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Building climate resilience for fish and fisheries in the CLLMM region

September 2026



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Executive Summary

The Coorong, Lower Lakes and Murray Mouth (CLLMM) region is a nationally significant aquatic ecosystem supporting internationally recognised wetlands, diverse fish communities, and important cultural, commercial and recreational values. Long-term catchment development has contributed to declining ecosystem condition in this region, requiring ongoing management interventions to maintain ecosystem health and improve resilience.

Climate change is expected to place increasing pressure on the region primarily through declining freshwater inflow, rising temperatures and accelerated sea-level rise. These interacting pressures are anticipated to alter hydrology, salinity, connectivity and habitat availability, increasing risks to aquatic ecosystems, fish populations and the fisheries that depend on them. A conceptual model developed by Rees et al. (2022) summarises the environmental changes expected under climate change. The model broadly describes a transition between two states. Initially, a 'Hydrological Minimum' state occurs as freshwater inflow progressively declines, until increasing seawater exchange begins to replace the declining freshwater contribution to aquatic environments. This marks a transition towards a 'High Ocean Exchange' state, in which rising sea levels exert increasing influence on the hydrodynamics of CLLMM aquatic ecosystems. Considerable uncertainty remains around when, how and to what degree these environmental changes will affect aquatic ecosystems.

To improve understanding of these climate-induced environmental changes for fish and fisheries, this study had three objectives:

- explore the potential impacts of climate change on fish habitat and fish populations in the CLLMM region;
- improve understanding of climate change risks for the Lakes and Coorong Fishery; and
- consult with the fishing industry to improve understanding of climate change adaptation in the CLLMM region.

These objectives were addressed through three project phases. Phase 1 reviewed the ecology of the CLLMM and developed ecological profiles for representative fish species. Phase 2 applied a trait-based climate change vulnerability assessment (CCVA) to 11 fish species, combining species sensitivity with projected exposure to temperature, freshwater flow, salinity and connectivity at the 2050- and 2090-time horizons. Phase 3 involved consultation with the Lakes and Coorong Fishery to communicate projected climate impacts and obtain industry perspectives on climate-related risks, adaptation opportunities and future management.

The literature review indicated that climate change is likely to influence fish populations through changes in habitat conditions, ecological processes and life-history requirements, in addition to direct physiological effects. Freshwater flow emerged as an important environmental driver in the CLLMM, influencing salinity,

connectivity and habitat availability, as well as ecological processes including migration, reproduction, recruitment and ecosystem productivity. Many species with complex life cycles, including diadromous, marine-estuarine opportunist and potamodromous freshwater species, showed dependence on maintained connectivity and, in some cases, aquatic environments outside the CLLMM.

Available ecological information differed considerably among species and life stages. Information was generally greater for commercially valued species and juvenile and adult life stages, while knowledge of small-bodied species and early life stages was more limited. Marine-estuarine opportunist and estuarine species generally exhibited broad salinity tolerances, while freshwater species displayed higher temperature tolerances. Eggs and larvae generally had narrower environmental tolerance ranges than juveniles and adults. Important knowledge gaps remain around fish thermal tolerances and the effects of short-term extreme events, as well as the broader indirect effects of warming on productivity, food webs and habitat condition.

Climate projections indicated that environmental change is expected to intensify throughout the century. Reduced freshwater inflows, rising sea levels and increasing temperatures are projected to alter hydrology, salinity, connectivity and habitat availability across the Lower Lakes and Coorong. By 2090, runoff from the southern Murray–Darling Basin was projected to decline by up to 41%, with River Murray freshwater flow to the CLLMM declining by up to 67% relative to the 1976–2005 baseline. Median sea level was projected to increase by up to 0.51 m relative to the 1995–2014 baseline, while annual average daily air temperature was projected to increase by approximately 2.04 °C.

Hydrodynamic modelling was integral to translating these climate projections into environmental conditions relevant to fish. In the Coorong, reduced freshwater inflow substantially altered salinity regimes, primarily through the contraction of freshwater to low-brackish conditions and the expansion of more saline environments. In the Lower Lakes, salinity change became more pronounced towards 2090 as declining runoff combined with increasing Coorong water levels and sea-level influence. Under Dry and Extreme Dry conditions, reduced freshwater inflow and barrage closure decreased the head difference between the Lower Lakes and Coorong, allowing intermittent ingress of saline water into parts of the Lower Lakes. Salinisation of freshwater habitats was therefore projected to emerge progressively, with the frequency, duration and spatial extent of saline conditions increasing through time. More extreme drying could bring these changes forward, resulting in earlier and potentially more extensive salinisation, including marine-level salinities within parts of the Lower Lakes. Modelled water depths generally remained sufficient to maintain within-waterbody connectivity; however, increasing Coorong water levels reduced water exchange across the barrages and consequently may affect existing pathways for fish movement between the Coorong and Lower Lakes. Climate change exposure across the CLLMM region ranged from low to moderate and increased from 2050 to 2090, with marginally higher exposure in the Coorong than the Lower Lakes. Exposure was also strongly influenced by freshwater inflow, with differences between freshwater inflow levels comparable to those between time horizons. This suggests that the

severity of drying may be as influential as the progression of climate change through time. Consequently, periods of extreme drying occurring earlier in the century may provide insight into environmental exposure conditions that are likely to become more common under future climate conditions. Freshwater flow was the dominant driver of exposure in the Lower Lakes, while both freshwater inflow and salinity were the primary contributors in the Coorong. In contrast, temperature contributed little to overall exposure. Overall, these findings indicate that climate change impacts on aquatic environments within the CLLMM are likely to be driven principally by hydrological change, with the magnitude of future drying strongly influencing both the timing and severity of habitat alternation and associated risks to fish populations.

Fish vulnerability generally increased towards 2090 and under lower freshwater inflow. Several of the more vulnerable species shared periodic-type life-history characteristics, particularly black bream, greenback flounder and mulloway in the Coorong and golden perch in the Lower Lakes. For these species, vulnerability reflected a combination of life-history characteristics and dependence on environmental conditions that are strongly influenced by freshwater flow for reproduction, recruitment and movement. In contrast, opportunistic species generally exhibited lower vulnerability, due to broader environmental tolerances and greater ecological flexibility. Importantly, similar vulnerability or sensitivity scores did not necessarily indicate common mechanisms of risk, with species vulnerability arising from different combinations of life-history traits and environmental dependencies. These vulnerability patterns highlight potential climate-related risks to the Lakes and Coorong Fishery, with several commercially harvested species among the more vulnerable fishes assessed. Black bream, greenback flounder and mulloway in the Coorong, and golden perch in the Lower Lakes, displayed elevated vulnerability associated with combinations of life-history characteristics and dependence on environmental conditions influenced by freshwater flow. Projected changes in freshwater inflow, salinity and connectivity may affect processes important for reproduction, recruitment, movement and habitat use, with potential consequences for the distribution, abundance and availability of fishery species. While these findings do not predict future catches or fishery production, they identify species, life-history processes and environmental pathways through which climate change may pose greater risks to fishery sustainability. Projected environmental change may not, however, affect all species negatively. Increasing salinity in the Lower Lakes may reduce habitat suitability for freshwater species while creating additional habitat opportunities for some estuarine and marine-estuarine species. The extent to which these opportunities result in changes in species distribution, abundance or population establishment will depend on connectivity, movement opportunities and the availability of environmental conditions suitable for reproduction and recruitment. Consequently, future change may involve a redistribution of suitable habitat and fish assemblages across the CLLMM, rather than a uniform decline in habitat suitability.

The strong influence of freshwater inflow on exposure and vulnerability also highlights the importance of water management in influencing how climate impacts are expressed within the CLLMM. Freshwater inflows contribute to the maintenance of

salinity regimes, habitat conditions and connectivity, while Lower Lakes water-level management, barrage and fishway operations currently influence connectivity and environmental conditions across the system. These management mechanisms may therefore remain important for supporting fish populations as climate pressures increase, although their future effectiveness will depend on water availability, sea-level rise and the magnitude of environmental change.

The assessment demonstrated the value of integrating species ecological traits with projected environmental change to identify relative differences in vulnerability. However, it also highlighted methodological challenges associated with comparing exposure across multiple climate drivers. Temperature, freshwater flow, salinity and connectivity differ in their measurement, spatial and temporal scales, ecological significance and levels of uncertainty, making direct comparison inherently challenging. Despite these limitations, the CCVA provided a useful means of synthesising diverse information and identifying environmental processes contributing to climate risk. Future application of the assessment would therefore be most valuable when new climate projections, hydrodynamic modelling or ecological information become available to refine sensitivity and exposure estimates, or where additional species, environmental drivers or identified knowledge gaps are incorporated as understanding improves.

Preliminary consultation with the Lakes and Coorong Fishery provided initial insights into industry perspectives on climate change and adaptation. Fishers recognised freshwater inflow as an important influence on fish abundance and catch variability and described changes to fishing operations in response to environmental conditions. These observations provided an applied perspective on relationships identified through the vulnerability assessment and demonstrated that adaptation to environmental variability is already an important component of fishing operations. Consultation also established a basis for continued communication between science, management and industry as understanding of future climate impacts develops.

Overall, the project substantially addressed its three objectives. It improved understanding of how projected climate change may affect fish habitats and populations by identifying the environmental drivers, species characteristics and ecological processes contributing to vulnerability; identified potential climate-related risks relevant to the Lakes and Coorong Fishery, including several commercially harvested species with elevated vulnerability; and initiated industry consultation that provided insight into existing responses and adaptation to environmental variability. The findings provide a basis for directing further monitoring, research and adaptive management towards identified risks and knowledge gaps.

Nevertheless, considerable uncertainty remains around the timing, magnitude and ecological consequences of future environmental change. The CLLMM is a dynamic system, and climate change may also affect processes not represented in this assessment, including geomorphology, sediment dynamics and broader ecosystem processes. Future risks will also be influenced by changes in management objectives, water availability, policy settings and the effectiveness of adaptation measures, all of

which remain uncertain. Rather than representing a definitive assessment of climate risk, this work provides a foundation that can be progressively refined through targeted ecological research, environmental monitoring, hydrodynamic modelling and continued engagement with industry. Maintaining and updating this knowledge as environmental conditions, climate projections, management responses and ecological understanding evolve will be important for reducing uncertainty and supporting informed adaptation and management of fish populations and fisheries in the CLLMM.

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1 Introduction

Climate change is a pervasive global phenomenon that is reshaping physical, ecological, and societal systems across the planet (IPCC 2023). Augmentation of atmospheric greenhouse gas concentrations has led to measurable increases in global temperatures and widespread changes in climate patterns (IPCC 2023). Global surface temperature over the period 2011–2020 has risen by ~1.1 °C above pre-industrial (1850–1900) levels (IPCC 2023). Warming has significantly increased since the 1970's, the fastest rate increase in any 50 year period over the last two millennia (IPCC 2023). The role of human activity (e.g. fossil fuel combustion, deforestation, intensive agriculture) is well understood and unequivocally responsible for such rapid changes to the global climate (IPCC 2023; Koehn *et al.* 2011; Lough and Hobday 2011; Pecl *et al.* 2011).

Evolving patterns in the climate, even when considering temperature increases alone, have caused societal concern in several areas, in particular: extreme climatic conditions and damaging weather (e.g. droughts, heatwaves, intensified storms), infrastructure damage (e.g. coastal erosion, increased flooding), food and water security (e.g. drought-driven reduced water availability, agricultural and fisheries production) reduced human health and well-being (e.g. disease, population displacement) and ecosystem change (e.g. changes to habitat, biodiversity loss) (IPCC 2023). At the core of these concerns is the resulting shift in environmental conditions driven from the impact of a changing climate (IPCC 2022; IPCC 2023). Ecosystems and organisms that have evolved to depend on particular environmental conditions now face habitats and ecological settings that are rapidly transforming (Jackson 2021)

As changes to environmental conditions occur at an ever-increasing pace, species and their populations, and/or social dimensions that depend on these ecological values, need to increasingly adapt to persist. Understanding the mechanisms that confer resilience, and identifying successful strategies to support adaptation, are critical areas of research (Mason *et al.* 2022; Rodriguez-Dominguez *et al.* 2022). An important, intitial step towards developing and implementing successful adaptation strategies is understanding how larger-scale changes in climate affect environments on local scales relevant to management.

The Coorong, Lower Lakes and Murray Mouth region (here in CLLMM region) is an environmentally, culturally and economically important area with local, national and international significance (Rees et al., 2022). The CLLMM region supports diverse biotic communities associated with marine, estuarine, hypersaline, lacustrine, riverine and wetland habitats (Hayes *et al.* 2026; Phillips and Muller 2006). These habitats are located within two distinct spatial units – the Lower Lakes and the Coorong, with each spatial unit characterised by a fundamentally different aquatic environment. The environmental conditions and function of these aquatic environments to support biota are dependent on broader hydroclimatic dynamics that act across both aquatic environments. Hydrodynamics across the CLLMM region are driven by freshwater flow

regime (including environmental water deliveries), tidal exchange with the sea, operational management (e.g. barrage operations, Lower Lakes level management, Murray Mouth dredging) and other influential environmental factors (e.g. wind patterns) (Dang *et al.* 2026; Gibbs *et al.* 2018). Furthermore, freshwater flow is a key factor that provides important linkages between the Lower Lakes and the Coorong aquatic environments, affecting both the environmental conditions in the Lower Lakes and Coorong, as well as, the connection (e.g. barrage and fishway operations) between these aquatic environments (Bice and Ye 2009; Bice *et al.* 2023b; Ye *et al.* 2016; Zampatti *et al.* 2010).

Fish are a regionally highly valued biota of CLLMM region, contributing to biodiversity (Bice *et al.* 2018; Wedderburn *et al.* 2017), food-web dynamics (Giatas *et al.* 2018; Ye *et al.* 2020) and have important regional social and economic value (Bice *et al.* 2018; Department of Primary Industries and Regions 2022; Ferguson *et al.* 2018). The region supports a diverse fish community comprising freshwater, estuarine, diadromous and marine species. These populations rely on hydrodynamic process that influence environmental conditions and connectivity between the Lower Lakes and Coorong to complete critical life-history processes such as spawning, recruitment and migration. Understanding how environmental conditions and connectivity influence fish populations is therefore important for maintaining ecosystem function and biodiversity, supporting commercial and recreational fisheries, and sustaining cultural values, including those of the Ngarrindjeri people and other communities connected to the region. Therefore, maintaining healthy and resilient fish populations is fundamental to sustaining the environmental, cultural and economic values of the CLLMM region.

Climate change is a broad-scale phenomenon, yet its impacts are realised unevenly across landscapes. Certain environments become focal points where multiple climate-driven processes converge and interact, including declining freshwater runoff, increasing temperatures and sea-level rise, amplifying changes in environmental conditions and ecological processes. Coastal aquatic environments are particularly susceptible because they occur at the interface of freshwater, estuarine and marine systems, where these climate-driven processes converge. Consequently, the CLLMM region is exposed to multiple concurrent climate pressures that are expected to alter hydrodynamics, salinity, water temperature, connectivity and habitat availability across interconnected aquatic environments (Dang *et al.* 2026; Rees *et al.* 2022). The physiographic characteristics of the CLLMM region, together with the operational requirements associated with barrage management, further compound the complexity of environmental and ecological responses to climate change. As these climate pressures intensify throughout the century, the magnitude of environmental and ecological change is also expected to increase.

Towards understanding how climate change may influence the Lower Lakes and Coorong, this project investigated the potential effects of climate-driven environmental change on aquatic habitats, fish populations and the Lakes and Coorong Fishery within the CLLMM region.

The project addressed the following objectives:

- Explore potential impacts of climate change on fish habitat and fish populations in the CLLMM region;
- Improve understanding of climate change risks for the Lakes and Coorong Fishery; and
- Consult with fishing industry to gain understanding of adaptation climate change in CLLMM region.

2 Project approach

To achieve the objectives, the project was undertaken in three sequential phases. Each phase contributed to a specific or multiple project objectives and provided information that informed subsequent project phases, progressing from the synthesis of existing knowledge, then an assessment of fish species vulnerability to climate change and finally consultation with the Lakes and Coorong Fishery members.

A description of each phase and section of this report in provided as follows:

Phase 1 (Section 3) reviewed and synthesised the scientific literature relating to climate change, aquatic ecosystems and the biology and ecology of eleven fish species of ecological, cultural and fisheries importance within the CLLMM region. This phase established the scientific foundation for the project by drawing on existing conceptual knowledge of climate change impacts on the Lower Lakes and Coorong, as well as specific climate-induced environmental change affecting fishes and habitat in the CLLMM region.

In this phase, the purpose of reviewing literature included description of:

- The CLLMM study region, including its geographic location, key environmental characteristics, and the major environmental issues and pressures affecting the area;
- Climate change impacts affecting aquatic ecosystems relevant to Southern Australia and the CLLMM region. Adoption of a CLLMM region climate change conceptual model to establish future Coorong and Lower Lakes 2050 and 2090 climate change scenarios to be examined in Section's 4 and 5;
- Environmental requirements for fishes, identifying four environmental drivers (temperature, salinity freshwater flow and connectivity) altered by climate change affecting fish populations in freshwater and estuarine environments;
- Development of a conceptual model linking CLLMM region climate change pressures, environmental drivers (temperature, salinity freshwater flow and connectivity) and fish life-history processes governing population dynamics;
- Fishes life-history strategies and use of life-history traits towards understanding future environmental changes impacting fish. Introduced recognised CLLMM region fish functional groups, fish life-history characteristics and values of fish in the CLLMM region;
- Compile information and data on current state of knowledge on fish species life history, habitat requirements, environmental tolerances and ecological characteristics to develop species profiles and summary tables for subsequent use in Section 4. These can be found in the Appendices (1–5) of this report;
- Description of species life-cycle knowledge gaps, limitations and assumptions in using information for climate change sensitivity assessments.

Phase 2 (Section 4) involved applying a Climate Change Vulnerability Assessment (CCVA) framework to evaluate fish species vulnerability across four climate change scenarios: Coorong (2050, 2090) and Lower Lakes (2050, 2090).

Information and data compiled during Phase 1 was integrated with climate scenarios, projections and hydrodynamic modelling outputs in Phase 2 to describe projected changes in temperature, freshwater flow, salinity and habitat connectivity. For each environmental driver; the change between the baseline and 2050- or 2090-time horizon was used to determine the magnitude of climate change exposure.

Climate change is expected to lead to hydrologic extremes in both drought and floods. Given the importance of freshwater flow to maintaining aquatic ecosystem structure and function (Section 3); projected declines for Murray-Darling Basin runoff are concerning. To account for this impact, an assessment of the environmental drivers was undertaken at two levels of projected freshwater inflow: a 'higher' level representing a medium-size reduction in freshwater flow and a 'lower' level corresponding to a severe reduction in freshwater flow.

To apply existing CCVA frameworks to the CLLMM region, modifications were required to adjust measures of sensitivity to suite local environmental conditions. This entailed developing sensitivity metrics for temperature, salinity and freshwater flow for life-stages. Re-scaling of existing sensitivity metrics (e.g. age at maturity, juvenile and adult movement distance) to suite local species biology and ecology was also undertaken. Information used to conduct species sensitivity and environmental exposure scoring, and key information describing the assessment approach can be found in Section 4 and Appendices (6–11).

Phase 3 (Section 5) describes preliminary consultation with Lakes and Coorong Fishery commercial fishers to understand fisher's perspectives and identify potential opportunities for adaptation informed by both scientific evidence and industry knowledge. This phase included development of a questionnaire to collect commercial fisher responses to climate change (Appendix 12). Communication of climate change scenarios, potential future environmental changes arising and corresponding vulnerability of fish species for the Lower Lakes and Coorong.

Phase's 1 and 2 contributed to objectives 1, 2 and 3, while Phase 3 largely informed objectives 2 and 3. The findings from all three phases are synthesised in Section 6, where the implications of projected climate change for fish populations, fisheries and climate adaptation within the CLLMM region are discussed.

3 Background literature and species profiles

3.1 Study area: the Lower Lakes and Coorong

The CLLMM region lies at the terminus of the River Murray, where it meets Encounter Bay (Figure 1). It comprises interconnected estuarine, lacustrine, riverine and wetland systems and is listed as a Ramsar wetland of international importance (Phillips and Muller 2006). This study focuses on the Lower Lakes and Coorong, two major water bodies that support diverse fish assemblages and support commercial fishing.

Descriptions of the Lower Lakes and Coorong, including their environmental diversity, ecological processes, and management responses, have been extensively documented (Aldridge *et al.* 2018; Bice *et al.* 2018; Bice and Ye 2009; Gibbs *et al.* 2018; Kingsford *et al.* 2011; Lester *et al.* 2009; Mosley *et al.* 2024a; Mosley *et al.* 2024b; Muller *et al.* 2018; Nicol *et al.* 2018; Phillips and Muller 2006; Ryan 2018; Ye *et al.* 2016; Zampatti *et al.* 2010). Here, we summarise key features relevant to climate change impacts on the Lower Lakes and Coorong, particularly those influencing fish habitat.

The region has a Mediterranean climate, with hot, dry summers and cool winters (Ryan, 2018). Mean annual temperature at Goolwa is 11.3 °C (0.6–43.0 °C, BOM station 023849), and rainfall ranges from 350–850 mm, with winter runoff supporting streamflow and aquifer recharge (Phillips and Muller, 2006). Climate, geomorphology, and hydrology together control water regimes and habitat conditions (Phillips and Muller 2006); with the CLLMM region's location at the terminus of the Murray–Darling Basin making it highly sensitive to hydrological change (Phillips and Muller, 2006; Kingsford et al., 2011; Mallen-Cooper and Zampatti, 2018; Mosley et al., 2024b). Reduced inflows threaten ecological condition, while environmental water delivery remains critical for ecosystem function (Brookes *et al.* 2015; Mallen-Cooper and Zampatti 2018; Mosley *et al.* 2023). These pressures are likely to intensify under climate change (Lester *et al.* 2013; Mosley *et al.* 2024b; Rees *et al.* 2022).

Lakes Alexandrina and Albert are shallow (< 5 m) freshwater lakes with surface areas of ~65,000 ha and ~17,720 ha, and volumes of ~1620 GL and ~280 GL, respectively, at 0.75 m AHD (Gibbs *et al.* 2018). Freshwater in the Lower Lakes may be retained up to 0.83 m AHD before water starts flowing over 'spillways' located adjacent to each barrage (Gibbs et al., 2018). Lower Lakes hydrology is dominated by River Murray inflows, with groundwater contributing minimally to water balance but adding to salinity (Gibbs *et al.* 2018). Wind and evaporation drive water level variability in the Lower Lakes and Coorong (Gibbs et al., 2018). Evaporation (600–950 GL/year; mean ~800 GL/year) is relatively uniform seasonally, with annual losses influenced by rainfall variability (Heneker, 2010). Wind seiching redistributes water, raising levels in Lake Alexandrina under southerly winds and in Lake Albert under northerly and westerly winds (Gibbs et al., 2018).

Water levels affect fish habitat in the Lower Lakes. The lake area changes little above 0.3 m AHD, but levels below this threshold during drought lead to major wetland habitat loss (Gibbs *et al.* 2018; Hammer *et al.* 2013). Water levels maintained above

0.6 AHD ensure fringing wetlands contain suitable water levels, providing conditions (e.g. aquatic plants, water quality, decrease interactions with alien fishes) for native small-bodied fishes (Wedderburn *et al.* 2019). With reduced levels, littoral vegetation is diminished, therby altering plant communities (Frahn *et al.* 2013). Submerged macrophytes rarely persist below 0.0 m AHD, while emergent species require ≥ 0.2 m AHD (optimal > 0.4 m AHD) (Nicol *et al.* 2018). With reduced water levels, the decline or extirpation of native freshwater fishes can result with harsher environmental conditions (Hammer *et al.* 2013; Wedderburn *et al.* 2012; Zukowski *et al.* 2026).

Salinity in the Lower Lakes is typically low (median 742 µS/cm; < 3000 µS/cm) and controlled by inflows and outflows (Gibbs et al., 2018; Nicol et al., 2018). Reduced water levels increase salinity via evapo-concentration (Bice and Ye 2009), with values during the Millennium Drought reaching 6000 µS/cm in Lake Alexandrina and 20,000 µS/cm in Lake Albert. Lake Albert remains more saline due to groundwater inputs and its terminal nature, but both lakes freshen following high inflows (Gibbs *et al.* 2018).

Through its connection to the Coorong via the barrages, Lake Alexandrina can experience temporary 'reverse head' conditions (Coorong water level > lake level), allowing seawater intrusion and increasing salinity (Nicol *et al.* 2018). These incursions elevate salinity near the barrages; for example, during the Millennium Drought, salinity in the Goolwa Channel exceeded 25,000 µS/cm, while broader lake salinity was ~5,000 µS/cm (Muller *et al.* 2018). The frequency of such intrusions is expected to increase under climate change and sea-level rise (Nicol *et al.* 2018). Saline intrusions into Lake Alexandrina can facilitate colonisation by opportunistic, reef habitat-forming species such as the benthic polychaete tubeworm (*Ficopomatus enigmaticus*). These reefs provide habitat that can be used by a range of estuarine fish species in the Coorong (Bice and Ye 2009; Schroder *et al.* 2025).

The Coorong is a shallow lagoon running parallel to the Southern Ocean, separated by the Younghusband Peninsula. The Murray Mouth is located in then northern Coorong, links a north-western arm toward Goolwa and a south-eastern arm into the lagoons. Five barrages — Tauwitchere, Goolwa, Mundoo, Boundary Creek and Ewe Island — constructed in the 1930s separate the Coorong from the Lower Lakes, limiting seawater ingress under low flows (Phillips and Muller, 2006; Mosley et al., 2024b). The barrages create a sharp salinity boundary, while freshwater discharge, seawater inflows, and evaporation drive salinity patterns across the system, particularly near the Murray Mouth (Phillips and Muller 2006; Ye *et al.* 2019).

The Coorong comprises the Murray Estuary, North Lagoon, and South Lagoon. The Murray Estuary extends from the Goolwa Barrage to Pelican Point (~12 km south of the Murray Mouth) and is a shallow, wave- and wind-driven mixing zone with numerous

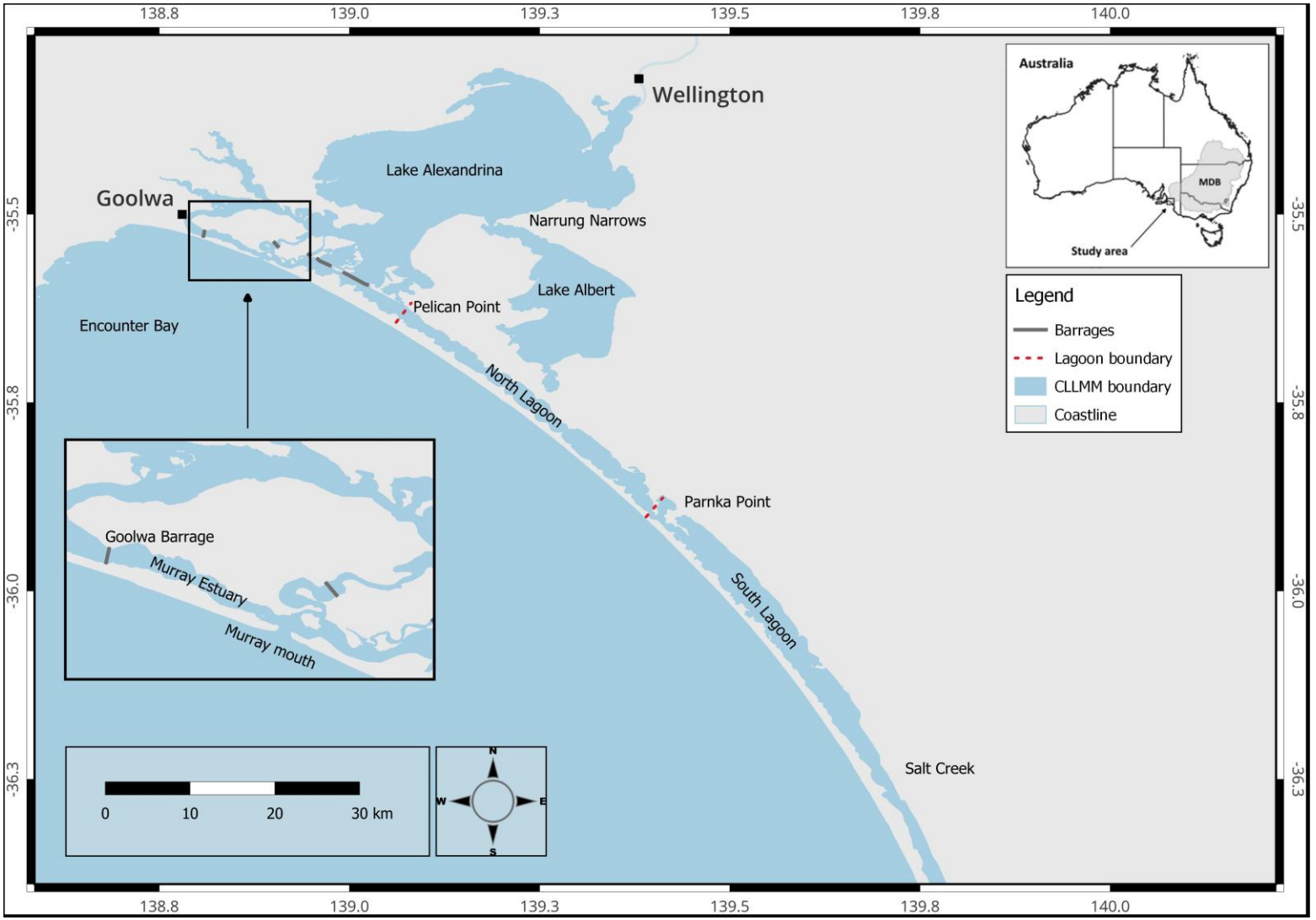


Figure 1. Map illustrating the study area: The Coorong, including the Murray Estuary, North and South Lagoons. Lower lakes: Lake Alexandrina and Lake Albert.

channels (< 5 m). Hydrodynamics are controlled by barrage discharge, tidal exchange,and wind mixing, producing salinity from freshwater during high flows to marine or low hypersaline conditions during low flows (Ye *et al.* 2016).

Freshwater releases maintain brackish conditions below the barrages and near the Murray Mouth, whereas flow cessation during extreme drought can result in marine to hypersaline conditions (Ye et al., 2016; Bice et al., 2023). During low flows, freshwater releases mix with saltwater to provide salinities of 5–35 psu and thus quality habitat for juvenile marine-estuarine opportunists such as mulloway (*Argyrosomus japonicus*; (Bice *et al.* 2023c). The presence of barrages also pose a negative influence, acting as barriers to fish movement, although installed fishways (since 2003) help improve connectivity, particularly for diadromous species such as congolli (*Pseudaphritis urvillii*). To operate, fishways require water levels ≥ 0.4 m AHD in Lake Alexandrina, with 0.5–0.85 m AHD supporting year-round operation (Muller *et al.* 2018).

The North Lagoon is shallow (~1.2 m) and narrow (~3 km) and extends ~50 km from Pelican Point to Parnka Point, where it is separated from the South Lagoon by a narrow constriction (Gibbs *et al.* 2018; Phillips and Muller 2006). Salinity is highly variable, ranging from freshwater during floods to hypersaline (~100 psu) during extended drought, driven by barrage discharge and tidal influence (Phillips and Muller, 2006; Schroder et al., 2025). The South Lagoon extends from Parnka Point to Salt Creek and is shallow (~1.4 m), long, and narrow (Gibbs *et al.* 2018). Limited freshwater input maintains hypersaline conditions (60 to > 120 psu) and promotes eutrophication, with reduced flushing driving nutrient accumulation and degradation of benthic communities (Brookes et al., 2018; Mosley et al., 2023).

3.2 Climate change: impacts on Australian aquatic ecosystems

Climate change is characterised by long-term changes in climatic conditions, including increasing temperatures, altered rainfall patterns, rising sea levels and increasing atmospheric carbon dioxide concentrations (IPCC 2023). These changes are expected to have widespread consequences for aquatic ecosystems. Although reduced pH associated with elevated atmospheric carbon dioxide is known to affect aquatic ecosystems, this project scope is limited to the climate pressures of warming, drying and sea-level rise.

Climate change is expected to affect Australian aquatic ecosystems through three principal climate pressures: warming, drying and sea-level rise (Balcombe *et al.* 2011; Jenkins *et al.* 2011). The magnitude and expression of these pressures vary geographically according to factors such as latitude, altitude and proximity to the coast (Jenkins *et al.* 2011). Climate models project continued warming across Australia, while Southern regions are expected to experience declining precipitation and more prolonged periods of drying interspersed with episodic wetter conditions (Chiew *et al.* 2025; Prosser *et al.* 2021; Rees *et al.* 2022).

Rainfall across the Murray-Darling basin will be characterised by greater variability and decline in cool-season rainfall (Chiew et al., 2025). With additional climate-induced temperature and evapouration increases, a trend of declining runoff to the end of

century is forecast for the Murray Darling Basin (Chiew et al., 2025). Future hydroclimates are expected to be characterised by increased hydrological variability, defined by more frequent and intense drought along with the occurrence of large flood events (Chiew et al., 2025).

Marine ecosystems of south-eastern Australia are projected to experience some of the most rapid climate-driven changes in the Southern Hemisphere (Pecl *et al.* 2011). Anticipated changes include increasing sea-surface temperatures, sea acidification, altered sea circulation, sea-level rise and more frequent marine heatwaves (Lough and Hobday 2011). Australia is warming about 1.4 times the Global average (CSIRO and Bureau of Meteorology 2025). Warming rates have been particularly high along Australia's south-eastern and south-western coasts (Lough and Hobday 2011), increasing the risk that temperate species may exceed their thermal limits and elevating the likelihood of events such as harmful algal blooms and fish kills (Glibert *et al.* 2022; Roberts *et al.* 2019).

Coastal aquatic ecosystems, including estuaries, rivers, wetlands and lakes, may be particularly sensitive to climate change because of their shallow, interconnected nature (Lauchlan and Nagelkerken 2020; Oczkowski *et al.* 2015). Estuaries are expected to experience warming, sea-level rise and altered salinity gradients associated with declining freshwater inflows (Gillanders *et al.* 2011; Khojasteh *et al.* 2021). These changes can affect habitat extent, salinity, dissolved oxygen and nutrient dynamics, with evidence linking reduced freshwater inflows to hypoxia in some Australian estuaries (Brookes *et al.* 2022; Chilton *et al.* 2021; Cottingham *et al.* 2018).

Freshwater ecosystems are similarly vulnerable to warming, drying and reduced water availability (Capon *et al.* 2021; Koehn *et al.* 2011). Small reductions in rainfall may be amplified into much larger declines in surface runoff (Chiew and Prosser 2011), altering flow regimes, reducing wetted habitat, disrupting ecological cues and degrading water quality (Koehn *et al.* 2011). These impacts are particularly relevant in highly regulated river systems such as the Murray River, where water resource development and climate change have contributed to substantial reductions in flow (CSIRO 2008; Prosser *et al.* 2021; State of the Climate 2024). About 61% of the historic mean annual discharge of the River Murray already lost through flow regulation and water extraction (CSIRO, 2008). In dry periods, environmental water provision can consist of up to 100% of the freshwater flow volume entering the Coorong (Brookes *et al.* 2022). With the majority of freshwater flow reaching the Murray Mouth originating from the Southern Murray–Darling Basin, further projected reductions in runoff are expected to compound existing hydrological stress. Events such as the Millennium Drought provide insight into the ecological consequences of prolonged drying and low-flow conditions (Gawne and Thompson 2023).

Coastal aquatic ecosystems such as those in the CLLMM region occur at the interface of marine, estuarine and freshwater environments and are therefore exposed to multiple interacting climate pressures. Compounded effects from sea-level rise, altered runoff regimes and warming may drive complex changes in environmental

conditions, with consequences for habitat availability, connectivity and ecological processes that underpin fish populations and fisheries (Rees et al., 2022).

3.3 Climate change and the CLLMM region

Preliminary investigations of climate change impacts on the CLLMM region's ecological, social and cultural values identified three co-occurring pressures on aquatic ecosystems: warming, drying and sea-level rise (Dunlop *et al.* 2022; Grigg *et al.* 2022; Rees *et al.* 2022). Rees et al. (2022) summarised their combined influence in a conceptual model, reproduced in Figure 2.

This model (herein, the 'Rees et al. model') proposes two climate-induced hydrological states: an initial 'Hydrological Minimum' followed by a 'High Ocean Exchange' state (Figure 2). The Hydrological Minimum represents the point at which increasing seawater contribution offsets freshwater losses from reduced precipitation and increased evaporation. With further sea level rise, increasingly there will be greater marine influence on waterbody hydrodynamics, this dominance reflected by the High Ocean Exchange state. These changes are supported by very high confidence in short- and long-term air temperature and sea-level projections, and high confidence in declining cool-season rainfall by the end of the century (Rees et al., 2022).

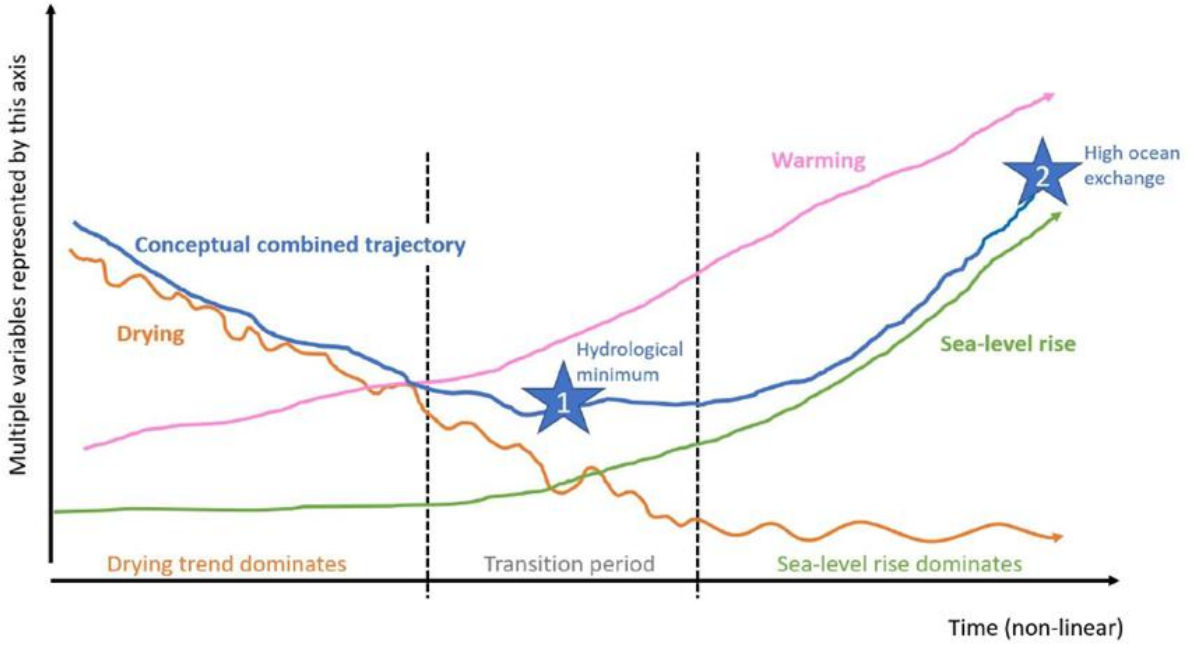


Figure 2. A conceptual description of climate change in the CLLMM region (source Rees et al. 2022). This conceptual model illustrates the transition from State 1 (Hydrological Minimum) to State 2 (High Ocean Exchange).

The Rees et al. model was informed by knowledge from a range of contributors, particularly environmental experts familiar with the CLLMM region and the ecological impacts of the Millennium Drought – a potential analogue for future climate change conditions (Rees *et al.* 2022). The model provides an accepted conceptual representation of future climate change in the CLLMM region and served as the

foundation for developing the climate change scenarios used in the vulnerability assessment (Section 4).

The Rees et al. model describes an earlier, shorter-term transition towards a 'Hydrological Minimum', followed by a longer-term transformation towards a 'High Ocean Exchange' state driven increasingly by sea-level rise. This progression represents substantial shifts in hydrology, salinity, connectivity and habitat conditions that are expected to affect fish (Rees et al., 2022). Although the sequence of these state transitions is conceptually defined, considerable uncertainty remains regarding their timing and ecological impacts. The Hydrological Minimum may occur around mid-century, with the High Ocean Exchange state developing towards late-century, while conditions during the Hydrological Minimum may resemble or exceed the severity of the Millennium Drought (Rees et al., 2022). Potential impacts of these environmental changes on Lower Lakes and Coorong fishes are summarised in Table 1, with responses expected to vary among species, life stages, waterbodies and the magnitude and direction of environmental change.

To provide further insight on the possible hydrological consequences arising from climate change on the CLLMM region; a concurrent study by (Dang *et al.* 2026) undertook hydrodynamic modelling to estimate several physicochemical and hydrological parameters (e.g. water temperature, salinity, water depth and water level) at 2050- and 2090-time horizons. They describe the following projections for the CLLMM region under moderate climate change (Radiative forcing of 4.5 W/m^2 by 2100). Median sea-level rise (Victor Harbour, SA) is expected to increase to + 0.22 m AHD at 2050 and to + 0.51 m by 2090, relative to a 1995-2014 baseline. The average annual mean daily South Australian temperature is projected to increase to + 1.28 °C by 2050 and reach + 2.04 °C by 2090 relative to 1995-2014 baseline. Seven percent of the days in the CLLMM region are likely to be above a daily mean 27 °C by 2090 (Dang *et al.* 2026). In addition is it reported that projected runoff from the Southern Murray-Darling Basin is expected to increasingly decline to 2090, with flow reductions up to 67% (Dang *et al.* 2026).

In line with the conceptual framework of Rees et al. (2022) and methods used by Dang et al. (2026), this study examines moderate-level climate change at mid- and late-century (2050 and 2090) on fish and fisheries in the CLLMM region. The 2050 time horizon provides a practical basis for assessing potential impacts on commercially important fish species because it represents a sufficiently near-term timeframe to inform climate adaptation planning while capturing projected environmental change (Champion *et al.* 2023). The 2090 time horizon was included to assess potential impacts under more advanced climate change, when the influence of sea-level rise and declining freshwater availability is expected to become more pronounced. CLLMM region management applying to baseline periods used in this study was assumed to remain unchanged for future time horizons. Future management responses to climate change remain uncertain (Dunlop *et al.* 2022) and testing future management scenarios in response to climate change is outside the scope of this study.

Table 1. Environmental changes during the Hydrological Minimum and Dominant Sea-level states for the Lower Lakes and Coorong described by Rees et al. (2022). Note: Species impacted considers current project fish species but may apply to other CLLMM region fish species. Species abbreviations: BH = bony herring; CC = common carp; GP = golden perch; RP = redfin perch; C = congolli; BB = black bream; M = mulloway; GF = greenback flounder; YM = yelloweye mullet; SS = sandy sprat; SH = Smallmouth hardyhead.

STATE	WATERBODY	ENVIRONMENTAL CHANGE	SPECIES IMPACTED
Hydrological Minimum	Lower Lakes	Median flows will be lower and periods of very low flows will be more frequent with longer duration in the Basin.	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	Water levels will fluctuate and frequently be lower than during the Millenium Drought. Increased temperature in water bodies.	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	As lake levels drop, more seawater may seep through or around barrages. Salinity in Lakes will increase due to reduced freshwater inputs and increased evapotranspiration.	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	There will be increased exposure of existing potential acid-generating sediments (ASS), which could decrease pH to acute levels and mobilise toxic metals from sediment when rewetted.	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	There will be reduced connectivity between Lakes Alexandrina and Albert, more frequent loss of connectivity between the lakes and the Coorong, and reduced connectivity to fringing wetlands.	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	There will be loss of submergent macrophytes, and emergent macrophytes may colonise the lake as waters retreat but will be in poor condition due to decreasing lake level and increasing salinity.	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	Decreased flows in streams flowing into Lower Lakes will lead to loss of habitat complexity and connectivity across freshwater areas	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	The tubeworm (*Ficopomatus enigmaticus*) will spread into the Goolwa Channel and Lake Alexandrina as salinity increases.	BH, CC, GP, RP, C, BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Lower Lakes	There will be a loss of fish habitat along with the loss of submergent macrophytes in the Lower Lakes.	BH, CC, GP, RP, C
Hydrological Minimum	Lower Lakes	Invasive fish species will be increasingly present, threatening survival of vulnerable small-bodied native fish due to predation and competition for habitat and food resources	CC, RP
Hydrological Minimum	Lower Lakes	There will be declines in salt-sensitive fish species and increases in salt tolerant species.	BH, CC, GP, RP, C, BB, M, GF, YM, C, SS, SH

Table 1. (cont'd)

STATE	WATERBODY	ENVIRONMENTAL CHANGE	SPECIES IMPACTED
Hydrological Minimum	Lower Lakes	Operation of barrage fishways will significantly reduce or cease, particularly affecting diadromous species movement and recruitment, due to insufficient hydrological connectivity, flow velocity and chemical signal gradients, resulting in decreased abundances of congolli, lamprey and galaxias.	C, BH, SS, SH, GP, YM
Hydrological Minimum	Lower Lakes	In general, fish reproduction and recruitment will be reduced by temperature increases (the extent varying among species).	BH, CC, GP, RP, C
Hydrological Minimum	Coorong	Sedimentation will increase in the Murray Mouth, leading to increased potential for the Murray Mouth to close, and decreasing exchange between the sea, Murray estuary and Coorong.	BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Coorong	There will be more frequent and longer periods of disconnection between the North and South Coorong Lagoons and there will be minimal freshwater inputs from Lake Alexandrina.	BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Coorong	There will be greater accumulation of salt and nutrients throughout the Coorong due to increasing influx of salt, increased evaporation and reduced flushing.	BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Coorong	The South Lagoon will increasingly experience prolonged periods of hypersalinity, eutrophication and low water levels.	BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Coorong	The North Lagoon will begin to experience impacts like those that occurred in the South Lagoon during the Millenium Drought.	BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Coorong	There will be changes to the planktonic basal food resources, combined with direct effects on invertebrate communities that will also drive changes to food web structures. Similar responses will occur with benthic food resources, with marine invertebrates dominating. Biomass and diversity of benthic food resources will be reduced.	BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Coorong	There will be decreasing inundation of saltmarsh and mudflats and persistent decreases in submerged vegetation (seagrass and macrophyte communities) throughout the Coorong and particularly in the South Lagoon. Habitat and food losses for fishes but with increasing habitat for species tolerant of hypersaline conditions.	BB, M, GF, YM, C, SS, SH
Hydrological Minimum	Coorong	Changes to water regime within the North Lagoon could lead to increasing development of a character similar to the current state of the South Lagoon, supporting fish that prefer higher salinities.	
Hydrological Minimum	Coorong	Decreasing connectivity across the entire system will impact fishes that require hydrological connectivity for movement and recruitment.	BB, M, GF, YM, C, SS, SH

Table 1. (cont'd)

STATE	WATERBODY	ENVIRONMENTAL CHANGE	SPECIES IMPACTED
Hydrological Minimum	Coorong	There will be continuing change in fish communities, with increases in marine and salt-tolerant taxa.	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Lower Lakes	Increased sea level will reduce the ability of the Lower Lakes to flow to the sea, increase salinity and alter carbon and nutrient dynamics, leading to water quality declines.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	A salinity gradient will form towards the River Murray.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	There will be occasional periods of flushing during episodes of high River Murray flows, leading to much lower salinities.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	Increasing salinity will affect all biota, from phytoplankton through to top consumers such as fish.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	There will be continued loss of salt-sensitive taxa which will be replaced by more salt-tolerant taxa, including some estuarine and marine species.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	Frequent seawater exchange will increase connectivity, favouring many estuarine and diadromous fish species.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	There will be gradual establishment of tidal and estuarine ecosystems in the Lower Lakes, such as seagrass, mud flats and salt marshes.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	Seawater will inundate freshwater wetlands and the diverse habitat associated with tributaries, leading to an overall loss in diversity of habitat for higher trophic organisms.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	There will be loss of fish habitat, particularly from loss of macrophytes in the Lower Lakes.	BH, CC, GP, RP, C
High Ocean Exchange	Lower Lakes	Fish assemblages in the freshwater sections are strongly influenced by the flow from the River Murray, maintaining suitable habitat, particularly vegetation.	BH, CC, GP, RP, C
High Ocean Exchange	Coorong	Increasing sea level results in increases in water levels in the Murray estuary and Coorong	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Coorong	Connection between North and South Lagoons will improve, allowing a flux of nutrients and salinity out of the South Lagoon and along the Coorong.	BB, M, GF, YM, C, SS, SH

Table 1. (cont'd)

STATE	WATERBODY	ENVIRONMENTAL CHANGE	SPECIES IMPACTED
High Ocean Exchange	Coorong	The estuarine conditions that presently exist through the Murray estuary towards the North Lagoon will become increasingly marine, depending on the volume and salinity of flows from Lake Alexandrina.	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Coorong	Higher water levels will potentially increase habitat area and suitable conditions for seagrass and macrophytes, which inhabit the Murray estuary and the Coorong.	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Coorong	High sea-level rise may lead to loss of mudflats and saltmarsh in the Coorong, depending on extent of new habitat formed and inundation by tidal regime.	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Coorong	Sea-level rise and aridification, affecting dunes, may influence erosion which may result in change depth profile of the North and South Lagoons.	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Coorong	Increasing water levels and tidal influence results in increase diversity of habitats compared to drying phase.	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Coorong	There will be continuing change in fish communities, with increases in marine and salt-tolerant taxa. Fish diversity in the Coorong will be primarily comprised of marine taxa.	BB, M, GF, YM, C, SS, SH
High Ocean Exchange	Coorong	There will be changes to planktonic basal food resources, combined with direct effects on invertebrate communities that will also change food web structures. Similar responses will occur with benthic food resources, with marine invertebrates dominating. Biomass and diversity of benthic food resources will be reduced.	BB, M, GF, YM, C, SS, SH

3.4 Climate change: environmental drivers and effects on fishes

Climate change presents unique challenges for aquatic ecosystems, fish populations, fisheries and ultimately people (Fulton *et al.* 2020; Pörtner and Peck 2010). Climate change may affect fishes in a number of ways, but changes to environments inhabitated by fish are reported, such as increased exposure to harsher environmetnal conditions (Nagelkerken *et al.* 2025; Nagelkerken *et al.* 2023). These changes may manifest across broad spatial and temporal scales, varying from smaller and shorter-scale, but extreme impacts (e.g. heatwaves) to larger, longer, incremental change (e.g. warming) (Gillanders *et al.* 2011; Gillanders *et al.* 2022; Nagelkerken *et al.* 2023). Some changes, such as increased rainfall variability, can prompt cascading changes in other environmental processes (e.g. flow regime, sediment and nutrient transport); subsequently influencing key habitat features (e.g. water temperature, salinity, dissolved oxygen) that aquatic biota have evolved with to depend upon or tolerate (Ficke *et al.* 2007; Lauchlan and Nagelkerken 2020). Consequently, shifts in fundamental environmental conditions affect fish distribution and abundance (Lauchlan and Nagelkerken 2020; Pecl *et al.* 2011; Poloczanska *et al.* 2013), which in turn affects productivity, food webs, and ultimately fisheries.

