Aerial imagery assessment

Anecdotal observations of shoreline erosion in the middle estuary, particularly the Montrose shoreline area, as a result of the MONA Roma ferry operation in this area, have been reported to the DEP. The Montrose shoreline includes soft to cohesive sediment types that are potentially susceptible to erosion and mobilisation under the impact of wave action, including boat wake.

Aerial imagery was obtained from the NRE Tas Aerial Photo Viewer website. Three images for the period prior to commencement of MONA Roma ferry operation were selected (2002, 2006 and 2009), and three after commencement (2012, 2015 and 2019), in order to assess for any changes to the shoreline that might be time-correlated with the introduction of ferry operation in the area.

Four sites were selected along the Montrose shoreline for identification of changes detectable on the aerial images. Cohesive cobbly intertidal substrate at Montrose is identifiable as a dark-toned surface in all the aerial images. However, beginning in 2012, and more widespread in 2015 and 2019 images, large areas of light-coloured material appear on the shoreline, which in parts show progressive landwards movement through the years. Comparison of the current shoreline with aerial imagery strongly suggests the light-coloured materials are loose cobble berms still present on the shoreline today. The reason for their appearance in 2012 is unclear; however, one possible explanation is that they were winnowed out of the erodible cobbly shore substrate by increased boat wake action. Alternatively, it may have been artificially deposited to protect the shore from increased wave (boat wake) action. Recent enquiries with local land managers and stakeholders failed to confirm the latter explanation.

Recommendations

The trial demonstrated that photo-point monitoring can be quickly and simply established without any on-site markings, and easily conducted, as per the methods outlined in Sharples (2022). In the case of new ferry routes, or other potential disturbances proposed which may generate changed wave impacts on shorelines in the Derwent Estuary, it is recommended to take the following steps to set up photo-point monitoring:

1. Determine proposed boat route and identify closest shorelines with highest (most direct, closest) exposure to boat wakes.
2. On the potentially exposed shores, identify the shoreline types more and less susceptible to wave erosion using existing geological mapping and field inspections.
3. Set up and commence photo-point monitoring at susceptible locations as soon as possible in advance of the commencement of the newly identified disturbances to accurately assess baseline condition and identify any shoreline changes resulting from the new disturbance.

9.8 Foreshore vegetation

Small to large areas of remnant native vegetation can be found along the foreshore of the Derwent Estuary. Foreshore vegetation is dominated by Tasmanian Blue Gum (*Eucalyptus globulus*), Silver Peppermint (*Eucalyptus tenuiramis*), Black Peppermint (*Eucalyptus amygdalina*), Black Gum (*Eucalyptus ovata*), White Peppermint (*Eucalyptus pulchella*), White Gum (*Eucalyptus viminalis*) forest and woodland areas, Drooping Sheoak (*Allocasuarina verticillata*) forest, Bursaria – Acacia woodland, scrub and lowland native grasslands. These native woodland areas provide important habitat and food sources for a diverse range of native fauna including insects, reptiles, birds and mammals. Larger Eucalyptus trees, especially Tasmanian Blue Gum provide extremely important habitat for the EPBC Act listed Critically Endangered Swift Parrot (*Lathamus discolor*).

North of the Bridgewater Bridge, the estuarine foreshore is dominated by the freshwater wetland areas of Dromedary Marsh and Murphys Flats, with Woolly Tea Tree (*Leptospermum scoparium*) scrubland occurring above the wetland area, and Bursaria – Acacia woodland and scrub further upland. North of these wetland areas the foreshore is more developed with agricultural, industrial and urban land which is impacted by weed species including Karamu (*Coprosma robusta*), African Boxthorn (*Lycium ferocissimum*), Gorse (*Ulex europaeus*), Willow (*Salix* spp.) and Blackberry (*Rubus fruticosus* aggregate) (DEP, 2019) which will be discussed further in **Section 9.9**. Karamu, Willow and Blackberry are capable of spreading into wetland areas, which is of concern due to the proximity of infestations in the upper estuary to the high-value wetland areas of Dromedary Marsh and Murphys Flats.

Eucalyptus ovata – Callitris oblonga woodland areas can be found in riparian areas of the upper Derwent Estuary. This woodland community is listed as a Vulnerable Ecological Community under the EPBC Act. This forest is characterised by a Eucalyptus overstorey, typically made up of *Eucalyptus ovata*, but generally also includes *Eucalyptus viminalis* and *Eucalyptus amygdalina*; a midstory of *Callitris oblonga subsp*. *oblonga*, and a shrubby understorey of *Bursaria spinosa*, *Melaleuca gibbosa* and *Acacia dealbata* (Harris and Kirkpatrick, 1991; Zacharek, 2000). This riparian vegetation benefits riverine ecosystems by limiting sunlight and maintaining temperatures, regulating nutrients, stabilising riverine banks, and providing habitat for species within riverine areas (Askey-Doran *et al*., 1999).

South of the Bridgewater Bridge the estuary's foreshore is dominated by industrial and urban developed areas. Several areas of large intact foreshore vegetation include Mount Direction Conservation Area, Queens Domain, Bedlam Walls, Alum Cliffs, Sandford and Tinderbox. Hotspot areas for threatened flora along the estuary's foreshore include Gagebrook-Old Beach, Cornelian Bay and Bedlam Walls.

Other foreshore areas containing a large proportion of one or two threatened species include the upper Derwent Estuary riparian zone, Green Point, Clarence Rivulet, Queens Domain, Lauderdale saltmarsh and South Arm (DEP, 2010c). Bedlam Walls supports a large population of Risdon Peppermint (*Eucalyptus risdonii*), this species is listed as Rare under TSP Act. This small, flowering Eucalyptus is endemic to the Meehan Range, and is found along gravelly, dry hills on the eastern shore of the Derwent Estuary.

Bedlam Walls and Queens Domain are important areas for lowland native grassland, which is listed as Critically Endangered under the EPBC Act. These areas would have been historically managed by Aboriginal fire regimes, however, since colonisation lowland native grassland areas have been impacted by grazing, land clearing for pasture and agriculture, inappropriate fire regimes, weeds, and urban development. The native grasslands of Queens Domain were historically impacted by high-intensity stock grazing, which was removed from the Domain in the 1920s. Since this disturbance was eliminated there has been a steady increase in tree cover on the Domain (Sorensen and Kirkpatrick, 2021), and in recent decades there has been a large increase in Drooping Sheoaks. In dense areas, these Sheoaks shade out native grasslands and outcompete other native vegetation. The City of Hobart (CoH) is involved in management of these native grassland areas of Queens Domain by mechanical thinning of Sheoaks and periodic burning of vegetation. Management of the Sheoaks has resulted in regrowth of grassland vegetation, increasing the number of grazing wallabies, pademelons and bandicoots on the Domain. This grazing activity further increases biodiversity by creating habitat for insects and promoting new plant growth.

9.8.1 TasVeg 5.0

TasVeg is a comprehensive digital map of Tasmania's vegetation communities. NRE Tas plans to release an updated version of TasVeg (version 5.0) at the end of 2025, the next major revision in this data since 2020. Vegetation across Tasmania has been defined into over 150 communities by experts and the Tasmanian Vegetation Monitoring and Mapping Program. Photographic interpretation of the most current aerial photography is the primary method of data collection of TasVeg updates, with field verification of representative areas undertaken where practical. TasVeg also incorporates updated mapping supplied by external stakeholders where accuracy of data had been confirmed.

Seabird Rookery Complex is listed as a Threatened Native Vegetation Community under the *Nature Conservation Act 2002*. This community is quite unique in that it is comprised of different vegetation types, with the defining feature being the use of the area by nesting or foraging seabirds for at least part of the year, as evidenced by the presence of the birds or by burrows, runs, or bioturbaceous soils. This community will be included in the TasVeg 5.0 update and will include areas within the Derwent Estuary where seabirds have been observed to be nesting. This allows for better protection of seabird nesting habitat, particularly in areas on private land, which are often subject to clearing without prior consultation with seabird conservation programs.

9.9 Weeds

A weed is an exotic plant species that has been introduced to Tasmania, either deliberately or accidentally, and become naturalised (Rozefelds *et al*., 1999). The Derwent Estuary foreshore supports a variety of introduced terrestrial weed species that have become established and threaten the survival of native plants and animals. The most devastating weeds are those that outcompete native vegetation and can impact valuable social, agricultural and environmental assets. There are currently 144 declared weed species listed under the *Tasmanian Weed Management Act 1999* with 42 found to be present along the Derwent Estuary foreshore. These declared weeds have a special legal status that requires landholders and managers to actively control and, in some cases, eradicate them.

The DEP is actively involved in several projects including mapping and prioritising weed management along the estuary foreshore, facilitating collaboration between land managers and applying for funding opportunities to better manage and control weeds. Projects of high priority in the region include monitoring for Rice Grass (*Spartina anglica*) across the estuary and providing funding for upper estuary Karamu (*Coprosma robusta*) control.

9.9.1 Rice Grass

Rice Grass (*Spartina anglica*) is an invasive intertidal plant, introduced to Tasmania between 1930–1970 for the purposes of land reclamation and riverbank stabilisation. It became established in seven regions of the states coastal and estuarine areas (Hedge *et al*., 1999), including the Derwent Estuary. This perennial grass forms dense infestations, which can alter the intertidal hydrology and sediment dynamics of coastal and estuarine areas. Over time, Rice Grass beds become raised compared to their surroundings, permanently altering the morphology of intertidal areas. The extensive infestation of Rice Grass in the Tamar Estuary has had significant negative impacts on the estuary's coastline and vegetation assemblages, with previously unvegetated areas transformed into marsh terraces, and sandy banks into tussocked mudflats (Sheehan and Ellison, 2014). Infestations can impact biodiversity as well as hydrology and geomorphology, have been shown to reduce benthic macrofaunal species diversity (Cutajar *et al*., 2012) and significantly impact the diversity and abundance of native fish species in Rice Grass-dominated mud flats compared to native vegetated marshes (Harrison-Day *et al*., 2022).

In the 1990s, the total area of Rice Grass infestation in the Derwent Estuary was estimated to be approximately one hectare (DPIPWE, 2002a). The Rice Grass Advisory Group was formed in 1995 to provide advice and direction on the management of all infested regions in Tasmania, and a management strategy was subsequently implemented (DPIPWE, 2002a). Between 1995 and 2002, substantial mapping, treatment and follow-up surveys were implemented by Parks and Wildlife Services' Derwent District Ranger John Megalos and his team (Megalos, 1995). From 2003–2016 annual surveys and treatment for small volumes of Rice Grass were conducted, with the DEP leading management efforts since 2009. In spring 2016, two small Rice Grass infestations were located on the northern side of Dogshear Point, Claremont and another two adjacent to Montrose Bay High School. However, no Rice Grass has been observed in the Derwent Estuary since 2016, despite regular monitoring.

A 2017 review of Rice Grass in the Derwent Estuary (DEP, 2017), proposed to divide the foreshore survey area (between Bridgewater Bridge and Bowen Bridge) into four sections, with one section to be surveyed annually on a four-year rotation. This methodology was followed from 2018 to 2023 (with no Rice Grass survey occurring in 2022), with no infestations observed. Given that no evidence of Rice Grass has been observed in the estuary, under rigorous surveying since 2016, and the low viability of its seeds, the survey effort for Rice Grass in the Derwent Estuary was further reduced to previous 'hotspot' locations for the 2024 survey. Again, no Rice Grass was observed during this survey.

Many organisations and individuals have generously donated their time to assist in Rice Grass surveys in the Derwent Estuary over the years. These include Clarence City Council (CCC), Glenorchy City Council (GCC), Tasmanian Parks and Wildlife Service (PWS), NRM South, Tasmanian Herbarium (TMAG), EPA, UTAS and Threatened Plants Tasmania.

Recent infestations of Rice Grass found in Hastings Bay, Huon Valley highlights the need to stay vigilant, as there is a potential for re-introduction to the Derwent Estuary. This infestation was reported by local oyster lease holders, with origin still unknown (pers comms M. Joy, Huon Valley Council, 2024). Before spring 2025, we aim to liaise with stakeholders in the upper estuary to ensure that local users of the foreshore are aware of the potential for Rice Grass to be re-introduced to the estuary, and to report any suspicious sightings to their local council or the DEP.

9.9.2 Tasmanian Weeds Action Fund

The Tasmanian Government has invested over $7 million to projects dedicated to the management and control of priority weed species in Tasmania between 2019 and 2027 through the Weeds Action Fund (WAF). The WAF aims to provide a more strategic and targeted approach to tackling high-priority weeds impacting valuable agricultural and environmental assets by engaging with landowners, local government and the broader community to make sustainable, long-term and effective actions to address the eradication of high priority weeds across the state.

The DEP was successful in gaining funding through the WAF Stage 2 small grants round to undertake Strategic Weed Assessment and Prioritisation Project (SWAPP) in the Derwent Estuary. The Derwent Catchment Project (DCP) was also successful in gaining funding to support control of Karamu in the upper estuary. Support from additional funding through the Tasmanian WAF has enabled organisations such as the DEP to undertake strategic weed prioritisation and management projects, and increased control efforts for priority weed species throughout the estuary foreshore and tributaries.

WAF Stage 3 will fund six projects across the state from 2025 to 2027. The focus of one of these projects will be on invasive emergent weeds that have established or expanded in distribution across Tasmania. One of the target species, Spiny Rush (*Juncus acutus*) is known to occur within the Derwent Estuary at Windermere Bay saltmarsh. The DEP will collaborate with the successful candidate on developing a Spiny Rush management plan for Windermere Bay.

9.9.3 Karamu

Karamu (*Coprosma robusta*), native to New Zealand, is a declared weed in Tasmania under the *Tasmanian Weed Management Act 1999*. This evergreen shrub can invade relatively undisturbed native vegetation, particularly in damp and wet forested areas, as well as on the banks of waterways. Karamu infestations are restricted to a few sites in Tasmania, meaning complete eradication of this weed species is a possibility. Known infestations are in the upper Derwent Estuary, Kermandie River, Fern Tree, Guy Fawkes Rivulet, and several isolated infestations near Kingston, Middleton and Blackmans Bay. The infestation in the upper Derwent Estuary is the largest in Tasmania (DPIPWE, 2006), and its control in this area is a high priority given the proximity to high conservation value wetlands of Murphys Flat and Dromedary Marsh.

In the upper Derwent Estuary, dedicated surveys and control efforts for Karamu have been conducted since 2010 (DEP, 2015c). In 2017, a comprehensive survey was undertaken to accurately map Karamu infestations in the upper estuary. The main infestation occurs on the banks of the Derwent between Bryn Estyn and downstream of Boyer, all tributaries in the New Norfolk area have been surveyed and Karamu infestations were found in the Lachlan River and Sorell Creek. In 2017, the Derwent Catchment Project (DCP) published the Weed Management Plan: Karamu in the Derwent Catchment 2017 – 2025, which has guided annual control works and follow-up surveys across 16 management

Derwent Catchment Project undertaking Karamu (*Coprosma robusta*) control works in the upper Derwent Estuary. Image: Derwent Catchment Project.

zones. Management zones were defined by factors of commonality i.e. density of infestation, land tenure and accessibility. Co-investment for annual treatment of Karamu in the upper estuary is made by DEP, Derwent Valley Council, Property Services Tasmania, Department of State Growth, DCP and PWS. On-ground control works are conducted by the DCP, with support from PWS and Inland Fisheries Service for boat survey work.