Understanding how fish depend on their environment is essential to assessing anthropogenic impacts (Koehn and Kennard 2013; Pratchett *et al.* 2011). Fishes have evolved with different environment attributes, that assist them in completing their life cycle (Mims and Olden 2012; Olden and Kennard 2010; Teichert *et al.* 2017; Wang *et al.* 2020; Winemiller and Rose 1992). This notion of a species dependence on the external environment (i.e. resources or conditions) is what underpins the concept of an organism's 'habitat' (Hall *et al.* 1997; Koehn and Kennard 2013).

Fish habitat comprises many environmental factors that can differ across aquatic ecosystems (Koehn and Kennard 2013). Fishes have complex relationships with environmental factors, with requirements changing with life-stage (i.e. ontogenetic shifts). For riverine and estuarine inhabiting fishes, principal external abiotic factors (e.g. freshwater flow) enable survival, reproduction, and thus population dynamics (Gillson 2011; Ye *et al.* 2016). Some abiotic factors are more critical than others, having disproportional influence, and considered 'drivers' of ecological patterns (Bice *et al.* 2018; Brookes *et al.* 2022; Gillanders *et al.* 2022). For instance, the quality and quantity of water are fundamental components of fish habitat that dictates where and how fish may live (Bice and Ye 2009; Brookes *et al.* 2022).

Climate change affects fish populations primarily through its influence on several 'environmental drivers' including temperature, freshwater flows, salinity and connectivity (Gillanders et al., 2011; Koehn et al., 2011; Morrongiello et al., 2011; Hallett et al., 2017; Brookes et al., 2022; Gillanders et al., 2022; Nagelkerken et al., 2023; Whitfield et al., 2023). Changes in recruitment, growth, survival, abundance and reproductive processes, including maturation, fecundity and spawning, have all been linked to variation in environmental conditions such as temperature, salinity and dissolved oxygen (Gillanders et al., 2011). The following sections (3.4.1–3.4.4) describe

the relationships between fish and the four key environmental drivers affected by climate change (temperature, freshwater flow, salinity and connectivity), including their associated effects on fish.

In some climate change studies, drivers of environmental change are considered to be primary climate factors such as air temperature, rainfall and sea-level rise that cause top-down 'pressures' or 'stressors' (Hallett *et al.* 2017; Lauchlan and Nagelkerken 2020). Drivers, pressures or associated prefix terms (e.g. primary, secondary) are used differently when describing causal factors throughout the literature, resulting in confusion (Oesterwind *et al.* 2016). In this study we consider 'pressures' to describe large-scale climate change factors (e.g. warming, drying, sea-level rise) forcing on ecosystems and 'drivers' to be environmental factors that subsequently control or influence fishes life history or population attributes, which is in-line with the use of these terms elsewhere (e.g. (Koehn *et al.* 2011) (Gillanders *et al.* 2022). This study examines four environmental drivers: temperature, freshwater flows, salinity and habitat connectivity that often are reported to affect fish populations and are in turn affected by climate change (Brookes *et al.* 2022; Gillanders *et al.* 2011; Gillanders *et al.* 2022; Hallett *et al.* 2017; Koehn *et al.* 2011; Morrongiello *et al.* 2011; Nagelkerken *et al.* 2023; Whitfield *et al.* 2023).

The links between climate change and the four environmental drivers understudy and their effects on fish populations are outlined in a conceptual model (Figure 1). This model recognises that climate pressures: warming, drying and sea-level rise act through environmental drivers to influence fish populations. Climate pressures may also interact, potentially amplifying effects of environmental drivers on fish populations. For example, warming may exacerbate drying through increased evaporation across broad areas of the Murray–Darling Basin. Consequently, larger reductions in runoff arising from outside the CLLMM region may propagate through connected landscapes, downstream to influence environmental conditions within the region.

Fish are sensitive to environmental drivers, as their physiology (e.g. tolerance), behaviour (e.g. movement) and life-history requirements (e.g. spawning) depend on habitat attributes such as water temperature, flow-regimes or salinity gradients (Brookes *et al.* 2022; Earl and Ward 2014; Gillanders *et al.* 2011; Gillanders *et al.* 2022; Morrongiello *et al.* 2011; Nagelkerken *et al.* 2023; Whitfield *et al.* 2023). Importantly, the conceptual model in Figure 1 also highlights that fish life-history and subsequent population dynamics respond to a suite environmental drivers (i.e. diversity), intermediate factors moderating individual drivers (e.g. habitat volume) or, interactions among multiple environmental drivers (i.e. complexity). Some drivers are essential, connectivity for instance may moderate impacts from other environmental drivers (e.g. temperature, salinity), therby facilitating fish access to use favourable habitat and escape from unfavourable conditions (Koehn *et al.* 2011; Lauchlan and Nagelkerken 2020). The interaction among drivers is also less understood aspect of climate change impacts on biota (Gillanders *et al.* 2011; Nagelkerken *et al.* 2023). While the conceptual model recognises environmental drivers rarely operate in isolation, assessments of exposure is often undertaken by scoring environmental drivers, individually, on a practical basis to evaluate vulnerability (see Section 4).

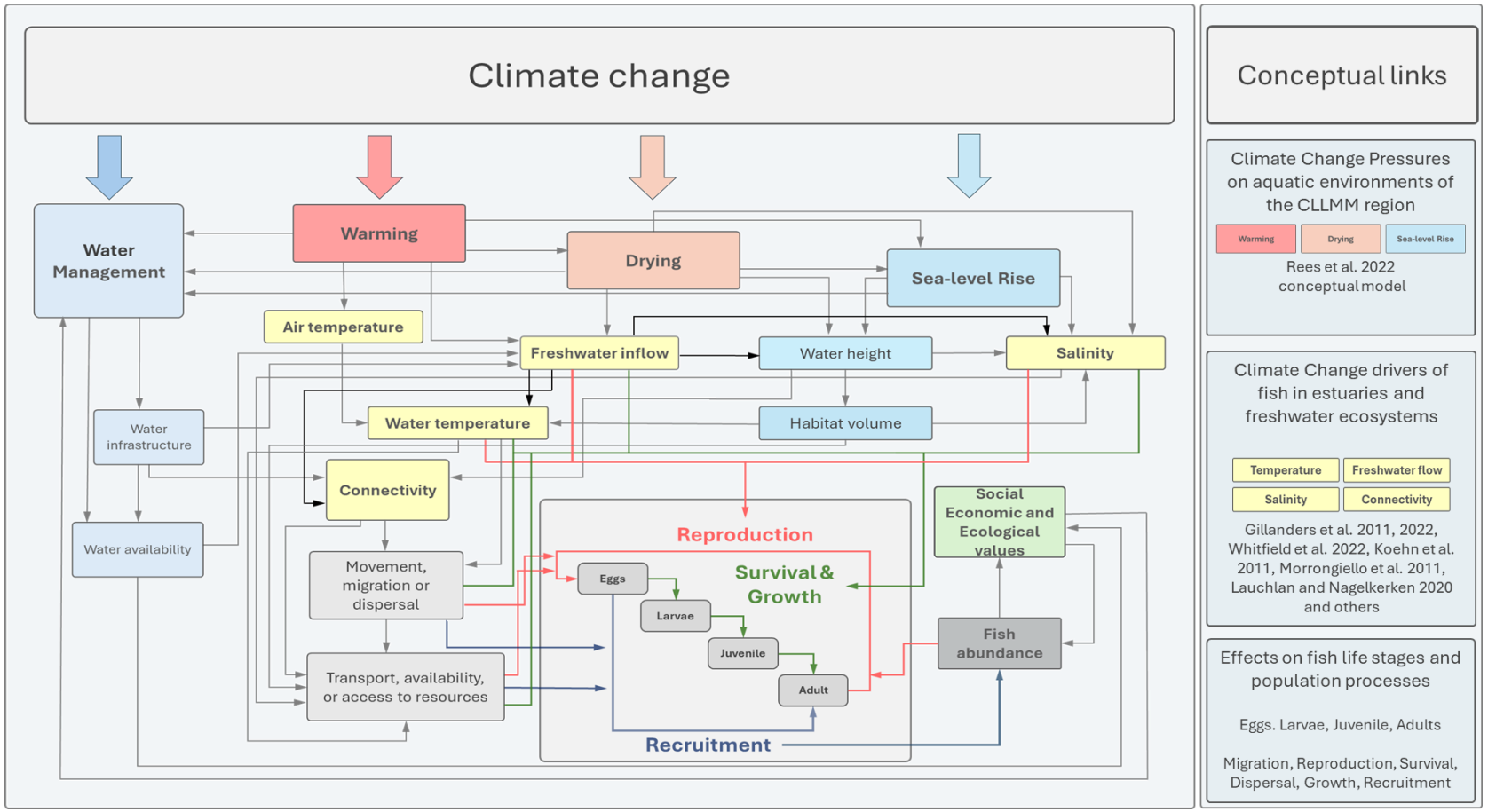


Figure 3. A conceptual model linking climate change pressures (warming, drying, and sea-level rise) to environmental drivers (freshwater flow, connectivity, temperature, and salinity) and fish population dynamics (survival, growth, reproduction, and recruitment). Freshwater flow affects other drivers, both directly and indirectly, through intermediary factors (e.g. habitat volume); further influencing the movement, transport, and availability of resources (e.g. nutrients). Water and fish are recognised for multiple values and their dependence on management. Arrows signify linkages and system complexity emerging from interacting drivers. Effect signs ('+' or '–') are not specified, as fish responses may vary under climate change, natural hydrological variability, and is species, life stage, and context dependent.

Freshwater is a key driver for this region, it has important direct effects on reproduction, survival and growth processes, and indirectly affects recruitment through facilitating connectivity or through resource provision. Freshwater may also influence other environmental drivers (temperature, salinity connectivity) that subsequently influence different aspects of life history. Water management is also acknowledged to be influenced by climate pressures and is known to influence multiple environmental drivers both directly (e.g. infrastructure on connectivity) or indirectly (e.g. lake water levels and salinity). Water, environment and fish population management are linked through human dimensions with consumption and demands for these resources means intensified intereactions as climate change progresses. The value of fish to maintaining ecosystem structure and function (i.e. food webs) are also identified and require consideration when examining climate change impacts on fish populations.

3.4.1 Temperature

Temperature describes thermal conditions of the aquatic environment. Within aquatic ecosystems, temperature varies over diurnal, seasonal and annual timescales in response to climatic conditions, solar radiation, water depth and hydrological processes. Climate change is increasing average air and water temperatures while increasing the frequency, duration and intensity of extreme heat events, altering thermal regimes across aquatic ecosystems.

Temperature is a primary regulator of fish physiology, influencing ecological performance through its effects on growth, reproduction, phenology and survival (Ern *et al.* 2023). These physiological processes operate within species-specific thermal tolerance limits, beyond which physiological performance declines and, if thermal thresholds are exceeded, mortality may occur (Smith *et al.* 2023). As ectotherms, fish cannot internally regulate body temperature and instead rely on behavioural thermoregulation by seeking suitable water temperatures (Jobling 1996). Consequently, movement is an important behavioural response to thermal stress, with many fish species shifting their distributions (e.g. range-shift) in response to rising temperatures from climate change (Booth *et al.* 2011).

Across a fish's life, temperature regulates both somatic growth and reproduction (Jobling, 1996). While somatic growth occurs over a broader thermal range than reproduction, warming beyond species-specific thermal optima generally promotes faster development but smaller adult body sizes due (Ikpewe *et al.* 2021), consistent with the Temperature–Size Rule (Baudron et al., 2014; Lindmark et al., 2022; Ben-Hasan et al., 2024). Increasing water temperature from climate change may impact fisheries productivity by limiting the production of larger, harvestable individuals.

Gonadal development is strongly regulated by temperature, with higher-than-optimal temperatures affecting every stage of reproductive development (Alix et al. 2020). Elevated temperatures can disrupt gonadal development, increasing oocyte atresia and resorption, while delaying, inhibiting or preventing ovulation and spawning once species-specific thermal thresholds are exceeded (Alix et al. 2020). Spawning and early life stages are characterised by relatively narrow thermal windows, making them among the most thermally sensitive periods in the fish life cycle and therefore

particularly vulnerable to warming (Alix et al. 2020). These negative influences of temperature on reproductive performance, may lead to recruitment failure in extreme cases (Jobling 1996).

Temperature is critical during the early life stages, influencing eggs and larval development through regulating embryonic duration, egg survival, size at hatching, developmental rate, pelagic duration and larval survival (Pankhurst and Munday 2011). Elevated temperatures accelerate embryonic development, shortening egg incubation periods and producing smaller larvae at hatching (Blaxter 1991; Korwin-Kossakowski 2008). Warmer temperatures also accelerate larval growth and development, promoting earlier settlement and reducing the time larvae spend in the water column, thereby limiting dispersal (Raventos *et al.* 2021). As larvae are generally more sensitive to environmental fluctuations than adults, temperature-driven changes during early life stages may have important consequences for recruitment and population replenishment (Pankhurst and Munday 2011).

3.4.2 Freshwater flow

Freshwater flow plays a central role in the structure and function of aquatic ecosystems, with important contributions to productivity extending into the marine environment (Broadley *et al.* 2022; Gillanders and Kingsford 2002). In riverine environments and downstream connected estuaries, flowing water is a major hydrodynamic force that shapes the physical environment through erosion, sediment transport and deposition, resulting in the formation of habitats such as deepwater channels, sandbars and mudflats that are colonised and utilised by a diverse array of organisms, including vegetation, invertebrates, fish and birds.

The importance of freshwater reaching the lower reaches of river systems is widely recognised because of its contribution to ecosystem productivity (Brookes *et al.* 2015; Gillanders and Kingsford 2002; Gillson 2011; Ye *et al.* 2016). Freshwater flow plays an instrumental role in transporting nutrients and other materials, including organic matter and zooplankton, from upstream catchments and laterally from floodplains into downstream rivers and estuaries, where these resources are incorporated into aquatic food webs (Bice *et al.* 2016; Brookes *et al.* 2015; Furst *et al.* 2014; Giatas *et al.* 2018; Leterme *et al.* 2018).

Freshwater flow also has important physical effects on estuarine ecosystems. Freshwater inflows moderate salinity gradients and influence hydrodynamic features such as salt-wedge stratification, which facilitates spawning and has important implications for the buoyancy of fish eggs (Booth *et al.* 2011; Hassell *et al.* 2008; Sherwood and Backhouse 1982; Ye *et al.* 2016; Ye *et al.* 2019). Freshwater discharge is also integral to maintaining an open Murray Mouth and supporting hydrological connectivity and water exchange between estuarine and marine environments (Gillanders and Kingsford 2002). Consequently, freshwater availability underpins many of the physical and chemical characteristics of fish habitat, including water temperature, salinity and habitat connectivity.

In freshwater ecosystems, fish have evolved life-history strategies in response to natural patterns of high and low flow (Matthews 1998). Elevated flows, including in-channel flow pulses and floodplain inundation, provide important environmental cues for spawning in some species, such as golden perch, whereas the absence of these flow events may result in spawning failure (Cheshire and Ye 2008). Flooding also increases habitat availability by inundating floodplains and riparian habitats, providing additional spawning substrates for species such as common carp and increasing access to food resources that enhance growth and recruitment (Koehn 2004). Freshwater flow further influences fish movement and dispersal by facilitating spawning migrations and the downstream transport of eggs and larvae through drift (Koehn *et al.* 2020; Koster *et al.* 2017). Conversely, prolonged reductions in freshwater flow during drought can disrupt these processes, reduce habitat connectivity and contribute to population declines or local loss of species, such as Congolli (Zampatti *et al.* 2010).

Within estuaries, freshwater influences fish populations through multiple ecological pathways. Freshwater discharge maintains brackish habitats that are important for the juveniles of many estuarine and marine-estuarine species while also regulating salinity gradients that influence habitat suitability (Gillanders and Kingsford, 2002; Ye et al., 2016). Increased freshwater inflows elevate turbidity, reducing visual predation risk for eggs, larvae and juveniles while potentially enhancing feeding opportunities (Griffiths, 1996; Ferguson et al., 2008). Freshwater also transports nutrients, detritus and zooplankton that increase food availability and support estuarine productivity (Earl et al., 2014; Stewart et al., 2020; Bice et al., 2016). In addition, freshwater discharge generates environmental and chemical cues that facilitate migration, larval orientation and recruitment into estuarine nursery habitats (Crawford, 1984), with stronger year classes often associated with years of elevated freshwater inflow (Ferguson et al., 2008; Stewart et al., 2020).

3.4.3 Salinity

Salinity describes the concentration of dissolved salts in water. In aquatic ecosystems, salinity varies naturally across space and time, forming gradients between freshwater, estuarine and marine environments that are regulated by the balance between freshwater inflows, evaporation, tidal exchange and seawater intrusion (Gibbs *et al.* 2018). Salinity patterns fluctuate over daily, seasonal and annual timescales in response to climatic conditions and hydrological processes (Gibbs *et al.* 2018). In estuaries, freshwater inflows generally reduce salinity and expand brackish habitats, whereas reduced freshwater flow, increased evaporation and sea-level rise promote more saline or hypersaline conditions (Ye *et al.* 2016). Consequently, climate change is expected to alter both the magnitude and spatial distribution of salinity, particularly within estuarine ecosystems where freshwater and marine waters interact (Gillanders *et al.* 2011; Lauchlan and Nagelkerken 2020; Whitfield *et al.* 2023).

Salinity is a major environmental factor influencing fish distributions in estuaries and is largely regulated by the balance between freshwater inflows, marine water exchange and other hydrological processes. As a fundamental characteristic of fish habitat,

salinity influences physiological processes, growth, survival, movement and reproduction across life stages, and consequently plays an important role in determining fish distribution and habitat use. Salinities above ~60 psu represent hypersaline conditions where many euryhaline fishes approach the physiological limits of osmoregulation, resulting in increased physiological stress and progressively reduced habitat suitability, fish diversity and abundance (Esbaugh 2024; Tweedley *et al.* 2019).

Fishes also exhibit preferences for particular salinity ranges that optimise physiological performance, resource availability and life-history processes. For example, greenback flounder (*Rhombosolea tapirina*) utilise a broad range of salinities but are commonly associated with transitional zones where freshwater and marine waters mix, supporting abundant benthic prey (Earl *et al.* 2017). Similarly, Mulloway (*Argyrosomus japonicus*) distribution within estuaries is linked to salinity gradients, with juveniles typically occupying low to intermediate salinities that provide favourable nursery habitat, growth conditions and refuge (Gray and McDonall 1993; Silberschneider and Gray 2008).

Salinity is particularly important for reproduction and early life stages, where specific salinity conditions are required for successful spawning, egg buoyancy, and larval survival. For black bream, spawning occurs within the salt wedge, where intermediate salinities optimise egg buoyancy and larval feeding (Newton 1996; Williams *et al.* 2013). Egg and larval survival is highest within defined salinity ranges, with both low and high salinity extremes reducing hatching success and increasing deformities (Haddy and Pankhurst 2000; Hassell *et al.* 2008). Similarly, for greenback flounder, egg fertilisation and buoyancy are strongly dependent on salinity, with optimal conditions occurring within marine to moderately brackish ranges (Hart 1995).

Previously, salinity has been identified as a major driver affecting the distribution of fish in the Lower Lakes and Coorong region and the subsequence structure of fish assemblages (Bice *et al.* 2018; Brookes *et al.* 2022; Ferguson and Ye 2016; McNeil *et al.* 2013; Noell *et al.* 2009; Zampatti *et al.* 2010). Further research aiming to understand different aspects of the variation in salinity and the distributions/changes in abundance of fish has focussed on studies compiling literature on the physicochemical tolerances and life-history of individual species (Bice 2010; Ye *et al.* 2010), in-situ observations and distributions with salinity (Noell *et al.* 2009) and laboratory assays determining fish physiological tolerances to salinity (McNeil *et al.* 2013; Wedderburn *et al.* 2016; Wedderburn *et al.* 2008). The relationship between salinity and food webs has also been described (Giatas *et al.* 2018). Reductions in freshwater along with increases in temperature and evaporation and in-conjunction with reduced water exchange results in evapo-concentration of salts and rising salinity, in places to hypersaline levels (Brookes *et al.* 2022; Tweedley *et al.* 2019).

3.4.4 Connectivity

Connectivity is a fundamental property of aquatic ecosystems, governing the movement of water, organisms, nutrients, and energy across spatial scales (Crook *et al.* 2015). In riverine, estuarine, and coastal systems, connectivity operates

longitudinally (upstream–downstream), laterally (channel–floodplain), and vertically (surface–groundwater exchanges), as well as between freshwater and marine environments. These linkages underpin key ecological processes such as migration, dispersal, recruitment, and recolonisation following disturbance. For many fish species, particularly diadromous and estuarine-dependent taxa, access to connected habitats is essential to complete life cycles, maintain genetic diversity, and exploit spatially variable resources (Franklin *et al.* 2024; Whitfield *et al.* 2023).

Disruption to connectivity — through barriers, flow regulation, or climate-driven changes such as reduced runoff and sea-level rise — can have profound ecological consequences. Reduced freshwater inflows may limit connectivity among habitats, alter salinity gradients, and disrupt environmental cues for spawning and movement, particularly for flow-dependent or periodic strategists (Bice *et al.* 2023c). Conversely, some changes (e.g. sea-level rise) may enhance connectivity between marine and estuarine systems, potentially facilitating range shifts or the colonisation of new habitats, although often with trade-offs such as loss of freshwater or biogenic habitats. As a result, connectivity is increasingly recognised as a key mediator of ecosystem resilience, influencing how aquatic communities respond to disturbance and climate change (Franklin *et al.* 2024).

Given its central role, incorporating connectivity into climate change vulnerability assessments is critical. It shapes both exposure (e.g. degree of habitat change or fragmentation) and sensitivity (e.g. reliance on connected habitats or migratory pathways) and can modulate adaptive capacity by enabling species to track suitable environmental conditions. In systems such as the Lower Lakes and Coorong, where hydrological variability affect salinity and habitat availability, understanding and maintaining connectivity is essential for sustaining fish populations and ecosystem function under future climate scenarios.

3.5 Fish life history and relationships to environment

There are close evolutionary links between fish biology and the environments that they live in (e.g. Olden and Kennard 2010; Sternberg and Kennard 2013; Thorson *et al.* 2023; Winemiller and Rose 1992). Fish are 'shaped' by their environments, constrained through adaptation and development of strategies (e.g. physiological, behavioural) to capitalise on the diversity of resources available, or to assist in coping with the dynamics of natural environment conditions (Crook *et al.* 2010; Webb 2015).

Anthropogenic or human-induced alterations to the natural environment, particularly those that can impact over large spatial and temporal scales and impose on key life cycle processes, are well-known to have significant ramifications for fish populations (Fausch *et al.* 2002). The degree of change brought about by anthropogenic pressures, can negatively impact upon individuals (e.g. reduced resources, increased environmental harshness) and affect different life-cycle processes (e.g. growth, reproduction, migration, recruitment), interference that ultimately culminates in an overwhelming detrimental impacts at the population level (Ames *et al.* 2020).

The capacity of fish to resist or recover from environmental change reflects their evolutionary history, expressed through behavioural, physiological, and life-history traits (Crook et al., 2010; Blewett et al., 2022). Life-history traits — encompassing reproduction (e.g. fecundity, egg size, timing), migration and dispersal (e.g. residency, diadromy, larval drift), and age and growth (e.g. longevity, body size, age at maturity) — represent adaptations to differing environmental conditions (Chessman 2013).

Trait combinations reflect trade-offs among these adaptations and define distinct ecological strategies, commonly conceptualised along a triangular continuum with three endpoints: Periodic, Opportunistic, and Equilibrium (Winemiller and Rose, 1992; Olden and Kennard, 2010). Fish species are typically classified according to one of these strategies (Bice et al., 2018; Ferguson et al., 2018).

Fish life-history traits provide key insights into environmental requirements and adaptation to variable conditions (Olden and Kennard 2010). They have been widely used to predict biological and ecological responses to environmental change in freshwater systems (Mims and Olden, 2012; Chessman, 2013), although their role in explaining species–environment relationships in estuarine systems is less well described (Teichert *et al.* 2017). In marine environments, life-history traits are increasingly used to identify species and life stages most sensitive to climate-driven change (Pecl et al., 2014; Hare et al., 2016; Moll et al., 2024).

3.6 CLLMM region fishes

Organisms in the CLLMM region exhibit diverse environmental dependencies across freshwater, estuarine, and marine ecosystems, with fishes spanning multiple environments (Lester *et al.* 2011). Fish are ecologically, socially, and economically important in the CLLMM region (Ferguson *et al.* 2018; Sarakinis and Earl 2024; Wedderburn *et al.* 2017; Ye *et al.* 2020). The CLLMM region supports a diverse fish assemblage of marine, freshwater, and estuarine origin, with 104 species recorded overall (93 in the Coorong and 45 in the Lower Lakes) (Bice *et al.* 2018; Wedderburn *et al.* 2016; Ye *et al.* 2016). However, many species are transient (e.g. marine stragglers) or rare (e.g. conservation-significant taxa), with only ~20 species occurring in high abundance. Eight finfish species (four freshwater and four marine–estuarine) are targeted by traditional, recreational, and commercial fisheries, with a similar number (4) contributing prominently to Coorong food webs.

Fishing has long been integral to the Lower Lakes and Coorong (Ferguson *et al.* 2018). For millennia, the Ngarrindjeri people — the Traditional Owners — have depended on these systems for food and cultural identity, harvesting species such as mulloway (*Argyrosomus japonicous*), black bream (*Acanthopagrus butcheri*), yelloweye mullet (*Aldrichetta forsteri*), Murray cod (Pondi; *Maccullochella peeli peeli*), and Pipi (Kuti; Latona (Donax) *deltoides*) using traditional methods (Department of Primary Industries and Regions 2022; Ferguson *et al.* 2018; Hatswell 2025; Sloan 2005). Practices were seasonal, aligned with tidal cycles, and included smoking and drying catches for trade (Ferguson *et al.* 2018). Archaeological evidence further demonstrates the longstanding harvest of freshwater and estuarine fishes across the Lower Murray and

Coorong, highlighting the deep history of Ngarrindjeri fishing and relationships with the aquatic environments of the region (Disspain *et al.* 2017).

Commercial fishing in the Lower Lakes and Coorong emerged in the late 19th and early 20th centuries alongside recreational and Indigenous fisheries. Early operators targeted mulloway, black bream, yelloweye mullet, and Goolwa cockles, adapting to seasonal and environmental conditions (Ferguson et al., 2018). Management frameworks introduced quotas, gear controls, and seasonal closures to maintain sustainability, with ongoing stock assessments informing adaptive management (Department of Primary Industries and Regions 2022; Sarakinis and Earl 2024; Sloan 2005).

Recreational fishing is widespread in the CLLMM region, targeting species such as mulloway, black bream, yelloweye mullet, Australian salmon, cockles, and common carp (Sloan, 2005; Earl, 2019; (Department of Primary Industries and Regions 2022); Beckmann et al., 2023). Alongside commercial and traditional fisheries, it underscores the strong connections between fish populations and human values.

Fish also play ecological roles in food webs, with species: bony herring (*Nematalosa erebi*), sandy sprat (*Hypherlophus vittatus*), congolli (*Pseudaphritis urvillii*) and yelloweye mullet forming key prey for piscivorous fish and birds (Giatas *et al.* 2018; Giatas and Ye 2015; Ye *et al.* 2020), and some fishes are hosts for the larval stage of freshwater mussels (Walker 1981; Wedderburn *et al.* 2026). The region also supports conservationally important fish taxa including: Yarra pygmy perch (*Nannoperca obscura*), Southern pygmy perch (*Nannoperca australis*), Murray hardyhead (*Craterocephalus fluviatilis*), Southern purple-spotted gudgeon (*Mogurnda adspersa*) and River blackfish (*Gadopsis marmoratus*), which are sensitive to a broad range of different environmental impacts, including climate change (Hammer *et al.* 2013).

3.6.1 CLLMM region fish functional groups

Ecological guilds are used to characterise environments where reproduction is undertaken and also where recruitment begins. Thus, it indicates groups of species based on functional habitat requirements (Ferguson *et al.* 2018). In this study, ecological guilds for CLLMM region fish species were adopted from classifications of Ye et al., (2016) and Bice et al., (2018). These classifications used in in this study are as following.

Freshwater straggler

Species found occasionally in estuaries (Bice *et al.* 2018). Ye et al., (2016) further subdivide into freshwater native, exotic and threatened groups. Most freshwater-occupying species are considered 'Freshwater Stragglers' with species origin described in parentheses (Appendix 2).

Freshwater-estuarine opportunist

Species found more frequently within estuaries than Freshwater Stragglers (Bice *et al.* 2018).

Diadromous

Species undertaking migrations between freshwater, estuarine and marine environments. These species can be further subdivided into anadromous and catadromous migrators (Bice *et al.* 2018; Ye *et al.* 2016) with migration type provided in parentheses (Appendix 2) .

Estuarine

Species that complete their entire life cycle within the estuary (Bice *et al.* 2018; Ye *et al.* 2016).

Marine-estuarine opportunist

Species that complete their life cycle within the marine environment, using estuaries as nurseries and for growth, before returning to the sea for reproduction (Bice *et al.* 2018; Ye *et al.* 2016).

Marine straggler

Species that complete their life cycle within the marine environment, but occupies estuaries on occassions (Bice *et al.* 2018; Ye *et al.* 2016).

3.7 Species profiles for climate change assessment

3.7.1 Species profiles

Species profiles are a fundamental component of climate change vulnerability assessments (Egan *et al.* 2020; Hare *et al.* 2016; Pecl *et al.* 2014). As the first step in the assessment, a literature review was undertaken to compile information on species biology, ecology, life-history traits, and environmental requirements relevant to climate vulnerability. This section describes the approach used to develop the species profiles.

Species selection

A literature review of species life-history and environmental requirements, for 11 CLLMM region fish species was undertaken (Appendix 1) with summary tables presented in Appendices 2–5. The literature review include the following species: black bream (*Acanthopagrus butcheri*), greenback flounder (*Rhombosolea tapirina*), mulloway (Argyrosomus japonicus), yelloweye mullet (*Aldrichetta forsteri*), congolli (*Pseudaphritis urvillii*), sandy sprat (*Hyperlophus vittatus*), smallmouth hardyhead (*Atherinosoma microstoma*), bony herring (*Nematalosa erebi*), common carp (*Cyprinus carpio*), redfin perch (*Perca fluviatilis*) and golden perch (*Macquaria ambigua ambigua*). While no formal species selection framework was applied, species included in this project represented a cross-section of CLLMM region fish fauna, encompassing different fish functional groups (ecological guilds), life-history strategy, habitat use, origin (native, exotic), trophic status, or other life-history characteristics (e.g. reproduction, see Appendices 1–5). These species also share important cultural, ecological or commericial values in the CLLMM region. In considering candidates, species with wide geographical coverage (e.g. potential

climate clines) with emperical research data or being utilised for aquaculture (e.g. experimental studies) were considered to provide valuable insight into life-stage environmental requirements (e.g. water temperature, salinity) and conditions arising from climate change.

The literature review involved a search of relevant international and Australian studies across the species distribution. This approach enabled the capture of widest variability in species traits across different populations exposed to environmental factors, which is valuable towards understanding species response. This information was used in conjunction with CLLMM region studies of environmental requirements and descriptions of fish habtiat use, as outline in the summaries of literature by: (Noell *et al.* 2009); (Bice *et al.* 2018; Bice 2010); (2016; Ye *et al.* 2010; 2013) and (Cheshire *et al.* 2013). Compilations of literature on some species (e.g. carp: Koehn 2000; golden perch: Koehn 2020a; redfin perch: Ning et al. 2025; mulloway: (Silberschneider and Gray 2008); black bream: (Norriss *et al.* 2002) were also available.

Life-history strategy for each species was classified according to the 'periodic', 'opportunist' and 'equilibrium' strategies following Winemiller and Rose (Winemiller and Rose 1992). For some CLLMM fish species, life-history strategy has been previously classified (Bice *et al.* 2018; Ferguson *et al.* 2018); these were adopted for this study. Furthermore, those species displaying a mix of different strategy type, were considered as 'Intermediate' following classifications by Ferguson et al. (2018).

Life cycle stages

Environmental impact studies often consider effects of environmental drivers (e.g. temperature) on fish life-history stages (e.g. eggs, larvae, juveniles and adults), to identify biologically sensitive or critical environmental periods (Dahlke *et al.* 2020; Moyano *et al.* 2017). For each species, information on physicochemical parameters for was obtained for key life stages (i.e. egg, larvae, juvenile and adult) and associated life-history processes: survival and growth, spawning and recruitment.

Habitat description

Habitat descriptions for each species and life stage were compiled from textbooks, peer-reviewed scientific papers and research reports. Habitat information was synthesised from studies across multiple spatial-scales, including macro-habitats (e.g. lakes, rivers, estuaries and marine waters), meso-habitats (e.g. pelagic, benthic, littoral zones) and, where available, micro-habitats (e.g. macrophytes, woody debris, substratum).

Supplementary information on environmental processes (e.g. hydrological impacts) expected to be affected climate change have also been doccumented, where reported, to assist in determining fish response to changing environmental conditions. Habitats supporting key life-history processes (e.g. spawning and recruitment), together with the sequence of environments occupied throughout the life cycle, were identified to describe periods and locations where climate change might affect local populations.

Environmental tolerance

Climate change has the potential to shift environmental conditions beyond the tolerance limits of aquatic species, increasing the likelihood of adverse ecological responses (Gillanders *et al.* 2022; Pörtner and Peck 2010). The degree of overlap between species tolerance ranges and projected environmental conditions provides biological context for assessing whether future conditions are likely to remain tolerable or pose greater biological risk. Where projected conditions extend beyond known tolerance limits, survival and persistence may be compromised.

A relative tolerance range was estimated for each species and life stage based on the lowest and highest reported temperature and salinity values. Data were generally available to characterise tolerance ranges associated with survival for most species and life stages. However, these ranges may be less informative for sub-lethal responses, such as growth, reproduction and recruitment, which often have narrower environmental requirements than those associated with survival.

Species environmental tolerance ranges were compiled using different sources of information. This included field observations, laboratory studies, aquaculture studies and published reviews (Table 2). Information on life-history traits and environmental requirements was available from literature reviews (e.g. textbooks, theses, book chapters) or biological studies where measurements were made (e.g. scientific reports, theses, published papers). Given the diversity of study types, methods and available information across life stages and species, compiling values from multiple literature sources provided the most practical means of estimating relative environmental tolerance.

Table 2. Study types for source information on egg, larvae, juvenile and adult life-stages for environmental tolerance.

STUDY KIND	FIELD OBSERVATION	AQUACULTURE TRIAL	LABORATORY EXPERIMENT	LITERATURE REVIEW
Black bream	✓	X	✓	✓
Bony herring	✓	X	X	✓
Common carp	✓	✓	✓	✓
Congolli	✓	X	✓	✓
Golden perch	✓	X	✓	✓
Greenback flounder	✓	✓	✓	✓
Mulloway	✓	✓	✓	✓
Redfin perch	✓	✓	✓	✓
Sandy sprat	✓	X	X	✓
Small mouth hardyhead	✓	X	✓	✓
Yelloweye mullet	✓	✓	✓	✓

Searches of fish species information extended globally. This was especially the case for introduced fish species (e.g. redfin perch), where much more is published on the biology and ecology in the northern hemisphere than Australian waters. Similarly, literature searches of Australian endemic species also included information broadly

across the species distribution. Incorporating evidence from diverse sources and locations where the species is known to be present, meant that for some traits like environmental tolerance, the widest estimate of range was potentially encountered.

Experiments and aquaculture trials

Experiments and aquaculture trials involving the manipulation of environmental parameters under controlled conditions, provided the strongest evidence of tolerance limits, and therefore elicidated the best estimate tolerance width. These studies were particularly valuable for identifying upper and lower lethal thresholds, although many focused on a single endpoint relevant to the study objectives (e.g. upper salinity limits, see McNeil *et al.* 2013). Estimates of tolerance range for experimental studies may also be influenced by acclimation history and exposure method, with gradual acclimation potentially resulting in greater tolerance than abrupt transfer between treatment conditions (Desforges *et al.* 2023).

Aquaculture studies complemented experimental data by identifying environmental conditions supporting growth, survival and early development. Although not designed to determine lethal thresholds, these studies provided valuable information on the environmental requirements of early life stages and optimum conditions for juvenile development.

Field observations and reviews of literature

Field observations provided information where experimental evidence was unavailable, particularly for poorly studied species or life stages. Records of species occurrence under seasonal or environmental conditions, together with descriptions of migration, locations of capture or observation of fish kills, were used to infer environmental tolerance where appropriate. In some cases, studies reporting fish kill events provided estimates of when extreme environmental conditions exceeded tolerance exceeded limits under (e.g. Hoeksema et al. 2006).

Literature reviews, including peer-review reports and textbooks were useful for summaries of environmental tolerance information from multiple sources.

Knowledge gaps

Knowledge gaps for environmental tolerance were noted across species, life stages, and key population processes. knowledge gaps were apparent for water temperature and salinity; where information was commonly: (i) absent entirely for a life stage or key life-history process, (ii) unavailable from experimental or tolerance studies but supported by field observations, or (iii) available for only a single life stage.

Where knowledge was absent, several approaches were adopted to provide estimates for the sensitivity assessment (Section 4). These included the substitution of widely accepted reference values (e.g. 35 psu for sea salinity), biologically reasonable assumptions – e.g. adopting a lower limit 0 psu for freshwater species; 35 psu for spawning and recruitment occuring in the sea; and, 2 mg/L as lower DO tolerance limit for all species. Expert opinion was also used where required, inline with applied recommendations used in other climate change vulnerability assessments (e.g.

Morrison et al., 2015). To account for variation in data quality and scoring consistency, a three-level confidence score based on Gonzalez et al., (2011) was assigned to each sensitivity metric. Further details on scoring confidence are provided in Section 4.

Where trait information, such as environmental tolerance was only available for a single life-stage (e.g. larvae) and absent for others (e.g. eggs), but other life-stages that may share locations or times with life-stages with known tolerance; then, the assumption of broadly similar environmental requirements was applied. This was considered separately for eggs and larval stages and separately for juveniles and adults.

Summary remarks

Several observations emerged from the species literature review. Experimental studies, literature reviews and field observations were the main sources of information on environmental tolerances. Although relatively few studies involved aquaculture trials, these provided useful estimates of tolerance ranges for early life stages and juveniles of some species (e.g. mulloway and greenback flounder).

Empirical data enabled environmental tolerance ranges to be constructed for species and life stages, although the strength of this information varied. Field observations provided limited information on how close the highest recorded values were to levels associated with mortality (e.g. LD_{50}). Without experimental data, it therefore remains uncertain whether these values approached physiological limits.

Environmental tolerances were also commonly examined in isolation, despite species being exposed to multiple environmental conditions simultaneously. Some experimental studies were exceptions, including McNeil et al. (2013), which examined the combined effects of temperature and salinity on tolerance. Such information is particularly valuable when integrated with environmental modelling (e.g. Brookes et al., 2022), as it provides a stronger basis for assessing how projected changes in multiple environmental conditions may affect species.

Environmental tolerance information was uneven across life stages and environmental parameters. Salinity tolerance was generally better characterised than temperature tolerance, particularly for juveniles and adults. For temperature, multiple values were often available to estimate upper limits at the species level, but information specific to individual life stages was limited. Much less was known about eggs and larvae for both parameters, with some estimates undefined or reliant on expert opinion. Dissolved oxygen and pH tolerances were also poorly documented for most species and life stages. These represent important knowledge gaps for understanding fish responses to climate change.

The limited information for early life stages is particularly important because these stages may be more environmentally vulnerable. Juveniles and adults were often reported to have broader environmental tolerances associated with the development of physiological functions, including osmoregulation. Greater knowledge of environmental tolerances during egg and larval stages would therefore improve understanding of potential climate impacts on recruitment.

The availability of information also differed among species. Larger, particularly commercially valued species had more readily available life-history information than small-bodied species with important ecological roles, including contributions to food webs (e.g. smallmouth hardyhead and sandy sprat). Even for smallmouth hardyhead, for which high salinity tolerance is relatively well understood and may confer some resilience, responses to freshwater flow and the influence of salinity on reproduction and early life stages remain poorly understood. Consequently, there may be unrealised benefits of freshwater flow for small-bodied species.

Some broader ecological patterns were also apparent among functional groups. Freshwater species generally appeared to have narrower salinity tolerance ranges but broader temperature ranges, whereas marine–estuarine opportunists and estuarine species displayed broader salinity tolerances. Diadromous, marine–estuarine opportunist, freshwater and estuarine species also displayed migration, redistribution or dispersal within and between aquatic environments. Movement in many species was associated with freshwater flow events, demonstrating the importance of connectivity both within the CLLMM and with adjacent environments.

Many species also showed associations with warmer seasonal periods for reproduction or recruitment, or occurred across broad geographical distributions. These characteristics may indicate some capacity to accommodate increasing temperatures, although the extent to which they translate into resilience to future warming remains uncertain. Marine–estuarine opportunists, diadromous and freshwater species also showed partial or complete dependence on environments outside the CLLMM, including the Southern Ocean and River Murray, for reproduction or recruitment.

4 Assessing fishes vulnerability to climate change

4.1 Introduction to climate change vulnerability assessments

Vulnerability assessments provide a structured approach to evaluate the potential influence of climate change on species, populations and ecosystems. They are useful for identifying risk from climate change impacts (Foden *et al.* 2013; Hare *et al.* 2016; Pecl *et al.* 2014; Williams *et al.* 2008), and are considered a practical approach where information is limited (Pacifici *et al.* 2015; Pecl *et al.* 2014). In the absence of robust stock and ecosystem projections, a risk-based approach to identifying emergent issues for policy and management provides a pragmatic first step in preparing fisheries for climate change adaptation (Fulton *et al.* 2020; Pecl *et al.* 2014).

There are different ways climate change vulnerability assessments can be undertaken (Pacifici *et al.* 2015). Trait-based vulnerability assessment are common applied to evaluate climate change impacts (Hare *et al.* 2016; Pacifici *et al.* 2015; Pecl *et al.* 2014), because life-history traits are often better predictors of vulnerability to climate change than taxonomy or distribution alone (Morrison *et al.* 2015; Pacifici *et al.* 2015). Traits associated with increased vulnerability include limited dispersal ability, slow reproductive rates, specialised habitat and dietary requirements, restricted distributions and narrow physiological tolerances (Hare *et al.* 2016; Morrison *et al.* 2015; Pacifici *et al.* 2015). Trait-based vulnerability assessments have been applied previously to both fish and fisheries studies of climate change assessment (e.g. Fulton *et al.* 2020; Hare *et al.* 2016; Pecl *et al.* 2014).

Climate change vulnerability assessments typically comprise of sensitivity and exposure components (Morrison *et al.* 2015; Pecl *et al.* 2014). Sensitivity reflects the biological traits that influence intrinsic resilience to climate change (Hare *et al.* 2016; Morrison *et al.* 2015; Pecl *et al.* 2014), while exposure considers the overlap between species distributions and the magnitude of projected environmental change (Morrison *et al.* 2015). A third component, adaptative capacity, assesses biological responses that may mitigate sensitivity or exposure (e.g. movement capacity). As many biological attributes contribute to both sensitivity and adaptive capacity, we followed (Morrison *et al.* 2015), by identifying an assessment of sensitivity also captures adaptive capacity.

In this section, a trait-based vulnerability assessment was undertaken for eleven fish species distributed across the CLLMM region (Appendix 1). The assessment species list included: black bream (*Acanthopagrus butcheri*), greenback flounder (*Rhombosolea tapirina*), mulloway (Argyrosomus japonicus), yelloweye mullet (*Aldrichetta forsteri*), congolli (*Pseudaphritis urvillii*), sandy sprat (*Hyperlophus vittatus*), smallmouth hardyhead (*Atherinosoma microstoma*), bony herring (*Nematalosa erebi*), common carp (*Cyprinus carpio*), redfin perch (*Perca fluviatilis*) and golden perch (*Macquaria ambigua ambigua*).

4.2 Method: sensitivity, exposure and vulnerability assessment

4.2.1 Sensitivity

Fulton et al. (2020) outlined a climate change vulnerability assessment framework for Australian fisheries, building on earlier approaches developed by Pecl et al. (2014) and Morrison et al. (2015). Although developed within a fisheries context, the framework was designed to be flexible and scalable, allowing its application to be adapted to different assessment objectives and taxa (Fulton et al., 2020). This adaptability is reflected in the broad consistency of sensitivity metrics used across climate change vulnerability assessments of marine, freshwater, and estuarine fishes. For example, fish life-history traits that are incorporated into marine fisheries species sensitivity assessments (e.g. age at maturity, fecundity, movement), have also included in climate change vulnerability assessments of freshwater or estuarine fish species (Chessman 2013; (Egan *et al.* 2020); Hare, 2016).

For this project, the sensitivity component of the Fulton et al. (2020) framework was modified to place greater emphasis on key environmental drivers relevant to the CLLMM region, including temperature, salinity and freshwater flows, and their influence across different life-history stages and processes (e.g. reproduction). Only sensitivity metrics relating to 'Abundance', 'Distribution' and 'Phenology' were included, as limited information was available for the 'Physiology' component. However, the 'oxygen sensitivity' metric was retained under 'Distribution' due to its relevance as a defining environmental constraint. For each species, a total of 18 sensitivity metrics were assessed (Table 3).

4.2.2 Exposure

Exposure is defined as a measure of the projected change in the environment due to climate change (Hare *et al.* 2016). Exposure assesses the extent to which a species' environment will change (Foden et al., 2013), and an important determinant is the degree of 'overlap' between species current and future environment occupancy. Climate change vulnerability studies may infer vulnerability from projected changes in species distribution across large-spatial scales (e.g. Egan *et al.* 2020). In contrast, the small spatial scale of this study, together with the dependence of some species on specific environments (e.g. estuaries) and environmental drivers assessed (e.g. salinity), required a greater understanding of species environmental tolerance to assess occupancy overlap with future environmental conditions.

In assessing exposure of envrionmental drivers — temperature, freshwater flow, salinity and connectivity — fourteen exposure attributes were assessed (Table 4.). Different kinds of information was accessible and used to assess exposure (Appendix 11). Typically, for single-component environmental drivers like temperature and freshwater flow, scoring was based on estimated values obtained from climate change projections (e.g. air temperature, runoff) as per previous vulnerability assessments (e.g. Egan *et al.* 2020; Hare *et al.* 2016; Morrison *et al.* 2015). Other environmental drivers such as salinity and connectivity were composite, derived from values of multiple factors (e.g. freshwater flow and sea-level rise). For these drivers, it was necessary to

employ hydrodynamic modelling to project future environmental conditions (see Dang et al., 2026).

4.2.3 Climate change scenarios

Climate change scenarios were developed separately for The Lower Lakes and Coorong. These scenarios described the projected environmental conditions at the 2050- and 2090-time horizons and formed the basis for assessing exposure in accordance with the Rees et al. (2022) model (Section 3.3). Four climate change assessment scenarios were evaluated: 2050 Lower Lakes, 2050 Coorong, 2090 Lower Lakes and 2090 Coorong. Within each scenario, exposure was assessed for four environmental drivers — temperature, freshwater flow, salinity and connectivity.

Considerable hydroclimatic variability is projected for both the 2050- and 2090-time horizons, reflecting the occurrence of both relatively dry and relatively wet future conditions across the climate model ensemble (Chiew *et al.* 2025; Zhang *et al.* 2020). These plausible future hydroclimatic conditions are captured by the range of ensemble runoff projections (e.g. 10th and 90th percentiles), corresponding to relatively drier and wetter future climates, respectively (Chiew *et al.* 2025). Given that freshwater inflow influences salinity, connectivity, and other environmental conditions, exposure included scoring under 'lower' and 'higher' freshwater inflow conditions.

Driver exposure scores for higher and lower freshwater inflow conditions are presented in table and figures (see Section 4.3.2.). However, for the purpose of calculating vulnerability for each scenario, driver exposure scores are averaged across lower and higher freshwater inflow levels.

Different modelling outputs were used to represent comparable lower and higher freshwater inflow conditions across environmental drivers. For temperature and freshwater flow, lower freshwater inflow was represented by the 10th percentile ensemble runoff projection, where as salinity and connectivity used the Extreme Dry flow state. Higher freshwater inflow was represented by the median (50th percentile) ensemble runoff projection for temperature and freshwater flow and the Dry flow state for salinity and connectivity.

Seasonal projections for runoff were used to assess freshwater flow exposure, as they were considered more informative (Appendix 7). Temperature exposure was assessed using both seasonal and annual projections because they represent different aspects of thermal change (Appendix 6). Air temperature projections were interpreted alongside modelling outputs for seasonal water temperature (Dang et al., 2026; Figure 32) and water depth (Dang et al., 2026; Table 7).

Salinity and connectivity were assessed using hydrodynamic model outputs for the Dry and Extreme Dry flow states within the 2050 and 2090 climate change scenarios (Dang et al., 2026). Connectivity assessments were also informed by projected annual runoff volumes (Appendix 8), Coorong water levels (Appendix 10), and modelled water depths (Dang et al., 2026).

Table 3. Sensitivity metrics and scoring criteria used to characterise climate change sensitivity of the eleven assessed fish species in the CLLMM region. Sensitivity was classified as Low (1), Moderate (2) or High (3). Note: final scores subject to confidence weighting.

METRIC	SENSITIVITY METRIC	ABBREVIATION	LOW SCORE	MEDIUM SCORE	HIGH SCORE
Abundance	Fecundity	FEC	> 20,000 eggs/year	100–20,000 eggs/year	< 100 eggs/year
	Recruitment period	REC	Consistent every 1–2 years	Occasional and variable	Highly Episodic
	Average Age at Maturity	AAM	< 2 years	2–4 years	> 4 years
	Generalist versus Specialist Habitat	GSH	No reliance on estuary or sea	Preference for estuary	Reliant on estuary or freshwater
	Generalist versus Specialist (Prey)	GSP	Large diversity of prey consumed	Limited number of prey types consumed	Dependent on a major prey type
Distribution	Larval Dispersal or Larval Duration	LDD	> 2 months (or > 100 km)	2–8 weeks (or 10–100 km)	< 2 weeks (or < 10 km)
	Juvenile/Adult Movement	JAM	> 100 km	10–100 km	< 10 km
	Eggs/Larvae Temperature range	ELT	Warm, majority of tolerance > 15 °C	Equal, spread of tolerance above and below 15 °C	Cool, majority of tolerance < 15 °C
	Juvenile/Adult Temperature upper limit	JAT	> 35 °C	31–34 °C	26–30 °C
	Eggs/Larvae Salinity range	ELS	Narrow and high/stable (26–40psu) or high (> 40 psu)	Broad, spans low–high and varying tolerance (0–40 psu)	Narrow and low or exposed to increase (0–20 psu)
	Juvenile tolerance to Rising Salinity	JRS	Widest tolerance (0–60 psu)	Wide tolerance (0–40 psu)	Narrow tolerance (0–20 psu)
	Juvenile/Adult tolerance to Hypersalinity	JAH	> 60–120 psu	40–60 psu	< 40 psu
	Local availability of Alternative Habitat	LAH	Substantial (multiple patch types available)	Limited (at least one other patch type available)	No other patches available

	Oxygen Sensitivity	OXS	< 2 mg/L	2–5 mg/L	> 5 mg/L
Phenology	Environmental Variable (excluding FW) as Reproductive cue	EVC	Weak correlation to environmental variable	Moderate correlation to environmental variable	Strong correlation to environmental variable
	Importance of FW to complete Life-history	FWL	Not important	FW identified for at least 1 aspect of life history	FW involved in 2 or more aspects of life history
	Temporal mismatch of life-cycle events	TML	> 4 months	2–4 months	< 2 months
	Migration	MIG	No migration	Migration known for part of population	Migration known ore common for whole population

4.2.4 Scoring

Scoring for both sensitivity and exposure followed a three-level scoring system (3, 2 and 1), where a score of 3 represented the highest sensitivity or exposure to climate change and a score of 1 represented the lowest. Exposure scores were assigned based on the projected magnitude of environmental change between the baseline and future time horizon for each environmental driver (e.g. runoff, temperature, salinity and water level).

Two complementary approaches were used to guide exposure scoring. The primary approach assessed whether projected environmental conditions approached or exceeded predefined ecological or management-relevant thresholds (Table 4; Appendix 11). These thresholds represented nominal environmental conditions considered likely to influence fish populations and were used as the principal basis for assigning exposure scores.

Where suitable baseline and projected climate data were available, a second approach was used to support scoring by evaluating the magnitude and uncertainty of projected environmental change relative to historical variability. This involved comparing the projected ensemble median (50th percentile) and projected model uncertainty (10th–90th percentile range) with the historical baseline variability (10th–90th percentile range). Comparing projected conditions with the historical range provides an indication of both the magnitude of projected environmental change and the confidence that future conditions depart from historical variability. The following criteria were used to guide exposure scoring:

- Score 1: The projected ensemble median (50th percentile) falls within the historical range of variability (baseline 10th–90th percentile range).
- Score 2: The projected ensemble median lies outside the historical range of variability, but the projected uncertainty (10th–90th percentile range) overlaps the baseline 10th–90th percentile range.
- Score 3: Both the projected ensemble median and projected uncertainty (10th–90th percentile range) lie outside the historical range of variability (baseline 10th–90th percentile range).

Exposure scoring assumed continuation of the current barrage operating objectives described by O'Connor et al. (2015). These assumptions provided the management context for interpreting future environmental conditions and included:

- Maintenance of Lower Lake water levels at 0.0 m AHD for 100% of the time,
- Maintenance of Lower Lake water levels at 0.4 m AHD for 95% of the time,
- Annual average barrage outflow of 4,000 GL/year over a three-year period, with no less than 3,150 GL/year in any individual year, to maintain Lower Lake water levels between 0.40 and 0.83 m AHD.

Table 4. Environmental driver metrics and scoring criteria used to assess climate change exposure in the CLLMM region. Exposure was classified as Low (1), Moderate (2) or High (3). Note: final exposure scores subject confidence weighting.

DRIVER	EXPOSURE METRIC	LOW SCORE	MEDIUM SCORE	HIGH SCORE
Temperature	Annual average daily maximum air temperature (°C)	< 25 °C	25 °C–30 °C	> 30 °C
	Seasonal average daily mean air temperature anomaly (°C)	< 2 °C	2–3 °C	> 3 °C
	Annual days when maximum daily air temperature 40 °C	< 10 days	10–20 days	> 20 days
Freshwater Flow	Summer projected runoff (% change)	< 15%	15–30%	> 30%
	Autumn projected runoff (% change)	< 15%	15–30%	> 30%
	Winter projected runoff (% change)	< 15%	15–30%	> 30%
	Spring projected runoff (% change)	< 15%	15–30%	> 30%
Salinity	Salinity (0–20 psu) (% relative change)	< 35%	35–70%	> 70%
	Salinity (20.1–40 psu) (% relative change)	< 35%	35–70%	> 70%
	Salinity (40.1–60 psu) (% relative change)	< 35%	35–70%	> 70%
Connectivity	Within waterbody connectivity (0.5 m Minimum waterbody depth)	Adequate: water level > 0.5 m AHD across Lower Lakes or Coorong	Sufficient: water level 0.5 m AHD across Lower Lakes or Coorong	Reduced: water level < 0.5 m AHD across Lower Lakes or Coorong
	Fishway connectivity (Lower Lakes ≥0.4 m AHD, 52 GL/year, head difference among the Lower Lakes and Coorong)	Projected annual runoff volume (> 4000 GL/year) and positve head difference among The Lower Lakes and Coorong	Projected annual runoff volume (2000–4000 GL/year) with positive or, reverse head difference among The Lower Lakes and Coorong	Projected annual runoff volume (< 2000 GL/year) or little-to-no head difference among The Lower Lakes and Coorong
	Barrage spillways inundation from rising sea-level (0.83 m AHD)	Projected median Coorong water level < 0.83 m AHD	Projected median Coorong water level 0.83 m AHD	Projected median Coorong median water level > 0.83 m AHD
	Murray Mouth maintenance (730–1090 GL/year)	Projected annual runoff volume (> 4000 GL/year)	Projected annual runoff volume (2000–4000 GL/year)	Projected annual runoff volume (< 2000 GL/year)

- Annual average barrage outflow of 2,000 GL/year over a three-year period, with no less than 650 GL/year in any individual year, to maintain Lower Lake water levels between 0.40 and 0.75 m AHD,

- Provision of at least 3,150 GL/year in one year, with an average of 4,000 GL/year over three years, to achieve fishway operation and Murray Mouth maintenance objectives,
- Provision of 52 GL/year to maintain fishway operation, and
- Provision of 730–1,090 GL/year, or dredging where necessary, to maintain an open Murray Mouth.

Additional hydrological information was also used to assist interpretation of exposure scores, including:

- Reverse head flow occurring when water levels in the Coorong exceed those of the Lower Lakes (> 0.4 m AHD),
- Disconnection between Lake Alexandrina and Lake Albert occurring at −0.5 m AHD,
- Mean water depths of ~1.2 m and ~1.4 m in the North and South lagoons, respectively, when Lower Lake water levels are 0.0 m AHD,
- Average annual evaporation from the Lower Lakes of 800 GL/year (Heneker 2010),
- Average annual consumptive use from the Lower Lakes of 53 GL/year (Heneker 2010), and
- Sea-level ingress to the Lower Lakes via spillway inundation when water levels reach ~0.83 m AHD.

4.2.5 Assessing uncertainty

Climate change vulnerability assessments commonly incorporate a measure of uncertainty to reflect confidence in the information used to assign sensitivity and exposure scores (Egan *et al.* 2020; Hare *et al.* 2016). In this study, confidence was assessed using the three-category system of Gonzalez et al., (2011), where information was classified as having high, medium or low confidence based on its quality, quantity and applicability (Table 5).

For sensitivity, confidence scoring considered the availability and relevance of biological information for each species and life stage. Where empirical information was limited (e.g. physiological tolerances for particular life stages), expert judgement was used to inform scoring and confidence was adjusted according to the quality and applicability of the available information. Consideration was also given to the applicability of biological information derived from other geographic regions to species occurring within the CLLMM. Exposure confidence similarly reflected the quality and applicability of the climate projections and environmental datasets used to assign exposure scores.

Similar to other climate change vulnerability assessments (e.g. (Egan *et al.* 2020; Hare *et al.* 2016), sensitivity and exposure scores were weighted according to confidence. Confidence scores were divided by three to produce weighting factors of 1.00, 0.67 and 0.33 for high, medium and low confidence, respectively. These weightings were multiplied by the corresponding sensitivity or exposure attribute score. For example, an attribute score of 3 was retained as 3 under high confidence, reduced to 2 under medium confidence and reduced to 1 under low confidence. This down-weighting

limited the contribution of less certain information to the vulnerability score, while giving greater influence to scores supported by higher-confidence information.

Table 5. Confidence criteria used to score the quality and applicability of information for sensitivity and exposure assessments.

VALUE	SCORE	CRITERIA
High	3	Published information, local or regional information available, experts confirm high certainty
Medium	2	Published information, local or regional information available, some conflicting/inconclusive conclusions, experts confirm some certainty
Low	1	No published information and perception based, inconclusive/conflicting, experts confirm uncertainty

4.2.6 Vulnerability assessment

Scoring of sensitivity and exposure was undertaken separately. Sensitivity was assessed first during a workshop held at SARDI (West Beach) on the 13th of February 2026. Participants included SARDI scientists familiar with fish biology and ecology, hydrology of the Lower Lakes and Coorong, the Lakes and Coorong Fishery, climate change and climate change vulnerability assessments.

Climate change vulnerability assessments commonly involve multiple experts independently scoring attributes over successive rounds, with discussion between rounds to refine scores and improve agreement (Morrison et al., 2015). In contrast, this assessment adopted a collective consensus-based approach, whereby a single sensitivity score and confidence score were agreed upon by the group for each attribute of each species.

The collective scoring approach facilitated discussion before scores were assigned, particularly regarding the interpretation and applicability of biological information from other geographic regions to local populations. To familiarise participants with the sensitivity attributes and scoring framework, assessments commenced with an information-rich species, allowing each attribute and its associated scoring criteria to be systematically discussed before progressing to the remaining species. Once participants were familiar with the assessment framework, scoring proceeded sequentially by sensitivity attribute across all species, improving efficiency while promoting consistency in the application of the scoring criteria. Earlier scores were revisited where necessary to ensure consistent interpretation and application of the scoring framework throughout the workshop.

Exposure scoring was undertaken over several days by two personnel who systematically evaluated the 14 exposure attributes across the four environmental drivers. Unlike sensitivity, exposure scoring primarily involved interpretation of quantitative climate projections and environmental modelling outputs, allowing scores to be informed directly by the available evidence. Scoring was initially completed using the best available information at the time and subsequently updated to incorporate newly available hydrodynamic modelling outputs. The

rationale for each exposure score and its associated confidence score was discussed, taking into consideration the quality, applicability and uncertainty of the available information.

4.2.7 Vulnerability scores

Species vulnerability was calculated as the product of sensitivity and exposure scores after both components had been weighted by their respective confidence scores (Morrison et al., 2015). Because the assessment considered four climate change scenarios, each evaluated under lower and higher freshwater inflow conditions, multiple exposure scores contributed to the final vulnerability assessment. Tables 8 and 9 present the exposure scores across these scenarios and freshwater inflow conditions.

Calculation of vulnerability involved three steps. First, the weighted scores for the 18 sensitivity attributes were averaged to produce a single sensitivity score for each species. Second, weighted scores for the 14 exposure attributes were averaged separately for lower and higher freshwater inflow conditions within each of the four climate change scenarios, producing eight scenario-specific exposure scores for each species. Third, vulnerability was calculated by multiplying the average sensitivity score by each corresponding scenario-specific exposure score.

Average sensitivity, exposure and vulnerability scores are presented to one decimal place in the Results. However, to assign vulnerability categories consistent with the three-level framework of Hare et al. (2016), average sensitivity and exposure scores were rounded to the nearest whole number before multiplication. This produced vulnerability scores ranging from 1 (lowest vulnerability) to 9 (highest vulnerability).

Climate change vulnerability assessments have adopted both three-level (low, moderate and high) and four-level (low, moderate, high and very high) classification systems (Morrison et al., 2015). Consistent with Hare et al. (2016), this assessment adopted a three-level classification, where vulnerability scores ranged from 1 to 9, with higher scores indicating greater climate change vulnerability.

4.3 Results of vulnerability assessment

4.3.1 Sensitivity

Sensitivity scores from 18 sensitivitity attributes recorded for all eleven species are summerised in Table 6. Average sensitivity scores was similar between the Lower Lakes species (1.6) and Coorong species (1.5). In the Lower Lakes, golden perch and redfin perch exhibited the highest senstivity scores, whereas greenback flounder, black bream and mulloway were the most sensitive Coorong species. Overall, golden perch had the highest sensitivity with a score of 2.0. Yelloweye mullet and smallmouth hardyhead shared the lowest sensitivity score of 1.2 (Table 6).

Table 6. Average weighted sensitivity scores for the eleven assessed species, calculated from 18 sensitivity attributes. *Average group score calculated excluding congolli.

LOWER LAKES		COORONG	
SPECIES	SCORE	SPECIES	SCORE
Golden perch	2.0	Greenback Flounder	1.8
Redfin perch	1.7	Black bream	1.8
Common carp	1.6	Mulloway	1.7
Congolli	1.5	Congolli	1.5
Bony Herring	1.4	Sandy sprat	1.3
		Smallmouth hardyhead	1.2
		Yelloweye mullet	1.2
Average	1.6	*Average	1.5

Lower Lakes species sensitivity

The distribution of weighted scores for all sensitivity metrics across the Lower Lakes species is presented in Figure 3. High sensitivity scores (2 and 3) were distributed across all Lower Lakes species, although redfin perch and golden perch exhibited the greatest proportion of high-scoring sensitivity attributes (Figure 3).

High scores

For all Lower Lakes species, the GSH metric received the maximum score, reflecting reliance on freshwater lake habitats and ecosystems. High sensitivity for ELS was also common, with all species except congolli receiving high scores. Maximum JRS scores were additionally recorded for redfin perch, common carp and bony herring. Golden perch and congolli also scored highly for MIG, reflecting dependence on migration to complete their reproductive strategy, while golden perch alone received the maximum score for FWL (Figure 3).

Moderate scores

Moderate sensitivity scores were most commonly associated with life-history traits. AAM received moderate scores for all species except bony herring, while LDD was moderate for golden perch, common carp and bony herring. EVC was also of moderate sensitivity for golden perch, redfin perch and common carp. FWL received moderate scores for all species except redfin perch, whereas GSP was moderate only for golden perch and redfin perch.

Low scores

Low sensitivity scores were most frequently associated with thermal tolerance and dispersal-related attributes. JAT, ELT, JAM, OXS and LAH generally received low scores across species, with the exception of LAH for golden perch and congolli. FEC was also assigned low sensitivity for all species except bony herring.

Coorong species sensitivity

The distribution of weighted sensitivity scores across the 18 sensitivity attributes for Coorong fish species is presented in Figure 4. Unlike Lower Lakes species, high sensitivity scores (2 and 3) were concentrated primarily among the three most sensitive species: greenback flounder, black bream and mulloway (Figure 4).

High scores

High sensitivity scores were concentrated among relatively few attributes and species (Figure 4). MIG received the maximum score for both mulloway and greenback flounder, reflecting reliance on migration to complete their life histories. Mulloway also received the highest score for AAM, while greenback flounder received the highest score for ELT. Black bream was assigned maximum scores for EVC and FWL. Smallmouth hardyhead was the only species to receive high sensitivity scores for GSH and FEC, the latter representing the highest fecundity sensitivity among all assessed species.

Moderate scores

Moderate sensitivity scores were distributed across a broader range of species and attributes. REC and GSH were assigned moderate scores for black bream, greenback flounder and mulloway, while AAM was also moderate for yelloweye mullet. Moderate scores for JAM were assigned to greenback flounder, black bream and sandy sprat. Black bream and mulloway also received moderate scores for JAH and TML. LDD was assigned a moderate score for all species except sandy sprat. Most species received low scores for GSP, with only mulloway and sandy sprat assigned moderate sensitivity.

Low scores

Low sensitivity scores were common across several physiological and life-history attributes. JRS, ELS and JAH generally received lower scores than those observed for Lower Lakes species, reflecting the broader salinity tolerances of Coorong fishes. Low scores were also common for OXS, FEC (except Smallmouth hardyhead), and FWL, indicating reduced reliance on freshwater compared with Lower Lakes species.

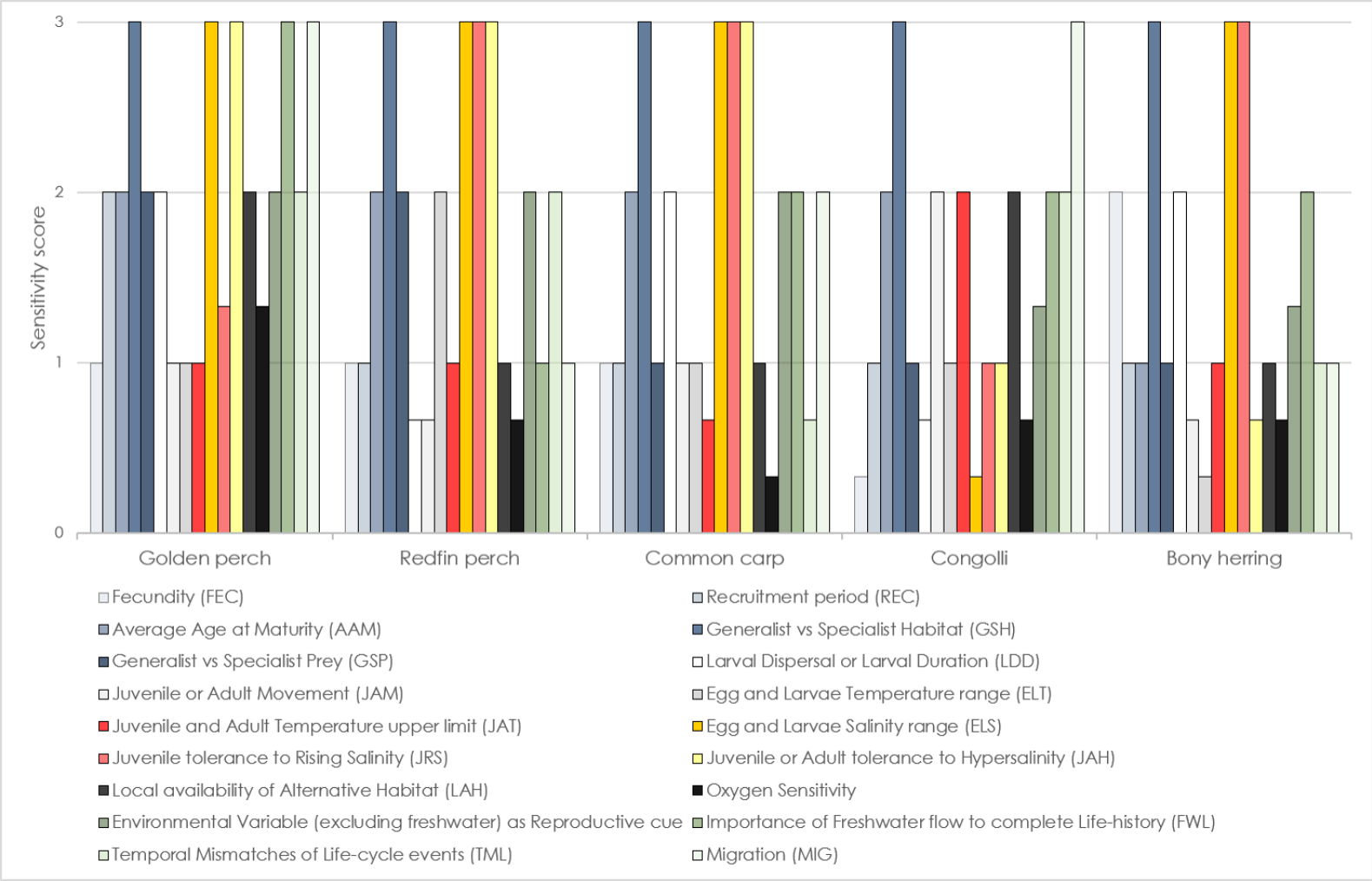


Figure 4. Weighted sensitivity scores for 18 attributes for each of five species of fish inhabiting the Lower Lakes.

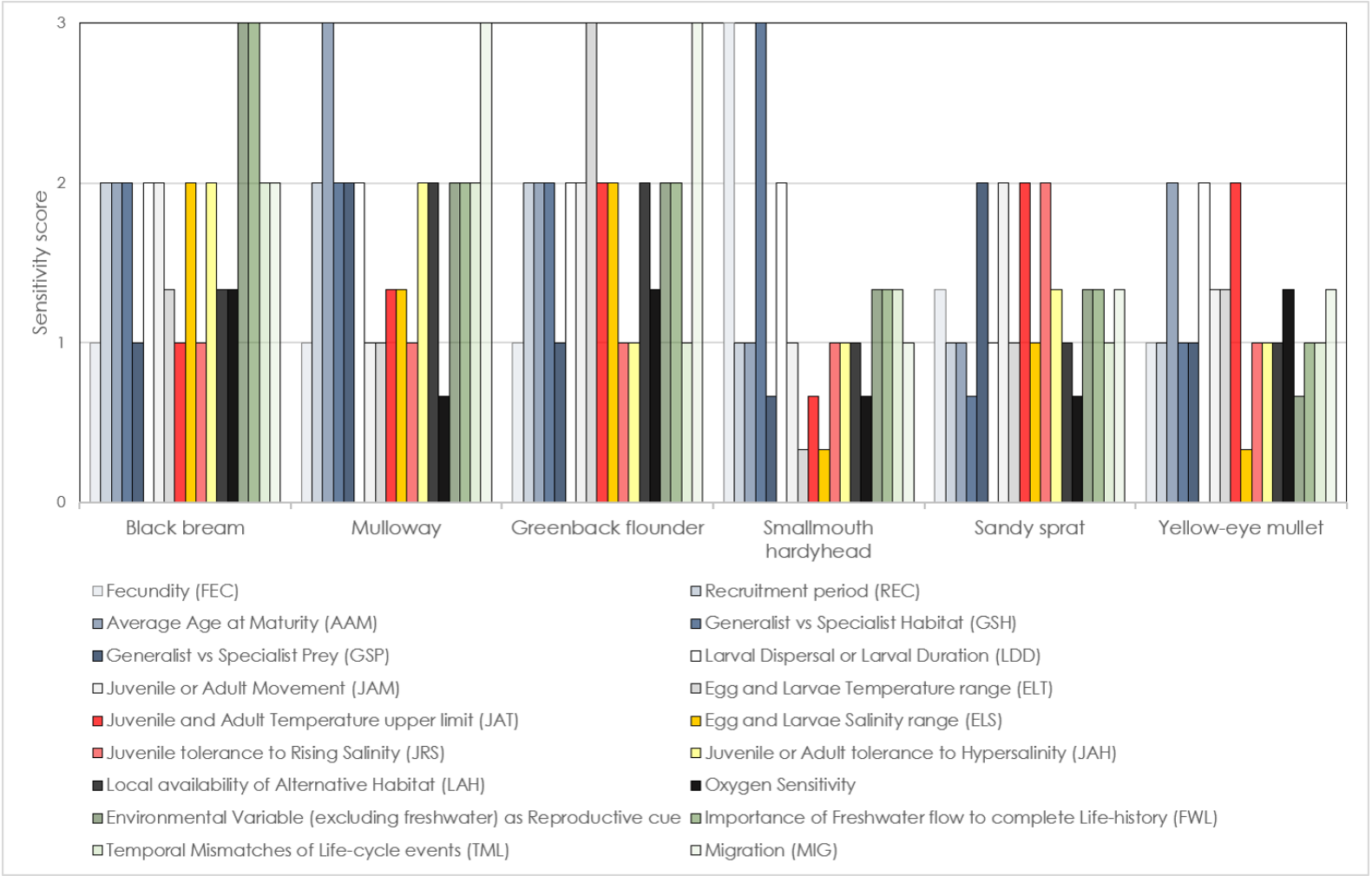


Figure 5. Weighted sensitivity scores for 18 attributes for each of five species of fish inhabiting the Coorong.

4.3.2 Exposure

Average exposure scores were lower for Lower Lakes than the Coorong under both climate change time horizons and at both freshwater inflow levels (Table 7). Exposure increased from 2050 to 2090 in both regions, with average scores increasing by 0.4 units (31%) for Lower Lakes and 0.4 units (27%) for Coorong. Lower Lakes average exposure scores were slightly greater between freshwater inflow levels within 2050 and 2090, than among time-horizons. Conversely, for the Coorong, differences between freshwater inflow levels was similar to differences obsreved between time-horizons (Table 7).

For Lower Lakes, freshwater flow contributed most to exposure scores, particularly under lower freshwater inflow conditions (Table 7). Connectivity, then salinity were next greatest contributors, with temperature contributing the least. For Cooong, freshwater flow and salinity were equally most influential drivers contributing to exposure scores, particularly under lower freshwater inflow conditions. Then connectivity contributed next to scores, while temperature also had the smallest influence on overall exposure (Table 7).

Table 7. Average weighted exposure scores for each environmental driver under lower and higher freshwater inflow conditions for the 2050- and 2090-time horizons. Summary averages across exposure variables for each freshwater inflow (FW) level (in bold) and for each time horizons (in underline).

	LOWER LAKES				COORONG			
	2050		2090		2050		2090	
DRIVER	LOWER	HIGHER	LOWER	HIGHER	LOWER	HIGHER	LOWER	HIGHER
Temperature	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
Freshwater flow	2.3	1.0	2.8	1.5	2.3	1.0	2.8	1.5
Salinity	1.1	1.1	1.7	1.7	2.0	1.7	2.0	2.0
Connectivity	1.5	1.0	2.0	1.5	1.3	1.0	2.0	2.0
Average (FW)	**1.5**	**1.0**	**1.9**	**1.4**	**1.6**	**1.2**	**1.9**	**1.6**
Average (TH)		1.2		1.6		1.4		1.8

Temperature

Temperature exposure remained low across all climate change scenarios (Table 7; Figure 6). Projected annual average daily maximum water temperatures remained below 24 °C, seasonal mean temperature anomalies were less than +2 °C, and extreme high-temperature days (> 40 °C) increased only slightly from two days per year at baseline to four days in 2050 and five days in 2090.

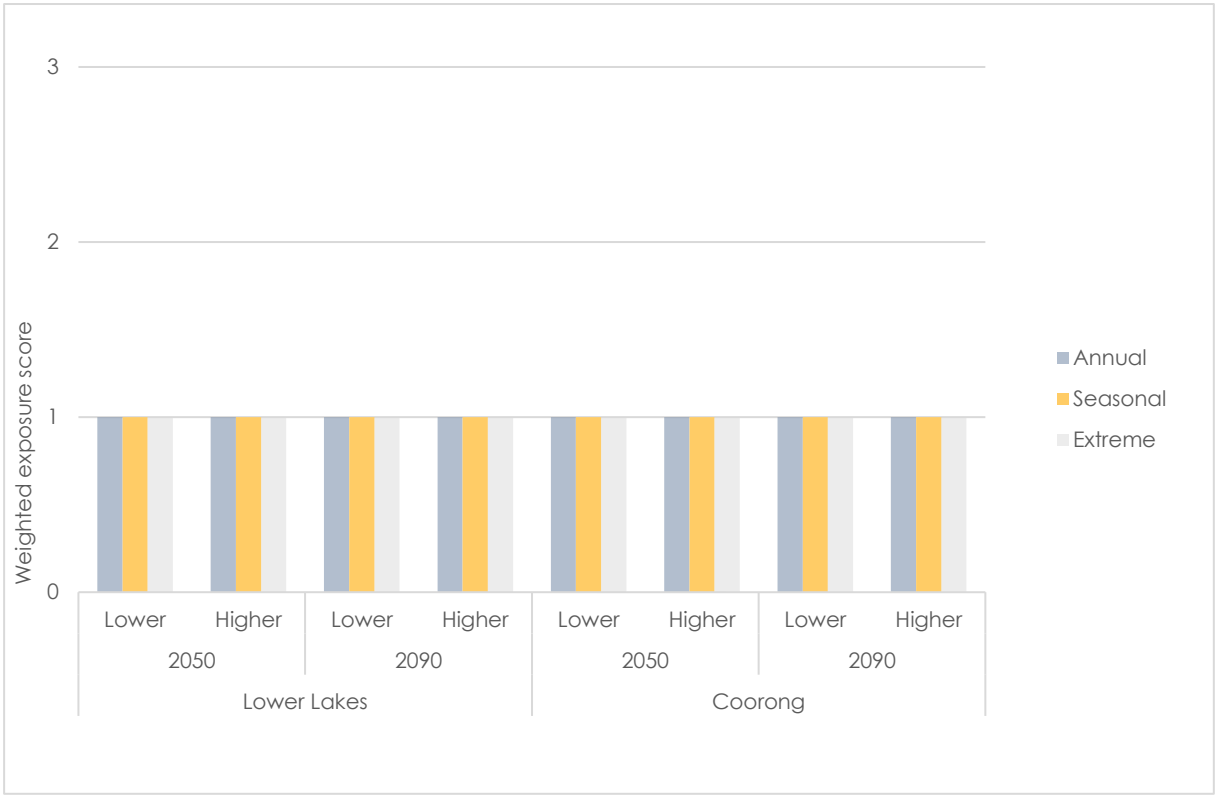


Figure 6. Distribution of weighted temperature exposure scores under lower and higher freshwater inflow conditions for the Lower Lakes and Coorong at the 2050 and 2090 climate change time horizons.

Freshwater flow

Freshwater flow exposure scores were strongly influenced by projected declines in runoff between 2050 and 2090 for both The Lower Lakes and Coorong (Figure 7). Under higher freshwater inflow conditions, seasonal exposure scores remained low across most seasons and both time horizons, with the exception of spring and autumn, which increased to moderate exposure in 2090. Under lower freshwater inflow conditions, exposure scores were consistently higher and generally increased from 2050 to 2090. Spring received the highest exposure score throughout both time horizons, whereas autumn and winter increased from moderate to high exposure by 2090. Summer remained the least affected season, maintaining a moderate exposure score under lower freshwater inflow conditions and a low score under higher freshwater inflow conditions.

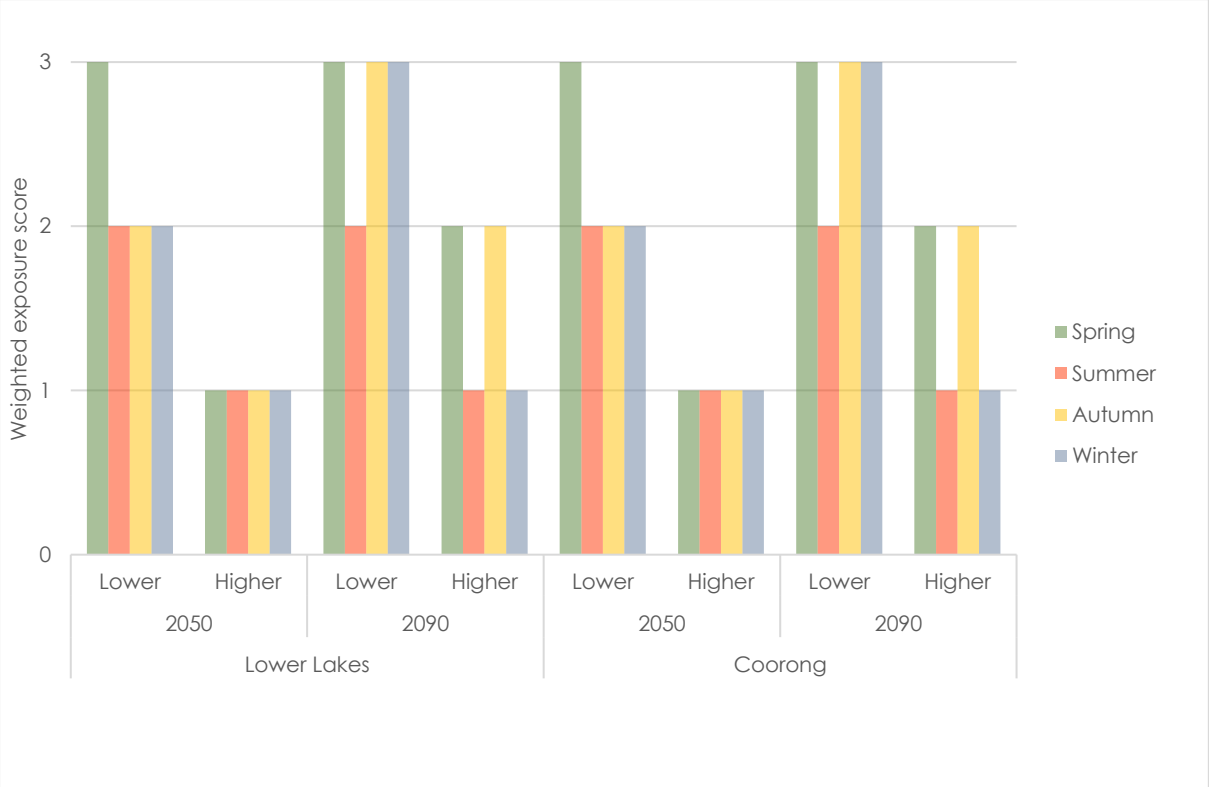


Figure 7. Distribution of weighted freshwater flow exposure scores under lower and higher freshwater inflow conditions for the Lower Lakes and Coorong at the 2050 and 2090 climate change time horizons.

Salinity

Salinity exposure was generally greater in the Coorong than the Lower Lakes and increased from 2050 to 2090 (Figure 8). Exposure was highest for the freshwater–brackish salinity range (0–20 psu), followed by the brackish range (20.1–40 psu), whereas exposure associated with salinities of 40.1–60 psu remained low across all climate change scenarios.

Coorong salinity

Median salinity differed little between the 2050 and 2090 time horizons for equivalent flow states (Appendix 9). Flow state exerted a much greater influence on Coorong salinity than projected sea-level rise and warming, particularly in the South Lagoon where hypersaline conditions (> 85 psu) occurred under the Extreme Dry flow state.

Exposure was greatest for the freshwater–brackish salinity range (0–20 psu), where hydrodynamic modelling indicated substantial change (Figure 8). Under the Base Case flow scenario, this salinity range comprised ~26% of the Coorong salinity profile. Relative to the Base Case, its proportional extent declined by 88% and 92% under the Extreme Dry flow state in 2050 and 2090, respectively, and by 42% and 70% under the Dry flow state. Consequently, exposure was high under lower freshwater inflow conditions in 2050 and under both freshwater inflow conditions by 2090.

Exposure for the brackish–marine salinity range (20.1–40 psu) was moderate, reflecting moderate increases in its proportional extent (Figure 8). This salinity range comprised

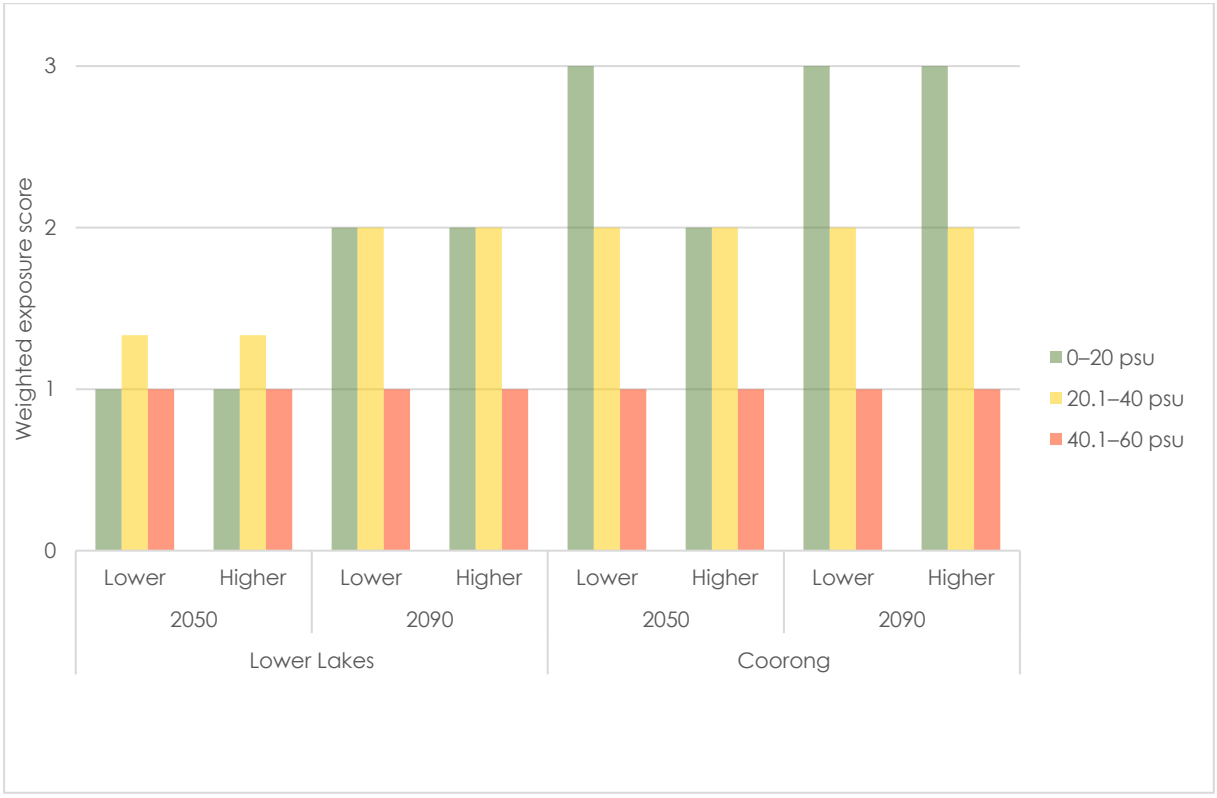


Figure 8. Distribution of weighted salinity exposure scores under lower and higher freshwater inflow conditions for the Lower Lakes and Coorong at the 2050 and 2090 climate change time horizons.

~23% of the Base Case salinity profile and increased by 49% under the Extreme Dry flow state in both time horizons, and by 34% and 59% under the Dry flow state in 2050 and 2090, respectively.

Exposure for the hypersaline salinity range (40.1–60 psu) remained low, reflecting relatively small changes in its proportional extent. This salinity range comprised ~13% of the Base Case salinity profile and increased by 18% and 20% under the Extreme Dry flow state in 2050 and 2090, repectively and by 30% and 54% under the Dry flow state.

Lower Lakes salinity

Exposure for the freshwater–brackish salinity range (0–20 psu) increased from low in 2050 to moderate in 2090 under both freshwater inflow conditions (Figure 8), reflecting projected reductions in the proportional extent of freshwater–brackish salinity. Under the Base Case flow scenario, freshwater–brackish salinities (0–20 psu) comprised ~99% of Lower Lakes salinity conditions. Relative to the Base Case, the proportional extent of this salinity range declined by 10% and 65% under the Extreme Dry flow state in 2050 and 2090, respectively, and by 9% and 52% under the Dry flow state.

Exposure for the brackish–marine salinity range (20.1–40 psu) increased from low in 2050 to moderate in 2090 under both lower and higher freshwater inflow conditions (Figure 8), reflecting projected increases in the proportional extent of this salinity range. Under the Base Case flow scenario, salinities of 20.1–40 psu comprised ~1% of Lower Lakes salinity conditions. Relative to the Base Case, the proportional extent of this

salinity range increased substantially, but largely reflected the influence of very small baseline proportion. Expansion of brackish–marine conditions were noted for 2090. Accordingly, confidence scores were adjusted to 2 for both time-horizons.

Exposure for the 40.1–60 psu salinity range remained low under both lower and higher freshwater inflow conditions throughout the 2050 and 2090 climate change scenarios (Figure 8), indicating little projected change in higher salinity conditions. Under the Base Case flow scenario, this salinity range was absent from the Lower Lakes. Small areas of 40.1–60 psu salinity were projected under the Extreme Dry (5% in 2050) and Dry (2% in 2050) flow states but were absent again by 2090, resulting in little overall change in exposure.

Connectivity

The median annual Lock 1 flow during the baseline period was 4523 GL/year. Applying projected changes in annual runoff resulted in estimated median annual flows of 4116 GL/year (−9%) and 3211 GL/year (−28%) for the 2050 higher and lower freshwater inflow scenarios, respectively, and 3799 GL/year (−16%) and 2894 GL/year (−36%) for the corresponding 2090 scenarios.

Connectivty within each waterbody, was the least exposed of the different types of connectivity evaluated, and was scored equally low across all climate change scenarios (Figure 8). Both depth and water levels were determined to be adequate to enable fish movement. Projected median annial water volumes were larger than the volume of the Lower Lakes at 0.4 m (1599 GL), supporting the assertion that Lower Lakes may be maintained at 0.0 m (100% time) and 0.4 m (95% time) AHD. Modelled water levels were likewise sufficient to maintain within-waterbody connectivity and generally increasing under future climate scenarios.