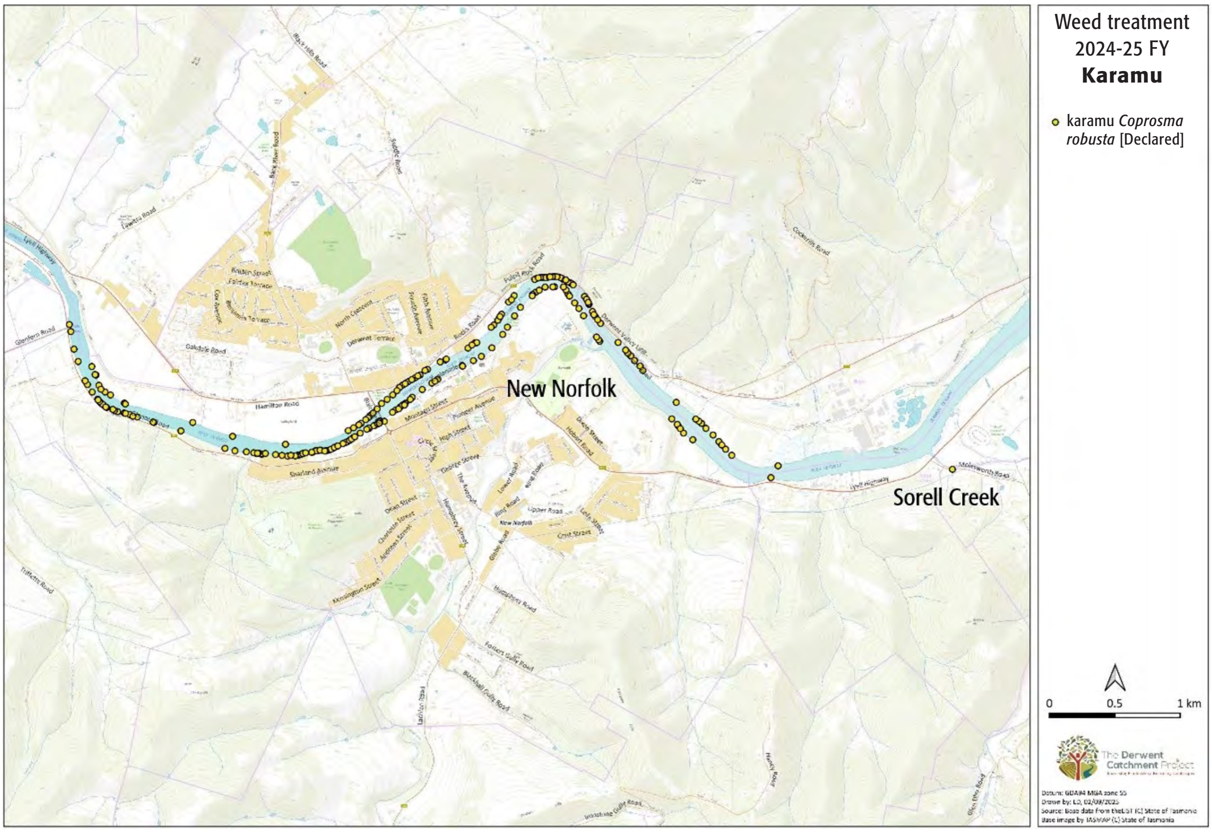
Detection of Karamu can be difficult, as this weed can grow within dense infestations of other weed species and native vegetation, making it difficult to spot and time-consuming to treat. Its presence over a mix of land tenures makes gaining approval for access difficult and time consuming. Some infestations occur on steep riverbanks, which is logistically challenging as boat access is required. In 2023 the DCP received funding from the WAF for Karamu control work. WAF funding enabled control works to be undertaken in difficult to access areas (requiring boat and rope access) in the upper estuary.

From 2018 to 2025, primary control and follow-up surveys were conducted across all management zones in the upper estuary. The 2023/24 Karamu control work focused predominantly on follow-up surveys with only six established plants identified and subsequently treated. In the 2024/25 season, the DCP conducted follow-up surveys for Karamu along the Derwent. During this survey multiple new Karamu plants were identified growing in previously concealed dense areas of riparian vegetation, which had been stripped away during a flood event in September 2024. Treatment of plants was conducted across all management zones during the 2024/25 surveys. Notably during surveys, a previously unrecorded population of Karamu was discovered and treated in Sorell Creek (Figure 9.20).

In March 2025, DCP staff conducted a riverine survey by boat to assess Karamu distribution along the river. While Karamu was recorded across all zones, most occurrences were of small plants that were thought to have emerged from the seed bank or as reshoots from individuals established along the waterline. Dense infestations previously identified around New Norfolk have been substantially reduced to low-density occurrences, and the program is showing an overall decline in Karamu numbers. Furthermore, recent flooding events have exposed more mature plants, improving their visibility and enabling more effective targeting during control activities.

The treatment program, particularly the intensive works supported through the WAF, has demonstrated that the investment of resources and effort is delivering strong results. However, achieving full eradication along the River Derwent remains challenging, as new seedlings and regrowth continue to emerge even after intensive control, requiring ongoing monitoring and follow-up treatment to prevent reinfestation. To maximise long-term success, a broader river restoration program would be necessary to remove Willows and reopen the riverbank. This would make the treatment of Karamu and other invasive weeds more effective, but would also require a comprehensive revegetation effort to re-establish native riparian vegetation, given that much of the current cover is dominated by invasive species.

Figure 9.20 Location of Karamu (*Coprosma robusta*) treatment in the upper Derwent Estuary and Sorell Creek during 2024/25 surveys. Image: Derwent Catchment Project.



9.9.4 Derwent Estuary strategic weed assessment and prioritisation project (SWAPP)

The Derwent Estuary Weeds Collaboration was established in 2018 to focus on weed management within the estuary's foreshore riparian zone. This group comprises representatives from GCC, Kingborough Council, CoH, CCC, Brighton Council, Derwent Valley Council, DCP, DEP, PWS, Property Services Tasmania, Tasmanian Herbarium (TMAG), Department of State Growth, NRM South and Biosecurity Tasmania.

In 2022, the DEP (on behalf of the Derwent Estuary Weed Collaboration) commissioned a SWAPP for the Derwent Estuary foreshore, funded by the WAF. Work was conducted by North Barker Ecosystem Services to provide a clear and strategic approach to the management of weeds in the estuary. The aim of the plan was to synthesise local knowledge, expertise and resources across the estuary to support multiple land managers targeting weeds, and to provide a clear and strategic approach to weed management within a whole-of-estuary framework.

Key actions included:

- Identify priority areas across the estuary that should be protected from invasive weeds, i.e. natural, cultural (not including Tasmanian Aboriginal), recreational and agricultural.
- Identify, map and maintain records of priority weed species and areas across the Derwent Estuary foreshore.
- Identify and develop priority projects with stakeholders to control and eradicate weeds and to protect and restore high-value habitats, e.g. wetlands tidal flats, inter-tidal zones, and native foreshore vegetation.

Output from the SWAPP were six spatial layers covering the whole of estuary foreshore (in 1 ha grids):

1. Natural values priority score,
2. Cultural (not including Tasmanian Aboriginal) values priority score,
3. Recreational values priority score,
4. Agricultural values priority score,
5. Combined values score, and
6. Location of known weeds, including information on species collection, declaration status, size and related data.

Value priority scores were based on the presence of reserve/conservation areas, cultural buildings, points of interest, public spaces, community buildings and agricultural land within the gridded cell. The combined values score included value classification from all four individual value priority scores. Higher scores relate to a higher level of value. Spatial layers used in the value classification were obtained from LISTmap.

The SWAPP highlighted the presence of 178 weed species, across 7,280 sites in the Derwent Estuary foreshore, including twelve of the 14 Tasmanian WAF priority weed species (Table 9.2).

The spatial information is hosted on the LISTMap for public access to help guide strategic weed management across the Derwent Estuary. This estuary wide map will encourage collaboration when targeting weed infestations that span multiple land management areas and jurisdictions. Where possible, this strategy will also leverage resources and funding for future weed management projects. Figure 9.21 shows an example of spatial output delivered by the SWAPP.

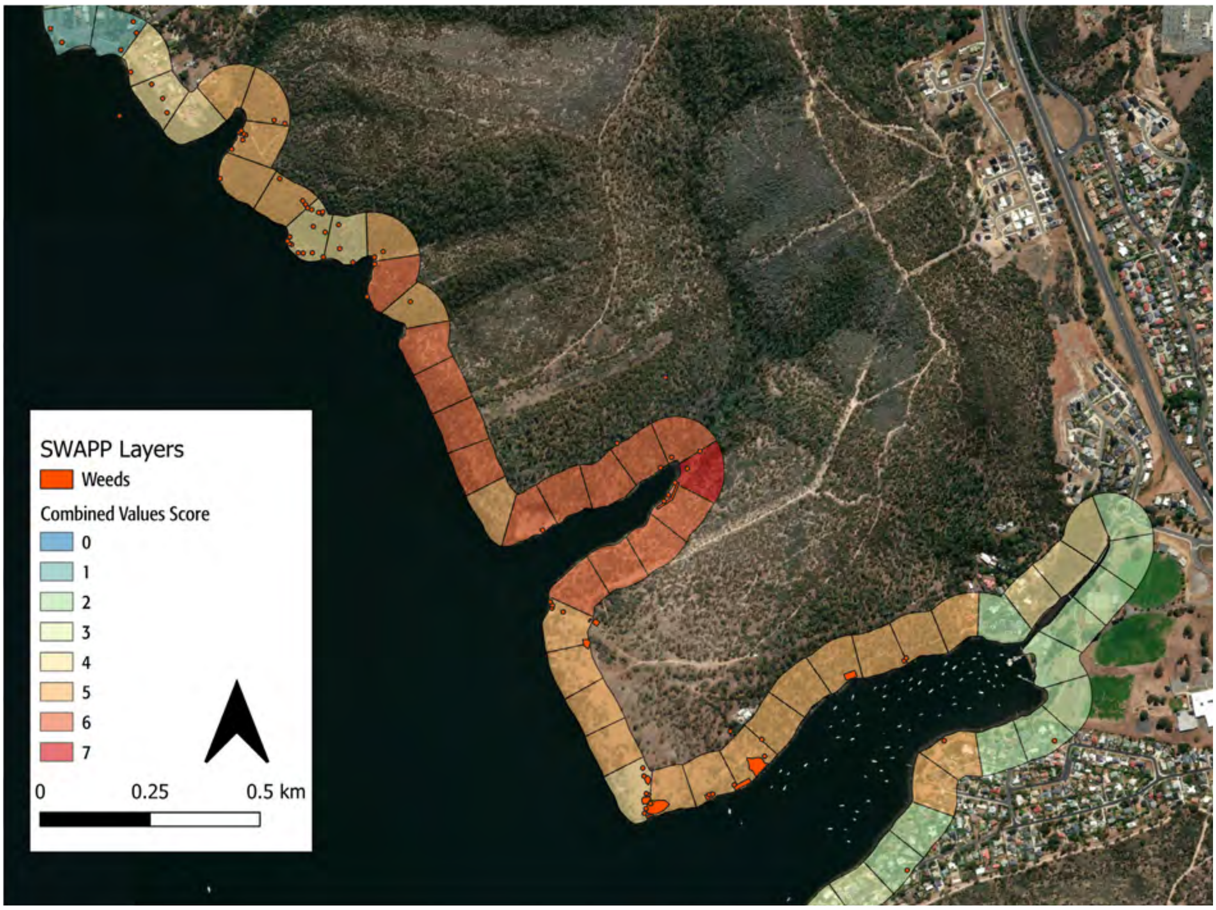
Table 9.2 Weeds Action Fund priority weed species mapped in the Derwent Estuary Strategic Weed Assessment and Prioritisation Project.

Common name	Scientific name
Espartillo	*Amelichloa caudata*
Chilean Needle Grass	*Nassella neesiana*
African Feather Grass	*Cenchrus macrourus*
Flax-leaved Broom	*Genista linifolia*
Spanish Broom	*Spartium junceum*
Bridal Creeper	*Asparagus asparagoides*
Paterson's Curse	*Echium plantagineum*
Serrated Tussock	*Nassella trichotoma*
Mediterranean Daisy	*Urospermum dalechampii*
St John's Wort	*Hypericum perforatum*
Golden Wattle	*Acacia pycnantha*
Spiny Rush	*Juncus acutus*

African Feather Grass (*Cenchrus macrourus*). Image: Roger Davis.

Spiny Rush (*Juncus acutus*) at Windermere Bay, Derwent Estuary. Image: Inger Visby.

Figure 9.21 Bedlam Walls, Clarence City Council high value area. Derwent Estuary Strategic Weed Assessment and Prioritisation Project spatial layer output for mapped weeds, and grid-combined-values scores.



9.9.5 Community Biochar Kiln Project

Provided by Bill Harvey, Landcare Tasmania

Biochar is a unique form of charcoal, and is created by burning organic material, such as wood chips, weeds, vegetation, leaf litter, etc. at very high temperatures with limited oxygen. This process, termed pyrolysis, is done under very controlled conditions in a biochar kiln, and is a safe alternative to open burns, with most of the carbon converted to stable, usable products. It is a controlled fire that is continuously fed, meaning no nuisance smoke is generated, reducing carbon dioxide emissions to the atmosphere. The free-standing kiln is portable, and means that soil is not sterilised beneath, like an open burn. The physical and chemical properties of biochar make it a great soil enhancer. Biochar increases soil structure, improves water retention and aggregation, improves porosity, regulates nitrogen leaching, and increases microbial activity.

Vegetation burning in biochar kiln at community biochar day. Image: Bill Harvey.

Interest in biochar has increased in recent years as the benefits begin to be recognised. There has been uptake in production from agricultural, land care, local council and natural resource management sectors, as a means of disposing of weeds and other organic materials. As well as increased demand for this product for application in residential gardens and commercial farms for soil enhancement.

Some additional uses of biochar that are now being explored include a lower carbon footprint alternative to cement and asphalt, bioremediation of contaminated sites, processing sewage biosolids to destroy nano- and micro-plastic, per- and polyfluoroalkyl substances and other contaminants, water filtration and nutrient removal. As well as improving gut health in cows and horses for healthier animals that produce less methane.

Landcare Tasmania is involved in a program to encourage small-scale, community biochar production and recognises the many benefits it has to offer. These are as follows:

- Working as a group, participants reduce the fire hazard on host properties by turning green waste into biochar.
- The biochar is distributed to participants to add to gardens, properties and farms for soil enhancement.
- The biochar process does not require a permit in Tasmania as it is a controlled fire, not an open burn.
- Making biochar from weeds *in situ* prevents the spread of weeds by dealing with them on site rather than spreading seed and material through transport.
- Making biochar is a positive, fun and healthy, social activity with benefits for physical and mental health.
- Community kilns are freely available to borrow from a kiln library, rather than groups, farmers and individuals needing to purchase kilns.
- Collected data demonstrates the effectiveness of the program, volume of biochar produced, and tonnes of carbon dioxide sequestered.

It is hoped that the community kiln program can make a significant contribution to managing weeds and improving biodiversity outcomes around the Derwent Estuary and elsewhere.

Outcomes

Two thirds of all southern Tasmanian biochar session were conducted in the Derwent Estuary area converting many tonnes of weeds, including African Boxthorn (Lycium ferocissimum), Blackberry (Rubus fruticosus aggregate), Briar Rose (Rosa rubiginosa), other weeds and organic material into biochar. The GCC has purchased a portable 290 L kiln, which is being used regularly to turn African Boxthorn into biochar at Prince of Wales Bay. Already the program has been a great success, with biochar sessions reducing weeds, and bush fire risk in southern Tasmania, while engaging people in a physical, outdoor, community-led activity (Table 9.3).

Table 9.3 Outcomes of the Landcare Tasmania southern community Biochar kiln project from April 2023 to July 2025.

Outcomes	N
Number of participants	750
Number of biochar sessions	80
Number of locations in southern Tasmania[1]	45
Volume of biochar produced	42,830 L
Mass of CO_2 sequestered[2]	64 t

[1]From Bridgewater to Woodbridge. [2]Sequestered for up to 1000 years.

10

Estuarine fauna

10. Estuarine fauna

Timtumili Minanya/River Derwent, including the Derwent Estuary supports a diverse range of fauna from microscopic planktonic zooplankton to large migratory mammals. The Derwent Estuary supports resident estuarine, marine, wetland, intertidal and terrestrial species, seasonal visitors, and occasional vagrants that visit the estuary to rest and/or forage for food. The estuary is also important habitat for many threatened species, including the Australian Grayling (*Prototroctes maraena*), Australasian Bittern (*Botaurus poiciloptilus*), Curlew Sandpiper (*Calidris ferruginea*), Southern Right Whale (*Eubalaena australis*) and many more (see (DEP, 2015c) for full list). Some species have most of their known population distributed within the Derwent Estuary, including the Derwent River Seastar (*Patiriella littoralis*) (possibly extinct) and the Spotted Handfish (*Brachionichthys hirsutus*).

10.1 Zooplankton

Zooplankton are a group of small animals that live in the water column. They are either free floating or weak active swimmers that are transported throughout marine and estuarine environments via currents. These animals can range in size from 2 μm to >200 cm. Zooplankton are a broad group including microscopic animals (such as sea snails, krill, and copepods), the planktonic larval phase of larger invertebrates and fish (including rock lobsters and eels) and soft bodied animals like jelly fish and salps.

A comprehensive assessment of zooplankton species present in the Derwent Estuary was investigated by Taw (1978), and provides a good baseline for observed species composition. Zooplankton assemblages were found to be variable throughout the year, being influenced by both sub-tropical and sub-Antarctic oceanic currents, as well as freshwater flows from the River Derwent. Certain species can suddenly appear in large blooms when food is abundant and conditions are favourable, such as the recent salp bloom in November 2024, followed by Moon Jellyfish (*Aurelia aurita*) in January 2025.