Fishway connectivity depended not only on annual freshwater inflow but also on the head difference between the Lower Lakes and the Coorong required for effective fishway operation. Projected annual inflows remained sufficient to support management objectives associated with fishway operation. However, modelling indicating greater increases in Coorong water levels progressively reduced the positive head difference and, under some scenarios, resulted in reverse hydraulic gradients. Consequently, exposure increased from low–moderate in 2050 to high under lower freshwater inflows in 2090.

Connectivity at the Murray Mouth was scored as low to moderate (Figure 8). Projected annual runoff volumes appeared to achieve objectives for maintaining Murray Mouth condition across all climate change scenarios. Exposure scores increased from low under higher freshwater inflows in 2050 to moderate under lower freshwater inflows in both time horizons and under higher freshwater inflows by 2090. Scoring also assumed that artificial dredging would continue under future declining runoff.

Exposure scores for the inundation of barrage spillways at the 2050 time horizon were low reflecting Coorong water height modelling was not of sufficient magnitude to submerge spillways at either lower or higher freshwater inflow conditions (Figure 8). By 2090, projected Coorong water levels of ~0.75–0.90 m AHD overlapped the barrage

spillway crest elevation (0.83 m AHD), substantially increasing the likelihood of spillway inundation and marine water ingress into the Lower Lakes. Consequently, barrage spillway exposure was scored as high under both freshwater inflow scenarios.

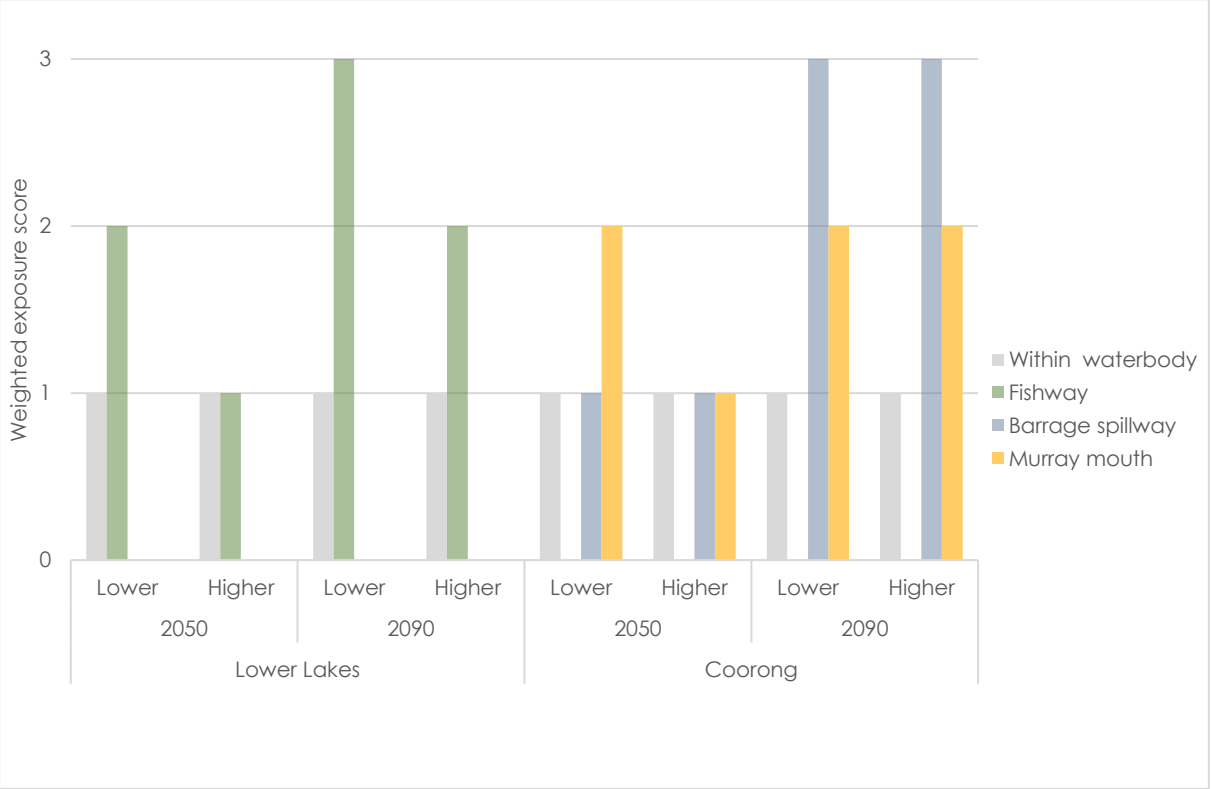


Figure 9. Distribution of weighted connectivity exposure scores under lower and higher freshwater inflow conditions for the Lower Lakes and Coorong at the 2050 and 2090 climate change time horizons.

4.3.3 Vulnerability

The average species vulnerability was similar across the four climate change scenarios (Tables 8 and 9). At 2050, the Coorong was slightly greater than the Lower Lakes (2.1 versus 2.0) but by 2090, the Lower Lakes was slightly greater than the Coroong (2.7 versus 2.6).

For 2050 and 2090 Lower Lakes scenarios, golden perch was the most vulnerable species (2.5 and 3.3; overall mean = 2.9), followed by redfin perch (2.1 and 2.8; overall mean = 2.4) (Table 8). Bony herring was the least vulnerable species (1.8 and 1.3; overall mean = 2.0). Species ranked order based on average vulnerability scores was identical between the 2050 and 2090 time horizons, with vulnerability consistently higher under lower freshwater inflow conditions (Table 8).

For 2050 and 2090 Coorong scenarios, black bream, greenback flounder and mulloway were most vulnerable with average vulnerability scores of 2.4–2.5 in 2050 and 3.1–3.2 in 2090 (overall mean: 2.8). Yelloweye mullet was the least vulnerable (1.7 and 2.1, respectively; overall mean: 1.9) (Table 9). Species ranked order based on average vulnerability scores was identical between the 2050 and 2090 time horizons, with vulnerability consistently higher under lower freshwater inflow conditions.

All species displayed increased vulnerability from 2050 to 2090, for both the Lower Lakes and Coorong (Tables 8 and 9). However, the magnitude of increase differed between freshwater inflow conditions and waterbodies. Under lower freshwater inflow conditions, the temporal change in average vulnerability scores was nearly three times larger in the Lower Lakes (0.8; Table 8) than in the Coorong (0.3; Table 9). Under higher freshwater inflow conditions, the pattern was reversed, with a greater change in the Coorong (0.7; Table 9) than in the Lower Lakes (0.5; Table 8). Differences in Lower Lakes vulnerability scores from 2050–2090 were largely due to a change in fishway connectivity and barrage spillway under lower freshwater inflow (Figure's 8 and 9), while differences in vulnerability under high freshwater inflows were being driven by both changes to salinity (particularly 0–20 psu), Murray mouth and Barrage spillway exposure scores (Figure's 8 and 9).

Table 8. Vulnerability scores for the five fish species inhabiting the Lower Lakes exposed to four environmental drivers under lower and higher freshwater inflow conditions (FW) for the 2050- and 2090-time horizons (TH). Summary averages for species across each freshwater inflow levels (in bold), time horizons (in underline) and overall (in bold and underline).

LOWER LAKES							
	2050			2090			TH
SPECIES	LOWER	HIGHER	AVERAGE	LOWER	HIGHER	AVERAGE	AVERAGE
Golden perch	3.0	2.0	**2.5**	3.8	2.8	**3.3**	2.9
Redfin perch	2.5	1.7	**2.1**	3.2	2.4	**2.8**	2.4
Common carp	2.4	1.6	**2.0**	3.1	2.3	**2.7**	2.4
Congolli	2.2	1.5	**1.8**	2.8	2.1	**2.4**	2.1
Bony herring	1.4	2.1	**1.8**	2.7	2.0	**2.3**	2.0
Average (FW)	2.3	1.8	**2.0**	3.1	2.3	**2.7**	**2.4**

Table 9. Vulnerability scores for the five fish species inhabiting the Lower Lakes exposed to four environmental drivers under lower and higher freshwater inflow conditions (FW) for the 2050- and 2090-time horizons (TH). Summary averages for species across freshwater inflow levels (in bold), time horizons (in underline) and overall (in bold and underline). *Average calculated excluding congolli.

	COORONG						
	2050			2090			TH
SPECIES	LOWER	HIGHER	AVERAGE	LOWER	HIGHER	AVERAGE	AVERAGE
Black bream	3.0	2.1	**2.5**	3.4	2.9	**3.2**	2.8
Greenback flounder	2.9	2.1	**2.5**	3.4	2.9	**3.1**	2.8
Mulloway	2.9	2.0	**2.4**	3.3	2.8	**3.1**	2.8
Congolli	2.4	1.7	**2.1**	2.8	2.4	**2.6**	2.3
Sandy sprat	2.1	1.5	**1.8**	2.4	2.1	**2.3**	2.0
Smallmouth hardyhead	2.0	1.4	**1.7**	2.3	2.0	**2.1**	1.9
Yelloweye mullet	2.0	1.4	**1.7**	2.3	1.9	**2.1**	1.9
*Average (FW)	2.5	1.7	**2.1**	2.8	2.4	**2.6**	**2.4**

4.4 Discussion

This study assessed the vulnerability of eleven fish species across four climate change scenarios — defined by differences in four key environmental drivers: temperature, freshwater flow, salinity and connectivity. Overall, fishes vulnerability ranged from low to moderate, but increased from 2050 to 2090 for both the Lower Lakes and Coorong (Figure 10). While sensitivity varied among species, differences in vulnerability were largely driven by changes in exposure, particularly declines in runoff from the Southern Murray-Darling Basin.

4.4.1 Impacts on environmental conditions for fish

In this study, assessment of climate-change exposure involved examining projected changes in environmental drivers under each scenario using available climate projections and supporting hydrodynamic modelling. In interpreting exposure, several limitations should be considered. Unlike many climate-change vulnerability assessments where exposure is only represented by climate projections (e.g. temperature), the exposure framework developed for the CLLMM incorporated drivers of differing complexity. Temperature and freshwater flow represented relatively simple estimates obtained directly from published climate projections, whereas salinity and connectivity were compound environmental drivers integrating multiple climate, hydrodynamic and management processes. This complexity also presented challenges for assigning confidence scores, particularly for connectivity, which

required greater expert interpretation. Consequently, exposure scores for each driver should be interpreted as 'relative' measures of projected environmental change.

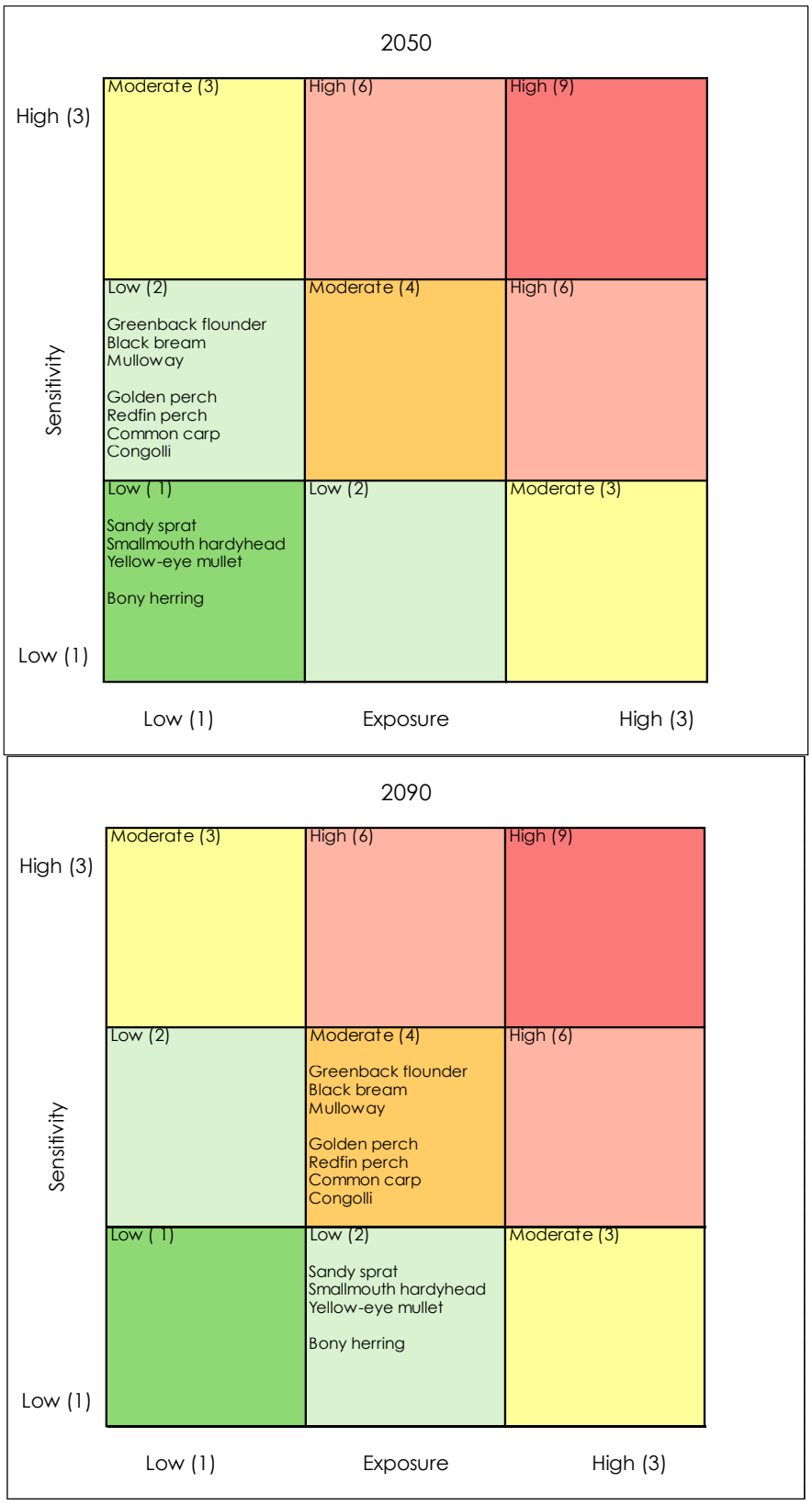


Figure 10. Vulnerability matrix for the eleven fish species assessed in the CLLMM region under the 2050 and 2090 climate change scenarios. Vulnerability scores range from 1 (green, lowest vulnerability) to 9 (red, highest vulnerability).

Exposure levels were determined as low to moderate. Exposure differed across time, space and for levels of freshwater inflow. Differences in exposure between levels of freshwater inflow were comparable to those between time horizons, suggesting that the degree of drying may be as important as the progression of climate change through time in determining exposure. Consequently, periods of extreme drying occurring earlier in the century may provide insight into environmental exposure conditions that could become increasingly characteristic under later climate conditions.

At the broader spatial scale, overall climate exposure was marginally higher in the Coorong than the Lower Lakes at both 2050 and 2090. However, differences in overall exposure between time horizons were greater than those between water bodies, indicating that projected changes through time were more pronounced than spatial differences in overall exposure. Differences between the Coorong and Lower Lakes were instead more evident among individual environmental drivers, reflecting the contrasting hydrological and salinity regimes of the two water bodies.

Reductions in freshwater flow emerged as the primary driver of exposure across both the Lower Lakes and the Coorong. Freshwater flow influences multiple interacting environmental processes, including salinity regimes, habitat availability and connectivity pathways, all of which are critical to fish survival, movement, and reproduction. In contrast, projected temperature increases were relatively small, even under late-century scenarios. Average daily maximum temperatures did not exceed levels considered to induce thermal stress for most species, and increases in extreme heat days were modest. Results of this study suggest that hydrological change is likely to be the dominant driver of fish vulnerability in the CLLMM region although elevated temperatures may also impose sub-lethal energetic costs or interact with other stressors, these. This contrasts with broader climate change narratives that emphasise warming as the primary ecological driver and instead highlights the importance of region-specific processes in shallow, flow-regulated systems.

The influence of hydrological change was particularly evident in projected salinity conditions. In the Coorong, the degree of drying exerted a greater influence on salinity than differences between the 2050 and 2090 time horizons. The principal change was a substantial contraction of freshwater to low-brackish conditions (0–20 psu), accompanied by expansion of brackish to marine conditions. This suggests that climate change may alter the relative availability of different levels of salinity rather than simply increase salinity uniformly throughout the Coorong. Such changes are important for fish because salinity gradients influence habitat use, reproduction and recruitment, particularly for species that depend on freshwater-influenced estuarine environments. Extreme Dry conditions also produced hypersalinity exceeding 85 psu in the South Lagoon, demonstrating the continued importance of freshwater inflow in moderating the most saline parts of the system.

The Lower Lakes exhibited a different trajectory, with salinity and connectivity exposure becoming more pronounced towards 2090. Freshwater to low-brackish conditions remained dominant but contracted substantially under future conditions,

while brackish to marine conditions expanded. At the same time, increasing Coorong water levels reduced the head difference across the barrages and, by 2090, overlapped the approximate elevation of barrage spillways. These changes potentially increase marine influence on the Lower Lakes while altering the operation and effectiveness of existing connection pathways. Within-waterbody connectivity, however, remained comparatively secure because projected water levels and depths were considered sufficient for fish movement. Climate change may therefore affect connectivity less through loss of aquatic habitat within the lakes than through changes at the interfaces connecting freshwater, estuarine and marine environments.

The observed increase in exposure from 2050 to 2090 is consistent with ecological trajectories described by Rees et al. model, which characterise the CLLMM region as transitioning towards increasingly marine-influenced conditions under climate change. Freshwater flow exposure scores indicated greater declines in runoff by 2090, suggesting that the hydrological constraints intensify over time. While 2050 scenarios indicated some buffering through management and residual freshwater inputs, 2090 scenarios reflected stronger sea-level rise influences and increased marine connectivity, particularly in the Coorong.

The assessment of climate-change effects on the environment for fish in this study largely focused on hydrological alterations associated with sea-level rise and reduced freshwater inflow, for which climate projections and modelling were available (e.g. Dang et al., 2026). These projections represent conditions at discrete future time horizons and were used to assess changes in the physicochemical environment for fish. However, climate change has broader and potentially interacting effects on aquatic environments and the biota on which fish depend. For example, sea-level rise may contribute to coastal erosion and landward dune migration along the Younghusband Peninsula (DaSilva *et al.* 2024). At an existing erosion hotspot, DaSilva et al. (2024) documented rapid landward dune transgression and a negative sediment budget, with eroded sediment transported both landward and offshore. Such geomorphic responses illustrate the potential for rapid physical changes within a dynamic system that were not represented in this assessment, introducing additional uncertainty in the longer-term application of these results.

4.4.2 Species vulnerability to climate change exposure

Across the four climate change scenarios, variation in fish vulnerability primarily reflected: (1) species differences in life-history strategy; and (2) species influenced by freshwater flow dynamics. Overall vulnerability remained low to moderate as it reflected low to moderate environmental driver exposure scores for both the Lower Lakes and Coorong.

The least vulnerable species: smallmouth hardyhead, sandy sprat, yelloweye mullet and bony herring, consistently scored low for sensitivity. These are small, highly abundant and widely dispersed species, reflecting greater environmental tolerance and capacity to use changing environments (Ye *et al.* 2016). Their life-history exhibits

limited dependence on specific environmental conditions for reproduction (e.g. freshwater flow), but have high fecundity, extended spawning periods, and shorter lifespans. Such attributes are characteristic of an opportunistic life-history strategy (Bice *et al.* 2018; Ferguson *et al.* 2018; Winemiller and Rose 1992) and associated with unpredictable environments, or capacity of rapid recolonisation following disturbance (Chessman 2013; Matthews 1998). Collectively, these traits are likely to contribute to stronger ecological or fisheries species resilience to climate change.

The finding of yelloweye mullet and bony herring to be among the least vulnerable species is consistent with previous trait-based vulnerability assessments (Chessman 2013; Egan *et al.* 2020). In the present study, vulnerability was calculated using different methods, but the emergence of similar patterns suggests to a degree of assessment robustness. This consistency is notable as vulnerability outcomes depend on both data availability and trait selection (Foden *et al.* 2013; Pacifici *et al.* 2015), suggesting that the relative ranking of these species may reflect underlying biological characteristics rather than the specific assessment framework alone.

Life-history information was generally sufficient to support the vulnerability assessment, although information was limited for some opportunistic, Coorong species, particularly smallmouth hardyhead and sandy sprat. Under the confidence framework applied, attributes supported by sparse or conflicting information, or greater reliance on expert judgement, received lower confidence scores and were subsequently down-weighted in the calculation of sensitivity. Consequently, while sensitivity rankings were considered representative for species with greater knowledge availability, the influence of data limitations on small-bodied species cannot be discounted and may have partly contributed to their lower sensitivity and vulnerability scores.

Overall, the most vulnerable species — greenback flounder, black bream and mulloway in the Coorong, and golden perch, redfin perch and common carp in the Lower Lakes — appeared to share a combination of life-history traits reflecting a periodic-type strategy and increased sensitivity to freshwater flow. Congolli also sharing 'intermediate' traits is also considered within this more vulnerable group of fishes. This group of fishes have life histories characterised by an older age at maturity, high fecundity, migration requirements, reliance on environmental cues and planktonic larval dispersal. These traits are closely associated with reproduction and recruitment and reflect adaptation to environmental conditions that are predictable in their timing or occurrence (Matthews 1998; Winemiller and Rose 1992). Periodic life-history strategies also provide a longer-term response to interannual variability in recruitment conditions, with longevity and repeated reproductive opportunities allowing populations to persist through unfavourable years and capitalise on periods favourable for recruitment (Winemiller and Rose, 1992; Ferguson et al., 2013).

Coorong species vulnerability

The more sensitive Coorong species displayed relatively limited separation in average sensitivity scores, although differences of up to approximately one sensitivity unit represented substantial differences on the three-point scoring scale. Despite application of the 'logic rule' to improve separation (Morrison et al., 2015), among-

species discrimination did not improve and instead increased clustering. However, similar aggregate scores arose from different combinations of sensitive traits. Black bream sensitivity reflected its estuarine association and dependence on suitable conditions for spawning and recruitment, whereas greenback flounder and mulloway were more sensitive through movement and connectivity requirements between estuarine and marine environments. Mulloway's relatively late maturity further contributed to its sensitivity. Thus, similar aggregate sensitivity scores did not necessarily reflect common mechanisms of sensitivity or climate risk.

Dependence on freshwater flow was particularly apparent among the more vulnerable Coorong species. Their sensitivity reflected freshwater-dependent processes, including the importance of flow for life-history functions, use of freshwater-influenced estuarine environments and lower salinity tolerance during early life stages. Projected reductions in freshwater flow may therefore affect environmental conditions important for reproduction and recruitment, increasing the vulnerability of these periodic strategists. Historical changes in the Coorong fish assemblage provide some support for this pattern. Ferguson et al. (2013) reported a substantial decline in the contribution of periodic strategists to estuarine fishery catches, from approximately 52% in 1984 to 12% in 2008, concurrent with declining freshwater inflows and increasing dominance of opportunistic strategists. The authors suggested that opportunistic species may be favoured by their capacity to recruit under a wider range of environmental conditions, whereas periodic strategists rely on longevity and high fecundity to persist through periods of unfavourable conditions and capitalise on periods favourable for recruitment.

Lower Lakes species vulnerability

Golden perch, redfin perch and common carp were the three most sensitive species assessed for the Lower Lakes. The current study finding of higher vulnerability contrasts with previous studies that have characterised them as relatively tolerant to hydrological variability (Crook et al., 2010b; Chessman, 2013). This discrepancy may relate to among-study methodological differences, given the present study incorporated a broader suite of sensitivity metrics and life-stage-specific salinity and temperature tolerance. In addition, vulnerability scores for the CLLMM region reflect the location-specific risk of Lower Lakes salinisation from sea-level rise, particularly under reduced runoff. This exposure pathway is not considered in vulnerability assessments of freshwater fishes elsewhere in the Murray–Darling Basin.

Species profiles for golden perch, common carp and redfin perch indicate that projected changes in the Lower Lakes may affect these freshwater species in different ways. Although all three are tolerant of warm-water environments and relatively high temperatures, declining freshwater flow and increasing salinity associated with reduced runoff and sea-level rise present greater risks. These changes may affect reproduction, recruitment, early life-history stages and the distribution and movement of individuals, depending on species-specific life histories and environmental tolerances. The combined influence of declining freshwater availability and increasing

salinity is therefore likely to be important in determining the future persistence and distribution of these species within the Lower Lakes.

The Lower Lakes provide important habitat for golden perch, with populations potentially supported by both local recruitment and dispersal from elsewhere in the Murray–Darling Basin. Reproduction and recruitment are strongly influenced by freshwater flow, with spawning occurring in the lower River Murray (Ye *et al.* 2025) and Lower Lakes age structures dominated by cohorts associated with elevated flows (Ferguson and Ye 2016). Consequently, projected reductions in runoff may affect reproduction, recruitment and dispersal into the Lower Lakes. However, Lower Lakes-specific cohorts have also occurred during comparatively low-flow periods, suggesting that localised spawning and recruitment may occur downstream (Bice *et al.* 2023a). Increasing salinisation may present an additional risk to early life-history stages, potentially affecting both local recruitment and individuals dispersed downstream. These changes also have implications for the Lakes and Coorong Fishery, the primary commercial fishery for golden perch in South Australia, which targets the species in the Lower Lakes (Sarakinis and Earl 2025). Further understanding of the contribution of local and upstream recruitment, and their responses to changing flow and salinity, would improve predictions of future population and fishery vulnerability.

The Lower Lakes provide suitable freshwater habitat for common carp and support substantial abundances of the species. Given their relatively low salinity tolerance and restriction to freshwater–low-brackish conditions (< 15 psu), increasing salinisation of the Lower Lakes from sea-level rise may progressively reduce suitable habitat and promote redistribution towards the Lower River Murray and other inflowing tributaries. Loss of freshwater macrophytes with increasing salinity may further reduce spawning opportunities, particularly under lower-flow conditions.

Redfin perch were also more vulnerable under climate change, reflecting their reliance on freshwater environments and sensitivity to increasing temperature and salinity, particularly during early life-history stages. Although adults exhibit some capacity to tolerate elevated temperatures (Sandström *et al.* 1997) and occur in some saline environments (Christensen *et al.* 2021), these tolerances may not extend across life-history stages. For instance, larvae are more sensitive to salinity, with significant mortality occurring at low-brackish salinities (~10 psu). Increasing salinity within the Lower Lakes, particularly under reduced freshwater inflow, may therefore reduce suitable habitat for larvae and subsequently limit recruitment.

For other Lower Lakes species, such as bony herring, their broad distribution across inland Australia, including tropical and intermittently flowing dryland river systems such as the Lake Eyre Basin, suggests comparatively low sensitivity to increasing temperature and reduced freshwater flow. Bony herring tolerate temperatures up to approximately 38 °C, while reproduction in the lower River Murray can occur independently of flooding. Increasing Lower Lakes salinity may present a greater risk, particularly to early life-history stages, although their salinity tolerance remains poorly understood. Juveniles and adults appear considerably more tolerant, with salinities of

at least 39 psu reported (Lintermans 2007) and occasional observations at substantially higher salinities in the Coorong (Ye *et al.* 2012).

For congolli, a mid-range vulnerability score reflected broad tolerance to environmental conditions but greater sensitivity to freshwater flow and connectivity, differentiating the species from those with lower dependence on freshwater processes. As a diadromous species with intermediate life-history characteristics, congolli may respond differently to reduced freshwater inflow and sea-level rise than other Lower Lakes species because connectivity between freshwater and estuarine environments is required to complete its life cycle. Fishways therefore provide important connective pathways across the barrages, with reduced connectivity under low freshwater inflows representing a substantial risk, as demonstrated by declines in diadromous fish populations during the Millennium Drought. Although fishways require relatively small volumes of water to operate, and future runoff volumes may remain sufficient to facilitate their operation, sea-level rise presents a different risk through its potential effects on fishway functionality. Conversely, higher sea levels, particularly by 2090, may create alternative connectivity pathways through increased water levels and inundation of low-lying areas, including barrage spillways separating the Lower Lakes and Coorong.

4.4.3 Lower Lakes salinisation and fish species response

Under current management arrangements that limit saltwater incursion during extended low-flow periods, defensive barrage operation is likely to result in increased salinisation of the Lower Lakes to brackish, marine or potentially low-hypersaline conditions (Dang *et al.* 2026). Such widespread salinisation would favour a shift away from freshwater fish species dominance towards species associated with more saline environments (Rees *et al.* 2022). Increasing salinity may create habitat opportunities for marine-estuarine opportunists, including yelloweye mullet and sandy sprat, and estuarine species such as smallmouth hardyhead and black bream. Succession will, however, depend on species life-history characteristics and available connection pathways, with species that currently use fishways expected to colonise sooner.

For freshwater fishes, the effects of salinisation may occur before adult survival is directly compromised. Successful reproduction occurs across narrower environmental ranges than those associated with survival. Fish eggs and larvae life-stages are generally more sensitive and thus vulnerable to salinity than more developed juveniles and adults (Hart *et al.* 1991). Consequently, increasing salinity may constrain reproduction, early-life survival and recruitment, reducing the capacity of freshwater fishes to maintain local populations and potentially affecting commercially targeted species.

Increasing salinisation of the Lower Lakes may also have complex ecological implications for introduced freshwater species such as redfin perch and common carp. Both species are widely distributed across the southern Murray–Darling Basin and their abundances can be characterised by substantial fluctuations in recruitment. As introduced species, they are associated with several density-dependent ecological impacts, including disturbance of aquatic habitats by common carp, predation on

native fishes by redfin perch, competition for resources and disease transmission. Declining abundance within the Lower Lakes may therefore reduce some of these impacts locally. Conversely, redistribution towards the Lower River Murray or other freshwater habitats in response to increasing salinity could shift or intensify ecological impacts elsewhere. Such responses represent a broader ecological consequence of Lower Lakes salinisation that extends beyond effects on individual species.

The expansion of estuarine and marine-estuarine species into the Lower Lakes is likely to occur progressively, with the timing of colonisation differing among species according to their capacity to use existing or future connection pathways. Yelloweye mullet, smallmouth hardyhead and sandy sprat are prominent components of Coorong fish assemblages, characterised by high abundance and broad salinity tolerance that enables distributions across much of the Coorong. Smallmouth hardyhead and sandy sprat, and yelloweye mullet to a lesser extent, also regularly use fishways. Their capacity to use existing connection pathways may therefore enable these species to be among the first to expand into the Lower Lakes as salinity increases with sea-level rise.

For larger periodic species, including black bream, greenback flounder and mulloway, broad salinity tolerance may similarly provide opportunities for increased use of the Lower Lakes, although their less frequent use of fishways may constrain colonisation. Greater distribution may therefore depend on alternative connection pathways, potentially arising later in the century as higher sea levels increase inundation of low-lying land or infrastructure, consistent with the conceptual changes described by Rees et al. (2022).

Considerable uncertainty remains around the magnitude and persistence of these assemblage changes. While some species may initially expand into the Lower Lakes, in absence of suitable ongoing connectivity, sustained populations may also depend on successful reproduction and recruitment under the altered salinity conditions. Species potentially capable of reproducing and recruiting at higher salinities, such as black bream, may have greater potential to establish resident populations. Successful establishment may, however, depend on the development of suitable salinity gradients, both vertically through the water column and longitudinally across the Lower Lakes. Barrage operations will remain influential in determining connectivity and habitat conditions. Monitoring fish distributions, movement and environmental conditions will therefore be important for distinguishing temporary redistribution from longer-term population establishment and potential fishery opportunities.

4.4.4 Climate change and fisheries species

Collectively, the most vulnerable species — black bream, geenback flounder, mulloway, golden perch, redfin perch and common carp — are all landed by the Lakes and Coorong Fishery, indicating a potential pathway through which increasing climate vulnerability may translate into greater risk. Increase risk to these species was identified at 2090 when vulnerability was considered at moderate levels.

The emergence of moderate vulnerability among seven species by 2090 indicates that climate risk may become increasingly apparent as environmental change progresses. These species represented contrasting life-history strategies, including freshwater, estuarine-resident, marine-estuarine and diadromous species, suggesting that increasing vulnerability was not restricted to a particular ecological group. Rather, different biological sensitivities interacted with increasing exposure to freshwater flow, salinity and connectivity changes to produce similar overall vulnerability outcomes.

Similar vulnerability among species should not necessarily be interpreted as indicating common mechanisms of climate risk. Black bream, greenback flounder and mulloway had similar overall vulnerability despite differences in the sensitivity attributes contributing to their scores. Similarly, the comparatively high vulnerability of golden perch reflected a freshwater life history and strong association with flow, contrasting with the estuarine and marine-associated life histories of the more vulnerable Coorong species group. Consequently, climate change may produce comparable levels of vulnerability through different ecological pathways, with implications for how individual species respond and the management actions that may be effective.

Collectively, these results indicate that climate vulnerability has implications for both fisheries and broader ecological values. The predominance of harvested species among the more vulnerable species highlights potential consequences for the Lakes and Coorong Fishery, while the increasing vulnerability of congolli demonstrates that these risks extend to species whose life cycles depend on connectivity across freshwater, estuarine and marine environments.

4.4.5 Vulnerability insights for climate change adaptation

Differences in fish vulnerability between the Lower Lakes and Coorong reflected contrasting hydrological dynamics and management regimes rather than substantial differences in overall vulnerability. The Lower Lakes are partially buffered by active water-level and barrage management, including objectives to maintain water levels above 0.0 m AHD for 100% of the time (O'Connor et al., 2015), which contributed to low exposure for within-waterbody connectivity across the assessed conditions. In contrast, environmental conditions in the Coorong were more directly responsive to changes in freshwater inflow, particularly through changes in salinity. However, increasing sea level and declining freshwater inflow towards 2090 may progressively challenge the capacity of existing management to maintain current environmental conditions, particularly at the interface between the Lower Lakes and Coorong.

From a management perspective, the strong influence of freshwater inflow on exposure and vulnerability highlights the importance of water management in mitigating climate-change impacts on fishes in the CLLMM. Maintaining sufficient flows can support ecological processes and influence salinity and connectivity conditions important for fish reproduction, recruitment and movement. Existing interventions, including management of Lower Lakes water levels, barrage operations, fishways and Murray Mouth dredging, may therefore provide important mechanisms for buffering some climate impacts. However, their effectiveness will depend on the magnitude of

future reductions in freshwater inflow and increasing sea-level influence, highlighting the importance of adaptive management as environmental conditions change.

5 Consultation with the Lakes and Coorong Fishery

5.1 Introduction

The Lakes and Coorong Fishery (LCF) is a multi-species commercial fishery that operates in the estuary of the River Murray and Coorong lagoons (the Coorong Estuary), the freshwater lower lakes of the River Murray (Lake Alexandrina and Lake Albert), and the nearshore marine environment adjacent to the Coorong Estuary (Primary Industries and Regions South Australia 2022). Fishery finfish production by weight of catch mainly comprises bony herring, carp, yelloweye mullet, mulloway, and golden perch (Sarakinis and Earl 2025). Other species such as greenback flounder, redfin perch, and black bream, have contributed significantly to the overall catch in some years. The use of fishing gear differs depending on the location of fishing and the species being targeted, although large-mesh gillnet (LMGN) and small-mesh gillnet (SMGN) are the main gear types used by fishers targeting finfish species. Around 98% of all finfish catches each year in the LCF are accounted for by the (i) freshwater LMGN sector in the Lower Lakes, (ii) estuarine LMGN sector and (iii) estuarine SMGN sector in the Coorong Estuary.

Freshwater flow from the River Murray is a major driver of ecological processes and biological responses in the CLLMM region, including the population dynamics and abundance of fish species that support the LCF (Bice *et al.* 2018; Ye *et al.* 2016). The identification of freshwater flow as the primary driver of vulnerability in the CLLMM region, and its influence on salinity, connectivity, and key life-history processes of fish species (Section 4), highlights the hydrological alterations from climate change may further affect the future sustainability of the LCF.

The following chapter builds on key findings from Section 4, including climate change scenarios, and consultation with LCF fishers to gather preliminary insights on approaches to facilitate dialogue on climate change between stakholders and industry . Such increased communication and engagement with industry aims to improve the capacity to anticipate, prepare for, and respond to climate change.

5.2 Methods

Consultation between the project team and industry was initiated through the Southern Fishermen's Association, which assisted in communicating the study to licence holders and invited participants. Participating fishers were consulted both in-person and by phone using a two-stage survey. First, a semi-structured discussion with fishers was completed to gain initial perspectives on potential future changes in the CLLMM region under climate change, including environmental and socio-economic impacts, and the future health of fish stocks that support the LCF. Second, climate change scenarios ('2050 Lower Lakes' and '2050 Coorong') were presented to fishers and responses on potential adaptation of fishing practices under each scenario were collected using a three-point Likert-type likelihood scale (i.e., 'unlikely', 'neutral', 'likely', see Appendix 12). R esponses were collected for each gillnet sector.

5.3 Results

Surveys were completed by four commercial fishers in March 2026, representing ~12% of active licences operating in the LCF (Sarakinis and Earl 2025). Given the limited sample size, results should be interpretated with caution and considered preliminary, providing an initial indication of how the commercial fishing industry in the CLLMM region may respond to climate change.

5.3.1 Future CLLMM – a commercial fisher's perspective

Responses to the semi-structured discussion indicate that industry participants consider the abundance of key target species in the LCF to be primarily influenced by freshwater flow. Fishers reported that increased discharge from the Murray barrages is associated with cooler water conditions and improved catch rates of estuarine species (e.g. mulloway, black bream, greenback flounder, and yelloweye mullet), whereas reduced flows and warmer conditions are associated with shifts towards more marine-dominated species.

Concerns by the LCF regarding climate change in the CLLMM region are therefore largely centred on future freshwater flow regimes and their influence on ecological conditions and processes, including potential changes to fish migratory behaviours, fishing seasons, and the occurrence of algal blooms. From an economic perspective, fishers identified interactions with long-nosed fur seals, rising operational costs, and market variability as key pressures that are likely to be exacerbated under future climate change stressors. Overall, fishers suggested that climate-driven changes in freshwater flow regimes are likely to have cascading ecological and economic consequences. Historical patterns also indicate that interannual variability in species abundance remains closely linked to periods of drought and flooding.

5.3.2 Fisher responses to climate change scenarios

Responses to the likelihood component of the survey indicate a generally consistent pattern of moderate adaptive capacity, with most responses recorded as either 'neutral' or 'likely' for both The Lower Lakes and Coorong climate change scenarios (Table 10). Fishers indicated that they are unlikely to stop targeting primary species (i.e., golden perch, mulloway, and yelloweye mullet), largely reflecting economic considerations including cost-benefit trade-offs relative to secondary and tertiary species, as well as operational constraints such as long-nosed fur seal interactions, which have reduced the viability of some previously productive fishing areas (e.g. greenback flounder near the Murray Mouth).

Fishers also reported that they are already highly responsive to changing environmental conditions, with adaptive strategies primarily driven by catch success and profitability. In the Coorong Estuary, responses to climate change are expected to manifest through increased optimisation of fishing strategies, including changes in gear selection, gillnet sector, timing of effort and seasonal targeting, as well as investment in technology and assets to improve efficiency and maintain profitability. These strategies are consistent with those already used by fishers in response to

variability in catch rates and environmental conditions. In the Lower Lakes, responses were more variable, although investment in technology and assets was commonly identified. Temporal and spatial shifts of fishing activity in response to climate change were more variable, likely reflecting the relative environmental stability of this system compared to the estuarine sectors.

Across both systems, key drivers of adaptive capacity were primarily economic and operational rather than environmental. Fishers indicated a preference for improving fish handling practices to enhance product quality and value (i.e., higher wholesale price) as a means of maintaining viability in the event of declining stocks. Participants also expressed support for reviewing the current harvest strategy, which regulates fishing effort through a Total Allowable Commercial Effort (TACE), as a possible adaptive management response to potential climate-driven changes in fish populations. Diversification of market channels was generally viewed neutrally, reflecting its limited sensitivity to environmental change.

Responses to increased information provision and stakeholder engagement were mixed. While some fishers saw value in enhanced communication and support for adaptation planning, others noted uncertainty around future conditions and suggested that limitations in predictive certainty may reduce the influence of external advice on current fishing practices.

Table 10. Commercial fisher responses to climate change scenarios in the CLLMM region. Survey includes a three-point Likert likelihood scale, with responses given as either 'unlikely', 'neutral', or 'likely', and collected for each Lakes and Coorong Fishery gillnet sector.

	2050 LOWER LAKES			2050 COORONG					
	FRESHWATER LARGE-MESH GILLNET SECTOR			ESTUARINE LARGE-MESH GILLNET SECTOR			ESTUARINE SMALL-MESH GILLNET SECTOR		
RESPONSE	UNLIKELY	NEUTRAL	LIKELY	UNLIKELY	NEUTRAL	LIKELY	UNLIKELY	NEUTRAL	LIKELY
Change the amount of fishing effort	0	4	0	1	2	1	1	1	2
Move to another fishing block within the sector	1	2	2	1	1	2	1	1	2
Stop fishing for primary species	3	1	0	4	0	0	4	0	0
Switch gear/sector	1	2	1	1	1	2	1	1	2
Adjust timing of fishing period	1	2	1	1	1	2	1	1	2
Adjust length of fishing season	1	1	2	1	1	2	1	1	2
Invest in new technology or assets	0	0	4	0	0	4	0	0	4
Adjusted harvest strategy	1	2	1	1	1	2	1	1	2
Improve fish handling	1	0	3	1	0	3	1	0	3
Adjust sale price	1	2	1	1	2	1	1	2	0
Diversity markets (local, intra-/inter-state)	1	3	0	1	3	0	1	3	0
Seek information about adaptation options	2	1	1	2	1	1	2	1	1
Communication with concerned stakeholders	2	1	1	1	1	2	1	2	1

5.4 Discussion

The survey results identified alignment between the primary drivers of vulnerability in the CLLMM region (Section 4) and key concerns raised by commercial fishers, most notably freshwater flow. Fishers recognised freshwater flow to the Lower Lakes and Coorong as the main environmental influence on the future health of fish stocks that support the LCF, highlighting potential vulnerability of the fishery to future shifts in freshwater flows and increased frequency of extreme weather events, such as droughts and floods.

Climate change scenarios were generally considered secondary to current economic pressures, including operational costs, market variability, and interactions with long-nosed fur seals. Fishers indicated that these pressures may be further exacerbated under future environmental change and potential reductions in fishable biomasses of key species. The current adaptability and efficiency of the LCF fleet, both temporally and spatially, is suggested to be primarily driven by economic factors rather than environmental conditions. This suggests that while adaptive responses to climate change are possible, they may not be the primary driver of change in the near term. Given the limited data these findings should be considered preliminary and interpreted with caution.

This initial survey provides insight into fisher perspectives on climate change and the associated environmental and economic pressures facing the commercial fishing industry. Building on these preliminary findings, an ongoing survey program involving both new and previously surveyed fishers, and a more comprehensive survey framework, would improve understanding climate change impacts on commercial fishing in the CLLMM region and help inform the long-term sustainability of the LCF.

6 Conclusion

This study explored how projected climate change may affect fish and fisheries of the CLLMM region, and improved understanding of the environmental conditions that may contribute to future climate vulnerability. The assessment indicates that climate change is unlikely to affect fish species or regions uniformly. Rather, future vulnerability will depend on the interaction between species-specific environmental requirements, the magnitude of environmental change and, importantly, future freshwater flow.

Across the environmental drivers considered, changes associated with freshwater flow emerged as an important source of future exposure. Projected reductions in runoff may influence freshwater inflows to the CLLMM and the capacity to maintain environmental conditions important for fish, including salinity regimes and connectivity among freshwater, estuarine and marine habitats. These effects were generally more pronounced under lower freshwater inflow and towards 2090. The assessment therefore indicates that climate risk arises not from individual climate pressures in isolation, but from their combined influence on the environmental conditions and habitats upon which fish populations depend.

The nature of this risk differed between the Lower Lakes and Coorong and among species. The Lower Lakes generally showed lower exposure than the Coorong, where greater environmental change was evident with reductions in freshwater flow interacted with salinity and connectivity. Species vulnerability similarly reflected differences in life history, habitat use and environmental requirements. Consequently, climate change may alter the relative suitability of the CLLMM for different species rather than produce a uniform response across the fish community. Maintaining the diversity of environmental conditions required by freshwater, estuarine and diadromous species is therefore likely to become increasingly challenging as climate pressures increase.

For the Lakes and Coorong Fishery, these ecological changes have the potential to affect the distribution, abundance and availability of fishery resources and increase variability in fishing opportunities. Consultation with fishers provided an applied perspective on this risk. Fishers recognised freshwater inflow as an important influence on fish abundance and catch variability and described changes to fishing operations in response to environmental conditions. These observations demonstrate that adaptation is already an important component of how the fishery responds to environmental variability, although the magnitude and rate of future climate change may present additional challenges.

The assessment also highlights important uncertainties. Climate projections, hydrological responses, ecological responses and future management actions cannot be predicted precisely, and vulnerability scores should therefore be interpreted as indicators of relative climate risk rather than forecasts of future population abundance or fishery production. The assessment provides a framework that can be updated as climate projections, hydrodynamic modelling, ecological knowledge and observations of fish populations improve.

Overall, this study identifies freshwater flow as a primary influence on future climate vulnerability of fish and fisheries in the CLLMM and demonstrates how its effects interact with salinity, connectivity and species-specific environmental requirements. The findings provide an improved basis for identifying where climate risks may emerge, which species and environments may be more vulnerable, and where monitoring and adaptive management can be directed as environmental conditions change. Continued integration of climate information, ecological monitoring, management and fisher knowledge will be important for supporting the long-term resilience of fish populations and the Lakes and Coorong Fishery.

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8 Appendices

Appendix 1. Life-history profiles of eleven CLLMM region fish species.

Black bream (Acanthopagrus butcheri)

Life history

Black bream are classified as 'estuarine' guild group of fish, completing their entire life cycle within the estuary (Potter and Hyndes, 1999; Norriss et al., 2002; Bice et al., 2018a; Jenkins et al., 2018; Ye et al., 2020a; Bice, 2023; Sarakinis et al., 2024). They generally reside within the estuary but may also move into nearby marine environments for periods of time (Neira et al., 1998; Burridge and Versace, 2007; Bice, 2023; Sarakinis et al., 2024). In Western Australia, they have also been stocked into various lakes (Norriss et al., 2002). Black bream have a periodic life-history strategy (Ferguson et al., 2018) which is characterised by high longevity and delayed/late maturity, prioritising growth and a large body size to allow the production of many small eggs (Winemiller and Rose, 1992). Black share similar life history requirements with Yellowfin bream (*Acanthopagrus australis*), and other estuarine dwelling members of the Sparidae family (Ochwada-Doyle et al., 2012).

Distribution, population structure and utility

Black bream are endemic to Southern Australia, extending from Myall lakes, New South Wales to Shark Bay, Western Australia, including Tasmania (Rowland, 1984; Norriss et al., 2002; Sakabe, 2009). Limited gene flow has been described across large extent of its distribution, with distinct subpopulation's occurring at the level of individual estuaries (Burridge and Versace, 2007; Sarakinis et al., 2024). In South Australia, populations exist across the Fleurieu Peninsula, St Vincent's Gulf, Spencer Gulf and Kangaroo Island (Earl et al., 2016). In estuaries, black bream use both shallow and deeper waters, and are associated with a range of different habitat attributes including sand, rock, shellfish, seagrass, woody debris or artificial structures (Norriss *et al.* 2002). In the marine environment, black bream are associated with reefs (Norriss et al., 2002).