10.2 Benthic infauna

Benthic infauna are a group of invertebrates that live within the sediment in freshwater, marine and estuarine environments. These communities carry out many important ecosystem processes, including nutrient cycling, organic matter decomposition, oxygenation and stabilisation of sediments (O'Meara *et al*., 2020), decomposition of pollutants (Oyetibo *et al*., 2017) and are an important food source for many higher trophic organisms (Snelgrove, 1997). Benthic infauna diversity and community composition has been used as an indicator of ecosystem health in estuarine systems globally (Diaz *et al*., 2008; Birk *et al*., 2012; Tweedley *et al*., 2012). Degraded systems generally have a lower diversity of infauna, dominated by more tolerant species, such as polychaete worms, compared to healthier systems that support a broad diversity of benthic infauna (Puente and Diaz, 2008). A comprehensive assessment of macrobenthic community structure across the Derwent Estuary was conducted in 2002–2003 (Macleod and Helidoniotis, 2005). High concentrations of metal and organic carbon in sediments were found to influence benthic infauna community composition in localised areas, however, they were not the overriding determinant in benthic community composition throughout the whole system. The general trend of benthic infauna abundance and diversity in the Derwent Estuary found was decreasing abundance and increasing diversity down the estuary. Low diversity with high abundance is generally associated with impacted conditions (Pearson and Rosenberg, 1978; Stark, 1998), and sites with peak abundance and low diversity were almost all impacted sites.

Benthic foraminifera (forams) are single-celled meiobenthic infauna, which comprise a significant portion of infaunal biomass and are sensitive to environmental influences (Gooday, 2003; Quilty, 2013). Forams have been utilised as bioindicators for marine pollution globally (Abd Malek and Frontalini, 2024). A student project is currently being conducted to investigate baseline ecology of forams in the Derwent Estuary and assess if forams can be used as a bioindicator for heavy metal pollution in the estuary.

10.3 Fish

Approximately 150 finfish species have been documented in the middle and lower zones of the Derwent Estuary (see DEP (2010) for full list). These fish communities can be broadly classified as pelagic (inhabit the middle and upper layers of the water column), demersal (inhabit soft bottom sediments and/or bottom water layers), or reef species. Some species, such as Southern Sand Flathead (*Platycephalus bassensis*), Black Bream (*Acanthopagrus butcheri*) and Spotted Handfish (*Brachionichthys hirsutus*) are permanent residents of the estuary, while others are transitory or seasonal migrants.

10.3.1 Southern Sand Flathead

Southern Sand Flathead (*Platycephalus bassensis*) is the dominant recreationally targeted fish species in Tasmania. Within the Derwent Estuary, Southern Sand Flathead show strong site fidelity (Tracey *et al*., 2011).

Fish have been known to make downstream migrations in spring, however, return to home ranges in early summer. Peak spawning is reported to occur between October and December in southeast Tasmania (Jordan, 2001), therefore, it is likely this downstream movement is related to spawning activities.

This species is demersal, inhabiting sandy, soft sediments in shallow coastal embayment's. Southern Sand Flathead are benthic feeders (Jordan, 2001) and as such feed on prey that is often exposed to higher metal levels. The general sedentary behaviour and feeding mechanisms of Southern Sand Flathead has implications for bioaccumulation of heavy metals (Verdouw *et al*., 2011), particularly for fish inhabiting the middle and upper Derwent Estuary, which has the highest levels of bottom water metal contamination (DEP, 2010a). Recreational fishers targeting Flathead in the Derwent Estuary should be aware of the potential health implications of consuming seafood caught from the estuary (see **Section 8**).

Stock assessment for Southern Sand Flathead in 2014/15 classified this species as Depleted. Subsequently changes to minimum size and bag limits were implemented in 2015, 2023 and 2024. Commercial take of Southern Sand Flathead was banned in November 2023, although this was only estimated to be 1% of the combined total harvest (Krueck *et al*., 2025). In late 2024, the size limits for Southern Sand Flathead were amended to be minimum size 35 cm and maximum size of 40 cm. Inclusion of a maximum size limit is to protect larger and older individuals from being removed, increasing the size/age structure and reproductive potential of the population. This has been shown to be an effective management tool for increasing biomass yield in fish stocks (Ahrens *et al*., 2020). The Department of Natural Resources and Environment Tasmania (NRE Tas) plans to review these output controls from November 2025 against stock recovery targets as part of a rebuilding plan for Southern Sand Flathead.

10.3.2 Black Bream

Black Bream (*Acanthopagrus butcheri*) are primarily an estuarine species found around Australia's southern coastline. This species is believed to complete their entire life cycle within a given estuarine system (Sarre and Potter, 1999). Tracey *et al*. (2011) found that tagged Black Bream individuals all remained within the Derwent Estuary, suggesting strong fidelity to the Derwent. High movement rates within the estuary were recorded during spring, summer and autumn, with movement rates dramatically decreasing over winter, suggesting overwintering behaviour. Almost all tagged individuals moved upstream in spring, which correlates well with the known spring/summer spawning season for Black Bream in Australia (Haddy and Pankhurst, 1998; Tracey *et al*., 2011).

Black Bream caught in the Derwent Estuary have been found to have elevated levels of mercury, over two times above the maximum Food Safety Guideline level (**Section 8**). This is thought to be due to the species site fidelity, and longevity, being known to live to over 30 years of age (Jenkins *et al*., 2006; Verdouw *et al*., 2011). Therefore, it is recommended not to consume any Black Bream caught in the Derwent Estuary.

10.3.3 Brown Trout

Brown Trout (*Salmo trutta*) is an introduced species to Tasmania. This species is widely distributed throughout inland lakes, rivers, and streams throughout Tasmania, as well as estuarine areas. It is a popular recreationally targeted species and populations in various inland waters are managed by the Inland Fisheries Service (IFS). This exotic species can be highly detrimental to native fish populations, either through direct predation or outcompeting them for resources (Crowl *et al*., 1992; Ault and White, 1994; McDowall, 2006).

Brown Trout are common in the Derwent Estuary (Tracey *et al*., 2011). Recreational fishers targeting trout in the Derwent Estuary should be aware of the potential health implications of consuming seafood caught from the estuary (see **Section 8**).

10.3.4 Whitebait

Whitebait are a group of small, transparent native fish species that occur in schools (Table 10.1). Whitebait are mostly juveniles, including Australian Grayling listed as Vulnerable under the *Environmental Protection and Biodiversity Conservation Act 1999* (EPBC Act). Tasmanian Whitebait (*Lovettia sealii*) included in the schools are adults. These anadromous fish migrate from the sea upstream into the River Derwent and inland streams between spring and early summer to spawn.

Table 10.1 Fish species that can be found within whitebait schools in the Derwent Estuary, and their listing under the EPBC Act.

Common name	Scientific name	EPBC Act listing
Australian Grayling	*Prototroctes maraena*	Vulnerable
Tasmanian Whitebait	*Lovettia sealii*	Not listed
Common Galaxias	*Galaxias maculatus*	Not listed
Climbing Galaxias	*Galaxias brevipinnis*	Not listed
Spotted Galaxias	*Galaxias truttaceus*	Not listed
Tasmanian Mudfish	*Neochanna cleaveri*	Not listed
Tasmanian Smelt	*Retropinna tasmanica*	Not listed
Small-mouthed Hardyhead	*Atherinosoma microstoma*	Not listed

The Tasmanian whitebait fishery was opened in 1941, and by 1947 the catch had peaked at approximately 515 tonnes (Roughley, 1966). In 1974, the whitebait fishery was closed following several years of low catches mainly due to unsustainable fishing practices, including targeting spawning adults, which were removed before they had a chance to spawn upstream. By 1990 stocks were considered sufficiently recovered to support a small recreational fishery, which continues today. The Tasmanian whitebait season is from 1 October to 11 November each year, with regulations on netting equipment, daily and seasonal catch limits.

The sustainable management of the recreational fishery is aimed at protecting populations of Tasmanian Whitebait, which are particularly vulnerable to overfishing. This species has only a one-year lifecycle, therefore any impacts on spawning efforts could have serious implications for the population. The timing of the season and selection of open rivers is designed to target the runs of juvenile Common Galaxias and Spotted Galaxias, rather than Tasmanian Whitebait. By targeting whitebait species in the latter-part of the seasonal run, the majority of adult Tasmanian Whitebait have an opportunity to spawn while the anglers keen to target whitebait, have the chance to lawfully take fish.

10.3.5 Diadromous fish

Diadromous fish are species which migrate between marine and freshwater environments during their lifecycle (Myers, 1949). Four diadromous fish species utilise the Derwent Estuary as a migratory pathway between freshwater and marine environments: the Australian Grayling (*Prototroctes maraena*), Southern Shortfin Eel (*Anguilla australis*), Pouch Lamprey (*Geotria australis*) and Shorthead Lamprey (*Mordacia mordax*).

The Australian Grayling is a native fish, endemic to southeast Australia, and is predominantly a freshwater species. Adults will spawn in freshwater and larvae are swept downstream into coastal areas, juveniles will remain in marine waters for approximately 6 months, before migrating back to inland waters. As this migration between freshwater and marine waters is not for the purposes of breeding, Australian Grayling are considered amphidromous. This species is particularly vulnerable to a range of threats including changes in water quality, particularly sedimentation and siltation, predation from introduced species, instream barriers impacting migration movements (Backhouse *et al*., 2008), changes to riverine flow (due to dams, diversions and irrigation takes), and climate change (Koster *et al*., 2020).

Southern Shortfin Eels are a catadromous species, meaning they spend the majority of their lives in coastal and lowland freshwater areas and migrate downstream to marine environments to spawn (Myers, 1949; McDowall, 1988). They reach sexual maturity between 6 to 20 years of age. Southern Shortfin Eels are distributed along coastal river systems of southeast Australia, being found as far north as southern Queensland. In Tasmania, Shortfin Eels can be found in most major catchments across the state. They undertake a significant migration of over 3,000 km north to spawning grounds within the Coral Sea, off the coast of northern Queensland (Koster *et al*., 2021). After spawning adults will die, and larvae are transported by currents from spawning grounds to the continental shelf. They then metamorphose into transparent glass eels, after which they can actively swim towards freshwater environments. Once they begin to move up into freshwater systems, they quicky develop into fully pigmented elvers and adjust to freshwater where they will spend the majority of their lives.

Southern Shortfin Eel can live up to 24 years of age, grow to 1 m in length and 3 kg in size. They are one of the largest, native, inland freshwater fish in Tasmania, and would have been an important cultural resource for Tasmanian Aboriginal people. Eel are targeted by both recreational and commercial fishers within Tasmania.

Pouch Lamprey and Shorthead Lamprey on the other hand are anadromous, meaning they spend the majority of their lives in the marine environment, migrating to freshwater systems as adults to breed (Myers, 1949; McDowall, 1988). Lamprey are an eel-like fish, which are common in inland freshwater areas around Tasmania. They are morphologically primitive, as they lack a true jaw, instead possessing an oral sucking disc, which is armoured with tooth plates arranged in a radial pattern (Gomon and Bray, 2023). Lampreys have several distinct life stages; during their larval phase they can be found buried in sediments of freshwater streams and rivers, filter feeding on detritus and micro-organisms (Moore and Mallatt, 1980). At three to seven years of age, lamprey metamorphose into adults, developing eyes, sucking disc and enlarged dorsal fins, and migrate downstream to marine environments. Little is known about the marine phase of lamprey; however, it is inferred that they are parasitic to larger fishes, using their sucking disc and rasping teeth to feed on fluids and flesh of their host. Once fully grown, they will return to freshwater systems, where they become sexually mature, spawn and die (Miller *et al*., 2021).

Threats

The biggest threats to diadromous fish species are large instream barriers (such as dams, weirs and flood gates) and regulation of riverine flow (Boubee *et al*., 2003; Miles *et al*., 2014). These structures interrupt important life-cycle migration events and have contributed to local extinctions of diadromous species (van Rijssel *et al*., 2024). Regulation of riverine flow can disrupt important environmental cues (Perkin and Wilson, 2021; Koster *et al*., 2024), meaning migrations may occur at incorrect times, or be missed completely, reducing survival chances during migratory events (Harris, 1984). Migratory fish species can enter freshwater areas of the Derwent catchment via the Plenty River, Tyenna River and Styx River; however, they cannot naturally migrate through the River Derwent due to Meadowbank Dam.

Meadowbank Dam fish trap

Since 2007 a static shore-based fish trap has been seasonally installed at the base of Meadowbank Dam. This trap facilitates the movement of juvenile Southern Shortfin Eels (elvers) and adult Pouch Lamprey upstream of Meadowbank Dam. No Shorthead Lampreys have been observed in the Meadowbank fish trap (pers comms J. Yick, IFS, 2025). Transfer of elvers to upstream locations has been shown to be successful in maintaining eel populations in systems impacted by hydroelectrical infrastructure, including dams and power stations (Boubee *et al*., 2003). IFS is responsible for seasonal deployment (October to March) of this fish trap. Trap deployment is dependent on Meadowbank Dam flows, access and resource availability. As the timing of deployment may not be consistent, comparing total annual catches should be done with caution. As a result, total catch is not an indication of these species stock status, but a reflection of total volume being translocated.

Since the 2014/15 season over 7,000 kg of elvers have been transferred from below Meadowbank Dam to Lake Meadowbank and other inland Tasmanian freshwater systems (Figure 10.1). During the 2024/25 season the average count of elvers captured was 287/kg (Mawbey, 2025). Therefore, cumulatively the total number of elvers transferred upstream of Meadowbank Dam since 2014/15 is likely to be over two million individuals.

During the 2024/25 season Pouch Lamprey catch volume was low compared to other years (Figure 10.2). Low catches were attributed to road-access restrictions, meaning that the Meadowbank trap was not set until the end of November, missing the peak lamprey migration timing (Mawbey, 2025).

Figure 10.1 Total Southern Shortfin Eel (*Anguilla australis*) elver capture from Meadowbank trap 2014/15 season to 2024/25 season. Total catch numbers are volume of elvers translocated upstream of Meadowbank Dam each season and are not indicative of overall population numbers. Data supplied by Inland Fisheries Service.

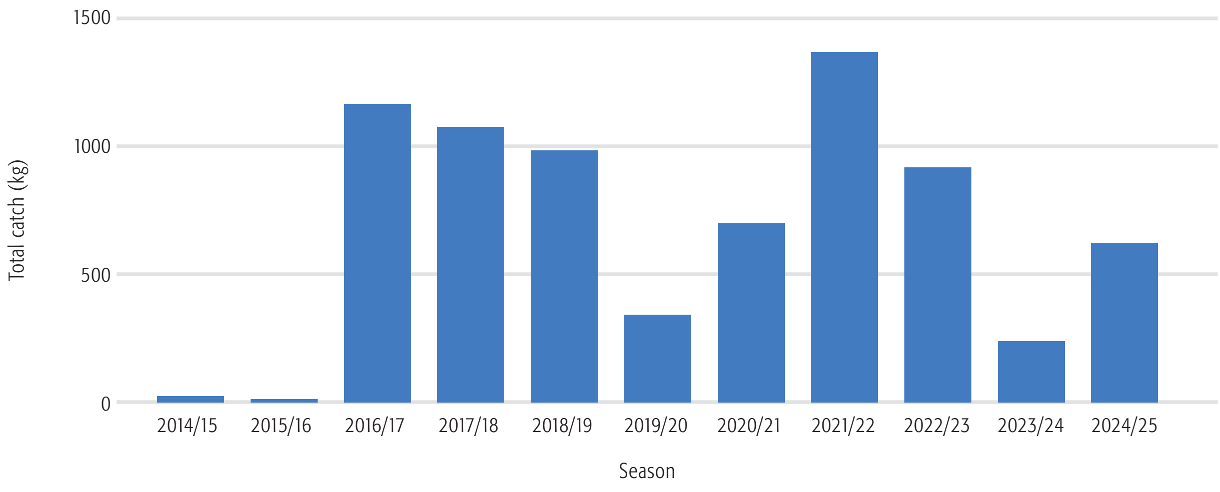
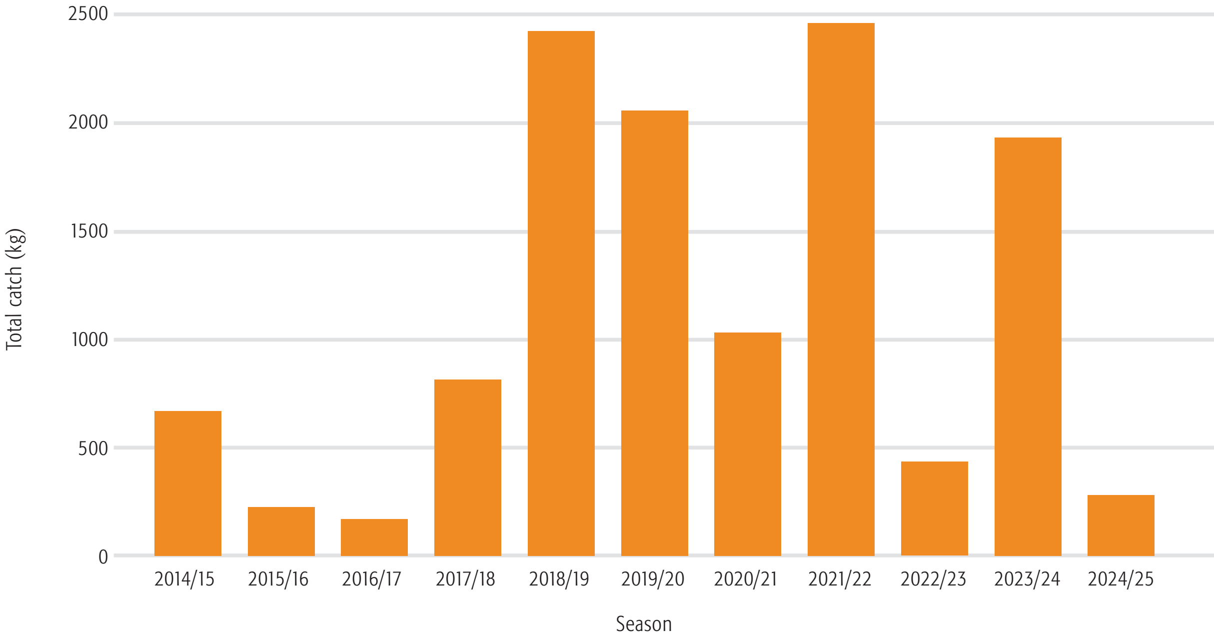


Figure 10.2 Total Pouch Lamprey (*Geotria australis*) capture from Meadowbank trap 2014/15 season to 2024/25 season. Total catch numbers are volume of lamprey translocated upstream of Meadowbank Dam each season and are not indicative of population numbers. Data supplied by Inland Fisheries Service.



10.3.6 Spotted Handfish

Spotted Handfish (*Brachionichthys hirsutus*) are a cryptic, demersal fish, that is endemic to coastal waters of south-east Tasmania (Last and Gledhill, 2009). They are small and stout, with adults being relatively sedentary. Handfish are unique in that their pectoral fins resemble hands, which they use to 'walk' along the bottom of the seabed.

Spotted Handfish may live up to 10 years of age, however fish generally live to approximately five years (Bessell, 2018). Females reach sexual maturity around two to three years of age (Bruce *et al*., 1998), which leaves only a short window for reproduction. The Spotted Handfish breeding season is between September and October, during which time adults will aggregate to find a mate (Bruce and Green, 1998). Egg masses are then laid onto benthic structures, such as stalked ascidians, seagrass, sponges, and polychaete worm tubes, which are guarded by females until they hatch (Bruce *et al*., 1997; Ross, 2001).

The current known range of Spotted Handfish is coastal regions in south-east Tasmania. Some local populations are restricted to fragmented colonies within the Derwent Estuary (Wong *et al*., 2018) and D'Entrecasteaux Channel (Wong and Lynch, 2017). Nine established local populations are studied within the Derwent Estuary and have been consistently surveyed since 2015 (Lynch, Green, *et al*., 2022) to inform trends in populations. This species was previously abundant throughout embayment's on Tasmania's east coast and up to Bass Strait (Last *et al*., 2007), and this southward restriction in their population is thought to be due to anthropogenic and climate change impacts in these coastal areas (Lynch, Green, *et al*., 2022). Once abundant in nearshore areas of the lower Derwent Estuary, significant declines in Spotted Handfish were observed in the late 1980s, and extensive surveys were undertaken throughout the estuary in 1994 and 1996 to assess their population. These surveys identified several fragmented colonies across the lower estuary, much less than their former range (Barrett *et al*., 1996). This resulted in Spotted Handfish being the first bony, marine fish species to be listed as Critically Endangered on the International Union of Conservation of Nature Red List and subsequently listed under the EPBC Act and Endangered under the TSP Act. There have been four consecutive recovery plans for the Spotted Handfish; 1997–2001 (Bruce and Green, 1998); 2002–2006 (DPIPWE, 2002b); 2005–2014 (DEH, 2005) and 2015–2025 (Department of the Environment, 2015). The National Handfish Recovery Team (NHRT) is in the process of drafting the new conservation action plan to be finalised by the end of 2025.

Spotted Handfish are particularly vulnerable to threats due to their local endemism, limited dispersibility, life history cycle, direct recruitment of juveniles and shallow, nearshore habitat range (Bruce *et al*., 1997). Within the Derwent Estuary, it is thought that initial declines observed in this species were due to habitat degradation from scallop dredging fishery that operated in the region until 1967 (Edgar and Samson, 2004), and from significant loss in seagrass within the lower estuary (Lynch, Soo, *et al*., 2022). The introduction of the invasive Northern Pacific Seastar (*Asterias amurensis*) has further degraded spawning habitat for the Spotted Handfish by reducing the abundance of stalked ascidians through direct predation (Lynch, Green, *et al*., 2022). Other threats include urbanisation (Lynch, Green, *et al*., 2022) causing local pollution, elevated nutrients and sedimentation through stormwater outflows, local filamentous algal blooms (Stuart-Smith *et al*., 2021), and chain moorings (Lynch *et al*., 2015; Wong *et al*., 2018). Geonomics conducted on seven sites within the Derwent Estuary found these Spotted Handfish populations to be genetically isolated from one another (Appleyard *et al*., 2021). This further exacerbates the risks to Spotted Handfish populations, as species that are demographically distinct face a higher risk of local extinction (Lynch, Green, *et al*., 2022). Increasing water temperatures due to climate change in coastal areas is also considered a significant threat to Handfish species (Stuart-Smith *et al*., 2020).

Due to significant losses in Spotted Handfish spawning habitat in the Derwent Estuary, artificial spawning habitat (ASH) was developed to supplement this natural habitat. Since 1998 over 15,000 ASH have been 'planted' in areas with low stalked ascidian densities (Lynch, Green, *et al*., 2022). There has been good uptake in use of ASH for spawning activities (Green, 2012; Edgar *et al*., 2017) and has been found to benefit Spotted Handfish populations (Lynch, Green, *et al*., 2022). Initially ASH was made from plastic, however in 2017 the Commonwealth Scientific and Industrial Research Organisation (CSIRO) collaborated with ceramic artist, Jane Bamford, to develop ceramic ASH for the captive breeding population. After a Spotted Handfish spawned on ceramic ASH in captivity, a design for ceramic ASH for the Derwent Estuary populations was developed based on the plastic ASH CSIRO had previously been using (Lynch, Soo, *et al*., 2022). The current design of ceramic ASH for deployment is made from Southern Ice Porcelain, rolled into a single stem of 9 mm diameter. The porcelain is fired in oxidation to 1280°C for vitrification to maximise durability, making the ASH suitable for long-term deployment in the marine environment. Biomimicry principals were used including using white porcelain to resemble the appearance of stalked ascidians, providing a suitable structure for Spotted Handfish to spawn in the wild. Jane Bamford has made 7,000 ceramic ASH for deployment in the Derwent Estuary, and continues to create ASH for the captive breeding program. Investigation into spawning habitat preference by Spotted Handfish found that ceramic ASH was preferred to plastic, however, fish still prefer to spawn on natural habitat if available (Hormann, 2019).

Figure 10.3 Ceramic ASH for the Derwent Estuary Spotted Handfish (*Brachionichthys hirsutus*) populations, created by Jane Bamford 2019. Image: Peter Whyte.

In 2017, CSIRO collected brood stock of 20 adults from eight local populations in the Derwent Estuary (Lynch *et al*., 2017). Two satellite populations were established at Seahorse World Tasmania and SEA LIFE Melbourne Aquarium, as a conservation strategy to ensure redundancy for this critically endangered species. In collaboration with aquarium industry partners captive breeding techniques have been refined, resulting in successful breeding events in 2018, 2022, 2023, and 2024. Currently SEA LIFE Melbourne Aquarium has a captive population of nine adults, and one juvenile from a successful breeding event in 2023. Seahorse World in Tasmania maintains 15 adults and 30 juveniles from a successful breeding event in September 2024.

Derwent Estuary Spotted Handfish monitoring and restoration project 2024–2025

Provided by Carlie Devine, Felicity McEnnulty and Leah Soo, CSIRO

Nine local populations within the Derwent Estuary have been consistently surveyed since 2015, with survey data dating back to 1997 (Lynch, Green, *et al*., 2022). Regular surveying of critically endangered species is essential to make robust assessments of populations for conservation management interventions (Nichols and Williams, 2006). This is particularly important for Spotted Handfish, where there have been historic rapid collapses of local populations. Insecurity in funding has been noted as a key reason for gaps in monitoring and conservation efforts for this species (Lynch, Green, *et al*., 2022).

From 2015 onwards, a standardised general underwater visual census (GUVC) survey method, conducted outside Spotted Handfish breeding season, has been used to monitor Spotted Handfish populations in the Derwent Estuary (Lynch, Green, *et al*., 2022). Currently, underwater remote operated vehicles (ROVs) are being used as a tool to survey for Spotted Handfish in south-east Tasmania for development approvals. However, the effectiveness of ROVs in surveying for Spotted Handfish has not yet been thoroughly investigated.

In 2023, the Derwent Estuary Program (DEP) received funding for conservation actions for the threatened Spotted Handfish through the Australian Government's Urban Rivers and Catchment Program. Funding was delivered to this project with co-investment from CSIRO, and in-kind contributions from SEA LIFE Melbourne Aquarium and Seahorse World Tasmania.

The aims of this project were to:

- Conduct monitoring of Spotted Handfish at nine colony locations in the Derwent Estuary.
- Conduct ascidian counts at nine colony locations in the Derwent Estuary to determine sites for ASH deployment.
- Deploy 1,500 ASH across sites where ascidian densities are observed to be low.
- Undertake ROV surveys and investigate suitability of this tool for future monitoring of Spotted Handfish.
- Release captive-bred Spotted Handfish juveniles to the Derwent Estuary.
- Deliver a report to outline the updated population status of the Spotted Handfish and future conservation strategies for its protection upon completion of the project.

Spotted Handfish surveys

GUVC surveys for Spotted Handfish at nine colony locations in the Derwent Estuary were conducted between February and July 2024. A total of 68 Spotted Handfish were observed across all nine sites during monitoring (Table 10.2). Fish were observed in shallow waters from 5.5 to 12.1 m. The largest fish recorded was 124 mm in length and the smallest fish was 60 mm. The surveys recorded a total of eight juvenile fish, with juveniles being determined as fish <71 mm in length.

Table 10.2 Survey counts of Spotted Handfish (*Brachionichthys hirsutus*), conducted between February and July 2024, in the Derwent Estuary. Average fish length and average fish depth at each site is also listed.

Site	Fish count	Average fish length (mm)	Average fish depth (m)
Site 1	9	94	8.5
Site 2	12	103	6.7
Site 3	5	75	6.7
Site 4	10	98	6.6
Site 5	4	97	8.3
Site 6	7	83	8.1
Site 7	1	98	8.1
Site 8	12	93	7.9
Site 9	8	92	7.2

The 2024 survey of Spotted Handfish colonies in the Derwent Estuary contributes to the long-term dataset of this species. There is interannual variability in modelled population densities of Spotted Handfish at each site (Figure 10.4). At some sites there is an observable decline in Spotted Handfish densities between 1998 and 2024, however from 2014 onwards there appears to have been a stabilisation in population across most sites (Lynch, Green, *et al*., 2022). Unfortunately, site seven may soon become locally extinct (Devine *et al*., 2024).

Figure 10.4 Time-series (1997–2024) of modelled estimated densities of Spotted Handfish (*Brachionichthys hirsutus*) per hectare with standard error bars at nine sites within the Derwent Estuary. Annotated demarcation (dashed green line) distinguishes monitoring methods of set-line transect versus GUVC surveys. Image: CSIRO.

ASH installation in the Derwent Estuary

ASH was deployed at five sites on the eastern shore of the Derwent Estuary where ascidian densities were found to be low, and where multiple fish had been sighted during surveys. A total of 1,495 ASH were planted in July 2024.

Deployed ASH were inspected in September and October 2024 for robustness and use by Spotted Handfish for spawning. In terms of robustness in September, 99.6% of the ASH were observed to be intact, and 88.1% were still intact in October. This demonstrates the robustness of the planted ASH in the marine environment. In September, 9% of ASH was observed to be in use by brooding fish or occupied by spawning or gravid females accompanied by males. There were no nearby egg clutches sighted on natural stalked ascidians. In October 4.7% of ASH were observed to be in use, again, no egg clutches were observed on nearby stalked ascidians. The uptake of ASH by Spotted Handfish at these sites clearly demonstrates the need for this ASH.

Figure 10.5 Spotted Handfish (*Brachionichthys hirsutus*) using ceramic ASH in Derwent Estuary. Image: Carlie Devine, CSIRO.

ROV surveys

ROV footage was collected alongside GUVC surveys of Spotted Handfish in the Derwent Estuary, between February and July 2024. This footage is currently being analysed by CSIRO researchers. Thorough investigation into the effectiveness of ROV surveys as a tool for Spotted Handfish monitoring, as well as recommendations for standardised methodologies, will be published in the final conservation package report to be delivered at the end of 2025.

Captive breeding program and release of head-started juveniles

In 2017, CSIRO, in collaboration with industry partners Seahorse World and SEA LIFE Melbourne Aquarium, initiated a captive breeding program for Spotted Handfish. After several successful breeding attempts, CSIRO in collaboration with the NHRT began working on a release strategy for captive bred juveniles into the wild. This involved obtaining the necessary approvals and permits to transport and release captive-bred juveniles of a critically endangered species.

A Spotted Handfish captive breeding strategy (2023) was developed with contributions from the NHRT, CSIRO, SEA Life Melbourne Aquarium, Zoo and Aquarium Association Australasia, NRE Tas and Seahorse World. This strategy provides high-level direction and outlines general guiding principles to ensure consistency in captive breeding practices for Spotted Handfish. This strategy will be superseded by a Regional Species Plan, which will include detailed species management considerations, including husbandry techniques and required resources to manage Spotted Handfish, to be published at the end of 2025.

Captive-bred juvenile release

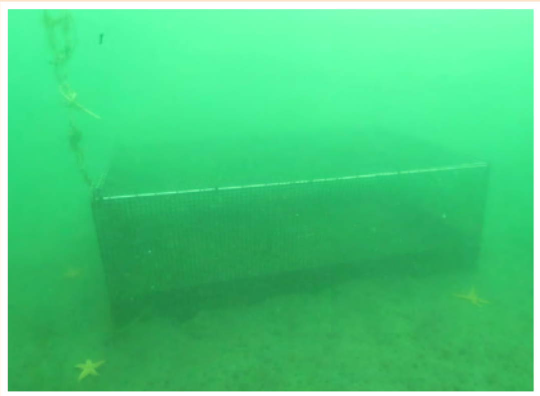
Wild-bred juvenile Spotted Handfish face high mortality rates from predators, disease, and environmental stress. Head-starting captive-bred juveniles is a conservation method that can improve survival rates by rearing them past their most vulnerable early stages before release. In captivity, they are provided with stable conditions, protection from predators, and live prey to develop natural foraging skills. This approach increases their chances of thriving in the wild and supports population recovery.

There is currently no established method for translocation of captive-bred, cryptic fish into the wild. In 2021, a 'hard release' of Red Handfish (*Thymichthys politus*) was trialled, resulting in rapid dispersal of individuals, making it difficult to assess the post-release survival of released individuals (Stuart-Smith *et al*., 2021).

A new 'soft release' translocation method was trialled for Spotted Handfish to the Derwent Estuary for this project. Where juveniles are initially released into 'handfish hotels', which are temporary structures designed to facilitate monitoring of released individuals during the crucial initial stages of release into the wild (Figure 10.6).

Fifteen captive-bred individuals from Seahorse World were randomly selected for release, the juveniles were seven months old and measured between 2.7 mm and 3.3 mm in length. These individuals were released to the Derwent Estuary in April 2025. Fish acclimatised well to the new environment, settled comfortably on the seafloor with their fins down, a sign they were relaxed and not in a defensive state, and in the following days, were observed foraging for food. Released fish gradually vacated the artificial 'handfish hotels', with one remaining inside for seven days and two fish observed outside the hotels ten days post release. The hotels were removed 14 days after release, with monitoring continuing at the site for a further six days and again months later. This new 'soft release' method trialled by CSIRO has provided valuable insights into best practices for releasing captive-bred Handfish into the wild and will help support the recovery of wild populations.

Figure 10.6 'Handfish hotels' installed in the Derwent Estuary for 'soft release' of captive bred Spotted Handfish (*Brachionichthys hirsutus*). Image: Carlie Devine, CSIRO.



Habitat mapping

Spotted Handfish have been shown to exhibit habitat preference for more complex microhabitat features, compared to flat sandy substrates and areas dominated by ephemeral, filamentous algae. These more complex microhabitat features have higher 3-D complexity, including depressions and ripples in sandy substrates. These more complex microhabitats provide cover for predator avoidance, increase foraging opportunities, and provided higher-quality spawning sites (Wong *et al*., 2018). Current habitat mapping of the Derwent Estuary lacks detailed classifications of 'sand' substrates (Lucieer, Lawler, and Pender, 2007) and available biota, such as stalked ascidians, which is a preferred spawning habitat. More in-depth mapping across nearshore areas within the Derwent Estuary will provide useful information for better management of critical habitat areas for Spotted Handfish. Specifically, this mapping could help guide more extensive Spotted Handfish surveys, management of anthropogenic impacts including developments, nutrient loading and boat moorings, and help guide conservation actions such as deployment of ASH. The DEP in collaboration with CSIRO will look for funding opportunities for a project to undertake this mapping work.