Diet

Black bream are omnivorous, consuming a variety of fish, invertebrates, detritus, algae and plant material the in diet (Ye et al., 2020b). An Increased proportion of animal matter occurs in diets of larvae and juveniles. Diet may also vary spatially between estuaries (Chuwen et al., 2007).
Larvae: copepods nauplii (Newton, 1996, Williams et al., 2013; McNaughton et al., 2022), cladocerans (Williams et al., 2013), polychaetes (Williams et al., 2013), copepodites (Williams et al., 2013).
Juveniles: detritus, amphipods, nereid polychaetes and algae (Potter and Hyndes, 1999; Sakabe, 2009)
Adults: algae, seagrass, molluscs, decapods, crustaceans, fish (Potter and Hyndes, 1999; Sakabe, 2009)

Age and growth

Max age: 32 years (Ye et al., 2024)
Reported to a maximum length of 600 mm (Morison et al., 1998). Flexible growth rate in WA, growth varies among estuaries (Sarre and Potter, 2000; Earl et al., 2016) and across years within estuaries (Cottingham et al., 2014). Environmental variability drives changes in growth over multiple years, with growth negatively related to low freshwater inflow and increased hypoxia extent (Cottingham et al., 2014).
Using a growth chronology approach, Doubleday et al., (2015) examined growth in black bream across two different climatic regions: a wetter and cooler Tasmania (across 21 years) and hotter and drier South Australia (across 13 years). Environmental predictors, including air temperature (as surrogate for water temperature), rainfall, river flow and Southern Oscillation Index (SOI) were investigated for their effects on growth. 'Estuary' was also included in growth models for Tasmania — this improve model fit for growth. Across populations, growth synchrony was evident, while prominent year-to-year growth variation also present suggest that growth was being driven by regional climatic variation (Doubleday et al., 2015).

For South Australian populations, black bream inter-annual growth was positively correlated with temperature and rainfall; growth was negatively correlated with temperature in Tasmania, possibly suggesting that climate warming may have greater negative impacts on growth for Tasmanian populations (Doubleday et al., 2015). However, these differences among broader regional areas could also be due to other confounding factors, such as effects of fishing possibly biasing fish samples towards faster-growing younger individuals. Thus, further investigation is required to confirm spatially driven growth patterns are attributed climatic effects on local environmental factors.
In a related study, Izzo et al., (2016) also used growth (otolith) chronology to examine long-term growth patterns of black bream (cohorts spanning 1978 to 2011) and relationships with several environmental predictors (salinity, air temperature, rainfall) in the Coorong estuary. Izzo et al., (2016) found for black bream, much of variation (75%) in growth were solely attributed to the fish age on otolith increment width. Environmental predictors only explained a small amount of additional variation (4%) among all species understudy. For Black bream, the best fitting model included salinity (range: 1–37 psu), but it only had a small, negative relationship with growth.

Size/age at Maturity

Size at Maturity: SAM_{50} Female: 289 mm (TL) Male: 340mm (TL) (Cheshire et al., 2013) (Coorong). SAM_{50} Female: 157 to 218mm (TL) Male: 145 to 212mm (TL) (Sarre and Potter, 1999) (WA estuaries).
Age at Maturity: 1.9–4.3 years for fish in WA estuaries (Sarre and Potter, 1999). 3+for fish in the Onkaparinga River, SA (Harbison 1973 *cited in* Earl et al., 2016).
Fecundity and eggs: Multiple spawner, releasing eggs in batches over the spawning season (Sarre and Potter, 1999). Fecundity is indeterminate (Earl et al., 2016) although a large female was estimated to contain 3 million eggs (Ye et al., 2024). Eggs are 0.7-0.8mm in size (Neira et al., 1998).

Movement

Black bream largely reside in estuaries, although is capable of extensive movement across its lifetime. Movement in black bream varies among individuals, with differing movement strategies evident within a population (Elsdon and Gillanders, 2005; Gillanders et al., 2015; Giraldo, 2015). Partial migration is observed where some remain as residents, while others frequently move to the marine environment or, beyond to other estuaries (Elsdon and Gillanders, 2005; Gillanders et al., 2015; Giraldo, 2015; Bice, 2023). Freshwater flow into estuaries is associated with movement, with reports that black bream leave the estuary during floodwaters (Starling 1988, Holt 1978 *cited in* Norriss et al., 2002; Bice, 2023; Ye et al., 2024). Downstream movements occur during large flows (Sakabe and Lyle, 2010; Williams et al., 2017; Bice, 2023) and upstream movements as flows subside (Sakabe and Lyle, 2010; Bice, 2023). Movements also occur with tidal cycles (Sakabe and Lyle, 2010). Black bream move extensively in Gippsland lakes, with an average of 8.7 km/day some fish travelling up to 2600 km/year (Hindell et al., 2008). Migration upstream into rivers likely for purpose of spawning (Hindell et al., 2008). Longer distance dispersal across multiple estuaries among the Fleurieu peninsula and Kangaroo island has also been observed using otolith microchemistry (Giraldo, 2015). While a reported tagged fish capture in an adjacent estuary (Hindmarsh River), is indicative that fish leaving the Coorong could also use nearby estuaries (Bice, 2023), movements between estuaries do not contribute significantly to gene flow across populations and perhaps reflect desire to return to natal waters (Sarakinis et al., 2024).

Spawning

Spring and summer in Tasmania (Sakabe et al., 2011), Spring and summer in Victoria (Hassell et al., 2008a; Jenkins et al., 2018) and South Australia (Earl et al., 2016; Ye et al., 2019). Gonad development peaks in Spring in the Gippsland lakes, Victoria (Walker and Neira, 2001), Moore River, Western Australia (Sarre and Potter, 1999) and the Little Swanport estuary, Tasmania (Sakabe, 2009). Gonad development is asynchronous with daily spawning across the spawning season (Haddy and Pankhurst, 1998). Spawning occurs during periods when estuaries may become increasingly saltier. Spawning occurs at the interface of saltwater and freshwater, along the 'salt-wedge' (Sherwood and Backhouse, 1982), upstream in deeper water where saltwater ingresses into deeper pools (Newton 1996). Under low flow conditions,

spawning may occur well upstream (Sakabe et al., 2011). With reduced freshwater flows and increased migration of the salt wedge upstream, spawning progressively moves upstream (Williams et al., 2020).
The timing of spawning can vary and may also be delayed until the right conditions are present (Jenkins et al., 2018; Ye et al., 2019), sometimes spawning is delayed until late summer (Jenkins et al., 2018; Ye et al., 2019) or autumn (Smith et al., 2009; Ye et al., 2019). It's postulated that if high freshwater flow events still occur during spring (Newton et al. 1996) and a halocline is absent, then spawning may be deferred. In WA estuaries with mouth closure, bream may not spawn until freshwater rains promote suitable hydrological conditions (Norriss et al., 2002).

Environmental requirements

Temperature: Lenanton (1977, *cited in* Norriss et al., 2002) observed black bream to live in environments with temperatures ranging from 9.5 °C–25.5 °C. In the Coorong where black bream reside, Noell et al. (2009) reported that between October 2006 and September 2008, mean monthly temperature in the Coorong ranged 11 °C–25 °C. McNeil et al., (2013) subsequently used proxies of 14 and 23 °C to represent anticipated winter and summer temperatures in the Coorong. Upper limit tolerance of black bream has been reported between 33–34 °C (Weng 1971 *cited in* Beerkens, 2016). Examination of anaerobic scope over the range of 21–30 °C revealed little energetic consumption (1.54) over the 10 °C range, indicating a broad and stable thermal tolerance response, suggesting black bream could be considered an exceptionally eurythermal species (Beerkens, 2016). Determination of thermal maximum (CTmax) of wild black bream from four different estuarine populations (Southern to northern region of southwest Western Australia) found that CTmax ranged between 35.57 (± 0.43 °C) to 36.32 (± 0.70 °C) for Southern to northern populations (Gomez Isaza and Rodgers, 2024). Notably, the CTmax varied for individuals among estuaries, particularly for Serpentine River, with individuals demonstrating wider ranges (~1.5-2 °C higher and 0.5–1.5 °C lower) than other estuaries (Gomez Isaza and Rodgers, 2024).
Temperature and photoperiod was more important for black bream gonadal development (vitellogenesis) than salinity (Haddy and Pankhurst, 2000). Spawning was identified as occurring at temperatures > 15 °C (see Williams et al., 2013 and references there-in). Several studies have reported spawning under a range of different temperatures: 17.5–28.5 °C (WA estuaries, Sarre and Potter, 1999), 15-22 °C (Gippsland lakes, Vic, Walker and Neira, 2001), 17–22 °C (Blackwood River, WA, Williams et al., 2020), which likely reflects both temporal variation over the duration of the spawning period as well as variation in geography of black bream populations. Williams et al. (2020) presented data on the minimum temperature associated with spawning, indicating minimum temperature varies considerably with populations in higher latitudes spawning at lower temperatures, thus populations in warmer locations spawn at higher minimum temperatures (Williams et al., 2020). In the Coorong, spawning occurs during Spring and Summer when water temperatures are elevated with successful spawning noted at temperatures of 18−25 °C (Ye et al., 2019). Eggs have been recorded in the field 14.9–23.1 °C (Newton, 1996) and larvae recorded in the field 9.3–23.1 °C (Newton, 1996).
Salinity: Black bream can tolerate hypersaline conditions with an observed upper limit in the Coorong of 60 psu (Noell et al., 2009). Juveniles (4-6 months old) kept at 22-24 °C can survive and grow in acclimated salinities from 0 to 48 psu, but growth was greatest at 24 psu (Partridge and Jenkins, 2002). Tolerance to low salinity through direct transfer may be a function of age, as older fish may have greater tolerance (Partridge and Jenkins, 2002). One observation of large black bream has been made at 114 psu (Partridge and Jenkins, 2002). Black bream may become stressed at ~60 psu and grow slower compared to fish in 36 psu (Partridge and Jenkins, 2002; Hoeksema et al., 2006). Black bream that are osmotically stressed show external signs of sores/legions and exhibit lethargy (Partridge and Jenkins, 2002; Hoeksema et al., 2006; McNeil et al., 2013b). Juvenile black bream from the Coorong have been found to tolerate up to a maximum of 90 psu (McNeil et al., 2013b) with LC_{50} of 84.7 psu at 14 °C and LC_{50} of 88.4 psu at 23 °C (McNeil et al., 2013b). Black bream have been observed in the Coorong in salinities up to 60.1 psu during the millennium drought (Noell et al., 2009). High salinity (83–85 psu) associated with mass fish kill of juvenile Black bream (age 0+and 1+) in Culham Inlet, WA (Hoeksema et al., 2006).
Spawning has been observed in salinities as low as 3.5–8 psu in the Moore River Estuary (WA) to as high as 41–45 psu in the Wellstead Estuary (WA) (Sarre and Potter, 2000). Williams et al (2020) indicated successful spawning at salinities > 15 psu. Eggs (2.1–30.5 psu) and larvae (2.8–34 psu) found in a broad range of salinities in the Hopkins River; however, most larvae were found in salinities greater than 27.5 psu (Newton

1996). Abundances of eggs and larvae have been observed in salinities of 20–30 (Newton, 1996; Nicholson et al., 2008; Williams et al., 2013). Black bream egg and larval abundance and survival is greatest at salinities between 10–20 psu (Jenkins et al., 2015). Intermediate salinities and high copepod concentrations at the halocline maximises the chance of successful hatching and larval survival (Williams et al., 2013).

Haddy and Pankhurst, (2000) reported that survival-to-hatch was high in eggs incubated between 15 to 35 psu, and survival was lower at salinities 5 and 10 psu. Larvae developed deformities < 15 psu (Haddy and Pankhurst, 2000). Very low survival of eggs in salinity < 10 psu (Hassell et al., 2008a). Newton et al. (1996) indicated spawning salinities of the magnitude of 20–25 psu was needed for black bream spawning. Spawning is likely to diminish upon the onset of higher salinities and reduced oxygen concentrations (Newton et al. 1996).

Oxygen: Field studies have identified that hypoxia is a significant threat to black bream eggs and larvae (Newton, 1996; Nicholson et al., 2008; Williams et al., 2013; Williams et al., 2020). Hypoxia has significant effects on black bream egg hatching (Hassell et al., 2008a). In hypoxic conditions (50% oxygen saturation) very few eggs hatched when salinity was < 35 psu and temperature at 16 °C (Hassell et al., 2008a). No hatching of eggs occurred at 30% saturation (2.1 mg/L; Hassell et al., 2008b). Time to hatching was not affect by hypoxia or salinity (Hassell et al., 2008a).

Cottingham et al (2014) identified that increased area of hypoxic water and reduced freshwater inflow in the Swan River estuary, WA, corresponded to reduced condition, slower growth and reduced size, and delayed maturity in black bream. These effects were potentially attributed to increased density-dependence and potentially a loss of food resources, owing to hypoxia driving fish from deeper waters to oxygenated shallow water (Cottingham et al., 2014). A subsequent study by Beerkens, 2016 set out to characterise aerobic scope of black bream across different temperatures, revealed the energetic capacity for growth was reduced below ~60% saturation and was zero at 30% saturation. Furthermore, acoustic tracking of the fish also revealed avoidance of deeper hypoxic zones and movement of fish towards normoxic, surface waters. As hypoxic waters subsided with the onset of winter, fish were observed to return to these deeper waters. These results, together, support the earlier assertion of Cottingham et al. (2014) that hypoxia was an important driver, responsible for declines in growth rate.

Hypoxia has been identified as a major contributing factor to several fish kills involving Black bream (Becker et al., 2009; Smith et al., 2009). Becker et al., (2009) observed extreme hypoxic (< 1 mg/L) levels of dissolved oxygen during an artificial mouth opening event which resulted in a fish kill that included black bream. Smith et al., (2009) also noted that extreme hypoxic dissolved oxygen levels (< 1 mg/L) occurred around the time a large fish kill involving juvenile and adult black bream in the inland saline lake, Lake Indoon, WA.

pH: Sarre et al., (2003) describe water quality for the culture of juvenile Black bream (mean: 74 ± 7.9 mm) in inland water bodies and reported values of pH between 8.3 – 9.1. Low pH (4-5) has been implemented with fish kills of black bream and 8 other estuarine species in Anglesea River, Victoria (Pope, 2010). The probable cause of the fish kills in Anglesea in September 2010 was a combination of pH stress, aluminium toxicity and suffocation through smothering of the gills by precipitated aluminium compounds. Other factors may also have contributed, though likely to a lesser degree, these include salinity stress and toxic effects from other metals as a function of low pH (Pope, 2010).

Recruitment

Black bream has episodic recruitment (Morison et al., 1998; Ye et al., 2024), which suggests that suitable conditions for spawning and survival from egg to larvae to juveniles occur infrequently and reflect variability in the estuarine environment (Jenkins et al., 2015). Spring temperatures may be important for year class strength (Earl et al., 2016) and increased temperatures in conjunction with an environmental water flow pulse, increased food and favourable salt wedge conditions would be beneficial to larval growth and survival and subsequently recruitment (Ye et al., 2019).

Eggs and larvae have been collected from estuaries, particularly river channels, with distribution of eggs and larvae closely affiliated with the salt-wedge (Ye et al., 2019). Eggs are buoyant and require specific salinity to float on the salt wedge (Newton, 1996; Hassell et al., 2008a), with salinities of 10-20 probably important to ensure buoyancy and the eggs remain pelagic (Williams et al., 2020). Ye et al., (2019) describes three important functions of the salt wedge to black bream spawning and recruitment: (1)

provides a cue for spawning and for locating spawning ground, (2) provides a cue for larvae and juveniles to locate suitable habitat, and (3) increases food availability for larvae.

Variability in freshwater flows affect recruitment success (Sarre and Potter, 2000; Jenkins et al., 2010; Sakabe et al., 2011; Ye et al., 2019). Salt wedge and stratification of the water column relates to freshwater flow, with higher recruitment corresponding to intermediate flows in the Gippsland lakes, Victoria (Jenkins et al. 2010). High freshwater discharge from rainfall events may flush the salt wedge from estuaries (Newton, 1996). In the Coorong, no recruits have been detected following high flow/flood years (2010/2011, 2016/2017, 2022/2023), likely due to disturbance of the salt wedge (Ye et al., 2024). Strong recruitment in the Coorong, the highest in the last 15 years of monitoring, was detected in 2021-2022, corresponding with low-moderate barrage discharges of 11,000–24,000 ML/day with warm temperatures (19–22 °C) during January and February 2022 (Ye et al., 2024). These results fit within the wider conceptual understanding that intermediate flow events facilitated salt wedge development were important to black bream recruitment (Jenkins et al., 2010; Williams et al., 2013; Ye et al., 2019; Ye et al., 2024).

Low oxygen is thought to result in reduced survivability of eggs and larvae, leading to recruitment impairment (Hassell et al., 2008a; Nicholson et al., 2008; Williams et al., 2020). Increased numbers of dead eggs collected in samples were possibly related to increase amounts of hypoxia in the Hopkins River compared to the Glenelg River (Nicholson et al., 2008). Nicholson et al. (2008) proposed that hypoxia (< 40% saturation) is influential to egg or larvae survival.

As a valuable fishery species, black bream recruitment is measured and assessed on an annual basis (Ye et al., 2024). Targets are set against recruitment, both in terms of number of fish captured per unit effort (> 0.35 YOY/net night) in a survey year and the spatial extent of recruits detected with a fixed proportion of sites (> 50%). Only twice in the last 15 years of monitoring have both these targets been met, once during a high flow year and another during a low flow year, while similar conditions in other years did not produce similar results. This highlights the need for further work to understand factors behind the variable nature of recruitment in the Coorong (Ye et al., 2024).

Bony herring (Nematalosa erebi)

Life history

Bony herring are large bodied native species distributed broadly across slow flowing and standing waters (Koehn and O'Connor, 1990; Puckridge and Walker, 1990; Lintermans, 2007; Pusey et al., 2017) and complete their life cycle in freshwater (Puckridge and Walker, 1990). Regionally, they have been considered a freshwater-estuarine opportunist (Bice et al., 2018a; Ferguson et al., 2018) given they can be found at times in the estuaries (Ye et al., 2012), even under high salinity (Ye et al., 2012; Bice et al., 2018a). Bony herring are considered to have an opportunist life-history strategy (Ferguson et al., 2018; Price et al., 2019) which is characterised by high growth rate, short longevity, small body size and maturity at a small size (Winemiller and Rose, 1992; Ferguson et al., 2018) with reproduction allowing them to re-populate quickly after disturbance or, where continuous high mortality occurs in the adult stage (Winemiller and Rose, 1992).

Distribution, population structure and utility

Bony herring are one of the most widely dispersed freshwater fish across Australia (Ye et al., 2010; Pusey et al., 2021). Their distribution expands the Murray-Darling Basin, Lake Eyre Basins, much of Northern Australia, including coastal and inland catchments of Queensland, Northern Territory, Western Australia and parts of Northern New South Wales (Pusey et al., 2021). Bony herring inhabit a Rivers, lakes, wetlands, estuaries, lagoons, floodplains, impoundments, open water, bare substrate, large woody debris or snags, shallow, deep, rocky areas (Koehn and O'Connor, 1990; Puckridge and Walker, 1990) tube worm's reefs (Schroder et al., 2025).

The distribution of Bony herring across large parts of the continent is likely to reflect tolerance to wide environmental conditions including temperature, salinity and hydrology (Merrick and Schmida, 1984; Lintermans, 2007). Although, bony herring have known sensitivity to winter low water temperatures (Puckridge et al., 1989) and likely cold water thermal pollution (Lintermans, 2007). Population structure is not well known within the Murray-Darling Basin, but one study investigating basin-scale connectivity (Hughes and Hillyer, 2006) identified that high degree of connectivity among drainages within the Murray-Darling and Lake Eyre Basins. Bony herring are an important forage fish species for large bodied native fish, particularly Murray cod in the lower Murray River (Puckridge and Walker, 1990; Lintermans, 2007) as well as, making important contributions to the diet of other large piscivores in the Coorong, such as Mulloway and aquatic birds (Lintermans, 2007; Giatas and Ye, 2015; 2016). In the lower Murray region, bony herring are a primary species taken as part of the Lakes and Coorong Fishery, mostly for bait supply to the Southern and Northern Zone Rock Lobster fisheries (Earl, 2020). Bony herring are not targeted by recreational fishers (Earl, 2020).

Diet

Bony Herring have a broad diet, consuming algae, detritus, plankton, insect larvae (Pusey et al., 2021) and plant matter (Balcombe et al., 2005; Sternberg et al., 2008) and have been previously considered a detritivore, algivore, planktivore or combination of (Pusey et al., 2017). Ontogenetic changes in diet have been observed with juveniles consuming zooplankton before switching to detritus as they increase in size (Pusey et al., 2021). Detritus derived from terrestrial vegetation is an important source of Carbon for adults in Northern Australia (Pusey et al., 2021), with algae also a significant source of Carbon for large individuals (Pusey et al., 2021).

Larvae: zooplankton (Puckridge and Walker, 1990)

Juveniles: microinvertebrates, zooplankton (Pusey et al., 2017)

Adults: detritus, algae, microcrustaceans, zooplankton, insect larvae (Medeiros and Arthington, 2014; Pusey et al., 2021)

Age and growth

Max age: 11 years (Ye et al., 2025)

Bony herring may attain a maximum length of 470 mm TL (Lintermans, 2007; Earl, 2020). By the end of their first year (age 0+), bony herring reach 80 mm (Briggs and McDowall, 1996).

Size/age at Maturity

Size at Maturity: SAM_{50} for Males is 159 mm TL and 199 mm TL for Females (Puckridge and Walker, 1990)

Age at Maturity: in second or third year in lower Murray (Puckridge and Walker, 1990). Males begin to mature in their first year in life (age 0+) and females begin to mature in their second year (age 1+) (Puckridge and Walker, 1990).

Fecundity and eggs: ranges from 33,000 for a 199 mm TL (88.9 g) fish, up to 880,000 for a 403 mm TL (595.4 g) fish (Puckridge and Walker, 1990). Mean egg size is reported to be 0.83 mm in diameter (Puckridge and Walker, 1990).

Environmental requirements

Despite being widely distributed, very little is known about environmental requirements of bony herring, especially for juveniles and larvae (Ye et al., 2010). The distribution of bony herring does suggest a tolerance of warmer water, while observations have also noted a tolerance to high water turbidity (McNeil et al., 2013a).

Temperature: Bony herring can tolerate temperature from 9 to 38 °C (Merrick and Schmida, 1984).

Salinity: Hart et al., (1991) describes most freshwater fish can cope with salinities up to around 10 psu, while Bony herring are thought to be much more salt tolerant. Reported to tolerate salinity up to 39 psu (Lintermans, 2007) and frequently occurs in brackish water in the Coorong estuary (Ye et al., 2012). Observed to be present in salinities as 72 psu (Ye et al., 2012).

Oxygen: Mortality of bony herring observed as part of three fish kill events in Darling River during 2018–2019 (Sheldon et al., 2022). During these events, it was considered that dissolved oxygen concentrations fell below a critical limit of 2 mg/L (Sheldon et al., 2022).

pH: Has a low tolerance to pH < 6 and absence from some catchments in Northern Australia thought to be due to this intolerance (Pusey et al., 2017).

Movement

Bony herring are known to have extensive movements with both adults and juveniles undertaking migration (Pusey et al., 2017). Large numbers of bony herring of different sizes have been observed moving though fishways (Mallen-Cooper, 1999; Stuart and Mallen-Cooper, 1999; Mallen-Cooper and Stuart, 2003; Zampatti et al., 2010). Upstream movements are noted (Stuart and Mallen-Cooper, 1999), as well as downstream and lateral movements to the floodplain (Geddes and Puckridge 1989 *cited in* Ye et al., 2010). It is unclear if movements are related to spawning (Ye et al., 2010).

Spawning

Spawning in the Southern Murray-Darling Basin occurs during summer from December to February at temperatures of 21–23 °C (Puckridge and Walker, 1990; Ye et al., 2010). In the tropics there is little seasonality to spawning (Puckridge and Walker, 1990) and in other locations spawning may occur multiple times throughout the year (Pratt et al., 2024). In lake Alexandrina, spawning has been observed to occur over bare sand substrate (Puckridge and Walker, 1990). Spawning has also been observed to occur in shallow backwaters (Llewellyn, 1983).

Recruitment

Bony herring eggs (in later stages) and larvae are pelagic and undergo drift during this early life period (Gilligan and Schiller, 2003). Although, unlike other large bodied native fish, recruitment does not seem to be impaired by hydrological modifications (Puckridge and Walker, 1990). Floods are not necessary to initiate spawning in bony herring and flooding is also not required to facilitate recruitment, however,

inundation of floodplains and subsequent improvement of feeding conditions for juveniles mean floods may enhance recruitment (Puckridge and Walker, 1990; Ferguson et al., 2013). Lower numbers of Bony herring larvae have been identified in some low flow years in the lower Murray River, suggesting that drought could have some impact on recruitment (Cheshire and Ye, 2008).

Greenback flounder (Rhombosolea tapirina)

Life history

Greenback flounder are classified as Marine-estuarine opportunists, given they are a marine species that enters estuaries during the juvenile and early adult life stages and use the marine environment also as habitat (Earl, 2014; Earl and Ye, 2016; Bice et al., 2018a). Ferguson et al., (2018) considers Greenback flounder to be intermediate between Opportunist and Periodic life-history strategies, as it is a medium-size species, that is fast growing, early maturing and high fecundity (Ferguson et al., 2013; Ye et al., 2021).

Distribution, population structure and utility

Distribution extends from south-eastern WA, around the Southern coasts of the continent and Tasmania, and up to Southern NSW (Earl, 2014) It also occurs in New Zealand (Sutton et al., 2010). Greenback flounder is targeted by commercial, recreational and first nations fishers across its distribution (Ferguson, 2007; Earl and Ye, 2016). Greenback flounder occupy a broad range of environments and habitat: sea, marine embayment's, estuaries, lagoons, salt marsh, sand and fine sediment bottoms, shallow water, deep water, open areas, macrophytes, beaches, tube worms' reefs (*Ficopomatus enigmaticus*) (Butler and Jernakoff, 1999; Earl, 2014; Ye et al., 2016; Prahalad et al., 2018; Earl, 2019; Schroder et al., 2025). The wider population and thus stock structure of Greenback flounder in SA is uncertain, and there has been evidence of fish moving between the Coorong estuary and the adjacent marine environment (Earl and Ye, 2016; Earl et al., 2017).

Diet

Greenback flounder is a carnivore, generalist consuming a variety of annelids, crustacea, molluscs, insects, fish and algae (Crawford, 1984; Earl, 2014). In the Coorong estuary, polychaetes and amphipods are common in the diet and large freshwater flows influence diet composition, likely as a change in prey availability (Earl, 2014).

Larvae: rotifers (Hart, 1995)

Juveniles: amphipods, copepods, mysids and polychaetes (Crawford, 1984)

Adults: polychaetes, crustacea, molluscs, fish, insects, algae (Earl, 2014)

Age and growth

Max age: Reaches a maximum age of 10 years (Sutton et al., 2010; Earl and Ye, 2016).

Reaching a maximum size of around 450 mm (Ferguson, 2007). Greenback flounder are a fast-growing fish attaining lengths around 220 mm TL in its first year of life (Earl et al., 2014). In the Coorong, the age structure consists of fish up to age 3+years and growth does not differ between sexes with spatial differences in growth marginal (Earl et al., 2014).

Size/age at Maturity

Size at Maturity: Estimated size at maturity for females and males in the Coorong estuary is 198 and 211 mm TL, respectively (Earl 2014).

Age at Maturity: Both males and females reach maturity early in their second year (Earl and Ye, 2016).

Fecundity and eggs: Multiple batches of eggs released over protracted season (Earl and Ye, 2016). Fecundity is indeterminate (Earl, 2014), although an estimate of 820,880 to 1,969,070 for individuals of 247 and 343 mm, respectively (Crawford, 1984). Eggs are pelagic. Egg size ranges between 0.73–0.87 mm (Crawford, 1984).

Environmental requirements

Temperature: Hart, (1995), Hart and Purser, (1995), Hart et al., (1996) examined temperature and salinity on eggs and larvae of Greenback flounder. Hart (1995) found that the incubation temperature for Greenback flounder eggs was optimum between 9-12°C and that incubation temperatures that were below 6 °C or greater than 21 °C resulted in complete mortality.

Larvae cultured at 18 °C were smaller than those cultured at 9, 12 or 15 °C, but were significantly larger than those cultured at 21 °C (Hart and Purser, 1995). Larvae held at 18 and 15 °C grew fastest. Larvae at 21 °C show poor growth 40-50 hours post hatch and complete mortality after yolk absorption (Hart and Purser, 1995). The optimal temperature for yolk absorption was 15°C (Hart and Purser, 1995). Increasing temperatures above 15 °C, yolk absorption accelerated but the time of first feed was not established at these temperatures (Hart and Purser, 1995).

In a subsequent experiment examining growth at temperatures of 15, 16, 17, 18, 19 or 20 °C, Hart *et al.* (1996) could not identify optimal temperatures for growth of larvae but noted that temperatures of 19-20 °C gave better growth rates than lower temperatures. Similarly, no clear temperature effect on survival trends were observed, except 16°C result in significantly lower survival of larvae than 17 or 19 °C (Hart et al., 1996).

May and Jenkins, (1992) observed relationships between air temperature and growth rates in newly settled Greenback flounder juveniles in shallow waters of Swan Bay, Victoria. Early in the settlement season, temperature positively correlated with otolith growths, but by later in the season, temperature was found to negatively correlated with otolith growth, suggesting temperature had exceeded a critical threshold for growth (May and Jenkins, 1992). Roper (1979, *cited in* Hart et al., 1996) reported that the upper thermal limit for juveniles was 32.6 °C.

The temperature tolerance of adults is not well understood, but there have been studies of the Coorong that describe a range of water temperatures with Greenback flounder present (e.g. Earl *et al.* 2014). Earl *et al.* (2014) report presence between water temperatures 27.8 °C and 11 °C over the years 2009-2010.

Salinity: Hart (1995) found that at 5psu, fertilisation of eggs did not occur and at 15psu a low fertilisation rate of 15%. Eggs commonly hatched between salinities of 15–45 psu, but hatch rates were contingent on temperature (Hart 1995). At 35 psu, eggs are buoyant and float on the surface, below 27-28 live eggs sink but can still develop (Hart, 1995). Hart *et al.* (1996) examined growth and survival of Greenback flounder larvae at 15, 25 and 35 psu over a period of 30 days. Hart *et al.* (1996) found growth did not differ among treatments, but at 15 psu, a significant but marginally lower larvae survival than 25 or 35 psu (91.3% vs 94.7% and 100%). Short term (up to 29 hours) salinity preference of larvae and juveniles has been examined by Crawford (1984) using an artificial salinity gradient (0–33 psu). Larvae were observed in similar proportions to use higher salinity (35%, 32–34 psu) low salinity end (30%, 0–5 psu), however, young juveniles that had undergone recent metamorphosis (up to 7 weeks) preferentially selected low salinities (0–5 psu) (Crawford 1984).

Juveniles can tolerate a wide range of changes in salinity and are often found in the upper reaches of estuaries (Last et al. 1983 *cited in* Ferguson, 2007). While drought breaking floods in the Coorong in 2010, which lead to a lowering of salinity across the Coorong estuary provided conditions that were well-received by Greenback flounder (Earl, 2014). Specifically, the reduction in salinity meant an increase in the size of the nursery area and thus recruitment success (Earl and Ye, 2016).

In a study examining the salinity tolerance of Greenback flounder acclimatised to seawater (33 psu), Girling et al., (2003) showed that juveniles were tolerant to salinities of 3, 7, 15 and 40 psu under direct transfer, but did not survive upon transfer to 0 psu. In another study investigating salinity tolerance of juvenile Greenback flounder using slow acclimatisation, McNeil et al. (2013) showed juveniles can tolerate up to a maximum of 92 psu at 14 °C (LC_{50} estimate of 87.5 psu), and 82 psu at 23 °C (LC_{50} estimate of 78.6 psu). The salinity tolerance range for juveniles might be narrower than adults due to a less developed osmoregulatory system (Hart, 1995).

A movement study examining the distribution of large (221–313 mm TL) Greenback flounder in the Coorong estuary, Earl et al., (2017) observed individuals to occupy a broad range of salinities 487-111,815 µS/cm (~0.3-67 psu). These salinities were largely found in the 'inner estuary' where hypersaline waters mixed with freshwater released from the barrages, resulting in a transitional zone that is rich in benthic fauna that are preferred prey for greenback flounder (Earl et al., 2017).

Oxygen: Low oxygen levels of 1.2 mg/L, higher water temperatures (28.5 °C) and higher salinity (60 psu) were found associated with a summer fish kill of yelloweye mullet in Lake George, South Australia (Hall, 1984).

Movement

Earl et al., (2017) employed acoustic telemetry to investigate movement in 20 large Greenback flounder (221–313 mm TL) over a period of 7 months across the Coorong during higher freshwater flows. They found individuals spread over a large area with fish exhibiting complex movement patterns that differed among individuals. Three kinds of movement pattern was identified: (1) movement within the estuary, (2) dispersal to the sea and (3) frequent return migrations from the estuary to the sea. Fish were also observed to make diurnal movements between deeper and shallower areas (Earl et al., 2017). Movements were also associated with the distribution of salinity with tagged fish spending most of their time (91%) below 60,000 µS/cm (~40 psu salinity) (Earl et al., 2017). Response to low salinity water from marine salinity during larval development is likely important for guiding larvae into estuarine nursery habitat (Crawford, 1984).

Spawning

Reproduction occurred during the cooler months from March to October in both estuaries and offshore environments (Kurth, 1957, Crawford, 1984). Gonadal development commenced in autumn when temperatures were below 20 °C and peaked in June when temperatures were ~12 °C. (Ye et al., 2021). Within the Coorong, spawning of greenback flounder occurs from March to October, peaking between April and July and most frequent between the temperatures of 11–14 °C (Earl 2014). Highest temperature report for spawning is 19°C (Earl 2014). Spawning occurs in deeper areas of estuaries and offshore (Earl, 2014). In Tasmania, sexually mature females are found in shallow water (5-10m) and males in deeper waters (10–25m) (Kurth, 1957; Crawford, 1984; Ferguson, 2007). Reproductive development and spawning does not appear to be affected by salinity, but salinity likely has a greater influence on recruitment (Earl, 2014).

Eggs are planktonic with fertilised eggs were buoyant in salinities above 27-28 psu (Hart, 1995). A salinity of 35–45 psu was found to be optimal for initial fertilisation with incubation able to be carried out in salinities ranging from 15–45 psu with no effect on subsequent hatch rates (Hart, 1995).

Recruitment

The contribution of recruits from the marine environment to the Coorong estuary is currently unknown (Earl and Ye, 2016). In the Coorong, recruitment of age 0+ occurs in summer and autumn each year (Earl et al., 2014). Post-larval recruits settle out into shallow, unvegetated sandy habitats between July and October (May and Jenkins, 1992). Growth in juveniles (age 0+years) is related to air temperatures positively correlated with growth for winter cohorts but negatively correlated with growth for spring cohorts suggesting that high temperatures may have deleterious effects on metabolism (May and Jenkins, 1992). Growth rate is relatively high within the Coorong, with individuals attaining around 65% of their asymptotic length (220-230 mm) in the first 12 months (Earl et al., 2014). Growth among sexes is similar, as is growth between the Murray estuary and North Lagoon (Earl et al., 2014).

During the Millenium drought (2002–2010) the biomass of greenback flounder was an order of magnitude lower (4 tonne/year) compared to peak levels (62 tonne/year) in 1990–92 (Ye et al., 2023). Drought-breaking freshwater inputs from flooding into the Coorong in 2010 reduced salinity and improved habitat quality, resulting in increased strength of age 1+years cohort that developed into a rare dominant mode of age 2+being retained (Earl et al., 2014). This cohort from 2010 dominating the age structure in 2012 was detected in fisheries catch until 2015. In drought years, young-of-year were almost completely

excluded from the North and South Lagoons due to elevated salinities but returned in higher abundances following post-drought years (Ye et al., 2023). It appears that freshwater flow and salinity are important for maintaining nursery habitat in the Coorong (Ye et al., 2023).

Mulloway (Argyrosomus japonicus)

Life history

Mulloway are classified as having a Periodic life-history strategy (Ferguson et al., 2018) which is characterised by high longevity, large body size and delayed/late maturity, allowing them to maximise lifetime fecundity (Winemiller and Rose, 1992). Mulloway are classified as part of the Marine-estuarine opportunist fish guild (Bice et al., 2018a). Large adults use the marine environment and can be found on nearshore reefs and surf beaches (Earl and Ward, 2014). Mulloway enter estuaries as young juveniles (Griffiths, 1996; Ferguson et al., 2008; Ferguson et al., 2014) but may also use the marine environment as nurseries or to complete other aspects of their life history (Hall, 1986; Griffiths, 1996; Bice et al., 2018a). Griffiths (1996) reported that *A. japonicus* of 150–400 mm can be found both in estuaries and in the surf zone, but fish between 20–150 mm were found exclusively in estuaries. Mulloway in estuaries are associated with deeper waters such as 'holes', channels, seagrass, mangroves and rocky areas (Taylor et al., 2006b; Russell et al., 2021). Ferguson et al., (2008) argued that the population of Mulloway associated with the Coorong depend on estuarine conditions, based on correlation between area of estuarine habitat and population size, as well as age structure and periods of increased freshwater flow into estuaries. Fishery catch also increases in years following significant flow events (Hall, 1986; Ferguson et al., 2008; Stewart et al., 2020).

Distribution and population structure

Mulloway are distributed across the Pacific and Indian Ocean waters, including the countries of Australia, India, Pakistan, China, Korea, Japan, India, Pakistan and Africa where it is also known by the common name 'Dusky Kob' (Griffiths, 1996; Silberschneider and Gray, 2008). Within Australia, Mulloway are found around the Southern coastline from Northwest Cape in WA to Burnett River in Queensland (Kailola et al 1993 *cited in* Silberschneider and Gray, 2008).

Previous genetic investigations of stock structure have observed differences between fish in Australia and South Africa (Silberschneider and Gray, 2008; Barnes et al., 2015). Four populations of mulloway are currently known to exist within its distribution across Australia, with individuals genetically grouping into a WA region, SA west coast region, SA-Vic region, and NSW-Qld region (Barnes et al., 2015).

Diet

Mulloway are carnivorous, consuming a wide variety of crustaceans, molluscs and fish (Hall, 1986; Taylor et al., 2006a; Giatas and Ye, 2015). With increasing size, there are clear shifts in the diet (Hall, 1986; Taylor et al., 2006a; Giatas and Ye, 2015). Small crustaceans (shrimp) are initially consumed before graduating to small fish species (e.g. Sandy sprat, gobies) and large crustaceans (prawns), with large individual consuming cephalopods (octopus) and larger fish species (e.g. mullet) (Giatas and Ye, 2015). Changes in diet also occur due to diet flexibility as a function of location (Taylor et al., 2006a).

Larvae: rotifers (Fielder and Bardsley, 1999, Battaglene and Talbot, 1994), copepods (Griffiths, 1996), Artemia (Battaglene and Talbot, 1994)

Juveniles: < 250 mm: mysid shrimps and prawns (Taylor et al., 2006a; Giatas and Ye, 2015), > 500 mm: fish and crustaceans (prawns) (Taylor et al., 2006a; Giatas and Ye, 2015)
Adults: fish, cephalopods, crustaceans (Giatas and Ye, 2015)

Age and growth

Max age: 42 years (Silberschneider and Gray, 2008; Ferguson et al., 2014)

The maximum reported length and age of *A. japonicus* is 1750 mm and 42 years, respectively (Silberschneider and Gray, 2008). In South Australia and Western Australia, growth differs between sexes with females reaching larger asymptotic size (Ferguson et al., 2014). Growth in South Australian population is lower than in Western Australia, New South Wales or South Africa (Ferguson et al., 2014). Juveniles grow rapidly attaining 350 mm TL in 1 year and 870–900 mm in five years in Australia and South Africa

(Silberschneider and Gray, 2008). Young larvae between 6–20 days old, had faster growth in low saline waters (5 psu) (Fielder and Bardsley, 1999).

Izzo et al 2016, examined inter-annual patterns in growth (otolith margin increment width) as a function of fish age and the effect of several environmental predictors (air temperature, rainfall, salinity, discharge and water level) of growth for Mulloway (ages 3–41 years) in the Coorong region. Most of the variance in growth was explained by intrinsic effects (age of fish) on otolith increment width, however, best fitting models also included two temporal random effects (cohort and year). Including cohort (birth year) as a random-effect and ordering individual cohort-levels by time (year) revealed periodic changes in growth. Environmental predictors explained a small amount of additional variation (4%) among all species studied, with mean air temperature over the annual growth year (November–October) positively related (not statistically significant) to mulloway growth. These results may indicate generalist fish like mulloway (an estuarine opportunist) have tolerance to environmentally unstable environments like estuaries (Izzo et al. 2016).
Nicolle et al., (2022) also examined inter-annual patterns in mulloway growth (otolith margin increment width), and effects of several environmental predictors on growth along the NSW coast, Australia. They applied a similar mixed-effects hierarchical statistical model to Izzo *et al.* 2016, but unlike Izzo *et al.* 2016 finding a positive relationship between interannual growth and temperature which was likely attributed to sampling fish from much broader geographical area (Nicolle et al 2022). No influence of rainfall or climate indices on growth, despite good relationships found with rainfall for recruitment over the sampling area. Although, given random effects of cohort were observed, it's possible these effects may indicate a growth response of individual cohorts to an external factor. Based on an optimum of 26 °C for growing, temperatures in the north of NSW in this study were typically 23–24 °C over the summer, suggesting that the growth of mulloway at the present is not approaching upper thermal limits. Nicolle et al. 2022 predict the mean annual growth rate of mulloway to increase in southeastern Australia by up to 8.9% by 2099 conditionally upon the temperature-growth relationship developed for the previous 40-year timeframe remaining true.

Size/age at Maturity:

Size at Maturity: SAM_{50} Female: 850 mm (TL) Male: 778 mm (TL) (South Australia, Ferguson et al., 2014), SAM_{50} Female: 1070 mm (TL) Male: 920 mm (TL) (South Africa, Griffiths, 1996). SAM_{50} Female: 680 mm (TL) Male: 510 mm (TL) (New South Wales, Silberschneider and Gray, 2008).

Age at Maturity:

Age at Maturity (50% individuals) for males and females is 2 and 3 years in eastern Australia, 5 and 6 years in WA and South Africa (Silberschneider and Gray, 2008). In South Australia, 5 and 6 years for males and females (Ferguson et al., 2014).

Fecundity and eggs: Fecundity Indeterminate in wild populations (Silberschneider and Gray, 2008) but Battaglene and Talbot, (1994) report two broodfish of ~10 kilograms shed around 1 million eggs. Eggs and larvae are pelagic (Russell et al., 2022). Egg size is 0.8 mm (Battaglene and Talbot, 1994).

Environmental requirements

Temperature: Mulloway are eurythermal (Harrison and Whitfield, 2006; Earl and Ward, 2014), given their propensity to be found in warm temperate to tropical estuaries (Harrison and Whitfield, 2006). In South Africa, mulloway have a natural temperature distribution of 12–28 °C (Collett et al., 2008), similar temperatures that are encountered broadly across their Australian distribution (14–24 °C Battaglene and Talbot, 1994, 14 °C Ferguson et al., 2014). In the Coorong, mulloway have been found over the range of 10–31 °C (Earl and Ward, 2014).

Populations in Southern Australia, form the most Southern distributions of the species and therefore winter temperatures are likely to be lower than optimal (25–-26 °C) for growth (Ferguson et al., 2014). Collett et al., (2008) investigated the optimal water temperature for the culture of juvenile mulloway, finding specific growth rates were the highest at 25.3 °C. The upper critical thermal limit of mulloway is unknown but in a

close relative the Meagre (*Argyrosomus regius*) inhabiting similar thermal ranges (14–26 °C), the upper thermal limit was observed to be 37.5 °C (Stavrakidis-Zachou et al., 2021). At 34 °C, there was poor growth performance and physiological status (Stavrakidis-Zachou et al., 2021).

The study of Nicolle et al. (2022) investing growth in mulloway across a temperate-subtropical climate found no indication of mulloway growth approaching their critical thermal maximum in the most-warmest environments (northern NSW and Southern QLD) under study.

Salinity: Silberschneider and Gray, (2008) indicate that distribution of mulloway in estuaries is linked to salinity, turbidity, freshwater flow and depth of water. Mulloway are euryhaline (Earl and Ward, 2014) as they can be found in different environments represented by differing salinity (or electrical conductivity) over their life cycle (Gray and McDonall, 1993; Griffiths, 1996; Fielder and Bardsley, 1999; Taylor et al., 2014).

Griffiths, (1996) reported that fish < 150 mm are found in the upper reaches of rivers where salinities are < 5psu. Gray and McDonall, (1993) also juvenile mulloway to be found upstream in low salinity waters (< 5 psu), but also near the estuary mouth where salinity was > 25 psu. Furthermore, most juvenile fish (10–20 cm TL) occurred at locations in an estuary where the salinity was 15 to 20 psu (Gray and McDonall, 1993).

In short studies examining the effects of salinity on growth and survival of larval and juvenile mulloway, Fielder and Bardsley, (1999) found that 6-day old larvae grew well over a 14-day period, exposed to a range of salinities (5, 12.5, 20, 27.5 and 35 psu) with fish in the 5 psu treatment obtaining a greater mean length compared to those in 12, 27,5 and 35 psu treatments. In a subsequent experiment involving juveniles (6.1 ±0.1 g/fish) stocked into different salinities (0.6, 5, 10, 20 and 35 psu) for 28 days, indicated that mulloway grown in 5 psu salinity had the greatest weight gain after 28 days, about 28.7% higher than fish grown in 35 psu (Fielder and Bardsley, 1999). However, immersion in freshwater (0.6 psu) resulted in the cessation of feeding and onset mortality after 3 days. Transferring remaining fish into low salinity (5 psu) resulted in full recovery (Fielder and Bardsley, 1999). No significant difference in salinity on growth, survival or food conversation ratio was observed between salinity treatments, although this study statistical power was considered low (Fielder and Bardsley, 1999).

Bernatzeder et al., (2010) also examined juvenile mulloway growth (< 150 mm, 9.6 ± 0.5 g/fish) at 3 salinity levels: 5 (hypoosmotic), 12 (isosmotic) and 35 psu (hyperosmotic) finding that fish held at higher salinity levels grew heavier on average, while reduced food conversion and growth was noted at 5 psu, attributed to greater energy expenditure (Bernatzeder et al., 2010). Contrary results found compared to Fielder and Bardsley (1999) may have been attributed to a slightly larger size individuals used and a longer trial period (9 weeks) (Bernatzeder et al., 2010). Bernatzeder et al., (2010) concluded that salinity alone does not explain the natural distribution of juveniles within estuaries.

Hall (1986) reported the ecological tolerance of Mulloway to salinity in the Coorong was below 60 psu and that Mulloway may not be found in salinities above 65 psu (Hall, 1984). Similarly, the maximum observed salinity in the field observed for Mulloway in the Coorong was 62.4 psu (Noell et al., 2009). In an experiment examining slow- acclimatization tolerance to increased salinity, McNeil et al., (2013) showed that juveniles can tolerate up to a maximum of 74 psu at 23 °C, with an LC_{50} of 64.4 psu at 14 °C and LC_{50} of 59.3 psu at 23 °C. In the same trial, juveniles were observed to stop feeding once salinity rose to 62 psu (14 °C) and 66 psu (23 °C) (McNeil et al., 2013b).