Recommendations

Ongoing population and habitat monitoring is essential in a rapidly changing environment particularly in the face of continued urbanisation in proximity to Spotted Handfish populations and the increasing frequency of marine heatwaves. In addition to biological data, climate-related information, such as salinity, nutrient levels, and temperature, is critical for assessing habitat suitability across sites. Regular assessments provide insight into species trends and habitat condition, helping to identify ecological refugia that support the persistence of wild populations. This information also underpins the strategic selection of future release sites for captive-bred individuals, ensuring that conservation efforts are both targeted and resilient to environmental change.

10.4 Shark, skates and rays

The Derwent Estuary, particularly Ralphs Bay is recognised as an important nursery area for sharks, skates and rays. Sheltered, near-shore coastal areas provide important habitat for juvenile sharks, skates and rays to feed and grow in their early years. The Derwent Estuary has been recognised as a Shark Refuge Area to protect breeding adults, and growing juveniles. Fishing restrictions apply within these Shark Refuge Areas, and no sharks, skates or ray of any kind can be taken. If caught accidentally in nets or line, they must be returned to the water as soon as possible to minimise harm to captured individuals. The effectiveness of these Shark Refuge Areas in protecting shark populations is uncertain, as the limited research investigating this has found their effectiveness to be relatively inconclusive (Stevens and West, 1997; McAllister *et al*., 2017).

More than ten species of shark can be found in the Derwent Estuary. The most common species being the Broadnose Sevengill Shark (*Notorynchus cepedianus*), Gummy Shark (*Mustelus antarcticus*), School Shark (*Galeorhinus galeus*), Draughtboard Shark (*Cephaloscyllium laticeps*), Whitespotted Dogfish (*Squalus acanthias*), and Australian Angelshark (*Squatina australis*). Different shark species are morphologically distinct, depending on which habitats they occupy and their feeding mechanisms. They fill different important ecological niches within food webs depending upon target species and where they sit within the food chain. Apex predators like the Broadnose Sevengill Shark, predate on mammals, other shark species and larger teleosts (Barnett *et al*., 2010). Some species, like the Gummy Shark, are more mid-level predators feeding on smaller teleosts and cephalopods; and others are benthic feeders consuming smaller benthic crustaceans, such as the Draughtboard Shark and Whitespotted Dogfish (Yick *et al*., 2012). Therefore, sharks are important in regulating populations within the Derwent Estuary across trophic levels, from other sharks, fishes, cephalopods and small benthic crustaceans.

10.5 Birds

A diverse range of bird species can be found across the Derwent Estuary's broad habitat types. Areas of particular importance to large numbers of birds and threatened bird species include upper estuary wetland areas and macrophyte beds, sheltered bays and wetlands of the middle estuary, Lauderdale saltmarsh and tidal flats. The South Arm area is recognised as a Key Biodiversity Area for migratory shorebirds. The Derwent Estuary provides important nesting grounds for seabirds including Silver Gulls (*Larus novaehollandiae*), Short-tailed Shearwater (*Puffinus tenuirostris*), and Little Penguin (*Eudyptula minor*) which breed at several locations along the foreshore of the lower estuary. Native foreshore vegetation supports several important woodland bird species including EPBC Act listed species Endangered Forty-spotted Pardalote (*Pardalotus quadragintus*) and Critically Endangered Swift Parrot (*Lathamus discolor*).

10.5.1 Upper estuary waterbirds

The expansive tidal wetland area and submerged macrophyte beds of the upper Derwent Estuary are inhabited by an abundant and diverse community of waterbirds and wetland birds. This area is located within the River Derwent Marine Conservation Area, and offers refuge to waterfowl from game hunting, which occurs across many other reserves and wetland areas in the state. The extensive submerged macrophyte beds support a large population of Black Swan (*Cygnus atratus*) that are present year-round. Before this area was designed as a Marine Conservation Area, it was proclaimed a 'sanctuary with respect to Black Swans' under the *Animals and Birds Protection Act 1919*. Other bird species that are common within this area include Eurasian Coot (*Fulica atra*), Grey Teal (*Anas gracilis*), Hoary Headed Grebe (*Poliocephalus poliocephalus*), White-faced Heron (*Egretta novaehollandiae*), Great Cormorant (*Phalacrocorax carbo*), Little Black Cormorant (*Phalacrocorax sulcirostris*) and Spotted Crake (*Porzana porzana*). Australian Pelican (*Pelecanus conspicillatus*) have been observed nesting in the reeds of Dromedary Marsh. Raptors have also been observed in this area predating on fish and other bird species including White-bellied Sea-Eagle (*Haliaeetus leucogaster*), Marsh Harrier (*Circus aeruginosus*) and Brown Falcon (*Falco berigora*).

The Great Crested Grebe (*Podiceps cristatus*) is also known to occur in the upper estuary wetlands. They feed on mainly small fish, but also aquatic invertebrates and insects that are abundant in these wetland areas. The Great Crested Grebe has quite a striking appearance when in breeding plumage with a black cap and erect crest, and an extensive ruff of black-edged, tawny-rufus feathers bordering a white face (Figure 10.7). They are also known for their elaborate mating displays. They are thought to only breed on Lake Dulverton near Oatlands, from November to March each year. They construct nests on heaps of floating vegetation anchored to reeds or drooping branches (Bryant and Jackson, 1999). This species is listed as Vulnerable under the TSP Act.

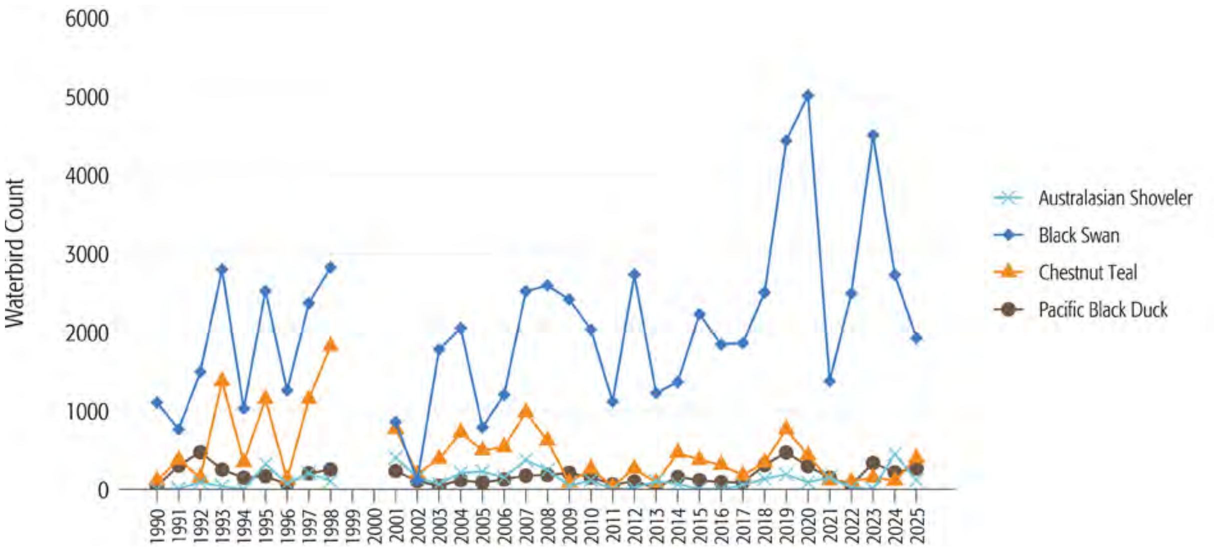
Figure 10.7 Great Crested Grebe (*Podiceps cristatus*) in breeding plumage. Image: Pierre Montieth.

For over three decades NRE Tas has been coordinating state-wide waterbird counts. These counts are conducted across 75 wetland areas of a variety of types (e.g. farm dams, inland reservoirs and estuaries) to give an indication of population trends. In the Derwent Estuary, counts are undertaken from south of Dromedary Marsh to the Bridgewater causeway and downstream to Green Point.

The waterbird species counted as part of this survey are Australasian Shoveler (*Anas rhynchotis*), Black Swan, Chestnut Teal (*Anas castanea*), and Pacific Black Duck (*Anas superciliosa*). All four species of waterbirds in the upper estuary exhibit interannual variability in counts (Figure 10.8). Waterbirds are highly mobile, and the upper Derwent is thought to be a drought refuge in low-flow years (e.g. summer 2018/19 and 2022/23), correlating with higher numbers observed in some species.

Black Swans are the most common waterbird in the upper estuary, with between 1,000 and 3,000 individuals regularly observed during counts. There is a general increasing trend in Black Swan population numbers in the upper Derwent, which is consistent with state-wide observations. Chestnut Teal counts in the upper Derwent in the last five years are lower compared to counts a decade ago, and much lower compared to the first eight years of survey data. This contrasts with state-wide counts, which show a slight increasing trend in population of Chestnut Teal since surveys began in 1985. Populations of Australian Shoveler and Pacific Black Duck appear to be relatively stable over the survey period.

Figure 10.8 Annual summer counts of Australasian Shoveler (*Anas rhynchotis*), Black Swan (*Cygnus atratus*), Chestnut Teal (*Anas castanea*), and Pacific Black Duck (*Anas superciliosa*) in the upper Derwent Estuary wetlands from 1990 to 2020. Data supplied by NRE Tas.



The Australasian Bittern

The Australasian Bittern (*Botaurus poiciloptilus*) is a secretive, stocky, heron-like bird, that inhabits inland and estuarine reed-dominated wetlands. Once abundant across south-east and south-west Australia, their habitat has been significantly impacted through drainage activities and altered hydrology within catchments. The Australian population in 2020 was estimated at 750–1800 individuals, with populations still declining (Herring *et al*., 2019), and Tasmanian population estimated to support 20–80 birds (Znidersic, unpublished). The major threats facing Australasian Bitterns is habitat loss caused by large-scale agriculture, and increased frequency and length of droughts, meaning many natural wetlands that support Bittern populations are dry for extended periods. The Australasian Bittern is currently listed as Endangered under the EPBC Act.

This elusive species requires large, relatively undisturbed wetland areas, where they breed in densely vegetated habitat, building nests in deep cover over shallow water. They are known to occur in the upper Derwent Estuary wetland areas of Dromedary Marsh and Murphys Flat. In summer 2022, the DEP collaborated with the Bookend Trust on their Call Trackers project. The aim of this project was to use sound recorders to find and monitor noisy but elusive species. Focus species for the project were the threatened Australasian Bittern and native bats. The Bittern has a distinctive booming call that can be heard over large distances. Acoustic monitoring is a great way to find and monitor a variety of species in an efficient and non-invasive manner. With advances in AI technology, audio files can be automatically processed and tagged for presence of Bittern and bat calls.

One acoustic recorder was installed in Dromedary Marsh, where Bitterns have been intermittently sighted. Over a 10-day period of deployment, Australasian Bittern calls were recorded on two occasions early in the morning. During field work, DEP staff observed Bitterns on the banks of the upper estuary on two occasions in 2023.

Figure 10.9 Australasian Bittern (*Botaurus poiciloptilus*) sighted in the upper Derwent Estuary in November 2023 by DEP staff.

10.5.2 Native Ducks

In early 2024, the DEP, in collaboration with the Pacific Black Duck Conservation Group, local councils and BirdLife Tasmania published the Ducks of Tasmania Booklet. This easy-to-read, pocket booklet has been created to help people identify and observe native ducks in Tasmania. The booklet contains useful information on where best to observe each species, alongside beautiful illustrations by Hobart-based artist Sam Lyne and promotes responsible behaviours towards local waterfowl (Figure 10.10).

Tasmania has 11 species of native ducks; some are year-round residents like the Pacific Black Duck (*Anas superciliosa*) and Australian Wood Duck (*Chenonetta* jubata). While others are occasional visitors like the Freckled Duck (*Stictonetta naevosa*) and Pink-eared Duck (*Malacorhynchus membranaceus*), who mainly inhabit inland wetland areas of mainland Australia.

Introduced ducks, like the Domestic Mallard (*Anas platyrhynchos domesticus*), are larger and more aggressive than most of the native duck species in Tasmania. These introduced ducks can form large flocks, particularly when they are fed by humans. This can lead to excessive duck faeces causing issues with water quality, and trampling of vegetation, which is important habitat for other species. Domestic Mallards are closely related to the native Pacific Black Duck and can crossbreed to create fertile hybrids. Pacific Black Ducks are now at risk of extinction in Tasmania, as a result of outcompeting and replacement by invasive hybrids.

Feeding ducks does not only benefit invasive duck species but can also be detrimental to native species causing illness and deformities in bone structure due to calcium deficiencies. Native duck species all have specialised diets, which they obtain through foraging in wetland and foreshore areas, reliance on feeding means these valuable skills can be lost. Feeding also means that waterfowl become more comfortable around humans and threats, making them more vulnerable to predators, such as feral cats and dogs.

As an alternative to feeding ducks, the Ducks of Tasmania booklet can be used by keen and amateur birdwatchers alike to observe the diverse duck population in Tasmania instead. Duck booklets can be obtained from local councils in Hobart, or on the DEP website Ducks page (www.derwentestuary.org.au/ducks/).

Figure 10.10 Chestnut Teal (*Anas castanea*) native duck from Ducks of Tasmania Booklet. Illustrated by Sam Lyne.

10.5.3 Shorebirds

Shorebirds are a diverse group of wading bird species found near intertidal habitats in coastal, estuarine and wetland areas; they forage for food by wading in shallow waters (Geering *et al*., 2007). They can be classified as either resident shorebirds, that are mainly territorial species inhabiting locations year-round, or migratory shorebirds. Australia supports approximately 37 species of migratory shorebirds outside of their breeding season. These species undertake annual migrations using the East Asian-Australasian Flyway (EAAF) from their Arctic breeding areas. Double-banded Plovers (*Charadrius bicinctus*) breed in New Zealand before migrating to southeast Australia, including Tasmania, for the winter; this east-west migration is unique among shorebirds worldwide. Australia offers important habitat to these migratory shorebird species to rest and build up critical energy reserves to undertake northward migrations of thousands of kilometres. The coastline and mudflats of the Derwent Estuary provide critical habitat for the shorebird populations of south-east Tasmania.

Resident Shorebirds

The coastline of south-east Tasmania is important for many species of resident shorebirds (Table 10.3), providing breeding, feeding and roosting (resting) habitats year-round for the breeding population, while also supporting non-breeding birds awaiting their opportunity to establish a breeding territory. Tasmania supports two important breeding populations of resident shorebirds, the Eastern Hooded Plover (*Thinornis cucullatus cucullatus*) and Australian Pied Oystercatcher (*Haematopus longirostris*) (Tasmanian Planning Commission, 2024). Tasmania supports more than 60%

of the Australian Eastern Hooded Plover population, with at least 750 breeding pairs found along coastal areas of Tasmania (Trebilco *et al*., 2021). The Eastern Hooded Plover is listed as Vulnerable under the EPBC Act, due to decreasing populations in southeast Australia. They inhabit and nest on flat, sandy ocean beaches in southeast Australia, which have been impacted by historic coastal development. Nesting occurs between August and March each year, which coincides with increased beach usage for human recreation. Birds are vulnerable to disturbance when nesting, and their small, speckled eggs blend into the sand, meaning they can be easily trampled.

Threats to resident shorebird populations include human disturbance in coastal areas including recreational activities, dogs and off-road vehicles (Schlacher *et al*., 2013; Trebilco *et al*., 2021). Coastal developments impact on beach size, structure and dune dynamics. Weeds, including Sea Spurge (*Euphorbia paralias*), Marram Grass (*Ammophila arenaria*) and Beach Daisy (*Arctotheca populifolia*) encroach on tidal flats and dune systems (Mitchell-Williams *et al*., 2022). Changing hydrological regimes and water extraction, particularly in inland areas has reduced crucial wetland habitat for Australian shorebird populations (Nebel *et al*., 2008). Increased flooding of nest sites as a result of climate change induced sea-level rise and increased storm surge is projected to have devastating impacts on shorebird populations, particularly in areas where there are no suitable upland areas for coastal retreat (Galbraith *et al*., 2002; van de Pol *et al*., 2024; Koivula *et al*., 2025) (see **Section 10.5.4** for investigation into Derwent Estuary seabird nest site exposure to storm surge inundation).

Table 10.3 Resident shorebird species known to inhabit the Derwent Estuary, and their listing under the EPBC Act.