Movement

Mulloway move between the sea and estuaries as part of their life history (Silberschneider and Gray, 2008) and are vulnerable to changes in environments (Barnes et al 2019). Larvae emigrate from the marine environment into estuaries, where juveniles reside for 4–5 years (Gray and McDonall, 1993; Ferguson et al., 2008). Mulloway move into nursery habitat in estuaries when individuals are < 1 year old and 20–30 mm in length (Griffiths, 1996) with more recent otolith microchemistry work identifying movement into estuaries occurs ~40 days old (Russell et al., 2022). A small proportion of fish may remain in the marine environment throughout their life (Russell et al., 2022), which may be attributed to a lack of accessibility to estuaries (Farmer, 2008; Russell et al., 2022). In the Coorong, emigration to the sea habitat occurs at ~4–5 years old (Ferguson et al., 2014).

Most tagging studies on mulloway have been undertaken in estuaries (Barnes et al., 2019). A compilation of studies using external tags has revealed some individuals are sedentary while others move considerable distances between estuaries (see studies in Silberschneider and Gray, 2008). Most tag-recapture data on movements from NSW coast have shown a high proportion of individuals (87%) were recaptured from the same estuary that they were tagged in (Silberschneider and Gray, 2008). Although use of otolith microchemistry has revealed movements between NSW estuaries was more frequent than thought from previous tagging studies (Russell et al., 2021).

Investigations into movement and habitat use by Mulloway based on acoustic tagging have shown that they (wild-caught: 520–730 mm TL and stocked: 368.3 ± 9.5 mm) used deeper holes in the Georges River estuary, NSW. These fish also moved between the Georges River and Botany Bay, with a movement rate of up to 16 km per day observed (Taylor et al., 2006b). In the Glenelg River, Western Victoria, acoustically tagged mulloway between 501–781 mm (TL) showed a preference for the lower reaches of the estuary with evidence of upstream movements during periods of low flow into the estuary (Lieschke, 2019). Liesckke (2019) also observed 13 fish to leave the estuary mouth between November and January with 7 fish re-entering the estuary between 4 and 18 months later. These emigrations were associated with a higher freshwater discharge (Lieschke, 2019). Smaller mulloway (470–890 mm TL) moved closer to the sea during periods of lower salinity from elevated discharge into estuaries (Taylor et al 2014).

Mulloway can undertake large-scale movements of several hundred km's between estuaries along the NSW coast (Silberschneider and Gray, 2008). Hall (1986) used external tags to mark ~5000 individuals between 1981-1985 in the Coorong to examine Mulloway movements. Five fish were detected moving between to St Vincents Gulf with no individuals detected moving from St Vincents Gulf to the Coorong, although marking a greater number of individuals in St Vincents Gulf would be required to demonstrate movements away from this area (Hall, 1986). Of 24 individuals acoustically tagged in the Glenelg River, western Victoria, four fish were noted to make larger-scale movements between the Glenelg River and Coorong estuaries (Lieschke, 2019). Barnes et al (2019) examined movement in 19 mulloway on the west coast of South Australia using popup satellite archival tags, finding that mulloway made horizontal movements seasonally, with greater median pop-up distance from deployment site in autumn (212 km) vs summer (51 km). Fish were also observed to move from western South Australia into Western Australian waters. This study also observed marine-inhabiting mulloway to encounter depths between the sea surface and 56.5 m and temperatures of between 15.1 and 27 °C, but more typically fish encountered 18–24 °C during summer and autumn periods (Barnes et al., 2019).

Spawning

Spawning season in eastern South Australia extends from October to January with a peak in November (Ferguson et al., 2014). The timing of spawning generally occurs when conditions maximise the survival of offspring, which the spring-spawning period of Mulloway in South Africa, corresponds with the period of spring/summer rainfall (Griffiths, 1996). In Western Australia, spawning occurs when water temperatures are above 19 °C (Farmer, 2008). Spawning occurs in the nearshore marine environment (Silberschneider and Gray, 2008), likely in the surf zone (Barnes et al., 2019), the estuary (Farmer, 2008; Silberschneider and Gray, 2008) or marine environment adjacent to estuaries (Griffiths, 1996; Silberschneider and Gray, 2008; Ferguson et al., 2014). In the Coorong region, Mulloway spawn near the Murray River mouth during spring and summer (Ferguson et al., 2014). Hall (1984) postulated that freshwater outflows during summer may promote aggregations of spawning fish near the mouths of estuaries as peak discharge corresponded with the timing of the spawning season.

Recruitment

Eggs and larvae have been found in nearshore marine waters (Griffiths, 1996). Mulloway larvae have also been observed in deeper, NSW coastal waters (20–60m) (Silberschneider and Gray, 2008). Juveniles emigrate from the marine environment to protected habitat in estuaries where they grew to maturity under conditions of relatively low natural mortality (Ferguson et al., 2018), although use of estuaries may not obligative in all populations, with a small proportion of fish remaining in the marine environment across their lifetime (Russel et al 2022).

Freshwater flow has been identified as an important contributor to population dynamics and fishery production in mulloway fisheries (Ferguson et al., 2008; Earl and Ward, 2014; Stewart et al., 2020). Freshwater flow relates to cohort strength in Mulloway, with years of elevated freshwater discharge corresponding to strong year classes (Ferguson et al., 2008). Ferguson et al. (2008) found that the variability in freshwater flows explained 28% and 35% of the variability in mulloway year class strength for 2001 and 2002, suggesting a link between higher freshwater inflows and stronger year classes in the population associated with the Coorong. Stewart et al. (2020) observed that relative year class strength is related (r^2 = 52%) to rainfall over a 16-year period in Southeastern Australia. Commercial landings over another 29-year dataset were related to rainfall two (R^2 = 21%) and three (R^2 = 44%) years earlier. Year class strength of mulloway corresponded well with year class strength of a key prey species (prawn), suggesting that rainfall influences recruitment of mulloway by affecting food availability (Stewart et al., 2020). Drought conditions are expected to affect recruitment success through the availability of nursery habitat (Earl and Ward, 2014; Stewart et al., 2020; Russell et al., 2021).

Loss of estuarine habitat from flow regulation and drought may result in recruitment failure, which may make mulloway populations vulnerable to overfishing (Ferguson et al., 2008). In NSW, the mulloway population is assessed as depleted as a combination of low stock spawning biomass and the effect of coastal droughts (Stewart et al., 2020; Russell et al., 2021). Associated with higher freshwater flows is greater turbidity and a salinity gradient, which only occurs during high flow years in the Coorong (Geddes 1987 *cited in* Ferguson *et al*. 2008). Both low salinity and high turbidity are important to young mulloway, as they depend on these environmental features for refuge from predators, including conspecific adults (Hall, 1986; Griffiths, 1996; Ferguson et al., 2008; Ferguson et al., 2018). Turbidity has been experimentally isolated as important for newly settled mulloway (21.9 ± 5.3 mm TL) that spend significantly more time in higher turbidity water compared to controls (James et al., 2022).

Yelloweye mullet (Aldrichetta forsteri)

Life history

Yelloweye mullet are classified as part of the 'Marine estuarine opportunists' guild of fish as they opportunistically use estuarine conditions during the juvenile period of their life cycle but do not need access to complete their life cycle (Bice et al., 2018a Ye et al., 2016). They may take advantage of favourable conditions in estuaries, such as the high productivity associated with floodwaters (Mosley et al., 2024b). Although tolerant to estuarine conditions, rapid changes in estuarine conditions can also have catastrophic effects (see Tweedley *et al*. 2024). They are known to leave estuaries (Orr, 2000) and reside in nearby marine environments (Chubb et al., 1981).

Yelloweye mullet have an opportunistic life-history strategy (Ferguson et al., 2018), which is characterised by high growth rate, short longevity, small body size and maturity at a small size (Winemiller and Rose, 1992; Ferguson et al., 2018) with reproduction allowing them to re-populate quickly after disturbance or where continuous high mortality occurs in the adult stage (Winemiller and Rose, 1992).

Distribution, population structure and utility

Yelloweye mullet (*Aldrichetta forsteri*) are distributed in bays, estuaries and open coastline from Shark Bay in Western Australia, throughout the Southern coastline including Tasmania, to Newcastle in New South Wales. They are also found in New Zealand waters (Thomson, 1957; Kailola et al., 1993; Curtis and Shima, 2005; Earl and Ferguson, 2013). The stock structure of Yelloweye mullet is not well understood, but there are believed to be two stocks: eastern and western Australian stocks (Thomson, 1957; Pellizarri, 2001; Earl and Ferguson, 2013). Yelloweye mullet live in brackish and inshore coastal waters and beaches overs sandy and muddy bottom in depths to about 20 m (Earl and Ferguson, 2013). They are found frequently in the lower reaches of rivers and larger fish tend to be found in deeper channels are opposed to smaller fish that are found over shallower areas (Earl and Ferguson, 2013). They can also be found in lakes and lagoons (Egan et al., 2020; Giatas et al., 2024). Yelloweye mullet are associated with seagrass, mangrove, saltmarsh or tube-worm reefs (Butler and Jernakoff, 1999; Crinall and Hindell, 2004; Payne and Gillanders, 2009; Prahalad et al., 2018; Schroder et al., 2025).

Diet

Yelloweye mullet are omnivorous (Lamontagne et al., 2007; Giatas and Ye, 2016) consuming a variety of crustaceans, molluscs, detritus, algae, insects, plant material and fish (Thomson, 1957; Platell et al., 2006; Giatas, 2012; Mosley et al., 2024a). Ontogenetic shifts in diet composition is known (Platell et al., 2006; Giatas, 2012). In the Coorong small mullet (< 150 mm) fed on post-larval fish, detritus, amphipods, molluscs, polychaete worms, copepods and mysid shrimp (Giatas, 2012). Large yelloweye mullet consume algae, detritus and polychaetes (Giatas, 2012; Mosley et al., 2024a). In response to significant flood event in the Coorong estuary, large Yelloweye mullet may opportunistically change their diet from a dominance of detritus to a dominance of polychaetes owing to the increased availability of polychaete worms post flood (Mosley et al., 2024a). Recent evidence of 'freshwater subsidiary' to the diet of Yelloweye mullet arising from extensive flooding of the Coorong in 2022 (Mosley et al., 2024a).

Larvae: unknown, probably pelagic zooplankton

Juveniles: copepods, polychaete worms, molluscs, insects, amphipods, post-larval fish, detritus, mysid shrimp (Platell et al., 2006; Giatas, 2012; Mosley et al., 2024a)

Adults: algae, detritus, crustaceans, molluscs, polychaete worms (Giatas, 2012; Giatas and Ye, 2016)

Age and growth

Max age: 10 years (Giatas et al., 2024).

Females attain a larger size at age than males (Curtis and Shima, 2005; Earl and Ferguson, 2013). Large female fish (up to 483 mm TL) grow in a land-locked population (Lake George, Giatas and Ye unpublished data). The specific growth rate of juvenile yelloweye mullet (120–150 mm FL) continually fed on an artificial diet at ambient seawater temperature was optimal at 17.2 °C (Flikac et al., 2020). Size patterns from fish surveys in the Coorong reveal that both juveniles and adults are present (Ye et al., 2012).

Size/age at Maturity

Size at Maturity: 242 mm for both sexes (Ferguson et al., 2018). SAM_{50} Female: 256 mm (TL) Male: 226 mm (TL) (Ye et al., 2013) (Coorong).

Age at Maturity: maturity occurs by their third year or third winter in the Coorong (Harris, 1968).

Fecundity and eggs: eggs are planktonic (Kingsford and Tricklebank, 1991). Fecundity varies from 125000 eggs in a 245 mm female to 630000 in 391 mm female (Thomson, 1957).

Environmental requirements

Temperature: Little is understood of the temperature thresholds of yelloweye mullet, although they are distributed across Southern Australia and New Zealand (Earl and Ferguson, 2013) and prefer temperate waters (Chubb et al., 1981). Yelloweye mullet are eurythermal (Earl and Ferguson, 2013; Flikac et al., 2020) and adapted to the nature of estuaries and near-shore environment, coping with temperatures up to 33 °C (Earl and Ferguson, 2013). In New Zealand, the preferred temperature range is indicated to be 13.5–20.2 °C with a mean of 17.1 °C (Egan et al., 2020).

Water temperatures in the Coorong over the period of 2006–2007 ranged 11.2–27.2 °C when yelloweye mullet was the second most dominant species in fish assemblages (Ye et al., 2013). Flikac et al., (2020) describes reduced feeding below 11 °C, but still relatively high feeding in yelloweye mullet in temperatures below 10 °C. Chubb et al (1981) describes for fish inhabiting Swan-Avon River system (Western Australia) that where yelloweye mullet exist in their usual summer habitats, water temperatures can range between 27–30 °C.

One study, Jones et al (1996) examined distribution and abundance of *Aldrichetta forsteri* and several other fish species in the Port River-Barker inlet exposed to thermal effluent from the Torrens Island power station This thermal effluent had previously been measured to range between 6–10 °C above inlet temperatures (Thomas et al. 1986 *cited in* Jones et al. 1996) and reported as high as 34 °C in February 1975 (Jones et al. 1996). Jones et al (1996) noted that during the summer-autumn period, *A. forsteri* avoided areas of elevated water temperature in the inner estuary but conversely attracted to the warmer waters of the inner-estuary during the winter-spring months.

Salinity: Recorded up to 95psu in the Coorong (Earl and Ferguson 2013). Ye *et al*. (2012) describes catches of juvenile and adult yelloweye mullet captured in the Coorong during 2011 (November and December) and 2012 (February and March). Catches were widely distributed over sites along the Murray Estuary and North Lagoon, with salinity during this period ranging between 0–74 psu (brackish to hypersaline conditions, Ye et al., 2012).

The upper tolerance to salinity for juvenile yelloweye mullet from the Coorong have been shown to be LC_{50} of 90.5 psu at 14 °C and LC_{50} of 82.4 psu at 23 °C (McNeil et al., 2013b). At 14 °C changes in feeding behaviour were noticed at 84 psu with feeding ceasing at 88 psu. At 23 °C changes in feeding behaviour were notice at 70 psu with feeding ceasing at 90 psu (McNeil et al., 2013b). Evidence of erratic scoring of behavioural indicator may suggest a negative stress response to increasing salinity at a high temperature level (McNeil et al., 2013b).

Oxygen: Low oxygen levels of 1.2 mg/L, higher water temperatures (28.5 °C) and higher salinity (60 psu) were found associated with a summer fish kill of yelloweye mullet in Lake George, South Australia (Hall, 1984).

Movement

Thomson (1957) found tagging yelloweye mullet problematic, finding tags were easily lost from fish. Of 1550 tagged, only 17 were recaptured with only 1 of those found outside the waters release. Hall (1984) tagged 145 yelloweye mullet in the Coorong between December 1982 and March 1983, finding only two recaptures with one located close the release point and the other ~15 km west of the Murray mouth. Hall (1984) proposed that seasonal (late summer to autumn) declines in catch of yelloweye mullet in the

Coorong may be attributed emigration from the Coorong for spawning, or possibility escaping higher temperature and salinity conditions. Schools of yelloweye mullet, presumed to be fish of spawning migrations occurs in coastal bays close to shore (Chubb et al., 1981). Yelloweye mullet migrate into estuaries as juveniles, where 0+fish are found in lower reaches of the estuary (Chubb et al., 1981). Abundances of juvenile and adult fish (ages 1+and 2+) were greater in the Swan-Avon River system during cooler months compared to summer when temperatures reach 27–30 °C possibly suggesting migration to sea in summer due to temperature (Chubb et al., 1981). Large yelloweye mullet leave intermittently close estuaries not long after they open, with openings corresponding to higher rainfall (Orr, 2000). Thomson 1996 (*cited in* Ye et al. 2013) stated that yelloweye mullet move into freshwater on rising tides.

Spawning

Thompson (1957) indicated that spawning occurs in eastern Australia during summer in Victoria and Tasmania and winter in Western Australia. Spawning occurs in summer-autumn in eastern Australia, but in western Australia occurs during winter Chubb et al., 1981. Harris (1968) noted that, although yelloweye mullet also have a relatively protracted breeding season of four months in South Australia, spawning in the summer rather than in the winter. In the Coorong, spawning occurs over summer, autumn and winter (Harris, 1968; Noell et al., 2009; Cheshire et al., 2013; Ye et al., 2013). Spawning in winter in WA, hypothesised to be different to summer and possibly timed with intermittent estuary openings after the onset of heavy rain (Chubb et al 1981). Spawning occurs in a range of locations including shallow coastal waters, coastal bays and estuaries (Chubb et al., 1981; Kingsford and Tricklebank, 1991; Earl and Ferguson, 2013). Spawning of mature land-locked individuals does not appear to occur in the Lake George population (Giatas et al., 2024).

Recruitment

Little is known of the early life history of yelloweye mullet (Pellizzari, 2001). Yelloweye mullet larvae are abundant around drift algae in open water around November to march in New Zealand whilst in its pelagic phase before settlement (Kingsford and Tricklebank, 1991; Hickford and Schiel, 2003). Larvae were detected in higher abundance, located close (2 km) to the coast near the Kowhai River in New Zealand during summer (Hickford and Schiel, 2003). Larval duration ranges between 19 to 25 days (Pellizzari, 2001). Most of the recruitment into the Coorong occurred during winter (Earl and Ferguson 2013). Pellizarri (2002) indicated that growth and settlement in yelloweye mullet is fast and possibly is a result from these juveniles seeking out shallower, warmer environment locations that on average are greater in temperature than the surrounding environment. From evidence of population age-structures, natural recruitment of juveniles into the land-locked Lake George Fishery in South Australia, only occurs through brief periods of connectivity to the Southern Ocean during years of elevated freshwater inflows (Giatas et al., 2024).

Congolli (Pseudaphritis urvillii)

Life history

Congolli are a diadromous species and classified as part of the 'Catadromous' guild of fish as they depend on movement to and from the marine/estuarine and freshwater environments to complete its life cycle (Bice et al., 2018a). Females are known to occupy freshwater reaches as adults and are associated with pools, rocks and woody debris (Hortle, 1978). Whereas males remain in the estuary or only occasionally enter freshwater (Schmarr et al., 2022) and associated with sandy or silty bottoms or seagrass (Jones et al., 2008; Bice, 2010; Schmarr et al., 2022).
Congolli have medium body size and a life-history that resembles a strategy that is intermediate between Periodic and Opportunist strategies. They have medium body size and longevity, with maturity at half-life, migration for reproduction, asynchronous spawning, consistent annual recruitment, rapid growth of young and semelparous spawning (Matthews, 1998; Ferguson et al., 2018).

Distribution, population structure and utility

Congolli are a medium-sized fish found throughout Southern Australia, distributed in coastal rivers between Streaky Bay South Australia to Clyde River in New South Wales, including coastal rivers around Tasmania (Andrews, 1996; Morris et al., 2001; Giatas and Ye, 2015). Schmidt et al. (2014) examined the genetic structure of Congolli, largely across several Victorian coastal catchments but included samples from the Coorong and 1 coastal river in northern Tasmania, and found no evidence of multiple stocks, suggesting connectivity across sub-populations across coastal catchments. Congolli has significant ecological and conservation values in the Coorong (Giatas and Ye, 2015) and plays important roles in the estuary food web as an important prey item of Mulloway (Giatas et al., 2018). Congolli were also previous landed in small numbers as part of the Lakes and Coorong Fishery (Ferguson et al., 2018).

Diet

Congolli are carnivorous, consuming a wide variety of prey, including crustaceans, molluscs, various orders of insect larvae, polychaete worms, fish and plant material (Hortle, 1978; Hortle and White, 1980; Giatas and Ye, 2015; 2016; Giatas et al., 2018). Congolli is facultative invertivore, as benthic invertebrates form a major component of the diet (Hortle and White, 1980; Giatas and Ye, 2016). Small congolli (< 80 mm) found to have a higher proportion of small amphipod species and polychaete worms, with a shift in diet in larger individuals (> 80 mm) including more fish and larger amphipod species (Giatas and Ye, 2015). For small Congolli, transitional changes in diet were observed from amphipods to polychaetes with higher salinity (Giatas and Ye, 2015). Diet also differs between individuals in the estuary compared to those in freshwater (Giatas and Ye, 2015), with individuals in freshwater containing greater proportions of insects and fish (Giatas and Ye, 2015).
Larvae: unknown, likely Zooplankton

Juveniles: insect larvae, amphipods, crabs, polychaete worms, fish (Hortle and White, 1980; Giatas and Ye, 2015)

Adults: insect larvae, amphipods, crabs, polychaete worms, fish (Hortle and White, 1980; Giatas and Ye, 2015)

Age and growth

Max age: known to live in captivity for 4–5 year (Andrews, 1996) more recent work has shown individuals to live to 7 years (Giatas et al., 2024).

Sexual dimorphism is evident with females growing larger than males (Hortle, 1978). Maximum size is about 350 mm for females (Crook et al., 2010), about 150 mm for males (Crook et al., 2010; Giatas and Ye, 2015). Late post-flexion larvae have been captured from July to September in Barker Inlet, South Australia, with settled juveniles captured in the Gippsland Lakes in November (Sutton and Bruce 1998 cited in Morris et al 2001). Congolli < 80 mm are young-of-year (age 0+)

Size/age at Maturity

Size at Maturity: SAM_{50} Female**:** 165 mm (Cheshire et al., 2013; Ye et al., 2013). Smallest, mature male was 103 mm total length (Cheshire et al., 2013).

Age at Maturity: Female Congolli mature 3–4 years, males 2–3 years of age (Hortle, 1978).

Fecundity and eggs: Fecundity and eggs are undescribed (Neira et al., 1998; Cheshire et al., 2013).

Environmental requirements

Temperature: Little is known about the environmental requirements of Congolli. Water temperatures in the Coorong over the period of 2006–2007 ranged between 11.2 and 27.2 °C when Congolli were observed inhabiting the estuary during a drought period (Ye et al., 2013). Female and male GSI was low over summer and increased during winter suggesting the onset of gonad development (Cheshire et al., 2013). It has been suggested that low water temperatures partially facilitate migration (Bice et al., 2018b). Congolli spawn in the marine environment but have life-histories where juveniles migrate into estuaries during spring when higher temperatures and abundant food provides favourable conditions for growth (Ye et al., 2013).

Salinity: Strong osmo-regulator can handle direct transfer from freshwater to saltwater and vice versa (Andrews, 1996; Cheshire et al., 2013). In the Coorong, Livore et al. (2013 cited in Giatas and Ye 2016) indicated that localised abundances at slightly higher levels of salinity (50–80 psu), while Noell et al., (2009) observed Congolli to be recorded at locations up to 74.1 psu. McNeil *et al.* (2013b) experimentally exposed juvenile Congolli (mean size: 39.8 ± 7.6 mm) to increasing salinity under two temperature regimes 14 and 23 °C. Juvenile Congolli were observed to tolerate salinity to a maximum of 104 psu with LC_{50} of 100.1 psu at 14 °C and LC_{50} of 94.1 psu at 23 °C (McNeil et al., 2013b), to be the second most salinity tolerant species in the study. Evidence of stress by incremental salinity increases first observed at 70 psu (McNeil et al., 2013b).

Movement

Adult females are catadromous, undertaking spawning migrations from freshwater to estuarine/marine environments (Hortle, 1978; Crook et al., 2010; Giatas and Ye, 2015). Males remain in the estuary, not moving into freshwater (Hortle, 1978; Crook et al., 2010). Has been recorded up to 120 km inland (Andrews, 1996). Identified to be severely impacted barriers in coastal rivers, where passage is available fish are suspected to migrate to the sea during winter (June–July) at sunset on the ebb tide, likely for the purpose of spawning (Bice et al., 2018b). No evidence of post-spawning migrations back to freshwater suggesting Congolli are semelparous (Crook et al., 2010; Schmidt et al., 2014; Bice et al., 2018b). Small congolli (< 60 mm) migrate from the marine environment to lower lakes between October and January (Zampatti et al., 2010; Giatas and Ye, 2015).

Spawning

Late July to November (Giatas and Ye, 2015) with stage 3 (and later) female gonads observed during autumn and winter, suggesting spawning likely occurs over winter (Cheshire et al., 2013). Asynchronous oocyte development observed in gonads, suggesting multiple spawning (Cheshire et al., 2013). Acoustic tagging of mature females have found Congolli moving to sea, suggesting spawning occurs in the sea (Crook et al., 2010; Bice et al., 2018b).

Recruitment

Very little is known about the early life history (Zampatti et al., 2010), except drought conditions, reduced upstream flows and barriers to connectivity can influence the strength of juveniles migrating into estuaries (Zampatti et al., 2010). Impacts of Millenium drought were significant with a loss of individuals over large parts of the estuary, with numbers not returning until important large flows returned to the system (Wedderburn et al., 2016; Bice et al., 2018b). Low numbers of larvae have been collected in estuarine larval surveys the vicinity of the Murray mouth during a prolonged drought (Bucater et al., 2013).

Carp (Cyprinus carpio)

Life history

Common carp are an alien freshwater species (Lintermans, 2007) and therefore considered as 'Freshwater exotic' (Ye et al 2016) or 'Freshwater straggler' (Bice et al., 2018a). In Australian freshwaters, carp life-history is considered flexible (Jones and Stuart, 2007; Diggle et al., 2012) and differs to that in their country of origin (Ferguson et al., 2018). In local freshwaters display characteristics more common with 'opportunistic' life-history strategists (Ferguson et al., 2018), characterised by a fast growth rate, earlier maturity, generalist diet and consistent annual recruitment. These features is advantageous to being able to re-populate quickly after disturbance or where continuous high mortality occurs in the adult stage (Winemiller and Rose, 1992). Carp contain multiple traits that make them a strong competitor to native fish and a powerful invader to colonisation (Koehn, 2004).

Distribution, population structure and utility

Introduced and successfully established throughout Europe, Asia, Africa, North America, South and Central America, Australia, New Zealand, Papua New Guinea and some islands of Oceania (Koehn, 2004; Stuart et al., 2021). Within Australia, Common carp are widely distributed across south-eastern Australia mostly across the Murray-Darling Basin, with other isolated populations located coastal river catchments, including South Australia (Stuart et al., 2021). Additional populations are known to occur in Western Australia and Tasmania (Koehn, 2004). Common carp inhabit several different environments including Rivers, lakes, wetlands, floodplains, irrigation channels, impoundments, estuaries and lagoons and associated with open, shallow, deep water, rocks, macrophytes, large woody debris and bare substrates, like silt and sand (Koehn et al., 2000; Bice, 2010; Schmarr et al., 2022). Four strains: Boolara, Yanco, Prospect and Koi are spread across the Murray-Darling Basin, with significant genetic structure evident across the Murray-Darling basin corresponding to known natural or artificial barriers to dispersal (Haynes et al., 2009). In Australia and other continental regions, Common carp can obtain 'superabundance' resulting in damage to aquatic ecosystems through the displacement of submerged vegetation, increased turbidity through resuspension of fine sediments and nutrient enrichment (Bajer and Sorensen, 2010; Vilizzi et al., 2014). Across Australia, commercial fishing of Common Carp is undertaken for the purpose of bait supplies for marine crustacean fisheries, with some export to international markets (Koehn et al., 2000). Overseas, particularly Asia and Europe, considerable consumption of carp occurs supporting greater commercial and recreational fishing activity (Koehn et al., 2000).

Diet

Common carp are generally considered omnivores/detritovores (Koehn, 2004), consuming a wide variety of micro and macroinvertebrates, algae, diatoms, detritus, plant material (e.g. seeds) and sometimes fish (Hume et al., 1983; Vilizzi, 1998; Khan, 2003). Ontogenetic shifts in diet are evident from larvae (planktivorous) to juveniles (benthivore) upon settlement (Vilizzi, 1998).
Larvae: phytoplankton, diatoms, algae, copepods, cladocerans, seeds (Vilizzi, 1998; Khan, 2003; Al-Lamy and Taher, 2016)

Juveniles: algae, diatoms, copepods, crustaceans, insect larvae, seeds (Vilizzi, 1998; Khan, 2003; Al-Lamy and Taher, 2016)

Adults: various crustaceans, molluscs, insect larvae (Koehn et al., 2000; Khan, 2003)

Age and growth

Max age: 34 years (Koehn et al., 2018)

Adults are known to attain a maximum length of 1,200 mm (Koehn et al., 2000). Larval growth is rapid (Koehn 2004). Juvenile onset is ~20–30 days post hatch at 20–25 mm standard length (Vilizzi and Walker, 1999). Growth rates of carp vary greatly between different regions, depending on temperature, food availability and population density. (Koehn et al., 2000). Carp respond well to higher temperatures with faster growth, earlier reproduction and increase reproductive investment, with larger latitudinal patterns

in population biology underpinned by temperature evident (Oyugi et al., 2011; Weber et al., 2015). Warm temperatures (25 °C) increased the growth of common carp juveniles, resulting in increased rate of gonadal development (Gehrig and Thwaites, 2013). In Tasmanian lakes where water temperatures are lower, carp grow and mature more slowly than the mainland (Diggle et al., 2012).

The continental-wide distribution of common carp populations means opportunities to examine population traits across natural temperature gradients (i.e. latitude), which can offer insight in potential population responses to increased warming associated with climate change (Weber et al., 2015). Weber et al (2015) examining 21 common carp populations across a 2700 km latitudinal gradient identified that trends in population life history characteristics from higher and colder areas to lower and warmer latitudes, fish growth increased but smaller maximum sizes attained, longevity reduced and mortality increased (Weber et al., 2015). Metabolic consequences associated with greater environmental temperatures may mean that increases in temperature associated with global warming promote shortened life spans and smaller body sizes (Weber et al., 2015).

Size/age at Maturity

Size at Maturity: Smallest males at 150 mm TL and females at 250 mm TL (Smith, 2005). Most carp mature (SAM_{95}) by 350 mm TL (Smith and Walker, 2004; Smith, 2005).

Age at Maturity: Carp display a wide range of ages at maturity depending on the climatic conditions and gender (Diggle et al., 2012). Water temperature is an important determinant of maturity (Smith and Walker, 2004; Li et al., 2024), with constant and high temperature (28 °C) more favourable to rapid maturity (11 months) than variable and low (28–23–18–13–18–23–28 °C) temperature (Li et al., 2024). Carp mature relatively slowly under lower temperatures (Diggle et al., 2012). In Tasmania, males take a minimum of three years to mature and females, four years compared to two years for males and three years for females on mainland Australia (Koehn et al., 2000; Diggle et al., 2012). In the tropics, carp can mature in 3-6 months (Smith and Walker, 2004).

Fecundity and eggs: Carp are asynchronous, multiple batch spawners (Smith and Walker, 2004). Relative fecundity estimated to be 0.163 million eggs/kg (Sivakumaran et al., 2003) and females produce over a one million eggs per year (Koehn et al., 2000). Eggs are 0.5 mm in diameter, demersal and sticky, attached to submerged vegetation (Lintermans, 2007). Incubation period of 48 hours at 25 °C (Koehn, 2004).

Environmental requirements

Temperature: Common carp thrive in warm water temperatures and 23–32 °C is reported for optimum growth (Sivakumaran et al., 2003; El-Gamal, 2009; Ahmad et al., 2011; Oyugi et al., 2012; Li et al., 2024). When temperature is < 15 or > 35 °C, growth slows significantly (Li et al 2024). At 10 °C carp stop feeding (Hao 2012 cited in Li et al 2024), but they can tolerate temperatures much lower (2 °C in Tasmanian lakes, Koehn 2000). For embryonic development, 26–28 °C is best for common carp (Korwin-Kossakowski, 2008; Sapkale et al., 2011), although percent hatchability decreases as temperature increases from 26–30 °C (Korwin-Kossakowski, 2008; Sapkale et al., 2011). Eggs that were incubated at higher temperature (24–28 °C) produced larvae that were longer and heavier than those hatched at 20 °C (Korwin-Kossakowski, 2008). No eggs have been observed to hatch at much higher temperatures (38 °C, El-Gamal, 2009) or low temperatures at 10–11 °C (Swee and McCrimmon, 1966).

For carp larvae, temperatures between 24 and 30 °C are considered optimal for growth (El-Gamal, 2009). Geldhauser (1995 *cited in* Réalis-Doyelle et al., 2018) indicated that the upper thermal tolerance of carp larvae is near 35 °C. For juveniles, temperatures for optimal growth is between 24 and 28 °C (Oyugi et al., 2012). The upper thermal limit for Common carp juveniles (40–72 mm) has been reported at 40.6 °C (Horoszewicz, 1973). Most work reporting critical thermal limits has been undertaken for eggs, larvae or juvenile but extremely little is known about the upper thermal limit of large (> 250 mm) or adult carp. One study that examined larger fish of the average size 190 (± 30 mm) indicated that an upper thermal limit of 32 °C as fish showed sign of high stress and could not survive hotter conditions for more than a few hours (Ahmad et al., 2011).

Among other factors (e.g. genetic), upper lethal temperatures depend on previous acclimatisation temperatures (Opuszynski et al., 1989) and acclimation to higher temperatures of 25 to 35°C can lead to an increase in upper extreme temperature (Golovanov and Smirnov, 2007).

Salinity: In the Murray-Darling Basin, Common carp are known to occur in wetlands containing salinities of 17.5 psu, which is close to their upper tolerance (Whiterod and Walker, 2006). In US waters, common carp have been captured as high as 17.6 psu (Schwarz 1964, *cited in* Mangat and Hundal, 2014). Mature fish from Lake Alexandrina and Albert survived direct transfer to salinities of 12.5 psu at 16–21 °C and with acclimation over 38 days, there was 50% survival at 15 psu (Geddes 1979 *cited in* Smith 2005).

Whiterod and Walker (2006) undertook direct transfer experiments of juvenile Common carp (31–61 mm TL) exposed to several salinity treatments (0.1, 5, 7.5, 10, 12.5, 15, 17.5 and 20 psu). Juveniles were sourced from the Murray River, a low salinity environment (0.04–0.1 psu) prior to direct transfer. Juveniles were also slowly acclimatised (rate of 2 psu/day) to 10 psu prior to direct transfer in a subsequent experiment. Direct transfer of juvenile carp from low salinity to salinity treatments < 10psu had similar survival outcomes, with 47–70% of individuals surviving after 7 days. Transfer of juveniles to 15psu and above resulting in total mortality within 24 hours. At the highest salinity treatment (20 psu), 100% mortality was recorded within 3 hours (Whiterod and Walker, 2006). Juveniles that were slowly acclimatised to 10 psu, prior to transfer to higher salinity treatments only had a marginal improvement to survival compared to no acclimatisation. The resulting LC_{50} for juvenile carp (10 psu acclimation) was 13 psu. At 20 psu, 100% mortality within 24hrs (Whiterod and Walker, 2006). Lower temperature may reduce the mortality of juvenile carp to salinity (Mangat and Hundal, 2014).

Malik et al., (2018) examined effects of salinity (0, 5, 10, 15 and 20 psu) on adult reproduction, finding total numbers of eggs produced was highest at 0 psu and fertilisation and hatchability of eggs similar between 0–10 psu. Malik et al., (2018) also investigated the salinity on the larval development of Common carp identifying larvae having a tolerance range of 0–10 psu.

Whiterod and Walker, (2006) estimate the critical intensity of where sperm are no longer viable to be 8.33 psu. Effect egg development and fertilisation does not occur at salinities (8–10 psu) (Whiterod and Walker 2006). Similarly, Malik et al., (2018) reported under experimental conditions that as salinity increases from 10 psu to 15psu, a considerable decrease in the total numbers of eggs (approx. 3.75-fold reduction), fertilised eggs (approx. 10-fold reduction), hatchlings (approx. 16.75-fold reduction) and fry (approx. 30-fold reduction).

Oxygen: For spawning, 4–6 mg/L is suitable (Malik et al., 2018)

pH: The present results suggest that water pH 7.5 at water temperature 26–28C is best for hatchability of eggs and pH 7.4 for growth of spawn of Cyprinus carpio at same water temperature (Sapkale et al., 2011). Eggs held at pH of 4.5 and 10.5 resulted in complete mortality after 24 hours Jezierska and Witeska, 1995). Some embryonic development is possible within the range of 5 – 9.5 (Jezierska and Witeska, 1995).

Movement

Carp aggregate over winter in the deeper waters of lakes, when water temperatures are at minimum (Diggle et al., 2012). Rising water levels, warming water temperatures associated with longer daylight trigger movement and aggregations of mature fish in Tasmanian lakes (Diggle et al., 2012). In the Murray River, carp typically move only short distances (< 5km, Stuart and Jones, 2006b), however, common carp can move over large spatial scales (100's km) in short time periods, with movement seeming to be related to temperature (Koehn et al., 2000; Stuart and Jones, 2006b). In Southern regions of the MDB, adult carp move between river and wetlands for spawning, movement activity increased with increasing temperature from winter minimums (10.6 °C) until mid-September when temperature reach 16 °C (Conallin et al., 2012). Carp are attracted to flowing water and move upstream to a source of flow (Conallin et al., 2012).

Spawning

Spawning is temperature and light dependent and occurs when temperatures of 15–17 °C are exceeded (Brown, 2003; Sivakumaran et al., 2003; Koehn, 2004; Smith, 2005; Gehrig and Thwaites, 2013) but can cease at higher temperatures around 28–29 °C (Swee and McCrimmon, 1966; Koehn et al., 2000). Spawning has been observed to occur in temperatures as low as 11 °C in Australia, however there was no evidence of successful recruitment observed from such low temperatures (Diggle et al., 2012). Spawning period begins from about mid-September in South Australia, with spawning extending over summer and autumn (Smith, 2005) and occasionally into early winter (Brown, 2003). In the cooler climate of Tasmania, the spawning period is shorter and restricted to spring-summer (October to February) (Diggle et al., 2012). Eggs are sticky upon contact with water and shed over submerged plants in shallow water (Koehn et al., 2000).

Recruitment

Eggs incubation time is short at warm temperatures (i.e. 25 °C) while larval growth is rapid enabling the escape of predators (Koehn 2004). High numbers of juvenile carp are evident after periods of flooding, when zooplankton proliferate (Koehn et al., 2000). Although, survival is density-dependent and mortality may exceed 98% in the first year of life (Koehn et al., 2000).

Hydrology, with increased water levels flooding has been identified as an important factor contributing to recruitment strength in the River Murray (Koehn et al., 2018; Stuart et al., 2021) as well as in Tasmania lakes (Diggle et al., 2012). Common carp favour shallow, floodplain areas for both spawning and recruitment (Koehn et al., 2000). For rivers, floodplain environments are recognised as important areas for carp recruitment in riverine ecosystems (King et al., 2003; Koehn, 2004; Crook and Gillanders, 2006; Stuart and Jones, 2006a). Immersion of the floodplain provides access to submerged terrestrial vegetation which is valued as a spawning substrate (Stuart and Jones, 2006a). Larvae are found on the forested floodplain during flood periods (Stuart and Jones, 2006a), while floodplain wetlands serve as nurseries for juveniles (Koehn et al., 2000; Stuart and Jones, 2006a). Stranding of large numbers of larvae and juveniles may occur in temporary shallow pools on the floodplain after floods have subsided (King et al., 2003). Drought provides reduced recruitment opportunity for carp, which can assist with management of populations (Koehn et al., 2000; Diggle et al., 2012).

Redfin perch (Perca fluviatilis)

Life history

Redfin perch are an introduced fish species, originating from Europe and widely distributed across the northern hemisphere (Lintermans, 2007; Ning et al., 2025). Much less is known about their biology and ecology in environments outside their natural distribution (Ning et al., 2025). In Australia, redfin perch is considered to complete its entire lifecycle in freshwater, and as such are considered as 'Freshwater exotic' (Ye et al 2016) or 'Freshwater straggler' (Bice et al., 2018a) where guild classifications extend to species presence in the Coorong estuary. However, in their home waters of northern hemisphere, populations of redfin perch are known to inhabit estuaries (e.g. western Baltic Sea, Christensen et al., 2021).

In Australian freshwaters, redfin perch have a Periodic life-history strategy in their country of origin, but in local freshwaters display characteristics more common with 'Opportunistic' type life history strategies (Ferguson et al., 2018). Opportunistic life-strategy are characterised by a fast growth rate, earlier maturity, generalist diet and consistent annual recruitment. These features is advantageous to being able to re-populate quickly after disturbance or, where continuous high mortality occurs in the adult stage (Winemiller and Rose, 1992).

Distribution, population structure and utility

Redfin perch are distributed largely across the lower and mid reaches of the Murray-Darling Basin (New South Wales, South Australia and Victoria). It is also found in Tasmania (McDowall, 1996) and Western Australia (Morgan et al., 2003) but not in Queensland or northwestern NSW, as their distribution is restricted by high water temperature (Weatherley, 1977; Lintermans, 2007). It has also been introduced to New Zealand and parts of Southern Africa (Ning et al., 2025). Redfin perch inhabit several different environments including Rivers, lakes, wetlands, impoundments, estuaries and lagoons and associated with open, shallow, deep water, rocks, macrophytes, large woody debris and bare substrates like sand (Morgan et al., 2003; Bice, 2010; Ning et al., 2025). Redfin perch are a piscivore that can predate on native fish, including many species of small-bodied fish native (Wedderburn and Barnes, 2016, Pen and Potter, 1992). For this reason, redfin perch are declared as 'noxious' under the South Australian Fisheries Management Act 2007 and must not be returned to water if caught. Redfin perch are taken as part of the Freshwater Large Gillnet Sector in the Lakes and Coorong Fishery (Ferguson et al., 2013), with catches ranging between 6–92 tonnes annually (Earl, 2020).

Diet

Redfin perch are carnivorous, eating a diversity of macroinvertebrates (Atyid shrimp, crayfish), fish, including other redfin perch (Morgan et al., 2003; Wilson et al., 2008; Wedderburn et al., 2014; Ning et al., 2025). Dietary items consumed in Australia is like home waters (Weatherley, 1977). As redfin perch increase in size, a switch in prey from planktonic crustaceans to benthic invertebrates is evident with decapod crustaceans constituting a large proportion of the diet volume by larger fish (> 120 mm) (Pen and Potter, 1992; Ning et al., 2025). Fish also features prominently in the diet, with consumption of small bodied native fish common (Pen and Potter, 1992; Wilson et al., 2008; Wedderburn et al., 2014) which is a concern for the conservation of rare and threatened small bodied native fish (Hutchison, 1991; Pen and Potter, 1992; Wedderburn et al., 2014). In Lake Alexandrina, redfin perch switch to piscivory at an early age (five months) and at sizes (93 mm TL) smaller than previously noted (Wedderburn and Barnes, 2016).

Larvae: rotifers, copepod nauplii and copepodites, cladocerans (Thorpe, 1977; Treasurer, 1990; Wang and Eckmann, 1994b)

Juveniles: copepods, insect larvae, crustaceans, fish (Morgan et al., 2003; Wedderburn et al., 2014; Wedderburn and Barnes, 2016)

Adults: crustaceans (decapods) and fish (Morgan et al., 2003; Wedderburn et al., 2014; Ning et al., 2025)

Age and growth

Max age: 22 years (Beverton and Holt, 1959)

Attains a maximum size of 600 mm (Ferguson et al., 2018; Ning et al., 2025). Females grow faster and larger than males (Jellyman, 1980). Growth is negligible between populations inhabiting brackish water and freshwater (Christensen et al., 2021).

Growth is faster in the Southern hemisphere where waters are warmer (Ning et al., 2025). In Lake Alexandrina, growth of age 0+redfin perch is rapid with individuals observed up to 147 mm TL to be less than 6 months of age (Wedderburn and Barnes, 2016). This is considerable difference in growth compared to other areas of their distribution and likely corresponds to an early switch to piscivory (Wedderburn and Barnes, 2016).

Cannibalism is common among redfin perch populations (Le Cren, 1992; Persson et al., 2000) and shapes population structure, especially where food resources become limiting (Persson et al., 2000). Stunting or slow growth is also prevalent among populations (Ylikarjula et al., 1999). Holmgren and Appelberg, (2001) identified that growth efficiency of redfin perch size classes (100–199 mm and > 200 mm) between lakes was better characterised by fish assemblage composition than abiotic factors. They indicated that their results likely reflected underlying biological interactions (e.g. competition and predation) that masked the effects of abiotic factors.

Size/age at Maturity

Size at Maturity: In Western Australia, the smallest, mature male was 66 mm TL and smallest, mature female was 94 mm TL (Morgan et al., 2003).

Age at Maturity: The age at maturity relates to grow rate and ambient water temperature, with faster growing populations maturing earlier (Morgan et al., 2003; Ning et al., 2025). Males: in their first or second year, females in third or fourth year (Morgan et al., 2003; Ning et al., 2025).

Fecundity and eggs: 2–3 mm in size and hatch after 1–3 weeks (Ning et al., 2025). Redfin perch are highly fecund with a reported fecundity up to 120,000–140,000 eggs/kg of body weight (Ning et al., 2025).

Environmental requirements

Temperature: Wang and Eckmann, (1994) examined the influence of temperature on the egg development and larval survival finding that hatching occurred between 6–22 °C. No hatching was observed at 24 °C, while individuals that hatched at 6–8 °C failed to reach swim up stage (Wang and Eckmann, 1994b). The optimum temperature for hatching ranges between 12–20 °C, while the optimum temperature for embryo and larval survival was 16–18 °C (Wang and Eckmann, 1994b).

For larvae, Souchon and Tissot, (2012) report thermal tolerances of larvae to range between 5–30 °C. Optimum temperatures to maximise growth has also been reported to as low as 11 (Souchon and Tissot, 2012), while 17 °C has been reported as optimum temperatures during perch rearing under aquaculture conditions (Toner and Rougeot, 2008).

For juveniles, Souchon and Tissot, (2012) report thermal tolerances to range between 8–36 °C. Optimum temperatures of 19–23 °C has been reported for rearing under aquaculture conditions (Toner and Rougeot, 2008). The highest upper lethal temperature reported for adults was 36.2 °C, an observation recorded for an individual originating from a heated lake in Russia (Filon 1972 cited in Souchon and Tissot, 2012).

Redfin perch are considered as a eurythermal species (Overton et al., 2008). Typically, 31 °C has been considered as the upper thermal limit for adult redfin perch inhabiting Australian ecosystems (Weatherley, 1977; Ning et al., 2025). A lower lethal thermal limit of 4 °C has been reported for redfin perch (Thorpe, 1977), although its possible adults may survive lower temperatures considering populations can persist in sub-freezing conditions in the Northern hemisphere (Ning et al., 2025). Optimum temperatures for adults have been reported to range between 16–27 °C. Observations of redfin perch existence in locations

where the exceedance of critical thermal values in surface waters may suggest that redfin perch behaviourally act to seek thermal refuge (Lake, 1967).

Salinity: In freshwater, redfin perch are stenohaline and tolerate salinities up to around 10–15 g/L (Thorpe, 1977; Hutchison and Armstrong, 1993; Christensen et al., 2019). However, there are some Northern hemisphere populations that naturally inhabit brackish waters, including estuaries (e.g. western Baltic sea, Christensen et al., 2016) and given this plasticity in tolerance, cannot be completely considered as stenohaline (Lutz, 1972). In these systems, it has been reported that redfin perch can tolerate salinities up to 17.5–18 psu, ~50% seawater (Thorpe, 1977; Weatherley, 1977; Christensen et al., 2019). However, Christensen et al., (2021) identified brackish water redfin perch from the western Baltic Sea experienced short-term exposure to salinities above these values (22 psu) on five occasions without apparent evidence of consequence. Genetic differences exist between populations of fish inhabiting freshwater and brackish water, although further work is required to understand to what extent salinity tolerance and osmoregulatory capability is a heritable (Christensen et al., 2019).