Common name	Scientific name	EPBC Act listing
Eastern Hooded Plover	*Thinornis cucullatus cucullatus*	Vulnerable
Australian Pied Oystercatcher	*Haematopus longirostris*	Not listed
Sooty Oystercatcher	*Haematopus fuliginosus*	Not listed
Masked Lapwing	*Vanellus miles*	Not listed
Red-capped Plover	*Charadrius ruficapillus*	Not listed
Black-fronted Dotterel	*Elseyornis melanops*	Not listed

Eastern Hooded Plovers (*Thinornis cucullatus cucullatus*). Image: Elis Simpson.

Australian Pied Oystercatcher in southeast Tasmania

Provided by Dr Mike Newman and Dr Eric J Woehler, Australasian Wader Studies Group

The south-east Tasmania Summer Wader Counts (SWCs) are highly coordinated counts that monitor the regional populations of breeding and non-breeding adults and recently fledged juveniles. The south-east Tasmania SWC has been undertaken annually since 1973 and is the longest continuous data set of its type in Australia.

The Australian Pied Oystercatcher is an obligate coastal-breeding species in Tasmania, with delayed breeding maturity. Adults are typically at least four years old before birds first breed. Oystercatchers are long-lived species that aggressively defend their territories. Some Australian Pied Oystercatchers in southeast Tasmania, including birds breeding in the Derwent Estuary, have been recorded to live more than 30 years (Newman and Woehler, 2017).

Between 1973 and 2000, there was an approximate four-fold increase in the regional population (Figure 10.11). Since 2000, numbers have remained relatively constant, but with greater interannual variation. These complex population statistics can be explained by well-documented environmental improvements in the Derwent Estuary (DEP, 2015c) and on the adjacent foreshores, which have improved both breeding productivity and the survival of pre-breeding birds.

Figure 10.11 Regional summer counts of Australian Pied Oystercatcher (*Haematopus longirostris*) in southeast Tasmania, including the Derwent Estuary, 1973–2025 inclusive. The species is a year-round resident. Figure: Australasian Wader Studies Group.

The size of the south-east Tasmanian breeding population is limited by the number of breeding territories along the coast. When monitoring commenced in 1973, Australian Pied Oystercatchers did not breed in the Derwent Estuary upstream of the Tasman Bridge. Now they breed and forage approximately 20 km farther upstream in response to improved environmental conditions.

However, this gain has been offset by the widespread loss of viable coastal breeding territories caused by increased recreational activities on sandy beaches elsewhere. The net population gain observed in southeast Tasmania since 1973 is therefore attributed primarily to increased survival of immature and non-breeding birds foraging in the Derwent Estuary as a consequence of improvements in foraging habitats resulting from pollution control measures.

Australian Pied Oystercatcher (*Haematopus longirostris*) walking on beach. Image: Hayley Alexander.

Migratory Shorebirds of southeast Tasmania

Provided by Dr Mike Newman and Dr Eric J Woehler, Australasian Wader Studies Group

Tasmania is the southernmost destination for migratory shorebirds from the Northern Hemisphere that use the EAAF. These migratory bird species from the Northern Hemisphere and New Zealand can be observed in coastal, estuarine and wetland areas of Tasmania year-round.

Eighteen of the 37 migratory shorebird species that visit Australia can be found in south-east Tasmania, with several species present in numbers of international significance (Table 10.4). Tasmania currently supports an estimated 15,000–20,000 migratory shorebirds each summer. Multiple species' populations are decreasing rapidly, reflected in their EPBC Act Threatened Species assessment as Endangered or Critically Endangered.

Eurasian Whimbrel (*Numenius phaeopus*) standing on rock. Image: Barry Madden.

Table 10.4 Migratory shorebirds that have historically been observed in the Derwent Estuary – Pittwater Area, and their listing under the EPBC Act.

Common name	Scientific name	EPBC Act listing
Eastern Curlew	*Numenius madagascariensis*	Critically Endangered
Curlew Sandpiper	*Calidris ferruginea*	Critically Endangered
Common Greenshank	*Tringa nebularia*	Endangered
Lesser Sand Plover	*Charadrius mongolus*	Endangered
Bar-tailed Godwit	*Limosa lapponica*	Endangered
Eurasian Whimbrel	*Numenius phaeopus*	Vulnerable
Great Knot	*Calidris tenuirostris*	Vulnerable
Red Knot	*Calidris canutus*	Vulnerable
Sharp-tailed Sandpiper	*Calidris acuminata*	Vulnerable
Terek Sandpiper	*Xenus cinereus*	Vulnerable
Grey Plover	*Pluvialis squatarola*	Vulnerable
Ruddy Turnstone	*Arenaria interpres*	Vulnerable
Latham's Snipe	*Gallinago hardwickii*	Vulnerable
Sanderling	*Calidris alba*	Not listed
Pacific Golden Plover	*Pluvialis fulva*	Not listed
Grey-tailed Tattler	*Heteroscelus brevipes*	Not listed
Red-necked Stint	*Calidris ruficollis*	Not listed
Double-banded Plover	*Charadrius bicinctus*	Not listed

Coordinated bi-annual (summer and winter) counts at known roosting areas in south-east Tasmania since 1973 have established the longest time series in Australia for monitoring migratory shorebirds. The South Arm – Pitt Water coastal complex has been recognised as a Key Biodiversity Area for shorebirds. The shorelines and intertidal sand flats within this area provide important feeding and roosting habitat for shorebirds. This area includes Lauderdale (Ralphs Bay), Clear Lagoon, Pipeclay Lagoon, Calverts Lagoon, South Arm Neck, Mortimer Bay, Orielton Lagoon, Sorell, Barilla Bay, Five Mile Beach, Seven Mile Beach and Carlton River. The Pittwater – Orielton Lagoon wetland area, is recognized as internationally significant under the Ramsar Convention. The South Arm – Pitt Water area supports high diversity both of resident and migratory shorebird species, is a priority site for Eastern Curlew and Eurasian Whimbrel, and has been regularly monitored on a long-term basis (Bryant, 2002).

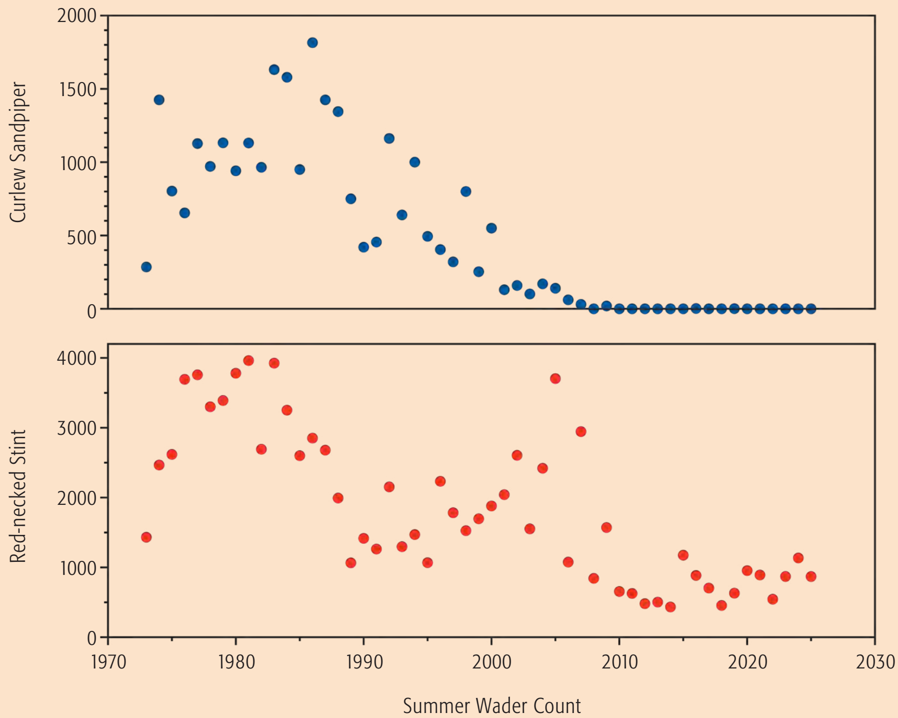
Figure 10.12 Non-breeding adult Red-necked Stint (*Calidris ruficollis*). Image: Ayuwat Jearwattanakanok.

Migratory shorebirds are impacted by many of the threats to resident shorebirds (listed above). However, due to their extensive migrations to the Northern Hemisphere are impacted by global threats, the biggest being loss of intertidal and wetland habitats. This is particularly prominent in the Yellow Sea. It is estimated that approximately two-thirds of intertidal habitat in the Yellow Sea area has been lost in the last 50 years as a result of coastal reclamation and increased sedimentation (Murray *et al*., 2014). The migratory shorebirds that rely on this area during their migratory journey have exhibited the biggest population decreases in Australia (Studds *et al*., 2017); with populations of Common Greenshank, Curlew Sandpiper and Far Eastern Curlew having decreased by more than 50% since 2013 (Rogers *et al*., 2023).

Although several factors affecting the quality of shorebird habitat in south-east Tasmania have changed over the last 60 years (Newman *et al*., 2020), they are not considered to have been the primary cause of the population decreases observed. There is strong evidence that habitat loss and environmental changes elsewhere in the EAAF are responsible, the effect of which is often first documented in south-east Tasmania because of its geographic position at the southern extremity of the flyway. South-east Tasmania provides an early indication of future population trends and responses elsewhere in the EAAF, including mainland Australia.

Within Tasmania, changes in species' populations provide indicators for national and international decreases. Changes in the numbers of migratory shorebirds at key habitats in south-east Tasmania over 50 years are clearly evident (Figure 10.13). The Curlew Sandpiper has not been observed in coordinated wader counts in south-east Tasmania since 2010. The population of Red-necked Stint observed in south-east Tasmania underwent a rapid decrease from 2005, however, juveniles are still observed returning to the region each year. It is critical to recognise and manage wetland complexes that support migratory shorebirds as a single system for management and conservation efforts, rather than as a collection of discrete sites. The loss or damage to one site can compromise the integrity of the entire complex. Shorebirds will move among sites depending on tides, winds, disturbance and require all sites to be available to them as their needs vary.

Figure 10.13 Regional summer counts of Red-necked Stint (*Calidris ruficollis*) (lower panel) and Curlew Sandpiper (*Calidris ferruginea*) (upper panel) in south-east Tasmania, including the Derwent Estuary, 1973–2025 inclusive. The species are annual migrants from the Northern Hemisphere. Figure: Australasian Wader Studies Group.



10.5.4 Seabirds

Gulls in southeast Tasmania

Provided by Dr Eric J Woehler, Australasian Seabird Group

Winter Kelp Gull (*Larus dominicanus*) and Pacific Gull (*Larus pacificus*) counts in south-east Tasmania commenced in 1980 by members of the Bird Observers' Association of Tasmania, later BirdLife Tasmania, under the initial coordination of Dr Bill Wakefield. Silver Gulls (*Larus novaehollandiae*) were incorporated in Winter Gull Counts (WGCs) in 1983. The 2025 WGC was the 42nd annual survey for south-east Tasmania (no WGCs were made from 1991 to 1994). It is believed that this is the longest time series for gull populations in Australia.

All gulls in coastal and near-coastal areas from Dover, northward throughout the d'Entrecasteaux Channel and Bruny Island foreshores, the Derwent Estuary, South Arm Peninsula, Sorell, Marion Bay and the Forestier and Tasman Peninsulas were counted (Figure 10.14). Nearshore sites in the Huon River, Storm Bay and d'Entrecasteaux Channel were counted by staff from Tassal and Huon Aquaculture. Kelp and Pacific Gull counts were classified into age classes (juvenile, immature/sub-adult and adult), while Silver Gulls were counted but not classified into age classes.

All three gull species' populations exhibit inter-annual variabilities, resulting from natural variability in their regional populations and regional movements among roosting and feeding sites during the non-breeding season. Current south-east Tasmanian populations for counted gulls remain higher than their respective initial counts in the 1980s (Figure 10.15).

Kelp Gull (*Larus dominicanus*) diving for food in the ocean. Image: Garret Skead.

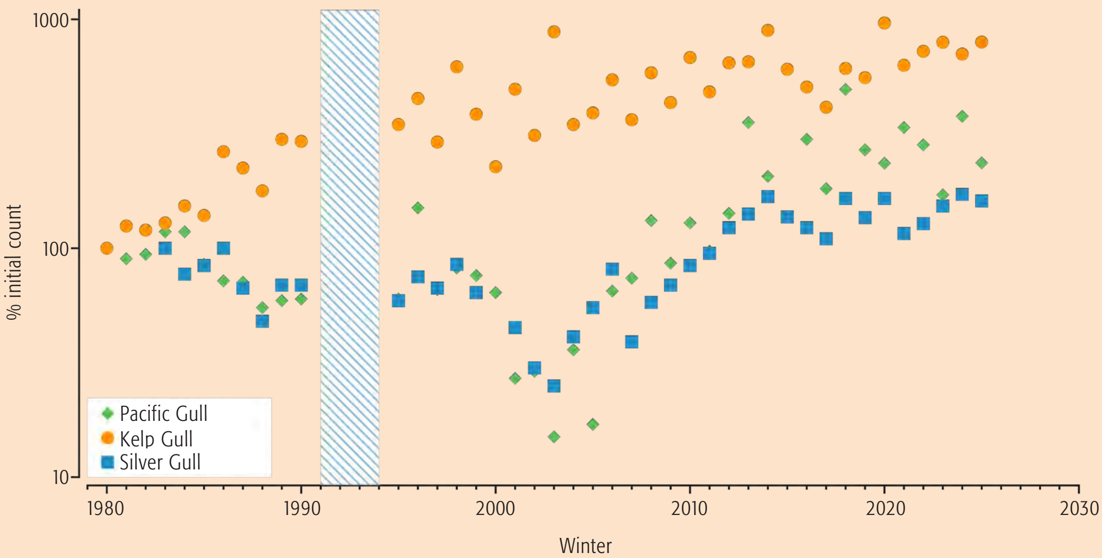
Figure 10.14 Indicative map showing approximate extents of coastal and near-shore areas counted during the 2025 Winter Gull Count (count areas in red). Grid is 10km. Image: Australasian Seabird Group.

Based on the 2025 WGC, the current regional population of Silver Gulls remains in excess of 16,000 birds, with substantial flocks observed along the Derwent foreshore from the Tasman Bridge to Kingston. Counts show that the regional populations of Pacific and Kelp Gulls are approximately 600 birds and 6,000 birds, respectively. Current analyses indicate that the rate of increase in the regional population of Kelp Gulls may be decreasing, possibly indicating that the population has reached the carrying capacity of southeast Tasmania.

Concern for the potential impact of the establishment of a breeding population of Kelp Gulls in southeast Tasmania on the resident population of Pacific Gulls was the primary reason for the initiation of the WGCs in 1980. On the basis of the 42-year data set, there is no evidence of any adverse effect arising from the regional Kelp Gull population in southeast Tasmania on the other two species of resident gulls.

In the 1990s, the tips in Hobart, Margate and Lauderdale supported high numbers of gulls, with counts often exceeding 2,000 gulls. Changing waste-management practices have reduced the opportunities for scavenging by gulls, and the Hobart and Glenorchy Tips typically support fewer than 100 gulls during recent WGCs.

Figure 10.15 Plot showing counts of Pacific (*Larus pacificus*) (green diamonds), Kelp (*Larus dominicanus*) (orange circles) and Silver (*Larus novaehollandiae*) (blue squares) Gulls, southeast Tasmania from Winter Gull Counts 1980-2025, expressed as the percentages relative to the initial species' counts in 1980 (Kelp and Pacific Gulls) and 1983 (Silver Gull). No counts were undertaken in 1991–1994, inclusive. Figure: Australasian Seabird Group.



Short-tailed Shearwater

Short-tailed Shearwaters (*Puffinus tenuirostris*) are the most abundant seabird in Australia. They are migratory seabirds, spending the winter months in the northern Pacific, before returning to breeding colonies in southern Australia at the end of September. Colonies can be found in coastal areas from New South Wales through to Western Australia, with the majority found on Bass Strait Islands and around Tasmania (Skira *et al*., 1996).

A small Short-tailed Shearwater breeding colony is found at Fort Direction, located on the southern tip of South Arm. This breeding rookery is subject to a small cultural harvest. NRE Tas undertakes annual monitoring of the rookery to assess Short-tailed Shearwater breeding productivity and inform decision-making in relation to the sustainable level of annual harvest of pre-fledging chicks. Surveys to determine percentage burrow occupancy and general condition of the birds are conducted in December for incubating adults and in March for pre-fledging chicks. Monitoring commenced in 2001, and results show a relatively high inter-annual variability in Shearwater burrow occupancy at Fort Direction.

Short-tailed Shearwater (*Puffinus tenuirostris*) flying. Image: Ramit Singal.

Little Penguins

Little Penguins (*Eudyptula minor*) are the smallest penguin species; fully grown adults can reach a height of 40 cm and weigh 1 kg. Tasmania is home to most of Australia's Little Penguins (Dann *et al*., 1996). They can be found in coastal areas around Tasmania, with 95% of Tasmania's population thought to breed on offshore islands. Little Penguins are not listed as threatened under state or national legislation. However, they are recognised as a species of conservation concern and listed as Protected Wildlife in the Wildlife (General) Regulations 2010 under the *Nature Conservation Act 2002*. However, burrowing seabird habitat used by Little Penguins is listed as a threatened vegetation community, Seabird Rookery Complex.

Little Penguins spend most of their time foraging in the marine environment. When they come ashore, they are active on land at night, crossing the shoreline at dusk and dawn (i.e. nocturnal). They are burrowing seabirds that will dig burrows under vegetation or into dirt and sand, or nest inside rock hollows, caves and under man-made structures that create a suitably sized space (e.g. culverts, raised houses, decks). Little Penguins exhibit high site fidelity (Agnew *et al*., 2016) and will return annually to their natal area for breeding activities (Reilly and Cullen, 1982). They reach sexual maturity at approximately two to three years of age and have biparental care of offspring. Mating pairs prepare a burrow for nesting, then females will lay one to two eggs and the pair alternates incubation and chick-rearing duties (Chiaradia and Kerry, 1999). Young chicks have at least one parent present to keep them warm and provide regular feeds, older chicks are left ashore while both parents forage during the day, returning at night to feed the chicks. In a year of good food availability, Little Penguin chicks are ready to fledge the burrows at approximately eight weeks of age.

Little Penguins are present within colonies almost year-round for various activities including burrow preparation, breeding activities, chick rearing and moulting. Onset and peaks of Little Penguin breeding are spatially and temporally variable. Within a single year, breeding phenology varies within and between colonies. Typically, the most active breeding period is spring to summer, with winter breeding widespread with fewer numbers (Salton *et al*., 2015; Allnutt, 2021). The exact timing of breeding is dependent upon many factors, most importantly prey availability (Berlincourt and Arnould, 2015).

Little Penguins have a relatively restricted foraging range (Hoskins *et al*., 2008), which is further constrained during breeding (Chiaradia and Kerry, 1999). During one summer breeding season, Little Penguins within the Derwent Estuary spent more time at sea foraging compared to Little Penguins at colonies in two more exposed areas of Storm Bay (Phillips *et al*., 2019). The Derwent Estuary penguins focused their foraging efforts around the northern end of Storm Bay, at the mouth of the Derwent Estuary, and spent more time in a highly energy-intensive behavioural state that is consistent with foraging. This suggests the Derwent Estuary penguins had to expend more energy per unit of time compared to penguins from other colonies to acquire adequate amounts of food, potentially due to less abundant prey or increased competition (Angel *et al*., 2016).

Little Penguins are vulnerable to threats both in marine and terrestrial environments. Within the marine environment, these risks include entanglement in fishing gear, oil spills, plastic ingestion, and disruptions to their food availability due to overfishing and rapid climate change impacts. On land, risks to Little Penguins include habitat loss and degradation, human disturbance, predation, pollution, and climate change impacts like heat stress and altered weather patterns. Some of these risks have a high likelihood in urban environments, like habitat loss and degradation, predation by domestic and feral animals, human disturbance, and pollution.

Recent work showed that small penguin colonies in Tasmania are at risk of colony collapse within 10–15 years if there are ongoing low levels of predator attacks (Blamey *et al*., 2024). Increased human disturbance where there is access to beaches through Little Penguin colonies, and particularly human activities occurring at night can affect the number of returning birds and their behaviours (Costello and Colombelli-Négrel, 2023). Tourism operations allowing access to Little Penguin colonies, if not managed well, can impact reproductive success of penguins (Seddon and Ellenberg, 2007; Agnew and Houston, 2020).

In Tasmania, and likely elsewhere, Little Penguin colonies surrounded by highly urban areas are more exposed to a group of human-manufactured chemicals called per- and polyfluoroalkyl substances (PFAS). These PFAS chemicals were found to be elevated in 'urban' penguins, and higher levels of PFAS in individual penguins was related to indicators of poor health (Wells *et al*., 2024).

On land, rapid climate change and weather impacts that threaten Little Penguin colonies include increased temperatures, reduced rainfall, and increased frequency of extreme rainfall events. These threats are expected to negatively impact adult Little Penguin survival (Ganendran *et al*., 2016). Penguin burrows located close to the intertidal zone are at risk of regular inundation from sea level rise and storm surge events (McFarlane, 2023). Increased spread of infectious diseases to wild bird populations, such as the current global outbreak of highly pathogenic avian influenza (H5N1), could cause devastating losses to Little Penguin populations (Ramey *et al*., 2022).

Derwent Estuary Little Penguins

There are several areas in the lower reaches of the Derwent Estuary with consistent Little Penguin activity. These aggregations range in size from several active burrows to larger colony areas with over 50 active burrows. Little Penguins were historically far more abundant in the estuary, with some evidence suggesting a decline in the population over the last 50 years (Stevenson and Woehler, 2007).

The monitoring and management of Little Penguins within the Derwent Estuary is a collaborative effort between local councils, scientists, NRE Tas Marine Conservation Program (MCP), the DEP and volunteers. Since 2004, the DEP has overseen the Penguin Advisory Group (PAG). The PAG facilitates regular monitoring of Derwent Estuary Little Penguin colonies, provides advice and supports on-ground works in colonies, installing artificial burrows, organising educational activities and documents, providing expert scientific advice and engaging in other data/information sharing as needed.

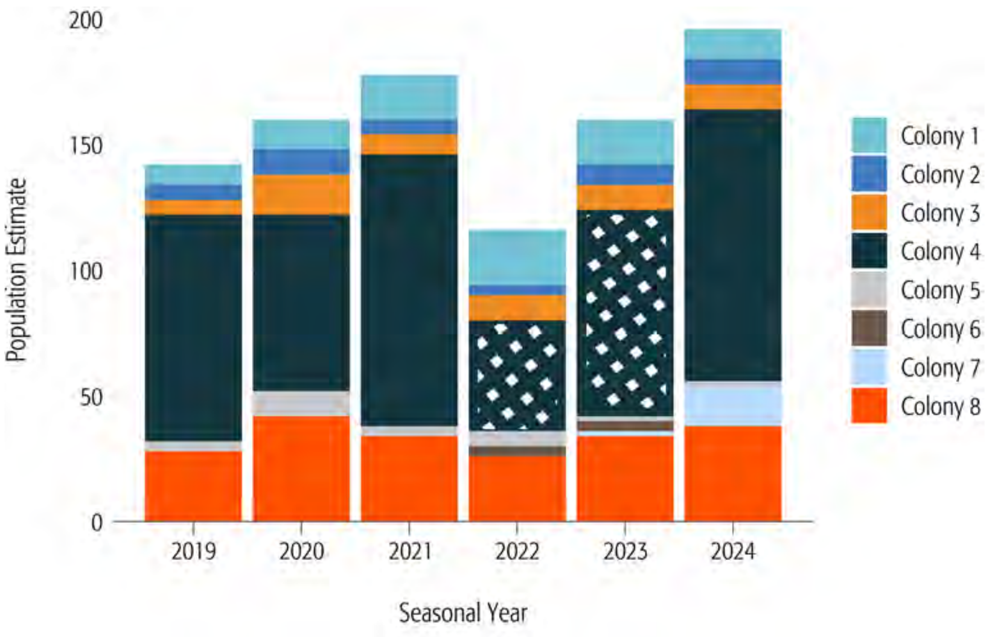
The PAG manages monthly monitoring at seven Little Penguin colonies within the Derwent Estuary and annually monitors several historic and emerging colonies. This monitoring is undertaken by volunteers during daylight hours to limit disturbance to Little Penguins. Information is collected on burrow occupancy (number of adults, eggs, and/or chicks present), nesting activity signs and moulting signs. Population estimates of Little Penguins within the Derwent Estuary are calculated based on the number of 'occupied' burrows observed during a breeding season, multiplied by two (assuming two penguins use each occupied burrow). To be classified as 'occupied' there needs to have been an observation of 'adult', 'egg', and/or 'chick'.

The population estimate of breeding Little Penguins in the Derwent Estuary based on colony surveys during the 2024 seasonal year is 196 individuals (Table 10.5). This is a 20% increase from 2023 population estimates. This increase is due to increased nesting activity observed at a newly discovered colony and increased survey effort at the largest Derwent Estuary colony. Survey effort has been variable between and within years, and between colonies. This is due to several reasons including volunteer availability, weather conditions, permits and colony access, which adds to the uncertainty in population estimates. The largest colony in the Derwent Estuary was only monitored twice in 2022 and three times in 2023 (out of a total possible 12 times), which has impacted population estimates for these years (Figure 10.16).

Table 10.5 Population estimates of breeding Little Penguins (*Eudyptula minor*) in the Derwent Estuary based on colony surveys, from 2019 to 2024. Seasonal year encompasses winter and summer breeding seasons, April to March of the following year. Survey effort for 2022 and 2023 seasonal year was greatly reduced at the largest colony in the Derwent, which has likely impacted population estimates here.

Season Year	Population Estimate
2024	196
2023	160
2022	116
2021	178
2020	160
2019	142

Figure 10.16 Population estimates of breeding Little Penguins (*Eudyptula minor*) in the Derwent Estuary based on colony surveys, from seasonal years 2019 to 2024. Seasonal year encompasses winter and summer breeding seasons, April to March of the following year. Survey effort for 2022 and 2023 seasonal year was greatly reduced at the largest colony in the Derwent (navy blue diagonal fill), which has likely impacted population estimates here.



During 2024 and 2025, important activities were completed to improve efficiency in routine Little Penguin monitoring and increase habitat for Little Penguins in the Derwent Estuary. These were as follows:

- Efforts to increase the number of volunteer Little Penguin monitors who participate in the monitoring program. Having a larger volunteer base ensures redundancy if volunteers are sick or unavailable to assist with monitoring. Throughout the year, nine new monitors were trained onsite to assist with monitoring activities. Annual volunteer training events are held to ensure consistency in monitoring methodologies, data collection, and safety when conducting monitoring for this program.
- Improvements were made to the *Little Penguin Monitoring App*, where data is collected during monitoring activities. This included hiding 'removed', 'destroyed' and 'damaged/requires work' burrows from the mobile application, flag burrows as inspected (with cross) for one week after an inspection was conducted and sorting inspection records by date.
- Replacement tags were installed on artificial burrows with missing tags. This ensures that data is collected accurately for each burrow and limits confusion, especially in areas with dense burrow numbers. Over 40 tags were installed across colonies in the Derwent Estuary in January and February 2025.
- Planning for revegetation of native species in colonies where undergrowth vegetation is sparse.
- Engaged new neighbours of Little Penguin colonies in the Derwent Estuary and distributed *Little Penguin Neighbour Newsletter*, with information on colony numbers and activities to neighbour mailing list.
- Successful in City of Hobart (CoH) Medium Creative Grant application to commission five new ceramic burrows for CoH Little Penguin colonies.

Threat mitigation

The PAG facilitates threat mitigation activities for Derwent Estuary Little Penguins, these are as follows:

- Regular camera trapping conducted by local councils to assess the need for more targeted pest management in Little Penguin colonies.
- Application to amend off-lead area in CoH Dog Management Strategy in an emerging colony area to limit the disturbance from dogs to Little Penguins.
- Education of the public and neighbours of Little Penguins to increase awareness of the impacts of human disturbance to Little Penguin behaviours and breeding success.

- *Avian Influenza (H5N1) Information Package* circulated to PAG members and volunteers to increase awareness of the signs of this disease in birds, when and where to report observations of disease, and best biosecurity practices to follow when interacting with Little Penguins for monitoring activities.
- Ambient temperature investigation in urban colonies during summer to assess temperature ranges experienced by exposed burrows. Findings from this investigation recommended increasing undergrowth vegetation in sparsely vegetated areas.
- Increasing suitable habitat in colonies through native planting efforts, combined with installation of artificial burrows in colonies with limited natural burrow habitat. This helps to limit burrow exposure to solar radiation and high temperatures. New artificial burrows are designed to ensure optimal temperature regulation inside the burrows for nesting Little Penguins. All new burrows are stress tested for temperature and humidity to ensure they are suitable for Little Penguin nesting before being installed in colonies.
- Routine colony monitoring to collect reliable, long-term data to assess Little Penguin population trends over time and inform colony management. This monitoring includes visiting historic and emerging colonies to accurately inform estuary wide population estimates.

Derwent Estuary nest site exposure to storm surge inundation study

McFarlane (2023) investigated the risk to Little Penguin nest sites within the Derwent Estuary to climate-induced sea level rise and storm surge over the next 80 years. Within the Derwent Estuary many nests are located in low-lying coastal areas, where there is little room for upland movement of colonies, due to upland topography, and urban coastal development/infrastructure.

Global mean sea level is projected to increase by between 0.6–1 m between 2020 and 2100 (Pörtner *et al*., 2022), which is significant enough to affect many coastal-dwelling species. Climate change will also increase the intensity and frequency of extreme coastal storm events (Bernier *et al*., 2024). The combination of extreme weather events, such as high winds, excessive rainfall, and large waves, with periods of abnormal high tide action are termed ‘storm surges’ (Bode and Hardy, 1997; McInnes *et al*., 2011). Storm surges are predicted to increase in prevalence and intensity by the year 2100 with the predicted rise in mean sea level, as well as increased risk of extreme weather events (Takayabu *et al*., 2015).

The aim of the study was to assess the risk of inundation from sea level rise and storm surge action to known Little Penguin nesting sites within the Derwent Estuary.

Results

The results showed that 18% of Little Penguin nest sites within the Derwent Estuary are at risk from inundation from storm surge. Combined with sea level rise predictions and increases in storm surge, the percentage of at-risk nest sites increases to over 27% in 2100 (Table 10.6). By 2100, three entire colonies, constituting seven of the 81 sites at risk were identified as having 100% of sites at risk of inundation.

Table 10.6 Number and percentage of at-risk nest sites within the Derwent Estuary for time points selected to assess impacts of sea level rise to 2100.

Year	Number of at risk sites	Total number of sites	Percentage of at risk sites
2022	54	294	18.4%
2050	64	294	21.8%
2100	81	294	27.6%

This analysis is based on the worst-case scenario of sea level rise under the IPCC sixth assessment AR6 report (2022) (SSP5-8.5 scenario), highest historical tide and one-in-one-hundred year extreme storm surge event. The likelihood of this scenario over less extreme alternatives is unknown. The analysis does provide evidence for a precautionary approach.

These studies provide valuable information for the PAG, local councils, and private land holders who actively manage Little Penguins nesting burrows. The research helps identify burrows that are at risk of inundation from sea level rise and storm surge. To follow on from this work, suitable upland areas can be mapped across colonies to help guide management of colony retreat with sea level rise (e.g. protection of vegetation and ensuring suitable habitat is available for penguins to occupy).

Art burrows for Little Penguins

In May 2025, the DEP on behalf of the PAG was successful in receiving funding from the CoH Medium Creative Grant to create five new ceramic Little Penguin burrows for CoH colonies. This innovative artistic project involves a group of four individuals in a unique collaboration – a writer (Katherine Johnson), a photographer (Peter Whyte), an ecologist (Ellie Green), and a ceramic artist (Jane Bamford). This collaborative project will focus on Little Penguin conservation and habitat creation in the Derwent Estuary. This project will have long-term benefits for the community, by conserving biodiversity within urban areas and fostering sense of place connection between the community and the Derwent Estuary. Little Penguin colonies within the CoH are highly susceptible to threats, particularly urbanisation, invasive weeds, reduced habitat availability, climate change (increased temperatures, sea level rise, etc.) and predation from domestic pets. Careful efforts to support their nesting activities will greatly benefit these colonies.

The design brief for these five new ceramic burrows has been informed by an initial ceramic Little Penguin burrow project undertaken in the Derwent Estuary in 2021. The project funded by Rural Arts in partnership with Kingborough Council involved a group of eight ceramic artists (six established and two emerging) to create ceramic Little Penguin burrows for the Derwent Estuary colonies. These ceramic burrows were installed in colonies for Little Penguin use in 2023. Design specifications were investigated alongside temperature and humidity data collected during stress testing of burrows and data collected on burrow occupancy by Little Penguins to refine ceramic burrow specification requirements for the five new burrows.

This 12-month project aims to deliver five new ceramic burrows created by ceramic artist Jane Bamford, to be installed in CoH colonies in autumn 2026. Jane's practice, and the new burrows will be photographed by Peter Whyte, to be displayed as prints at two public talks to discuss the project and this unique collaboration. Desktop and onsite analysis will be conducted to select the optimal location to install the burrows, and burrows will be stress tested to ensure they are safe for Little Penguin use in the environment. A feature length article will be written about the project by writer Katherine Johnson.

Figure 10.17 Jane Bamford creating ceramic Little Penguin (*Eudyptula minor*) burrows in studio for City of Hobart colonies. Image: Peter Whyte.