For eggs, *in-vivo* egg hatching studies suggested 7 psu as maximum upper limited for salinity (see studies within Christensen et al., 2016). However, spawning can occur in salinities that are higher than what eggs reportedly tolerate (Christensen et al., 2016). A subsequent hatching success experiment using eggs from brackish water and examining hatching success across a range of salinities (4, 7, 10 and 12 psu at 12 °C) indicated no statistical difference in the proportions hatched individuals among treatments and no observed larval deformities (Christensen et al., 2016). Rask (1983) reports that Norwegian lakes containing populations of redfin perch have water conductivities as low as 10 microsiemens (about 0.0 psu), which likely approximates the lower limit of tolerance. Eggs are affected less by salinity than post-hatch life stages (Tibblin et al., 2012).

Larvae survival (29 days after hatching) was observed in similar proportion across 0, 0.6, 1.2, 2.4 and 4.8 psu salinity treatments (Bein and Ribi, 1994). Most larvae died in the upper salinity treatment of ~9.8 psu, which was considered the upper limit (Bein and Ribi, 1994). Growth of larvae may decline in salinities above 4 psu (Overton et al., 2008).

For juveniles (~90 days old), mortality was observed to be greater for fish reared in brackish water (7 psu) compared to freshwater (tap water: conductivity not reported) (Tibblin et al. 2012). Dainys et al., 2019 report no significant differences in the growth of juveniles (over 60 days) under salinities of 0, 3 and 6 psu. Tibblin et al. (2012) report that previous studies indicated that salinities below 5 psu had positive effects, while salinities above 8 psu had negative effects on survival and growth.

Oxygen: The lower limit for oxygen has been indicated as 1.3 mg/L (Thorpe, 1977; Toner and Rougeot, 2008), while 13.5 mg/L has been identified as the upper limit for Redfin perch in natural environments (Toner and Rougeot, 2008). Toner and Rougeot, (2008) recommended that for the culture of juveniles, dissolved oxygen concentrations should be above 4 mg/L.

pH: Hatching of eggs ceased to occur at pH of 3.5 (Rask, 1983). A lower pH of 3.5 was also found to cause mortality in larvae, age 0+ juvenile and adults (Rask, 1983 A pH range 6–12 is reported across their natural distribution and a pH range of 6–9 is recommended for culture (Toner and Rougeot, 2008).

Movement

In the first year of life living in lakes, redfin perch are known to make a series of movements residing in different habitats starting in pelagic zone for larval rearing after hatching, before moving back to the littoral zone as settled juveniles where during summer, they alternate between shallow and deeper water, at times stay close to cover avoiding predation from adults (Wang and Eckmann, 1994a). During winter age 0+redfin perch move from the littoral zone to deeper waters (Wang and Eckmann, 1994a). Adults have been observed to move from the littoral zone to deeper waters on a diel basis, while movements of adults also correspond to water temperature, light intensity, air pressure, wind, and lunar phase (Nakayama et al., 2018).

Anadromy has been observed seasonally in some Northern hemisphere populations, where annual movements of fish moving upstream to lower saline environments during winter and returning to brackish

water during summer have been identified likely because of the adverse physiological effects of high salinity at lower temperatures (Christensen et al., 2021). In brackish water populations, movement to freshwater has also been observed for the purpose of spawning (Thorpe, 1977; Bein and Ribi, 1994). Downstream movements as part of spawning activities into brackish environments have also been observed (Skovrind et al., 2013). In places such as the Baltic sea where cooling water discharges also occurs from power stations, migration towards these warmer effluent discharges for the purposes of spawning has been noted (Sandström et al., 1997).

In Australia, little is known about the movement of redfin perch, however, significant numbers (4365–36,853 individuals) of redfin perch have been detected around or moving through fishways in the Coorong estuary-lower lakes interface as part of fishway monitoring for the years 2018/2019–2021/2022 (Bice et al., 2019; 2020; 2021a; 2023).

Spawning

Spawning of redfin perch occurs in rivers, lakes, and estuary deltas (Urho, 1996). Spawning has been observed in brackish waters (Thorpe, 1977; Christensen et al., 2016) up to 9.6psu (Skovrind et al., 2013). Hatching of eggs from *in-vivo* translocation experiments have shown successful hatching up to 12 psu (Christensen et al., 2016). Eggs are laid in gelatinous ribbons, which is unique to *Perca species* (Ning et al., 2025) usually over submerged vegetation, however, in absence of aquatic plants, non-living matter may be used including bundles of sticks (Lake, 1967; Weatherley, 1977). Eggs can be laid in waters of several meters in depth (Lake, 1967; Sandström et al., 1997; Čech et al., 2009; Probst et al., 2009).

Onset of spawning has been reported to occur in the austral late winter and early spring (August and September) upon a cue of increasing water temperature, spawning beginning at 11–11.5 °C and continuing to about 15°C (Lake, 1967; Morgan et al., 2003). However, spawning is reported to be undertaken over a broader range of temperatures in the homeland waters of the Northern hemisphere (5–19 °C, Ning et al., 2025) and has been reported at ceasing at 24 °C (Sandström et al., 1997). Spawning is expected to occur earlier in the spawning season under the presence of warmer water temperatures (Sandström et al., 1997; Dainys et al., 2019). Gonad development (gonadogenesis) in mature females can progress under constant higher temperatures (18–22 °C), although egg development is halted prematurely (Abdulfatah et al., 2013). Temperature appears to modulate gonadogenesis in redfin perch, where alteration of photoperiod (i.e. constant levels) can stop gonadogenesis irrespective of exposure to different thermal conditions trialled (Abdulfatah et al., 2013).

Sandström et al., (1997) examined the effects of higher water temperature associated with thermal effluent from the Oskarshamn power station (Sweeden) on the spawning and embryonic development of redfin perch. Power station thermal discharges were up to about 10 °C above ambient conditions, while redfin perch were known to actively migrate towards and use these warmer waters (Sandström et al., 1997). Sandström *et al.* (1997) unexpectantly observed exposure to elevated temperatures during the final maturation and ovulation stage denatured and disintegrated the gelatinous egg ribbon. Despite high fertilization success and survival to first stage of development, loss of gelatinous structure in disintegrated egg ribbons resulted in the exposure of eggs to pathogens and subsequently caused in high mortality (Sandström et al., 1997). Sandström *et al.* (1997) concluded there is a potential conflict between temperatures needed for optimal reproduction and adult thermal preference.

Recruitment

Both environmental (e.g. temperature) and biological (e.g. mortality from predation or disease) factors are recognised in affecting redfin perch recruitment (Buijse et al., 1992; Lintermans, 2007; Becker et al., 2016; Gwinn and Ingram, 2018; Kokkonen et al., 2019; Linløkken, 2023). In the native waters of the Northern hemisphere, recruitment failure has been reported to occur, with the period between oviposition and fish reaching 1 year of age identified as a sensitive period for survival in this species (Ning et al., 2025).

Ning et al., (2025) describes patterns in recruitment that relate to temperature and wind. Wind and associated waves can disturb deposited eggs, imposing a hydrodynamic stress that directly damages eggs and or results in egg mass dislodgement, both which can result in mass mortality (Rupp 1965 *cited*

in Probst et al., 2009). On shorelines where wave action is higher, adults deposit eggs on substrates in deeper water (Probst et al., 2009). Water temperature has strong and consistent relationships with year class strength (Böhling et al., 1991) with higher water temperatures corresponding to stronger year-class strength (Linløkken, 2023). Summer water temperatures, spawning biomass and stock size of a predator (Pike perch, *Sander lucioperca*) at spawning best explained variation in redfin perch recruitment (Kokkonen et al., 2019). Temperature is an important determinate of growth, especially in colder regions, with warm water temperatures facilitating faster growth in boreal environments (Ning et al., 2025). In a study examining larval fish assemblages in the lower Murray River, South Australia, Cheshire (2010) identified redfin perch larvae were rare in the main-channel larval assemblage and from the few captured, higher captures were noted during lower water temperatures.

Faster growth among some individuals within or between cohorts can give rise to cannibalism (Ning et al., 2025). Cannibalism is common in populations and can begin as early as the larval stage (Brabrand, 1995) and can have strong top-down regulation of redfin perch recruitment (Gwinn and Ingram, 2018). Removing cannibalistic adults can lead to strong recruitment in redfin perch (Closs et al., 2001).

Little has been undertaken on the response of redfin perch to hydrological modification, especially in rivers. One study, Janáč et al., (2010) examined raising water levels ('floods') on artificial oxbows and the spawning response of several fishes, including redfin perch, observing that numbers of age 0+ was independent of flood duration. In a similar study examining artificial lentic water bodies, Jurajda et al., (2004) also observed higher abundances of age 0+ individuals in non-flooded waterbodies as opposed to those that had a flood treatment.

Golden perch (Macquaria ambigua ambigua)

Life history

Golden perch are classified as part of the 'Freshwater straggler' guild of fish as they complete their entire life cycle within the freshwater environments but in the Southern Murray-Darling Basin, on occasions, small juveniles can also be found in nearby estuarine environments (Bice et al., 2018a). In freshwater, golden perch inhabit a range of different freshwater environments: including lakes, rivers, floodplain wetlands, river anabranches and have been stocked into large water reservoirs (Gill et al., 2003; Koehn et al., 2020). They also can be found in estuaries and coastal lagoons (Zampatti et al., 2010) and here can be associated with tubeworm reefs (Schroder et al., 2025). Golden perch also are found using a range of habitat components including large woody debris or snags, rocks macrophytes, bare substrate (Koehn and O'Connor, 1990; Bice, 2010; Koehn et al., 2020)

Golden perch have a periodic life-history strategy (Ferguson et al., 2018) which is characterised by high longevity, delayed/late maturity prioritising a large body size and high fecundity (Winemiller and Rose, 1992). Large body size is advantageous to competition and assists in mediating interactions with other species, enhancing survival (Winemiller 1992). Longevity and opportunistic growth are characteristics suited to semi-arid and temperate hydrology of the Murray-Darling Basin (Mallen-Cooper and Stuart, 2003). Golden perch may move across the riverine landscape in response to increases in discharge and water levels (Koehn et al., 2020). Significant migrations within freshwater (potamodromy) also occur in association with reproduction (O'Connor et al., 2005; Koster et al., 2017).

Distribution, population structure and utility

Golden perch are a large bodied native fish distributed throughout the Murray-Darling Basin, Lake Eyre Basin, Dawson and Fitzroy rivers (Ferguson and Ye, 2012). Golden perch have also been introduced to Western Australia for recreational fishing opportunities (Gill et al., 2003). Historically, golden perch have been targeted by indigenous, recreational and commercial fishers, with remaining commercial fishing within the Murray-Darling Basin restricted to the Lakes and Coorong fishery (Ferguson and Ye, 2012).

The natural range and abundance of Golden perch across the Murray-Darling Basin has declined (Ferguson and Ye, 2012), because of river flow regulation, barriers, thermal pollution and overfishing (Lintermans, 2007; Zampatti et al., 2021). Environmental water releases to promote more natural hydrological conditions (i.e. in-channel rises in flow) has been implemented to improve reproductive and recruitment conditions (King et al., 2009; Price et al., 2019) along with the modification of locks and weirs to include fishways and facilitate connectivity across lengthy river reaches (Stuart et al., 2008; Baumgartner et al., 2010). Stocking for the enhancement of existing populations and establishment of new populations for recreational fishing opportunities (e.g. water impoundments) has also been undertaken for numerous waterways across eastern Australia (Crook et al., 2016; Hunt and Jones, 2018).

Diet

Golden perch are opportunistic carnivores consuming a wide variety of different crustaceans, insects and fish (Baumgartner, 2007; Lintermans, 2007; Wedderburn et al., 2014). Diet varies as fish progress from larvae to juveniles, initially feeding on slow swimming cladocerans, copepod nauplii and copepodites before transgressing to other zooplankton between 12 – 24 mm TL (Ferguson and Ye, 2012). At 25mm insect larvae (Chironomidae) form an important part of the diet, while individuals at size 37mm TL ingest larger Daphnia (Ferguson and Ye, 2012). Adults feed on larger prey items, particularly crustaceans (Baumgartner, 2007) and fish, including various small and large bodied native and introduced species, are often found in the diet (Baumgartner, 2007; Ferguson and Ye, 2012; Wedderburn et al., 2014).

Larvae: copepodites, microcrustaceans, Cladocerans (Arumugam, 1990; Rowland, 1996; Ferguson and Ye, 2012)

Juveniles: macroinvertebrates and microcrustaceans (Lintermans, 2007)

Adults: decapods (Atyidae, Parastacidae), fish, macroinvertebrates (Lintermans, 2007; Wedderburn et al., 2014)

Age and growth

Max age: 26 years (Koehn et al., 2020)

Known to attain a maximum length of 760 mm (Lintermans, 2007). Studies of growth in otoliths and associated environmental parameters over the same growth periods, have revealed complex spatio-temporal growth patterns among individuals, for example differences in life-stages (juveniles and adults), life-history traits (movers vs. residents), dispersal direction (upstream vs. downstream), seasonal period (spring vs. summer) and interactions with larger-scale environmental components linked to climate (e.g. rainfall, discharge, temperature, SOI).

Izzo et al., (2016) used growth (otolith) chronology examining long-term growth patterns of Golden perch (cohorts spanning 1990 to 2008) captured from the River Murray lower lakes and investigate relationships of growth with different environmental predictors (salinity, air temperature, rainfall). A growth model that included mean rainfall over the specified growth year: October to September, best explained annual growth, with higher growth evident in high river flow years and reduced growth during drought periods (Izzo et al. 2016). Most of the variability in otolith growth was attributed to among-individuals, then among-year differences, with less growth variation among-cohorts (Izzo et al. 2016).
Barrow *et al.* (2021) examined the growth of individuals that have been classified as movers, or resident fish after having made at least one movement during their lifetime over four long river-reach scales: lower, mid and upper Murray River and Darling River. Barrow *et al.* (2021) identified that fish that made at least one among-reach movement in their lifetime had higher age-specific annual growth rates during their first eight years than those that did not move. They also observed that resident fish had higher growth variation among individuals up to five years old, postulating that these results taken together could be indicative either of the ability of movers to track stable environmental conditions or residents are exposed to more variable environmental conditions (Barrow et al., 2021).
In a follow up study, Barrow *et al.* (2024) examining movement type on growth (dispersal from natal reach: downstream, upstream, resident or hatchery) across natal and second year spring and summer periods and the influence of hydrology, across ten Murray-Darling mainstem and tributary reaches. Otolith growth in the second year of life was strongly correlated to dispersal with fish of all dispersal types (downstream, upstream and hatchery stocked fish) growing faster than resident fish that remained in their natal location (Barrow et al., 2024). In the natal year, increased spring discharge was associated with faster growth in dispersers over residents. However, increased summer discharge in the natal year corresponded to lower growth for all fish. In years of high spring flows, high recruitment is known to occur in the Southern Murray-Darling Basin (ref) and those fish that can spread downstream or upstream may be able to capitalise on productivity gains or, encounter better environmental conditions across the landscape (Barrow et al., 2021; Barrow et al., 2024). In the same token, low flows over summer is also a natural hydrological pattern for Southern Murray-Darling basin and higher growth during this period likely corresponds to increased concentrations of food resources with lower flow (Barrow et al., 2024).

Size/age at Maturity

Size at Maturity: > 203 mm CFL (Koehn et al., 2020)

Age at Maturity: 3 years for males and 4 years for females (Koehn et al., 2020)

Fecundity and eggs: up to 750,000 eggs (Koehn et al., 2020), 500,000 eggs for a 2.2 kg female (Koehn et al., 2020). Eggs diameter is around 3–4 mm in diameter when water hardened (Lintermans, 2007).

Environmental requirements

Temperature: Golden perch are a warmwater native fish, with a wide temperature tolerance between 4 and 37 °C (Llewellyn and MacDonald, 1980; Koehn et al., 2020).

Michie et al (2020) examined growth in juvenile Golden perch (31 days post hatch) across 4 experimental temperature treatments (13, 16, 20 and 24 °C) over a 30-day period. Fish somatic growth positively responded to temperature with the highest and lowest growth observed at the respective highest (24 °C) and lowest (13 °C) temperature treatments. Temperature significantly affected mouth gape, with smaller

mouth gape observed in smaller fish, consequentially, from reduced body growth in lower temperature treatments (Michie et al 2020). Exposure of juvenile Golden perch to colder water, particularly thermal pollution discharges from water reservoirs into warm water rivers, have the potential to significantly affect early juvenile period of Golden perch and thus, possible follow-on population-level effects.

Salinity: Golden perch tolerate salinity up to 33 psu (Langdon, 1987). Hart et al., (1991) describes most freshwater fish can cope with salinities up to around 10 psu. Clunie et al. (2002) report that post-hardened golden perch eggs LD_{50} salinity tolerance values of 15500 µS/cm (~8 psu).

Oxygen: Crook et al. (2024) observed that for adult Golden perch, movement in the Condamine River (Qld) was undertaken where dissolved oxygen concentrations were below 1 mg/L.

Movement

Golden perch are potamodromous, with adults known to undertake significant migrations across Murray-Darling Basin rivers (Reynolds, 1983; Bice et al., 2018a; Koehn et al., 2020). Golden perch exhibit complex movement behaviour among individuals with some fish undertaking spatially limited movements (Crook, 2004; Zampatti et al., 2018; Bice et al., 2021b; Crook et al., 2024; Harding et al., 2024), while others may undertake more extended movement during high flow events over 10's-100's kilometres (Zampatti et al., 2018; Koehn et al., 2020; Crook et al., 2024; Harding et al., 2024) to 1000's kilometres (Reynolds, 1983; Koehn et al., 2020). Golden perch also exhibit 'homing behaviour' moving to areas previously inhabited areas (O'Connor et al., 2005; Zampatti et al., 2018; Koehn et al., 2020; Crook et al., 2024), while more recent work suggesting philopatry – the returning of adults back to natal origins (Zampatti et al., 2021; Stuart et al., 2025) means consideration of connectivity over lifetime spatial scales is critical to managing and rehabilitating golden perch populations (Zampatti et al., 2018; Koehn et al., 2020; Zampatti et al., 2021).

Detected movements are undertaken across a range of fish ages (e.g. 1-2 years, Mallen-Cooper and Stuart, 2003, 2–8 years and up to around 15 years Barrow et al., 2021). Both upstream and downstream movements occur in equal proportions (Koehn et al., 2020), along with evidence of spatial aggregations of fish around mainstem-tributary junctions (O'Connor et al., 2005; Koehn et al., 2020). Downstream movements of golden perch to tributary junctions, could be reflective of fish seeking appropriate hydraulic conditions that may be lacking in other places (Koster et al 2017). Swimming performance of juveniles considerably reduces below 15 °C (Koehn et al., 2020).

Movement is believed to occur prior to and during the reproductive season (O'Connor et al., 2005; Koster et al., 2017) and often occurring with increases in flow (Koster et al., 2014; Koster et al., 2017; Thiem et al., 2020), particularly large flood events (Koehn et al., 2020). Rising or elevated water temperature has also been associated with movement patterns (O'Connor et al., 2005; Koster et al., 2014; Zampatti et al., 2018). Upstream movements correspond with temperatures > 20 °C (Koehn et al., 2020).

Spawning

Golden perch spawning is known to occur in the river mainstem with females releasing large eggs into the water column to drift downstream (Koehn et al., 2020). Larvae are also pelagic and drift in flow over extended areas (Stuart and Sharpe, 2020) until settlement (Koehn et al., 2020). Spawning occurs over spring and summer (October to February) in the Southern Murray Darling Basin (Koehn et al., 2020). Mostly October to April in the Northern Murray-Darling Basin but also can extend to much of the year (Koehn et al., 2020).

In the Southern Murray-Darling Basin, spawning commences upon elevated river flows and water temperatures in spring (King et al., 2009; Zampatti and Leigh, 2013a; Zampatti and Leigh, 2013b; Koehn et al., 2020). Spawning occurs in the river channel and is responsive to flow, especially flow increases with both small and large flood events known to promote spawning (King et al., 2009; Zampatti and Leigh, 2013a; Koehn et al., 2020). During the Millenium drought (2000–2010), no detection of golden perch larvae over consecutive years in the lower Murray River indicated a lack of spawning success in response to low flow conditions (Cheshire and Ye, 2008). In the Northern Murray-Darling Basin, spawning has also been observed to occurred in isolated waterholes with little or no flow (Koehn et al., 2020).

Across the Murray-Darling Basin, spawning occurs when water temperatures reach 17 °C and temperatures considered optimal above 19 °C (Koehn et al., 2020). Spawning has been observed at temperatures as high as 27 °C and noted as low as 14 °C in the Darling River (Koehn et al., 2020). Spawning detected when temps above 18 °C, but when flows are low and stable, no spawning may be detected (Koster et al 2017).

Recruitment

Large numbers of eggs and larvae are found during or after rises in flow or flooding in main channels, but large numbers of young-of-year (age 0+) are often not evident (Stuart and Sharpe, 2020). Floodplain lakes and terminal lakes are locations where high numbers of young-of-year have been collected (Stuart and Sharpe, 2020).

Recruitment of Golden perch is not uniform but varies spatially and temporally among different rivers in the Murray-Darling Basin and no one region appears a dominant recruitment source (Price et al., 2019). In the Southern basin, the lower Darling, lower, mid and upper Murray Rivers; as well as the Murrumbidgee River are identifiable recruitment sources. However, recruitment is episodic in the lower Darling, lower and upper Murray reaches (Price et al., 2019).

Recruitment is episodic (Ye et al., 2008; Koehn et al., 2020) but corresponds with years that have within-channel flow pulses (Koehn et al., 2020). Recruitment is noticeably higher in years with significant flooding (Ye, 2005; King et al., 2009; Zampatti and Leigh, 2013a; Koehn et al., 2020) with assessments of flow-driven recruitment for the Southern Murray-Darling Basin, indicating the higher the flow pulse during the spawning season, the greater the recruitment is likely to be (Price et al., 2019). Although, a higher flow-greater recruitment relationship is not observed for the Northern Murray-Darling Basin (Price et al., 2019) and recruitment has also been noted in years when floodplains have not been inundated (Mallen-Cooper and Stuart, 2003).

Temperatures between 24 to 28 °C have been identified as being best for Golden perch larvae survival (Price et al., 2019). Temperatures below 16 °C have been noted to result in significant larval mortality (Price et al., 2019). Food density significantly impacts the survival of Golden perch larvae, with higher food densities corresponding with higher rates survival (Price et al., 2019)

Sandy sprat (Hyperlophus vittatus)

Life history

Sandy sprat are a small bodied species that inhabit the marine environment as well as estuaries (Goh, 1992; Rogers and Ward, 2007). Sandy sprat are considered as a marine-estuarine opportunist (Gaughan et al., 1996; Bice et al., 2018a), found along inshore waters of both St Vincent and Spencer Gulf's as well as being found in the Coorong (Rogers and Ward, 2007). Sandy sprat complete their life-history entirely in the marine environment but may opportunistically use estuaries during their early life-history as nurseries and for grow out (Bice et al., 2016). In the Coorong, abundances of sandy sprat have been noted to decline during periods of drought and return in year of higher freshwater discharges (Bice et al., 2016).
Little is understood of the life-history of sandy sprat (Rogers and Ward, 2007), but for a small-bodied fish they are considered to have a short longevity (max age 4 years), moderate growth rate, maturity (50% of females) by the end of their first year, releasing batches of eggs on frequent intervals (5 days) during a lengthy (5 months) spawning season (Rogers and Ward, 2007). These traits are consistent with opportunistic types of life-strategies (Winemiller and Rose, 1992), favouring reproduction to re-populate quickly after disturbance or, where continuous high mortality occurs in the adult stage (Winemiller and Rose, 1992).

Distribution, population structure and utility

Sandy sprat are distributed around the Southern Australia coastline from Kalbarri (Western Australia) to Morton Bay (Queensland), except Tasmania (Gaughan et al., 1996; Neira et al., 1998). Sandy sprat is targeted as part of inshore commercial fisheries as mainly bait and human consumption (Gaughan et al., 1996; Rogers and Ward, 2007). Sandy sprat inhabit open water, bare substrate, shallow and deep water, tube worm's reefs (*Ficopomatus enigmaticus*) (Rogers and Ward, 2007; Bice et al., 2016; Schroder et al., 2025). In the Coorong estuary, sandy sprat is numerous has been found to dominate catches in fish surveys (Zampatti et al., 2010), and prominent in the diets of piscivorous fish (Giatas and Ye, 2015; 2016; Giatas et al., 2018). Sandy sprat are also known to be important prey item to sea birds (Gaughan et al., 1996; Bice et al., 2016). Despite the widespread distribution of sandy sprat along Australia's Southern coastline, commercial and ecological value's, very little is known about its population genetics and stock structure.

Diet

Sand sprat is considered zooplanktivorous (Giatas et al., 2018) and an omnivore, consuming a variety of different zooplankton including copepods, cladocerans, fish eggs, diatoms (Goh, 1992; Bice et al., 2016; Hossain et al., 2017). With increasing fish size, larger prey is ingested and larger crustacea may also be ingested during spring season (Goh, 1992). The diets of fish within estuaries compared to offshore regions may also contrast, with different proportions of prey items with size observed (Goh, 1992).
Larvae: unknown, likely zooplankton (e.g. rotifers, copepod nauplii) (Tregonning et al., 1996)
Juveniles: copepods, cladocerans, diatoms (Goh, 1992)
Adults: crustacea (calanoid, cyclopoid and harpacticoid copepods, cladocerans, amphipods, decapods, ostracods), fish eggs, diatoms and nematodes (Goh, 1992; Bice et al., 2016; Hossain et al., 2017)

Age and growth

Max age: 4 years (Rogers and Ward, 2007)
Slow larval growth (0.11 mm/day) and extended pre-metamorphic periods for fish from the Coorong is comparable to other locations where slow growth in larvae has been attributed to low sea surface temperature and salinities in inshore waters (Rogers and Ward, 2007). Variable freshwater discharge from barrage operations may be a driver of detected lower larval growth (Rogers and Ward, 2007). Size patterns of Sandy sprat in the Coorong largely reflect the presence of juveniles (< 60 mm TL) (Ye et al., 2012; Bice et al., 2016).

Size/age at Maturity

Size at Maturity: SAM_{50} females at 59mm and SAM_{50} females at 58mm in South Australia (Rogers and Ward, 2007). All individuals mature by 75mm in South Australia (Rogers and Ward, 2007). Maturity attained at 60mm in Western Australia (Goh, 1992).
Age at Maturity: Age at Maturity (50% individuals) for males and females is ~12 months (Rogers and Ward, 2007).
Fecundity and eggs: eggs are 0.83–0.95 mm in diameter (Tregonning et al., 1996) and pelagic (Rogers and Ward, 2007). Reported fecundities for female sizes (64–87 mm FL) of 743 to 5600 hydrated oocytes per batch Rogers and Ward, 2007.

Environmental requirements

Temperature: Little is known about the temperature requirements of sandy sprat, however, the distribution of sandy sprat from temperate to sub-tropical waters indicates population capacity to adapt to a range of temperature. Rogers and Ward (2007) identified that temperature had a significant association with density of eggs collected from icthyoplankton samples in South Australian coastal waters. Sea surface temperatures where eggs were collected ranged from 12.4 °C– 23.4 °C (Rogers and Ward, 2007). In Western Australia, Tregonning et al., (1996) reported eggs in ambient conditions (15.2–24.0 °C) reporting that eggs survive well with low mortality of hatchlings.
Salinity: The distribution of sandy sprat in the Coorong estuary appeared to be limited according to salinity (Hossain et al., 2016; Ye et al., 2016). Noell et al., (2009) observed the presence of sandy sprat during Coorong fish surveys between salinities of 12.9–69.2 psu. Hossain *et al.* (2016) identified under higher salinities (40–85 psu) in the South lagoon, sandy sprat were absent from fish assemblages but were abundant at sites within and adjacent to the Murray Mouth region where salinity was lower (below 40 psu) (Hossain et al., 2016). Following increases in freshwater discharges into the Coorong after the Millenium drought, southward range extensions of sandy sprat into the South Lagoon were noted along with other marine-estuarine opportunists (Ye et al., 2016). Bice *et al*. (2016) describes catches of Sandy sprat and conductivity around barrages in the Coorong. Numbers of juveniles at Tauwitchere and Goolwa barrages during November and December sampling in 2014. At the time of sampling, conductivity varied and ranged between 10000 to 50000 µS/cm (~6–33 psu) in the vicinity (Bice et al., 2016).

Movement

Little is understood about movement of sandy sprat, although in Western Australia, distributions across inshore waters and estuaries are represented by smaller size individuals (25–85 mm), while only larger fish (> 90 mm) have been observed in offshore waters (Goh, 1992. This suggests fish may migrate from inshore to offshore waters at ~90 mm in size (Goh, 1992).
Sandy sprat can be found in high numbers in estuaries, dominating fish catches by number (Potter et al., 1990; Noell et al., 2009; Zampatti et al., 2010). In three Western Australian estuaries adjacent to abundant coastal populations, numbers of sandy sprat eggs were low and considered possibly only present due to passive tidal movement (Gaughan et al., 1996). However, high abundances of larvae can be found in these estuaries (Gaughan et al., 1996), suggesting movement from the sea to use estuaries as part of their early life-history (Gaughan et al., 1996; Bice et al., 2016). Using a combination of stable isotope analysis techniques, Bice *et al.* (2016) identified sandy sprat that entered the Coorong as larvae or juveniles, likely benefitted for the purpose of feeding or as nurseries for growing out juvenile fish.
Freshwater may be an important trigger to movement along estuaries as schools of sandy sprat have been observed moving up the Swan River (WA), particularly when freshwater runoff reaches the estuary (Gaughan et al., 1996). Zampatti et al., (2010) also noted that during the lower freshwater inflows into the Coorong during drought, high abundances of sandy sprat were attempting to move upstream towards freshwater sources. Ye *et al*. (2016) indicated with the resumption of freshwater inflows to the Coorong following a significant drought period, reduced salinity and facilitated the southward range extension of sandy sprat to recolonise previous unavailable habitat, including the South Lagoon.

Spawning

Spawning occurs in the marine environment (Gaughan et al., 1996; Rogers and Ward, 2007) as detection of eggs in estuaries is rare (Gaughan et al., 1996). In Western Australia, spawning is reported to occur from May to September, peaking in June and July (Gaughan et al., 1996). Spawning occurs from behind the surf break to up to 14 kilometres offshore (Gaughan et al., 1996). In South Australia, spawning has been reported to occur over spring and summer, from October to February, with spawning peaking in November (Rogers and Ward, 2007). Spawning has been noted during summer in Victoria and during autumn in NSW (Rogers and Ward, 2007).

In Western Australia, Gaughan et al. (1996) report temperatures associated with eggs ranged between 15–22.6 °C with the highest concentration of eggs found in waters between 17–18.5 °C. Surface salinities associated with eggs were 35.4–36.4 psu (Gaughan et al., 1996). Tregonning et al., (1996) hatched eggs in ambient conditions (15.2–24.0 °C, mean: 17 °C) reporting that eggs survive well with low mortality. In South Australia, sea surface temperatures where eggs were collected ranged from 12.4 °C to 23.4 °C with a mean of 18.4 °C (Rogers and Ward, 2007).

Recruitment

Little is understood about the environmental influences on sandy sprat recruitment and year class strength. However, estuaries appear to be important environments used by sandy sprat larvae and juveniles for feeding and nursery habitats (Potter et al., 1990; Gaughan et al., 1996; Bice et al., 2016). Associations between freshwater discharges, distribution and abundance of individuals have been noted for the Coorong (Bice et al., 2016; Ye et al., 2016) and other estuaries (Gaughan et al., 1996). Significant dietary supplement from the transport of organic matter and freshwater zooplankton (*i.e.* trophic subsidy) was also noted, even when low volumes of freshwater entered the Coorong (Bice et al., 2016). Although it is well known that freshwater discharges can augment productivity in estuarine ecosystems (Gillanders and Kingsford, 2002; Gillson, 2011), the effects on sandy sprat recruitment are less clear but possibly beneficial. For instance, upon the resumption of freshwater flows following drought in the Coorong, Ye et al., (2016) reports enhanced recruitment and abundances of estuarine and marine opportunist species, especially small bodied species in the North Lagoon.

Smallmouth hardyhead (Atherinosoma microstoma)

Life history

Smallmouth hardyhead is a small-bodied, euryhaline species that are found across many estuarine and associated ecosystems (e.g. coastal lakes, rivers and wetlands) as well as in marine embayment's and hypersaline lagoons (Lui, 1969; Potter et al., 1986; Lintermans, 2007; Wedderburn et al., 2016; Ye et al., 2020a). Smallmouth hardyhead inhabits shallow water, bare substrate, shallow, macrophytes, salt lakes, salt marsh and tube worm reefs (*Ficopomatus enigmaticus*) (Lintermans, 2007; Jones et al., 2008; Prahalad et al., 2018; Schroder et al., 2025). Smallmouth hardyhead are 'solely estuarine' as it completes its entire life cycle in estuaries (Bice et al., 2018a; Ye et al., 2020a), however, this species can also successfully reproduce in hypersaline waters (Ye et al., 2020a) including enclosed lakes (Lui, 1969).
Smallmouth hardyhead is considered an annual species, having a short longevity of ~1 year (Ye et al., 2020a), rapid growth, batch spawner with spawning occurring over considerable part of its lifetime (4 months) before dying (Molsher et al., 1994). These traits are consistent with opportunistic types of life-strategies (Winemiller and Rose, 1992), favouring reproduction to re-populate quickly after disturbance or, where continuous high mortality occurs in the adult stage (Winemiller and Rose, 1992).

Distribution, population structure and utility

Smallmouth hardyhead is endemic to Southern Australia (Ivantsoff and Crowley, 1996). Its distribution extends from Spencer Gulf (SA) to the mid-coast of NSW, including Tasmania (Ivantsoff and Crowley, 1996; Gomon et al., 2008). Although, it has a broad distribution across southeastern Australia, little is known about the population structure of Smallmouth hardyhead. Several closely related species are recognised within the Atherinosoma genera across Southern Australia (Potter et al., 1986).
Smallmouth hardyhead are one of the most numerically dominant species in estuarine fish assemblages (Molsher et al., 1994; Becker and Laurenson, 2008; Ye et al., 2016; Schroder et al., 2025). Their abundance and small size, makes them one of several important prey species contributing to the Coorong food web for range of predatory fish and piscivorous birds (Deegan et al., 2010; Giatas and Ye, 2015; 2016; Giatas et al., 2018). Furthermore, the value of Smallmouth hardyhead to the Coorong food web is a function of its sustained abundance during low flow periods and is the only available prey fish species under extreme hypersaline conditions in the South lagoon (Giatas and Ye, 2016; Giatas et al., 2018).

Diet

Smallmouth hardyhead are classified as zoobenthivorous fish (Giatas et al 2018) that has an omnivorous diet. Despite its pelagic nature, it feeds on a range of benthic food types (Becker and Laurenson, 2007). A broad range of consumed items has been found in the diet including various Crustacea, Insect larvae, Gastropoda, Polychaeta, Arachnida and detritus (Becker and Laurenson, 2007; Hossain et al., 2017; Giatas et al., 2018).
Larvae: unknown, likely zooplankton (e.g. rotifers, copepod nauplii)
Juveniles: crustaceans (Copepoda, Amphipoda, Ostracoda), insects (Chironomidae, Diptera, Corixidae), nematodes, polychaete worms, Acanthocephala (Hossain et al., 2017).
Adults: crustaceans (copepods, decapods), insect larvae, gastropods, polychaete worms, Arachnida, detritus (Becker and Laurenson, 2007; Hossain et al., 2017).

Age and growth

Max age: 1 year (Molsher et al., 1994)
Smallmouth hardyhead is a short-lived species attaining a longevity of ~1 year (Molsher et al., 1994). Growth is rapid (Hossain, 2016) with larvae and fry (5–15 mm) appearing over October–November and by the following Spring had reached sizes of 36–54 mm (Molsher et al., 1994; Ivantsoff and Crowley, 1996). Adults are > 39 mm and attains a maximum size of 90 mm (Potter et al., 1986; Wedderburn et al., 2016).

Size/age at Maturity

Size at Maturity: In fish < 30mm gonads could not be clearly distinguished (Molsher et al., 1994), but sexual maturity is attained at 45mm (Ye et al 2020a).
Age at Maturity: Less than 1 year (Ye et al., 2020a).
Fecundity and eggs: eggs are 1.6–2.5mm in diameter and adhesive, being attached to macrophytes as a substrate (Lintermans, 2007).

Environmental requirements

Temperature: Little is known about thermal requirements of Smallmouth hardy head. In the Coorong estuary and lagoons where Smallmouth hardyhead reside, Noell et al., (2009) reported mean monthly temperature in the Coorong ranged 11 °C–25 °C for October 2006–September 2008. Similar temperatures were reported in Molsher's *et al.* (1994) study reporting Coorong minimum and maximum water temperatures as 11 and 24 °C, respectively during 1990–1991.
Members of the family Atherinidae are diverse, inhabiting a range of freshwater to marine ecosystems across Southern and Northern Australia (Potter et al., 1986; Ivantsoff and Crowley, 1996). Given the distribution of Smallmouth hardyhead extends from Tasmanian to mid New South Wales (~7 degrees of latitude), there is likely to be some population capacity to adapt to variation in temperature.
Salinity: Smallmouth hardyhead are a euryhaline species, a strong osmoregulator capable of tolerating from near freshwater (Potter et al., 1986; Ivantsoff and Crowley, 1996) to hypersaline conditions (Wedderburn et al., 2016; Ye et al., 2020a). Smallmouth hardyhead are among the most salt tolerant fish in the world (Molsher et al., 1994; Ye et al., 2016) with an upper LD_{50} (upper test range of 70–115 psu) of 108 psu and a lower LD_{50} (lower test range of 5–70 psu) of 3.3 psu (Lui, 1969). Several observations of fish in the field at extreme hypersalinity (> 120 psu) have been noted. They were captured in the Coorong at salinities of 133.5 psu at the height of the Millenium drought (Noell et al., 2009; McNeil et al., 2013b; Ye et al., 2020a). Wedderburn *et al.* (2016) reports observations of small numbers of fish at Salt Creek in the Coorong at salinities of 136.6–160 psu. Geddes *et al*. (1987, *cited in* Wedderburn *et al*. 2016) also reports observing individuals at 149 psu south of Salt Creek in the Coorong.
Adaptation to hypersaline conditions may confer a potential advantage, given access to resources, and lack of competitors and predators, when no other fish species can tolerate extreme hypersalinity (Ye et al., 2024). Despite capacity to tolerate hypersaline conditions, there is some evidence (e.g. abundance, distribution, reproduction and recruitment) to suggest a positive response to freshwater, or low saline conditions (Ye et al., 2023).
Lui (1969) experimentally tested the capacity of Smallmouth hardyhead to incremental changes in salinity (from acclimation 11 days at 70 psu), finding that fish tolerated a rising salinity of 100 psu before mortality was evident and above 120 psu complete mortality occurred. Below 2 psu, complete mortality was also observed (Lui 1969). Lui (1969) reported the biggest fish used in experiment to determine osmoregulation capacity was 3.5 cm, implying that results were applicable to the juvenile stage.
These upper critical values from Lui (1969) have been suggested to limit the Southern distribution of Smallmouth hardyhead in the Coorong in some years (Geddes 1987, cited in Molsher *et al.* 1994). Wedderburn *et al.* (2016) confirmed the impact of these values after observing Smallmouth hardyhead was extirpated from over 50% of its range in the Coorong once salinities reached 120 psu during the height of the Millenium drought. Wedderburn *et al* (2016) undertook salinity tolerance testing of adults (39–75 mm TL), over the range of 25 to 120 psu, finding osmoregulation is constant between salinities of 25–85 psu. Few adults were alive (but disoriented) at the maximum 120 psu before the experiment was terminated.

Movement

The extent of movement of Smallmouth hardyhead is unknown, but there is some evidence of aggregations around fishways and movement through barrages in the Coorong (Bice et al., 2019). Some work has attempted to identify spatial structuring of individuals across the Coorong salinity gradient (Gillanders and Munro, 2012). As a small-bodied fish, use of external tags are difficult to implement and alternative such as natural marks, like otolith microchemistry is likely to be useful (Campana, 1999; Gillanders et al., 2003). The otolith composition of elements, particularly single ratio compositions of

Barium–Strontium, Barium–Calcium or Strontium–Calcium can vary according to the composition of natural waters, which can serve as a useful tracer of fish movement across different natural waters (e.g. freshwater, estuaries, sea) (Campana, 1999; Gillanders and Munro, 2012).
In hypersaline environments, such as the Coorong, relying on single elements to reveal underlying past environmental histories of Smallmouth hardyhead was found to be troublesome, given that the effects of environmental processes (e.g. evaporation) resulted unusual relationships between Barium and salinity (Gillanders and Munro, 2012). Further use of multivariate elemental signatures in future work is required to before reconstructing movement of fish under hypersaline conditions (Gillanders and Munro, 2012).

Spawning

Smallmouth hardy head is a multiple batch spawner (asynchronous oocyte development (Ye et al., 2020a). In females, only one ovary develops, over late winter and early spring is likely to be driven by increases in light and temperature regime (Molsher et al., 1994). Smallmouth hardy head has a protracted spawning season of four months over spring-summer period (September to December), with female gonadal development (GSI) maximised between October and November (Molsher et al., 1994). Spawning occurs in estuaries (Molsher et al., 1994) and specific thermal requirements for spawning is unknown. Spawning occurs over macrophytes, such as *Ruppia sp.*, with eggs attach to plants (Lintermans, 2007; Ye et al., 2023). Smallmouth hardyhead dies after spawning is complete, approximately a year of commencing is life (Molsher *et al*. 1994).
Reproduction has been suggested to be not affected by high salinities after populations have been identified to exist in hypersaline environments (Molsher et al., 1994). Molsher *et al*. (1994) observed spawning to occur over salinities between 32–74 psu in the Coorong and didn't find any shortening in spawning season across a salinity gradient from the Murray estuary to South lagoon. Ye et al., (2023) reported a more recent study of Smallmouth hardyhead gonadal development changing across different salinities (10–80 psu) in the Coorong, with evidence of negative effects upon salinities becoming greater than 60 psu. At these salinities, detection of a sharp increase in the proportion of fish with a gonad containing more than 50% atretic oocytes (Ye et al., 2023).

Recruitment

New recruits are < 40 mm TL (Ye et al., 2023). Given the ecological value of Smallmouth hardyhead to the Coorong ecosystem, recruitment is measured and assessed on an annual basis (Ye et al., 2024). Ecological targets are set against recruitment, both in terms of number of fish captured per unit effort (800 fish per unit effort) in a survey year and the spatial extent of recruits detected with a fixed proportion of sites (> 75%) having a composition of catch containing a minimum proportion of recruits (> 60%). In years following high flows, the target reflecting spatial extent of recruitment is usually met (Ye et al., 2023; 2024). Smallmouth hardyhead are highly responsive to increases in freshwater flows, showing rapid increases in recruitment, abundance and distribution after high flow events (Ye *et al.* 2020a). Increases in freshwater may have multiple effects on recruitment including reducing salinity to tolerable conditions, whilst at the same time improving production of zooplankton (Ye et al., 2023). Additionally, increases in R*uppia sp.* coverage may have benefitted spawning and possibly enhanced recruitment in the Coorong (Ye *et al*. 2023).

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Appendix 2. Species taxonomy, utility, current distribution, allocated waterbody, ecological guild, life-history strategy, age, and growth parameters of eleven CLLMM region fish species. *Denotes water body allocated to species currently commonly inhabits for the purpose of assessing vulnerability (see Section 4).

Species	Family	Utility	Contemporary distribution outlined (Mosley et al. 2018)	*Waterbody	Ecological guild (Ye et al 2016, Bice et al. 2018a)	Life-history strategy (Ferguson et al. 2018)	Longevity Maximum age (years)	Growth Maximum length (mm)	Asymptotic length (L_{inf} mm)
Black bream	Sparidae	Social or Commercial value	Lake Alexandrina, Murray Estuary, North Lagoon, South Lagoon	Coorong	Estuarine	Periodic	32	600	540
Bony herring	Clupeidae	Social or Commercial value	Lake Alexandrina, Murray Estuary, North Lagoon, South Lagoon, Lake Albert	Lower Lakes	Freshwater-estuarine opportunist	Opportunistic	11	470	470
Common carp	Cyprinidae	Social or Commercial value	Lake Alexandrina, Lake Albert, Murray Estuary, North Lagoon	Lower Lakes	Freshwater-straggler (exotic)	Intermediate	34	1,200	1,200
Congolli	Bovichtidae	Ecological value	Lake Alexandrina, Lake Albert, Murray Estuary, North Lagoon, South Lagoon	Lower Lakes/ Coorong	Diadromous (catadromous)	Intermediate	7	350	360
Golden perch	Percichthyidae	Social or Commercial value	Lake Alexandrina, Lake Albert, Murray Estuary, North Lagoon	Lower Lakes	Freshwater-straggler (native)	Periodic	26	760	760
Greenback flounder	Pleuronectidae	Social or Commercial value	Murray Estuary, North Lagoon, South Lagoon	Coorong	Marine-estuarine opportunist	Intermediate	10	450	220–230
Mulloway	Sciaenidae	Social or Commercial value	Lake Alexandrina, Murray Estuary, North Lagoon	Coorong	Marine-estuarine opportunist	Periodic	42	1,750	1,380
Redfin perch	Percidae	Social or Commercial value	Lake Alexandrina, Lake Albert, Murray Estuary	Lower Lakes	Freshwater-straggler (exotic)/Freshwater-estuarine opportunist	Intermediate	22	600	25–35
Sandy sprat	Clupeidae	Ecological value	Lake Alexandrina, Murray Estuary, North Lagoon	Coorong	Marine-estuarine opportunist	Opportunistic	4	100	78.1
Smallmouth hardyhead	Atherinidae	Ecological value	Lake Alexandrina, Lake Albert, Murray Estuary, North Lagoon, South Lagoon	Coorong	Estuarine	Opportunistic	1	90	90
Yelloweye mullet	Mugilidae	Social or Commercial value	Lake Alexandrina, Murray Estuary, North Lagoon, South Lagoon	Coorong	Marine-estuarine opportunist	Opportunistic	10	483	429

Appendix 3. Environmental descriptions of eleven CLLMM region fish species. Note: In parenthesis denotes study kind for tolerance range: 'field obs.' = field observation; 'expt.' = laboratory experiment; 'aqua.' = aquaculture trial and 'rev.' = literature review. Temperature values refer to water temperature.