10.5.5 High pathogenicity Avian Influenza

Avian influenza is a contagious, viral respiratory disease that can infect both domestic and wild bird populations. The strains classified as highly pathogenic avian influenza (HPAI) are extremely contagious and can cause severe clinical symptoms. In 2020, a new strain of HPAI, H5N1 clade H5 2.3.4.4b emerged. This strain has proven to be extremely contagious and deadly, affecting millions of birds (both domestic and wild populations) around the world. HPAI H5N1 has so far not been detected in Tasmania, or Australia, however there is concern that this virus could be introduced through migratory shorebirds and seabirds entering Australia. The clinical signs of HPAI in wild birds are largely neurological, respiratory or gastrointestinal, including incoordination, diarrhea, respiratory problems and swelling.

HPAI poses a significant risk to threatened bird populations, colonial nesting seabirds where the disease could spread rapidly through populations, and species that are genetically susceptible to infection, such as Black Swans (*Cygnus atratus*) (Karawita *et al*., 2023). There is also a risk of infection to other species that predate on birds including eagles and seals, and scavengers of dead birds such as Endangered Tasmanian Devil (*Sarcophilus harrisii*).

HPAI is a nationally notifiable animal disease, meaning that anyone who suspects an animal might be infected with HPAI has a legal responsibility to report it as soon as possible. If a sick bird is observed to have any of the clinical signs of HPAI, it should be reported. If five or more dead birds are observed, please make a report. Avoid any physical contact with the sick or dead bird. Record observations including location, species, number of birds, signs of sickness, collect photos and videos. Report observations to the 24/7 Emergency Animal Disease hotline (1800 675 888) as soon as possible.

10.6 Marine mammals

Marine mammals can be regularly observed within the Derwent Estuary. Two species of dolphin have been recorded within the Derwent Estuary; these are the Bottlenose Dolphin (*Tursiops truncatus*) and Common Dolphin (*Delphinus delphis*). Observations of dolphin pods are becoming more common in the upper estuary above Bridgewater, and there have even been anecdotal sightings in New Norfolk.

Several species of seal have been recorded in the Derwent Estuary, the most common are the Australian Fur Seal (*Arctocephalus pusillus doriferus*) and Long-nosed Fur Seal (*Arctocephalus forsteri*), which is listed as rare under the TSP Act. Seals are commonly observed in coastal areas around Tasmania outside of the breeding season, congregating at various rocky outcrops known as haul-outs.

There have been more regular observations of seals hauling out around Tinderbox and Blackmans Bay area in the Derwent Estuary in recent years. In July 2025 50 seals (mostly comprising Long-nosed Fur Seal) were observed in a haul-out near Tinderbox, the most that has ever been observed at this site (pers comms M. Salton, MCP, 2025).

10.6.1 Whales

Resident whales may be seen in southeast Tasmania year-round, however the most commonly seen whale species belong to the group of migratory whales that are observed in south-east Tasmanian waters between May and December each year. Whales migrate north along coastal areas of Tasmania to warmer waters of northern Australia for breeding and calving, then return south with calves to the nutrient-rich waters of Antarctica to feed during the summer months. Humpback Whales (*Megaptera novaeangliae*) migrate as far as northern Queensland to calve in the protected waters of the inner Great Barrier Reef (Smith *et al*., 2012). Southern Right Whales (*Eubalaena australis*) that are observed within Tasmanian waters belong to the south-eastern sub population and will migrate to inshore waters off the coast of Tasmania, Victoria and occasionally NSW (Carroll *et al*., 2015; Stamation *et al*., 2020). Humpback Whales have recently been delisted as vulnerable migratory species, and Southern Right Whales are currently listed as endangered migratory species under the EPBC Act. These migratory whales can occasionally be observed taking refuge within the Derwent Estuary, sometimes for several weeks during their long migratory journeys.

In the early to mid-nineteenth century, the population of whales found within the Derwent Estuary was enough to sustain a whaling industry. The height of shore-based whaling activities by early colonialists was between 1820 and 1855 and had a strong focus on Southern Right Whales. By the mid-nineteenth century, the numbers of whales observed in inshore waters of Tasmania had become greatly reduced (Evans, 2006), and inshore whaling activities began to greatly decline. Reduction in the demand for whale oil with the advent of petroleum and kerosene for lighting, as well as competition from other profitable industries also contributed to the collapse of the Tasmanian whaling industry.

While early whaling activities significantly reduced coastal whale populations in south-east Australia, technological advances and the advent of factory ships led to an increase in offshore whaling. Indeed, extensive offshore illegal whaling, which occurred south of Australia in the summers of 1959 to 1961 has been attributed for causing the near extinction of migratory whale populations (Harrison and Woinarski, 2018). After international concern for whale populations was raised in 1986, the International Whaling Commission introduced an international moratorium on commercial whaling, setting catch limits to zero, which remains in place today (Rocha *et al*., 2014).

Since management measures were put in place to eliminate the threat of whaling to Humpback Whale populations, the species has been slowly recovering over the last five decades. It is believed that the population of eastern Australian Humpback Whales has now

Humpback Whale (*Megaptera novaeangliae*) in southeast Tasmanian waters. Image Drew Griffiths.

essentially recovered, with current population estimates at approximately 25,000 whales (Noad *et al*., 2019).

Southern Right Whales form two genetically distinct populations in south-west and south-east Australia (Carroll *et al*., 2011). The south-eastern population is showing slower rates of recovery compared to the south-west subpopulation, with the eastern population estimated to be 268 individuals as of 2017 (Stamation *et al*., 2020). This slow recovery is thought to be due to the species strong site fidelity (Harcourt *et al*., 2019), migratory routes (Carroll *et al*., 2015) and long calving cycle (Watson *et al*., 2021).

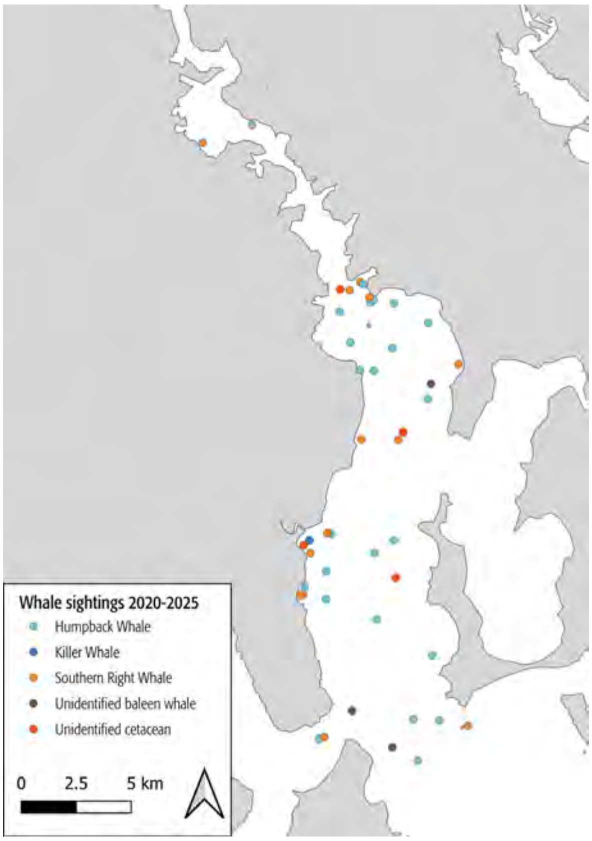
The strong recovery of Humpback Whales is reflected in the number and composition of whale sightings in the Derwent Estuary. Since 2018, there have been increased sightings of Humpback Whales within the Derwent, compared to pre-2018 (Table 10.7).

Whales are most commonly observed in embayment's south of the Tasman Bridge (Figure 10.17) and are often composed of smaller sub-adult whales. However, there have been several recent observations of whales north of the Bowen Bridge. In June 2025, a group of Humpback Whales was observed around Montrose Bay in the Derwent Estuary.

Over the last 10 years, there has been a reduction in the number of Southern Right Whales observed in south-east Tasmania, as well as coastal Victoria. The *Conservation Management Plan for the Southern Right Whale 2011–2021* provides a framework for collaborative efforts to help protect this species, and a *National Recovery Plan for the Southern Right Whale (Eubalaena australis)* was published in 2024. There is need for better monitoring of species distribution, abundance and habitat use, as well as data management and continuity between jurisdictions and stakeholders. This will help to better manage identified threats to Southern Right Whale populations across their range.

Observations of Killer Whale (*Orcinus orca*) and Long-finned Pilot Whale (*Globicephala melas*) are rare in the Derwent Estuary (Table 10.7). This is primarily due to these species frequenting the estuary in search of food, and being highly transitory within the estuary, compared to other migratory whale species which are taking refuge in the Derwent and may be present for prolonged periods of time.

Figure 10.18 Location of whale observations within the Derwent Estuary reported to NRE Tas Marine Conservation Program from 2020 to 2025. Data supplied by NRE Tas Marine Conservation Program.



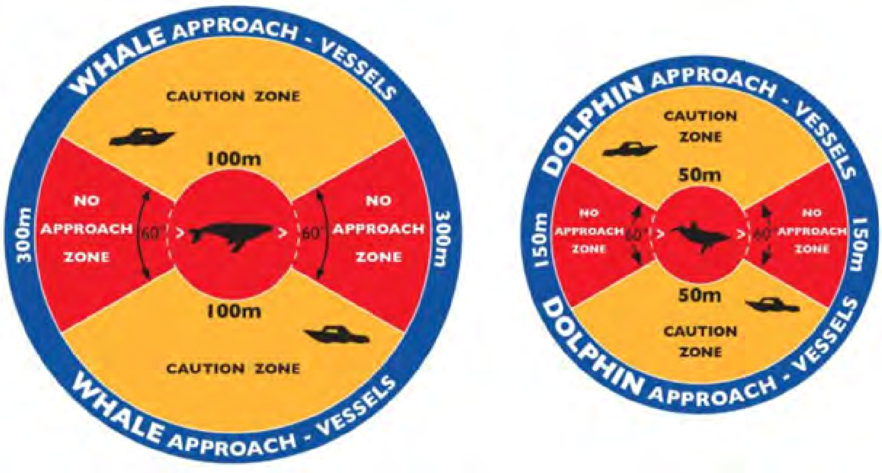
Although Humpback Whale populations are recovering well, this does not mean they are safe from threats that may cause population decline. Threats include expanding offshore development, pollution, entanglement and climate change impacting food sources. One major threat to whale populations is the increased incidence of vessel strikes with increasing maritime traffic in coastal southeast Australia (Mayaud *et al*., 2022). While out boating on the Derwent please keep a watchful eye out for whales.

In Tasmania, NRE Tas guidelines for approaching whales and dolphins state that a vessel should not approach any closer than 100 m to a whale (Figure 10.18). If the vessel is under steam, then 300 m distance should be maintained and avoid approaching the animal from in front or behind. For dolphins, the approach distances are to maintain a minimum distance of 50 m, and 150 m if the vessel is under stream. Please report any whale sightings to NRE Tas WHALES hotline (0427 WHALES or 0427 942 537).

Table 10.7 Observations of whales reported to NRE Tas Marine Conservation Program in the Derwent Estuary between 2003 and 2024. Records before 2016 did not necessarily distinguish between unique individuals, therefore multiple sightings of the same individual may have been made over a period of several months. Data supplied by NRE Tas Marine Conservation Program.

Year	Southern Right Whale *(Eubalaena australis)*	Humpback Whale *(Megaptera novaeangliae)*	Killer Whale *(Orcinus orca)*	Long-finned Pilot Whale *(Globicephala melas)*	Unidentified cetacean	Unidentified baleen whale	Total
2003	5	3	2	-	-	-	10
2004	3	1	-	-	-	-	4
2005	14	1	-	-	-	-	15
2006	4	1	-	-	-	-	5
2007	5	-	-	-	-	-	5
2008	3	3	1	-	-	-	7
2009	1	1	-	-	-	-	2
2010	9	-	6	-	-	-	15
2011	3	1	-	-	-	-	4
2012	7	-	-	-	-	-	7
2013	9	-	2	-	-	-	11
2014	3	3	-	-	-	-	6
2015	3	-	6	-	-	-	9
2016	-	2	-	-	-	1	3
2017	1	1	1	-	1	-	4
2018	3	2	-	5	-	-	10
2019	-	7	-	-	-	-	7
2020	1	-	-	-	2	-	3
2021	10	11	-	-	-	-	21
2022	-	-	-	-	-	3	3
2023	7	8	-	-	1	-	16
2024	5	10	2	-	2	1	20
Total	91	52	18	5	6	5	177

Figure 10.19 NRE Tas guidelines for cetacean vessel approach.



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Acronyms

Acronyms

AAD	Australian Antarctic Division
ADWF	Average Dry Weather Flow
AHD	Above Height Datum
AMR	Antimicrobial Resistance
ANZECC	Australian New Zealand Environmental and Conservation Council
ANZG	Australian and New Zealand Guidelines for Fresh and Marine Water Quality
ASH	Artificial Spawning Habitat
AST	Analytical Services Tasmania
AWQ	Ambient Water Quality
BC	Brighton Council
BOD	Biological Oxygen Demand
BoM	Bureau of Meteorology
CAP	Derwent Estuary Conservation Action Plan
CCC	Clarence City Council
CES	Combined Effluent Stream
COD	Chemical Oxygen Demand
CoH	City of Hobart
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CU	Colour Units
DCP	Derwent Catchment Project
DEP	Derwent Estuary Program
DGV	Default Guideline Values
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
DoH	Department of Health (Tasmanian Government)
DOM	Dissolved Organic Matter
DVC	Derwent Valley Council
EAAF	East Asian-Australasian Flyway
EAC	East Australian Current

eDNA	Environmental DNA
EMP	Electrostatic Mist Precipitator
EMPCA	*Environmental Management and Pollution Control Act 1994*
EPA	Environment Protection Authority
EPBC	*Act Environmental Protection and Biodiversity Conservation Act 1999*
EPN	Environment Protection Notice
E&SC	Erosion and Sediment Control
ETP	Effluent Treatment Plant
FIA	Flow Injection Analyser
FIB	Faecal Indicator Bacterium
FOGO	Food Organics and Garden Organics
FRDC	Fisheries Research and Development Corporation
FSANZ	Food Standards Australia New Zealand
GCC	Glenorchy City Council
GELs	Generally Expected Levels
GPT	Gross Pollutant Trap
GUVC	General Underwater Visual Census
HPAI	Highly Pathogenic Avian Influenza
IECA	International Erosion Control Association
IFS	Inland Fisheries Service
IMAS	Institute for Marine and Antarctic Studies
IPCC	International Panel on Climate Change
KC	Kingborough Council
LISTmap	Land Information Services Tasmania (theLIST website: www.thelist.tas.gov.au)
LoR	Limit of Reporting
MAST	Marine and Safety Authority of Tasmania
MCP	Marine Conservation Program (Tasmanian Government)
MHW	Marine Heatwave

MLs	Maximum Levels
MONA	Museum of Old and New Art
MPB	Microphytobenthos
MPN	Most Probable Number
NATA	National Association of Testing Authorities Australia
NCA	Natural Capital Accounting
NESP	National Environmental Science Program
NGT	Nature Glenelg Trust
NHRMC	National Health and Medical Research Council
NHRT	National Handfish Recovery Team
NN	New Norfolk
NOx	Nitrate and nitrite
NPI	National Pollutant Inventory
NPOC	Non-P urgeable Organic Carbon
NRE	Tas Department of Natural Resources and Environment Tasmania
NTB	New Town Bay
NTU	Nephelometric Turbidity Units
ODV	Ocean Data View
PAG	Penguin Advisory Group
PFAS	Per- and Polyfluoroalkyl Substances
POMS	Pacific Oyster Mortality Syndrome
PWB	Prince of Wales Bay
PWS	Tasmania Parks and Wildlife Service
QA	Quality Assurance
QC	Quality Control
RBN	Ralphs Bay North
ROV	Remote Operated Vehicle
RSET	Rod Surface Elevation Table
RWQ	Recreational Water Quality
SEEA-EA	System for Environmental-Economic Accounting- Ecosystem Accounting
SETP	Secondary Effluent Treatment Plant
SFA	Segmented Flow Analyser
STP	Sewage Treatment Plant
SWAPP	Strategic Weed Assessment and Prioritisation Plan
SWC	Summer Wader Count
SWTF	Stormwater Taskforce
TAC	Tasmanian Aboriginal Centre
TasPorts	Tasmanian Ports Corporation
TasVeg	Tasmanian Vegetation Map
TEER	Tamar Estuary and Esk Rivers Program
TN	Total Nitrogen
TP	Total Phosphorus
TSP	*Act Threatened Species Protection Act 1995*
TSS	Total Suspended Solids
UTAS	University of Tasmania
URCP	Urban Rivers and Catchment Program
WGC	Winter Gull Count
WHO	World Health Organization

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