Species	Response to Hydrological Conditions		Maximum Temperature (°C)	Maximum Salinity (psu)	Minimum DO (mg/L)	Minimum pH	Habitat Description or Occurrence			
	Low Flow	High Flow					Eggs	Larvae	Juveniles	Adults
Black bream	Low winter flow prevents flushing of low DO/stagnant water for new larvae. Reduce discharge and increased hypoxia impacts growth. Tolerates limited hypersalinity but likely moves to suitable conditions, Brackish–marine salinities used for development and growth. Low flow: spawning occurs upstream in estuaries. High salinity under drought implicated in fish kills.	High flows flush eggs from estuary, destroy salt wedge and delay spawning. Intermediate-size flow linked to strong recruitment. Growth variability corresponds with rainfall and temperature. Moves along estuary, can leave estuary on high flow. Migrates upstream for spawning. Spawning in some intermittently open estuaries after rainfall.	Observed at 28.5 °C	Reported up to 114 psu	50% sat. few hatched <40% sat. complete egg/larvae mortality. <1mg/L implicated with fish kills.	pH 4–5 implicated with fish kills	Estuary	Estuary	Estuary	Estuary
							Pelagic: float on halocline	Pelagic: above or below halocline	Benthic	Benthic
							Temperature (°C)			
							14.9–23.1(field obs.)	9.3–23.1(field obs.)	9.5–37(field obs., expt.)	9.5–37(field obs., expt.)
							Salinity (psu)			
							10–35(expt.)	15–34(expt., field obs.)	0–90(expt.)	3.5–114(field obs.)
Bony herring	Tolerant of low-flow conditions, but susceptible to low water temperatures over winter prompting fish kills	High flows providing beneficial production gains and greater recruitment.	Reported to up to 38 °C	Observed at 72 psu in the Coorong.	2	pH 4.8–8.6. Low tolerance pH <6	River, lake and wetland	River, lake and wetland	River, lake and wetland, estuary	River, lake and wetland, estuary
							Pelagic	Pelagic	Pelagic	Pelagic
							Temperature (°C)			
							Undefined — expert opinion informed	Undefined — expert opinion informed	Undefined — expert opinion informed	9–38(rev.)
							Salinity (psu)			
							Undefined — expert opinion informed	Undefined — expert opinion informed	Undefined — expert opinion informed	39 (rev.)
Common carp	Tolerant of low-flow conditions, but susceptible to brackish conditions.	Under high-flow conditions spawning and recruitment is maximised	Reported to up to 40.6 °C	Reported up to 20 psu	4	pH range 4.5–10.5	River, lake, creek wetland, floodplain	River, lake, creek wetland, floodplain	River, lake, wetland, floodplain, estuary	River, lake, wetland, floodplain
							Benthic: attached to plants/detritus	Pelagic	Pelagic and benthic	Pelagic and benthic
							Temperature (°C)			
							10–38(expt.)	3–35(expt.)	3–40.6(expt.)	2–32(field obs., expt.)
							Salinity (psu)			
							0–10(expt.)	0–10(expt.)	0–20(expt.)	0–15(expt.)

Species	Response to Hydrological Conditions		Maximum Temperature (°C)	Maximum Salinity (psu)	Minimum DO (mg/L)	Minimum pH	Habitat Description or Occurrence			
	Low Flow	High Flow					Eggs	Larvae	Juveniles	Adults
Golden perch	Low-flow conditions impact spawning and recruitment	High flow facilitates movement and dispersal. High flow triggers spawning and benefits recruitment.	Reported up to 37 °C	Reported up to 33 psu	< 1	Undefined — expert opinion informed	River, creek and floodplain	River, creek, floodplain, wetland	River, creek, lake floodplain, estuary	River, creek, lake floodplain, impoundment
							Pelagic	Pelagic	Benthic	Benthic
							Temperature (°C)			
							Undefined — expert opinion informed	13–28 (rev.)	Undefined — expert opinion informed	4–37 (rev.)
							Salinity (psu)			
							8 (rev.)	Undefined — expert opinion informed	33(rev.)	Undefined — expert opinion informed
Greenback flounder	In drought years, young-of year are nearly absent from Coorong lagoons and higher abundances of individuals are near estuary mouth region	Higher abundance and wider distribution of young-of-year in years following flooding after higher freshwater flows. Many adults tagged were located at salinities below 40 psu, indicating avoidance of hypersaline concentrations.	Reported up to 32.6 °C	Reported up to 92 psu	Undefined — expert opinion informed	Undefined — expert opinion informed	Estuary or sea	Estuary or sea	Estuary or sea	Estuary or sea
							Pelagic	Pelagic	Benthic	Benthic
							Temperature (°C)			
							6–21 (aqua.)	Optimal: 15 (aqua.) upper: 21 (aqua.)	32.6 (rev.)	11–27.8 (field obs.)
							Salinity (psu)			
							5–45 (aqua)	15–35 (aqua)	0–92 (expt.)	0.3–67 (field obs.)
Mulloway	Low flow and drought conditions reduce the amount of nursery habitat available for juveniles	High flows provide low salinity which is favoured by juveniles. High flow also increases turbidity, which is beneficial for ambush predator concealment. Salinity gradient and turbidity provide cues for adult aggregation and for larvae to locate estuary	*Argyrosomus sp.* reported up to 37.5 °C	Reported up to 74 psu	Undefined — expert opinion informed	Undefined — expert opinion informed	Sea	Nearshore and offshore in sea	Estuary or sea	Estuary or sea
							Pelagic	Pelagic	Benthic	Benthic
							Temperature (°C)			
							Optimal: 22.5–23.5 (aqua.)	Optimal: 13–26 (aqua.)	12–28 (field obs.)	10–31 (rev.)
							Salinity (psu)			
							Undefined — expert opinion informed	Low: 0.6, Optimal: 5–12.5 (aqua.) 35 (rev.)	0.6–74 (aqua., expt.)	Undefined — expert opinion informed

Species	Response to Hydrological Conditions		Maximum Temperature (°C)	Maximum Salinity (psu)	Minimum DO (mg/L)	Minimum pH	Habitat Description and Occurrence			
	Low Flow	High Flow					Eggs	Larvae	Juveniles	Adults
Redfin perch	Low flow likely increases salinity and potentially elevates water temperature which less suitable for egg and larvae survival.	High flow likely provides low salinity and cooler temperatures suit spawning. Potential recruitment benefit from freshwater flow assisting larvae distribution and facilitating low salinity conditions in Estuary.	Reported up to 36.2 °C	Reported up to 22 psu	1.3	pH range 3.5–12	River, lake, wetland, estuary	River, lake, wetland, estuary	River, lake, wetland, estuary	River, lake, wetland, estuary
							Benthic: attached to plants/detritus	Pelagic	Pelagic and benthic: littoral and	Benthic: littoral and deeper water
							Temperature (°C)			
							6–24(aqua.)	5–30(rev.)	8–36(rev.)	4–36(aqua., rev.)
							Salinity (psu)			
							0–12(field obs., expt.)	0–9.6(rev.)	0–7(field obs., expt.)	0–22 (field obs.)
Yelloweye mullet	During drought periods, species avoids extreme hypersaline conditions in the south lagoon	Higher abundance after flooding in Coorong, re-distribution into hypersaline lagoons indicated positive response to reduced salinity. Evidence of freshwater subsidy of diet after flooding in Coorong.	Reported up to 33 °C	Observed at 95 psu in the Coorong	1.2 mg/L associated with a fish kill	Undefined — expert opinion informed	Coastal bays and open sea, estuary	Coastal bays and open sea estuary	Nearshore sea and estuary	Nearshore sea and estuary
							Pelagic	Pelagic	Benthic	Benthic
							Temperature (°C)			
							Undefined — expert opinion informed	Undefined — expert opinion informed	10–33(aqua., rev.)	10–33(aqua., rev.)
							Salinity (psu)			
							Undefined — expert opinion informed	Undefined — expert opinion informed	Brackish (field)–90.5 (expt.)	Brackish (field obs.)–95 (field obs.)
Sandy sprat	Drought and elevated salinity reduce distribution in the Coorong. Higher abundances attempting to move upstream to freshwater sources during drought.	High freshwater flows correspond to lower salinity and extending southward distribution in the Coorong.	Observed at 24 °C	Observed at 69.2 psu	Undefined — expert opinion informed	Undefined — expert opinion informed	Nearshore and open sea, estuary	Nearshore and open sea, estuary	Nearshore and open sea, estuary	Nearshore and open sea, estuary
							Pelagic	Pelagic	Pelagic	Pelagic
							Temperature (°C)			
							15.2–24(field obs.)	Undefined — expert opinion informed	Undefined — expert opinion informed	Undefined — expert opinion informed
							Salinity (psu)			
							35.5–36.4(field obs.)	Undefined — expert opinion informed	6–69.2(field obs.)	12.9–69.2(field obs.)

Species	Response to Hydrological Conditions		Maximum Temperature (°C)	Maximum Salinity (psu)	Minimum DO (mg/L)	Minimum pH	Habitat Description and Occurrence			
	Low Flow	High Flow					Eggs	Larvae	Juveniles	Adults
Smallmouth hardyhead	Dominates fish assemblages in Coorong lagoons in low flow years due to having the highest salinity tolerance of all fish species inhabiting the Coorong.	Higher abundance after flooding suggest positive freshwater response. Freshwater flow reduces salinity in hypersaline environments likely benefits reproduction.	Highest temperature observed in field 25 °C	Observed in the field up to 160 psu	Undefined — expert opinion informed	Undefined — expert opinion informed	Estuary, hypersaline lake	Estuary, hypersaline lake	Estuary, hypersaline lake	Estuary, hypersaline lake
							Benthic: attached to vegetation	Pelagic	Benthic	Benthic
							Temperature (°C)			
							Undefined — expert opinion informed	Undefined — expert opinion informed	11–25(field obs.)	11–25(field obs.)
							Salinity (psu)			
							32–74(field obs.)	Undefined — expert opinion informed	2–115(expt.)	0–160(rev, field obs.)
Congolli	Recruitment failure under drought conditions. Lack of freshwater flow prevents connectivity between ecosystems due to artificial barriers	Migration occurs with flow pulses, possibly indicating flow as a cue or assists in facilitating migration.	Observed to be present in the Coorong at 27.2 °C	Maximum salinity reported is 104 psu	Undefined — expert opinion informed	Undefined — expert opinion informed	Undescribed, suspected in sea	Estuary and sea	Estuary	Freshwater (female), estuary
							Egg stage suspected in sea	Pelagic	Benthic	Benthic
							Temperature (°C)			
							Undefined — expert opinion informed	11.2–27.2(field obs.)	11.2–27.2(field obs.)	11.2–27.2 (field obs.)
							Salinity (psu)			
							Undefined — expert opinion informed	Undefined — expert opinion informed	0–104(expt.)	Undefined — expert opinion informed

Appendix 4. Reproductive information for eleven CLLMM region fish species. *Reference to age unspecified.

Species	Reproduction									
	Age at Maturity$_{50}$ (years) *Age at Maturity (years)		Spawner type	Fecundity	Location	Season	Months	Environmental		Hydrological
	Females	Males								
Black bream	1.9*	4.3*	Asynchronous, multiple spawner, batches across spawning season (Sarre and Potter 1999)	Fecundity indeterminate (Earl *et al.* 2016) ~3,000,000 eggs large female (Ye *et al.* 2024).	Upper reaches of estuaries	Spring–Summer	September–February	Temperature: 15–28.5 °C	Salinity: 15–45 psu	Intermediate flood level beneficial for Salt-wedge development and stratification critical. High flow delays spawning.
Bony herring	2	1	Synchronous or group-synchronous (single batch per season reported)	33,000– 880,000 eggs (199–403 mm TL) (Puckridge and Walker 1990).	Rivers, creeks, lakes and backwaters	Spring–Summer	September–February	Temperature: 21–23 °C	Salinity: Spawn in freshwater.	Spawning is not flow dependent
Common carp	3	2	Asynchronous, multiple batch spawners (Smith and Walker 2004)	~163,000 eggs/kg (Sivakumaran *et al.* 2003). > 1,000,000 eggs/year (Koehn *et al.* 2000)	Freshwater: shallow margins of rivers, lakes and floodplains	Spring–Summer	October–February	Temperature: 11–29 °C, ~17 oC is favourable for spawning.	Salinity: 0–10 psu, < 5 psu is optimum.	elevated water levels provide inundated vegetation for oviposition.
Congolli	3	2	Asynchronous	Fecundity and eggs are undescribed (Neira *et al.* 1998; Cheshire *et al.* 2013	Spawning likely in sea	Late Autumn–Winter	May–August	Temperature: unknown, GSI increase and migration in winter.	Salinity: Spawning likely in sea (~35 psu)	Migrates to sea late winter. freshwater outflows for spawning is unclear.
Golden perch	4	3	Asynchronous (2 egg stages	up to 750,000 eggs (Koehn *et al.* 2020), 500,000	Freshwater; river and creeks	Spring–Summer	September–January	Temperature: 14–27 °C Spawning	Salinity; little known but spawning	Flow pulses and large floods important for

			in ovary Koehn et al. 2020)	eggs for a 2.2 kg female (Koehn *et al.* 2020)				starts 17 °C, optimal above > 19 °C	described in freshwater.	spawning, coincide suitable temperature.
Greenback flounder	2	2	Multiple batches released over protracted season (Earl and Ye 2016).	Indeterminate (Earl 2014), estimate of 820,880–1,969,070 (247–343 mm) (Crawford 1984).	Estuary or Sea	Autumn–Winter	March–October (peak April–July)	Temperature: ~12 °C. < 6 or > 21 °C complete egg mortality.	Salinity: 15–45 psu	Spawning observed in brackish to slight hypersaline waters
Mulloway	5	6	Group-synchronous, indeterminate (Farmer 2008)	Indeterminate, Battaglene and Talbot (1994) two broodfish of ~10 kilograms shed ~1,000,000 eggs.	Sea	Spring–Early Summer	October–January (peak November)	Temperature: > 19 °C.	Salinity: Spawning occurs in the sea at marine salinities (~35 psu)	Mulloway in spawning condition congregate at the mouths of estuaries with elevated freshwater outflow
Redfin perch	3*	1*	Group-synchronous (single egg ribbon per spawning event)	Highly fecund ~120,000–140,000 eggs/kg (Ning et al. 2025).	Rivers, lakes. Also, estuaries	Late Winter–Spring	August–October	Temperature: 5–19 °C. 11–15 °C most common.	Salinity: Spawns in salinity up to 9.6 psu.	temperature cue, hydrological influences not well understood.
Sandy sprat	1	1	Multiple batch spawner	Females (64–87 mm FL), 743–5,600 hydrated oocytes/batch (Rogers and Ward 2007)	Nearshore marine environment	Spring–Summer	September–February	Temperature: 15–22.6 °C. Mean temperatures where eggs present 17–18.5 °C.	Salinity: Spawning occurs in the sea (~35 psu)	Spawns in sea, any dependence on freshwater for spawning is unknown.
Smallmouth	< 1	< 1	multiple batch spawner asynchronous oocyte development	5-40 ripe eggs in ovary (Molsher et al. 1994)	Estuary, hypersaline lakes	Spring–Autumn	September–April	Temperature: Little known about requirements	Salinity: < 74 psu. atretic gonads at > 60 psu	Unknown, but potential flow benefits on aquatic plants used for oviposition.

			(Ye et al. 2020a).							
Yelloweye mullet	3	2	Asynchronous, multiple batches released (Earl and Ferguson 2013)	125,000–630,000 eggs (245–391 mm female) (Thomson 1957)	Nearshore, coastal bays, also estuaries	Summer-Autumn-Winter	December–August (SA/Coorong)	Temperature: Little understood	Salinity: spawns in sea (~35 psu).	Coordinated with mouth opening after heavy rains is hypothesised for WA intermittent estuaries

Appendix 5. Summary table of recruitment information for eleven fish species inhabiting the Coorong and Lower Lakes.

		Recruitment		
Species	**Location**	**Environmental**		**Hydrological**
Black bream	Estuary	Salinity: 10–20 psu for eggs, egg survival low at < 10 psu	Salinity: larvae observed to tolerate up to 34 psu	Both high and low freshwater flow negatively affects recruitment. Freshwater flows into estuaries critical, increasing food resources, maintains salt-wedge conditions for recruitment
Bony herring	Rivers channels and backwaters	Temperature: Populations widely distributed, particularly northern Australia. Potential capacity to acclimatise to warmer water temperature	Salinity: observed in estuaries, sometimes at high salinity levels (i.e. > 35 psu). Effect of salinity on juveniles is not well understood.	Recruitment in River Murray occurs irrespective of high flows. Recruitment reduced during drought-level flow.
Common carp	Freshwater: shallow areas of lakes, rivers, floodplains and wetlands	Temperature: warm temperatures (~25 °C) favourable to egg and larval development.	Salinity: low salinity (ideally, < 5 psu) required for recruitment.	Recruitment reduced in drought years. Floods creates large floodplain wetland areas creating 'recruitment hotspots'
Congolli	Estuary	Temperature: juveniles migrate into estuaries during spring.	Salinity: Juveniles are excellent osmo-regulators. Higher abundances recorded in years with higher freshwater flows. Recruitment failure in extreme droughts years.	During drought, fewer larvae encountered in the Coorong estuary. In subsequent high flow years, abundant juveniles recorded moving upstream.
Golden perch	Rivers and creeks, floodplain and terminal lakes	Temperature: 24–28 °C suits recruitment. Increased larval mortality < 16 °C.	Salinity: little understood about effects of salinity on recruitment. Juveniles tolerant to 33 psu and occasionally found in Coorong estuary.	Floods increase recruitment In the Southern MDB. Recruitment occurs under lower flows in northern MDB. Warm-water and high flow produces greater food resources abundance — supports stronger recruitment.
Greenback flounder	Estuary or Sea	Temperature: juvenile upper limit of 32.6 °C	Salinity: higher marine salinities important for eggs and larvae, but juveniles have tolerance to low salinity (to 3 psu).	Freshwater flow may guide larvae into estuarine nursery habitat. High freshwater flow years, stronger recruitment is observed. High flow maintains nursery habitat, producing stronger recruitment.
Mulloway	Estuary or Sea	Temperature: Juveniles tolerate 12–28 °C	Optimum salinities for larval growth is brackish water (5–12.5 psu). For juveniles, survival and growth is similar at low to medium salinities (5–35 psu).	Larval growth greater under 5–12.5 psu than higher salinity. Juvenile recruitment correlated with freshwater flow. Freshwater assists in guiding newly settled juveniles to estuaries.
Redfin perch	Rivers, lakes and wetlands. Also, estuaries	Temperature: recruitment occurs over spring and summer. Warmer water beneficial for growth in larval (11–17 °C) and juvenile (19–23 °C).	Salinity: low salinity (ideally, < 5 psu) is favoured for recruitment.	Flooding does not appear to affect recruitment in artificial wetlands and channels. Little is understood about hydrology on recruitment. Higher flows reduce salinity, benefiting larval survival in freshwater populations.
Sandy sprat	Sea, nearshore marine environment, estuary	Temperature: little understood of requirements for early life-history period or later juvenile life stage.	Salinity: higher abundance in Coorong where salinity is < 40 psu. Recruitment occurs in sea at ~35 psu.	Movement and distribution correspond with higher freshwater flow in estuaries. Drought and low flow conditions increase hypersalinity, which is avoided. Freshwater subsidy in diet also detected, suggesting multiple benefits of freshwater to this species.
Smallmouth hardyhead	Estuary, hypersaline lakes	Temperature: little understood of requirements for early life-history period or later juvenile life stage.	Salinity: Recruitment can occur under hypersaline conditions when other species cannot tolerate conditions.	High freshwater flow benefits recruitment, Coorong recruitment targets met over a broader area in years with greater freshwater flow. Mechanism for recruitment benefit unclear, but increased productivity could be an explanation.
Yelloweye mullet	Estuary or Sea	Temperature: tolerant 10–33 °C. Recruitment originates in the sea; some individuals move into estuaries	Salinity: juveniles tolerate up to 90.5 psu. Juveniles move from the sea, opportunistically, into estuaries and exposed to variable salinity.	Juveniles opportunistically use estuaries. In the Coorong, higher freshwater flows increase southerly distributions with evidence of freshwater subsidy in diet. Higher freshwater flows assist in mouth-opening, facilitating connectivity to sea, enabling recruitment for a land-locked lake fishery (Lake George, SA).

Appendix 6. Projected average daily temperature (°C) for annual and seasonal periods at 2050- and 2090-time horizons.

AVERAGE DAILY TEMPERATURE (°C)	BASELINE (1986-2005)	PROJECTED ANOMALY (2050)	PROJECTED ANOMALY (2090)
Annual	Max: 21.6 Mean: 16.0 Min: 10.4	+1.3 +1.2 +1.2	+1.7 +1.7 +1.7
Summer	Max: 26.7 Mean: 20.2 Min: 13.7	+1.2 +1.2 +1.2	+1.5 +1.5 +1.5
Autumn	Max: 22.5 Mean: 17.1 Min: 11.6	+1.2 +1.3 +1.3	+1.6 +1.6 +1.7
Winter	Max: 16.1 Mean: 11.6 Min: 7.1	+1.2 +1.2 +1.2	+1.8 +1.8 +1.9
Spring	Max: 20.9 Mean: 15.1 Min: 9.3	+1.5 +1.3 +1.1	+1.9 +1.8 +1.6
No. of extreme temperature days 40°C or above	2	+2	+3

Appendix 7. Tables of published projected freshwater runoff for 2050- and 2085-time horizons.

ANNUAL RUNOFF (JAN-DEC)					
BASELINE (1976-2005)		EMISSION SCENARIO: RCP4.5			
1990		2050		2085	
	mm/year	± mm/year	% Change	± mm/year	% Change
10th percentile	34.7	-12.1	-28.8	-15.7	-36.2
50th percentile	35.8	-3.3	-8.9	-5.6	-16.1
90th percentile	40.7	4.6	13.4	-0.7	-2.0

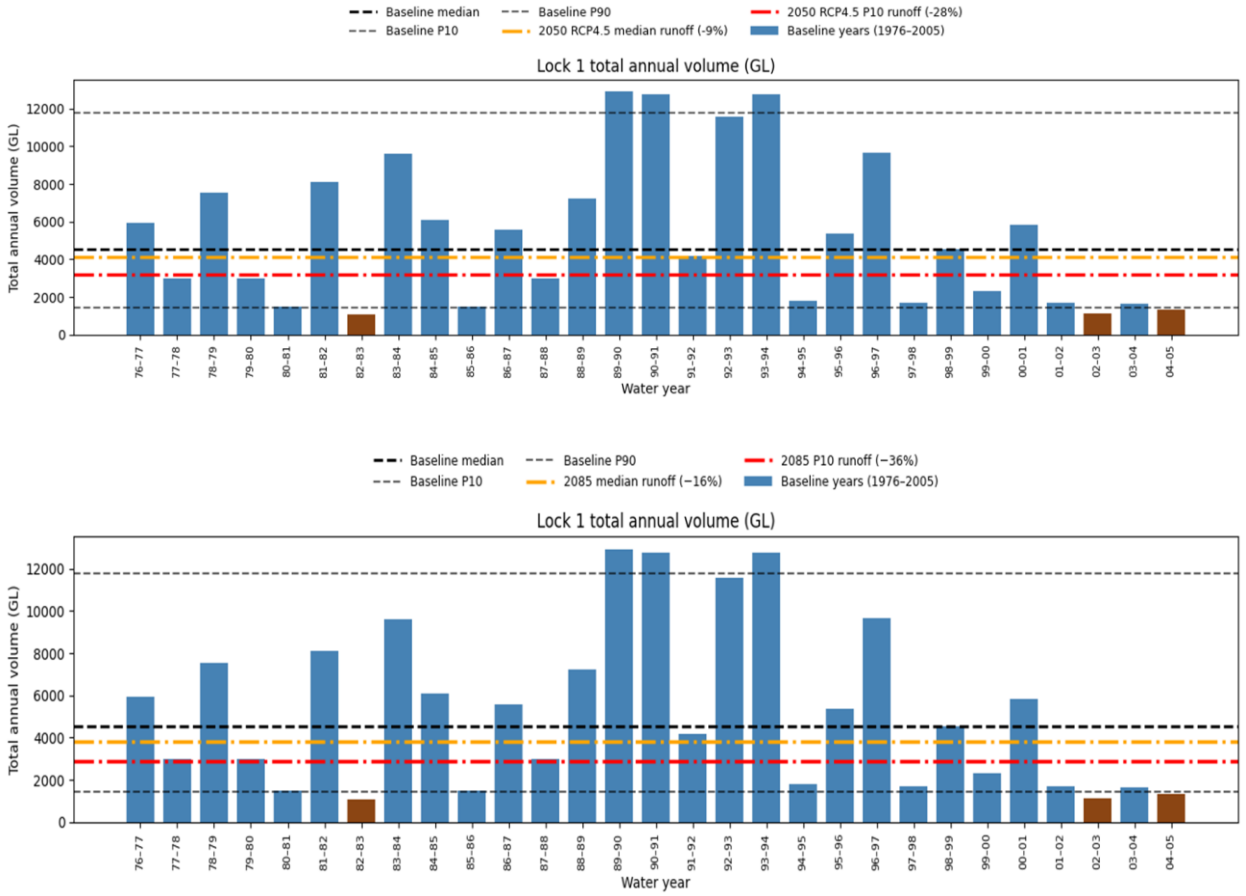
SUMMER SEASON RUNOFF (DEC-FEB)					
Baseline (1976-2005)		EMISSION SCENARIO: RCP4.5			
1990		2050		2085	
	mm/year	± mm/year	% Change	± mm/year	% Change
10th percentile	5.3	-1.2	-17.7	-1.4	-20.7
50th percentile	5.6	-0.3	-4.7	-0.7	-13.3
90th percentile	6.4	0.7	13.6	-0.2	-3.4

AUTUMN SEASON RUNOFF (MAR-MAY)					
BASELINE (1976-2005)		EMISSION SCENARIO: RCP4.5			
1990		2050		2085	
	mm/year	± mm/year	% Change	± mm/year	% Change
10th percentile	5.1	-1.4	-25.5	-1.9	-32.3
50th percentile	5.4	-0.6	-12.7	-1.4	-26.1
90th percentile	6.1	-0.2	-4.1	-0.4	-7.3

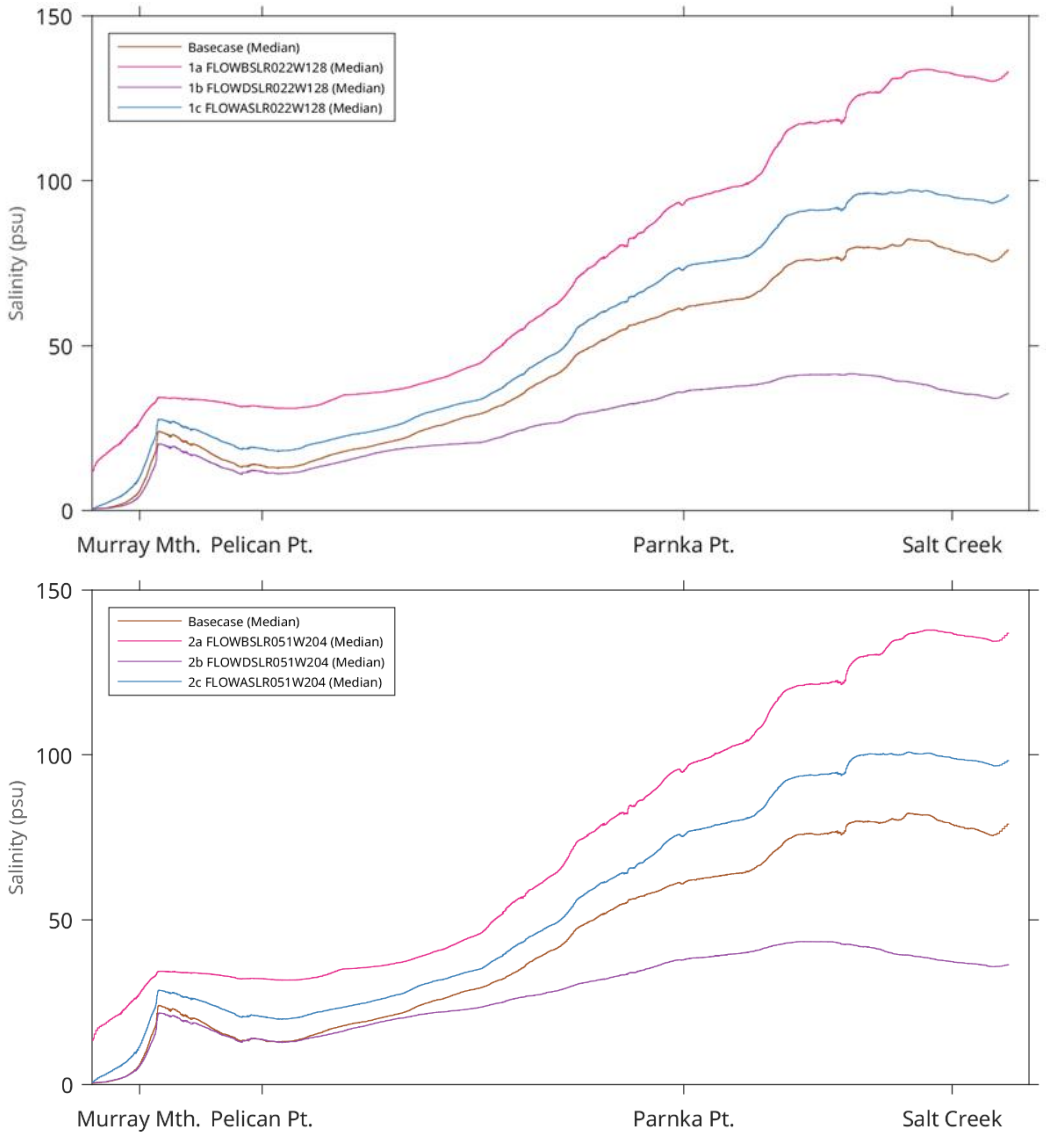
WINTER SEASON RUNOFF (JUN-AUG)					
BASELINE (1976-2005)		EMISSION SCENARIO: RCP4.5			
1990		2050		2085	
	mm/year	± mm/year	% Change	± mm/year	% Change
10th percentile	12.2	-4.1	-29.1	-5.3	-36.0
50th percentile	13.4	-2.0	-13.7	-1.5	-12.2
90th percentile	14.8	4.2	36.7	0.7	5.3

SPRING SEASON RUNOFF (SEPT-NOV)					
BASELINE (1976-2005)		EMISSION SCENARIO: RCP4.5			
1990		2050		2085	
	mm/year	± mm/year	% Change	± mm/year	% Change
10th percentile	11.0	-5.7	-37.9	-6.6	-46.6
50th percentile	11.8	-1.0	-8.5	-3.5	-28.0
90th percentile	13.9	1.3	10.9	0	-0.2

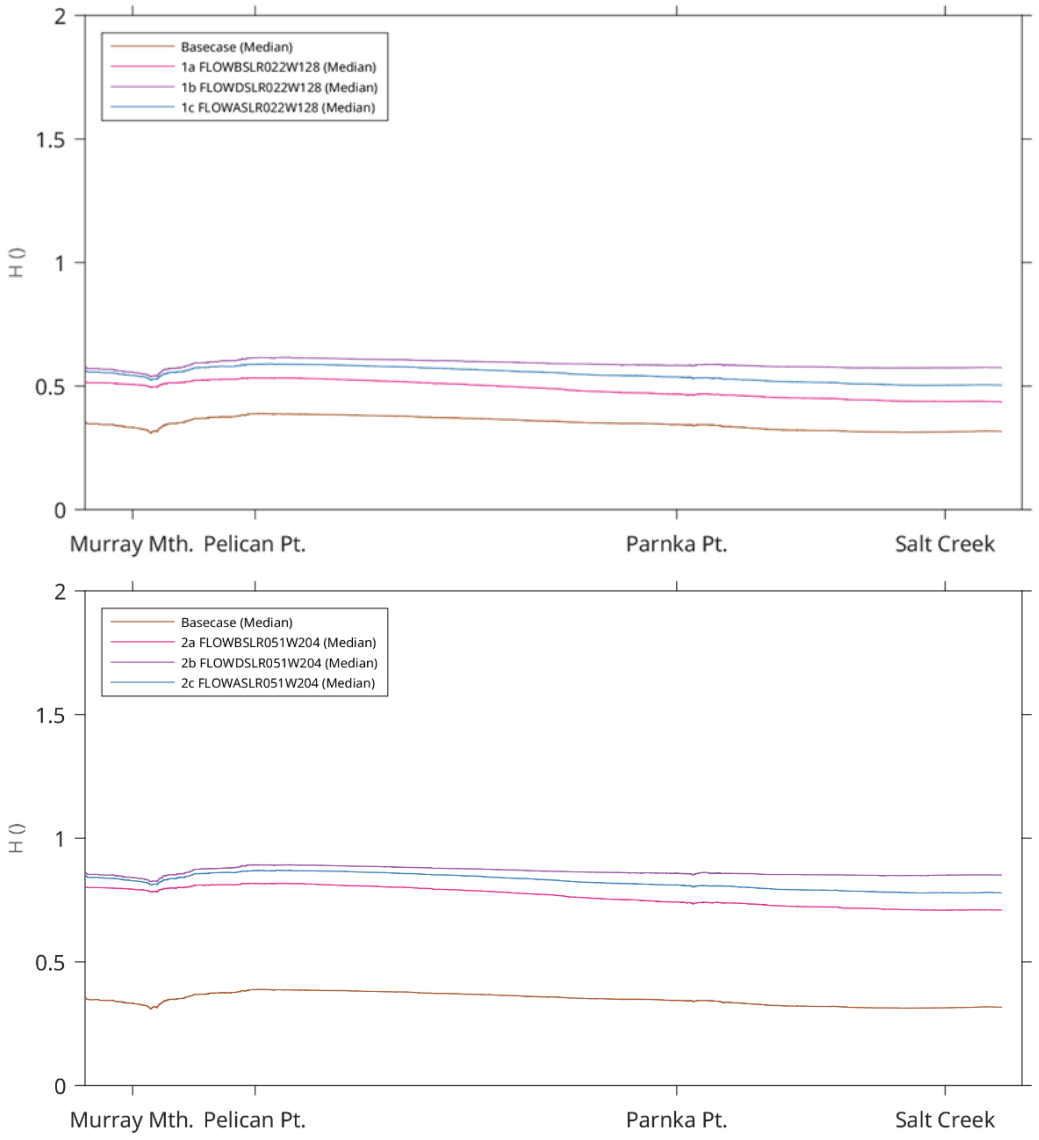
Appendix 8. River Murray Lock 1 (annual volume) for the Southern Murray-Darling Basin runoff projections baseline period (1976–2005). Median annual total volume (4523 GL/year) over the 1976–2005 period (thick black broken line) with upper and lower (fine black broken lines) represent 90th and 10th percentile annual volume levels. Brown hydrograph bars represent years with annual volumes < 10th percentile level. Colour lines represent shift in median annual volume corresponding to projected runoff ensemble 50th (yellow broken line) and 10th percentile (red broken line) change values. Top chart is 2050 projections; bottom chart is 2085 (2090) projections.



Appendix 9. Modelled salinity distribution (median value) as a function of flow state, sea-level at 2050 (upper chart) and 2090 (lower chart) time horizons.



Appendix 10. Modelled water height (median value) as a function of flow state, sea-level at 2050 (upper chart) and 2090 (lower chart) time horizons



Appendix 11. Rational for scoring four environmental drivers of climate change exposure

Temperature

Rees et al. (2022) suggest that CLLMM ecosystems will inevitably become warmer. That means greater energetic demands placed on fish and potentially, high temperatures that result in fish moving to seek suitable temperature (refugia) or range shifts. A conservative threshold of 30 °C was considered as a point at which extreme temperatures affecting fish begin.

Air temperature was used as a surrogate for water temperature for both the Lower Lakes and the Coorong. Given the shallow nature of these water bodies, air temperature corresponds well with water temperature and has been used in previous climate change assessments (Egan *et al.* 2020) and studies to understand water temperature influence on Coorong fish species (e.g. Izzo *et al.* 2016).

Climate change projections for air temperature (RCP 4.5, NARClim 1.5) for annual and seasonal periods (average data), and for extreme days > 40 °C), at 2050- and 2090-time horizons, were obtained online from the South Australian climate change projections viewer (Department for Environment and Water 2026) for the10 x 10 km grid cell located at coordinate 139.15E, 35.55S.

Temperature exposure for both the Lower lakes and the Coorong could be assessed using the same set of air temperature projections. Additionally, we cross-checked projected seasonal air temperature anomalies against modelled seasonal water temperatures across the Coorong and Lower Lakes as outlined in Figure 32 in Dang et al. (2026). Scores presented for temperature exposure are consistent for both estimates of future temperature exposure.

Both annual and seasonal average temperatures were assessed. Average annual daily mean maximum temperature characterise the chronic effect of long-term gradual warming. Average seasonal daily mean temperature reflect acute effect temporary elevated warming, for temperature-sensitive processes like reproduction, or periods when elevated temperature may have extreme effects (e.g. autumn low water levels). Assessment of seasonal temperature was based on the seasonal baseline plus the projected future anomaly. In line with Egan et al. (2020), the air temperature anomalies above + 3 °C by 2090 were considered to reflect high seasonal temperature exposure. Air temperature anomaly below + 2 °C by 2090 was considered to reflect low temperature exposure. Upon applying temperature scoring to summer, autumn, winter and spring seasons, temperature scores were equivalent. Thus, seasonal temperature is therefore adequately represented by one seasonal score for each scenario.

Freshwater flows

Runoff projections were used to gauge future freshwater flow. Runoff, instead of precipitation, was chosen, given evaporation is included in calculating runoff and for arid landscapes, considering evaporation is important given a small amount of precipitation will become runoff (Teng *et al.* 2012). Runoff is also useful because of the direct linkage to streamflow, as catchment runoff generated ultimately leaves the

basin as streamflow at the outlet (Davie 2019). Thus, over sufficiently long periods (decades), when changes in catchment storage are negligible, runoff approximates streamflow (Peel *et al.* 2010; Zhang *et al.* 2001). Accordingly, use of projected runoff for long-term (multi-decadal) periods with changes relative to a 1976–2005 baseline — it was assumed that historic median discharge approximated historic median runoff. A projected change in future runoff could be used to gauge an approximate water volume. Such methods to gauge climate change on long-term water availability may be useful when assessing the potential impact on the water, environment of food security (Teng *et al.* 2012).

Projected runoff for the Southern Murray-Darling Basin (RCP4.5, CMIP5 data) were obtained from the Australian Water Outlook website (Bureau of Meteorology 2026). Available annual, and all seasonal projections for the 50th (median) and 10th percentile values, at time horizons 2050 and 2085 (closest available to 2090) were downloaded and presented as relative change (%) from a 1976–2005 baseline period (see Appendix 7). Projected annual runoff was used to estimate total water volumes required to support other flow-dependent drivers such connectivity (e.g. fishways, see connectivity below). For freshwater flow, only seasonal runoff projections were used to score on exposure, given flow-dependent ecological processes are seasonal. Freshwater flow exposure scores were equal for both the Lower Lakes and the Coorong, as Southern Murray River Basin runoff projections apply to both water bodies.

Projected runoff change was scored against the following thresholds. Given River Murray average annual discharge has historically decreased by 61% (CSIRO, 2008), further flow reductions may have larger consequences. We considered that if runoff reduced by 30% or more over a period, it was considered high and scored a 3. Runoff reduction between 15 to 30% scored 2, while low reductions in runoff were considered up to 15%. Assumptions of current management with respect to keeping Lower Lakes at > 0.0 m AHD, 100% of the time (O'Connor *et al.* 2015) was also considered.

Projected ensemble median (50th percentile) and lower (10th percentile) annual runoff declined under both the 2050 and 2085 climate projections. Drought-level annual flow volumes were considered to represent flow volumes in impacts may become severe. The baseline 10th percentile annual flow volume was estimated at 1,440 GL/year, which closely corresponded to the annual flow threshold associated with drought conditions in the region (< 1,500 GL/year; Ryan, 2018).

To translate projected runoff, change into River Murray inflow, a total annual volume hydrograph (Lock 1, Murray River) was constructed for projected runoff baseline period. This hydrograph was developed using Lock 1 average daily discharge observations (1976–2005), corrected for missing values, using imputed means of available observations for the corresponding month and year (Appendix 8). Plotted lines representing the baseline annual total flow volume are overlaid for 10th, 50th (median) and 90th percentiles (black broken lines). With the baseline median annual runoff aligned on the median annual total volume, projected change in annual runoff (%) was plotted as a deviation from the baseline median annual volume and the corresponding water volume estimated. Projected annual total volume for the lower

(10th percentile, red broken line) and upper (50th percentile, yellow broken line) levels for freshwater inflow is presented in Appendix 8.

Salinity

Salinity is inversely related to freshwater volume, with salinity of CLLMM region water bodies being lowest during large floods and highest during drought (Bice and Ye 2009; Gibbs *et al.* 2018; Ye *et al.* 2016). The Coorong Dynamics Model and CLLMM-wide Model provides estimates of salinity along the Coorong and Lower Lakes, respectively, as a function of freshwater discharge using flow states (Dang *et al.* 2026).

Based on the species profile information, salinity between 0–60 psu was considered 'habitable' for most CLLMM species and that higher salinities would likely trigger movement towards lower saline conditions or potentially result in mortality. Given salinity tolerance is range bound with differing upper and lower limits among fish species and life stages, we assessed salinity exposure over three nominal ranges (0–20, 20.1–40 and 40.1–60 psu).

Outputs from hydrodynamic modelling were integrated into scoring two ways. For the Coorong (Goolwa Barrage to Salt Creek), Coorong Dynamic Model output and plots of median salinity were provided and the cumulative proportion of salinity within the categories (0–20, 20.1–40 and 40.1–60 psu) was calculated and compared between a Base Case flow period and two different flow states (Extreme Dry or Dry) for each time horizon (2050 and 2090). For each category, if the proportional amount of salinity changed 35% or less between the Base Case and the flow state, then this was considered as low and scored 1. A high level of change was considered as > 70% proportional change relative to the Base Case flow.

For the Lower Lakes, modelled salinity estimates with respect to 2050 and 2090 climate change scenarios for Dry and Extreme Dry flow states, were obtained from Figures 41 and 42 in Dang et al. (2026). In these charts, salinity is presented for Dry and Extreme Dry flow states for the locations: Lake Alexandrina, Lake Albert and Barrages US. To keep scoring comparisons consistent between the Coorong and Lower Lakes based on area, the following area proportions were considered for Lake Alexandrina (65%), Lake Albert (21%) and the remaining 14% considered to be Barrage US. Similar for the Coorong, distribution of salinity across the levels 0–20, 20.1–40 and 40.1–60 psu were undertaken for a Base Case flow period and Dry and Extreme Dry flow states at each of the time horizons 2050 and 2090. For each category, if the proportional amount of salinity changed 35% or less between the Base Case and the flow state, then this was considered as low and scored 1. A high level of change was considered as > 70% proportional change relative to the Base Case flow.

Connectivity

Connectivity can influence fish movement, both within (e.g. access to habitat patches) and between (e.g. estuary mouth open) water bodies, while also different degrees of connectivity can arise as water level rises both naturally (e.g. freshwater flow) or anthropogenically through accelerated sea-level rise. Connectivity may be also maintained artificially through dredging, such as keeping the mouth open of

estuaries under low flow periods. Fishways, are another example where artificially, engineering can be used to facilitate the movement of fish through the water infrastructure like weirs or barrages, although the efficiency is dependent on suitable hydraulic conditions (e.g. lake water level) to operate. Sea-level rise adds complexity to connectivity in estuaries, where in some instances additional water level submerging biogenic habitats like seagrass and saltmarsh is expected to lead to negative impacts (loss of access to important habitat). However, increasing sea-level will also contribute to greater connectivity between the Coorong and Lower Lakes (Rees et al., 2022), through active water movement through barrages or fishways (e.g. reverse head conditions) or through 'over-topping' and inundation of infrastructure such as barrage spillways.

In this study, connectivity consisted of scoring 'within' and 'between' waterbody attributes (Table 4). First, for within waterbody connectivity, a minimum passage depth of 0.5 m was considered to enable passage of up to the largest size fish, although it is recognised that juveniles and small-bodied fishes may move across shallower water, which was also taken into consideration during scoring. For the Lower Lakes, it was also considered that lake levels maintained at 0.0 m AHD would provide sufficient depth across the lake for fish to access other areas. For instance, complete disconnection of Lake Albert from Lake Alexandrina is identified to occur at -0.5 m AHD (Gibbs *et al.* 2018). For the Coorong, at 0.0 m AHD (sea-level), it is identified that the North and South lagoon's contain average depth of 1.2 and 1.4 m, respectively (Gibbs *et al.* 2018). Plots of modelled water levels along the Coorong at lower (Extreme Dry flow state) and higher (Dry flow state) flows, including expected sea-level rise, were examined as part of scoring (Appendix 8).

Three attributes were used to score connectivity between Lower Lakes and the Coorong (Table 4). First, water bodies are connected by barrage fishways. Fishways are instrumental for fish movement throughout the year, particularly during closure of barrages during low flow periods. This current management was assumed to occur in the future. The operation of fishways require at least 52 GL/year operational water volume (O'Connor *et al.* 2015) but this volume only considers only water moving through the fishway, not 'attraction flows'. The provision of total annual water volumes of at least 3150 GL/year in any year and an average of 4000 GL/year in a three-year period is expected to meet year round operational requirements of fishways (O'Connor *et al.* 2015). Fishways also require Lower Lakes water levels of 0.4 m AHD or greater to operate as higher Lower Lakes water levels enable freshwater to flow through the fishway into the Coorong. This 'head difference' was also taken into consideration using modelled water height estimates for Lower Lakes (Barrage US) and Coorong (Murray Estuary) from modelling outputs (P10 for lower and Median in Table 6 of Dang et al. (2026). Management of Lower Lake levels at 0.4 m for 95% of time, projected annual runoff and knowledge of Lower Lake volume and evaporation were taken into consideration as part of scoring.

The effect of rising sea-level rise is also likely to increase connectivity between the Coorong and Lower Lakes, particularly in the High ocean state (Rees *et al.* 2022). Although barrage infrastructure provides some resistance to rising sea-levels,

inundation of lower-lying spillway's set between the barrages may be subjected to increased breaching, thereby, providing potential connective pathways. To score the inundation of spillways (+0.83 m AHD), plots of modelled water levels for the Coorong were examined at the two flow-states for 2050 and 2090 scenarios (Appendix 10). Other water height modelling for the Lower Lakes in Table 6 of Dang et al. (2026) also was considered.

Finally, connectivity between the Coorong and the sea through an open Murray mouth is vital for fish to access productive estuarine environments and/or to complete life cycle. Open estuary mouths also critical for the export of salt loads, and for the exchange of fresh marine water which aids in the regulation of salinity (Brookes *et al.* 2009). Scoring of connectivity through mouth was undertaken using annual runoff projections in conjunction with specified annual flow volumes of 730–1070 GL/year needed to keep the mouth open (O'Connor *et al.* 2015). The provision of total annual water volumes of at least 3150 GL/year in any year and an average of 4000 GL/year in a three-year period is expected to meet maintenance requirements of the Murray Mouth (O'Connor *et al.* 2015). Current management to deploying dredging to keep the Murray mouth open during low-flow periods, was also considered during scoring.

Appendix 12. Survey questions provided to commercial fishers, pertaining the impact of climate change on the Coorong and Lower Lakes.

	SCENARIO 1: Lower Lakes Stays mostly freshwater and stable Hotter weather More intesnse floods and droughts			**SCENARIO 2: Coorong Estuary** Hotter and more variable Less freshwater Saltier conditions, especially in the South Lagoon						
Response	Lower Lakes large-mesh			Coorong large-mesh			Coorong small-mesh			Comments
	Unlikely	Neutral	Likely	Unlikely	Neutral	Likely	Unlikely	Neutral	Likely	
Change the amount of fishing effort	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Move to another fishing block within the sector	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Stop fishing for primary species	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Switch gear/sector	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Adjust timing of fishing period	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Adjust length of fishing season	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Invest in new technology or Assets	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Adjusted TACE	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Improve fish handling	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Adjust sale price	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Diversity markets (local, intra-/inter-state)	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Seek informaiton about adaptation options	☐	☐	☐	☐	☐	☐	☐	☐	☐	
Communication with concerned stakeholders	☐	☐	☐	☐	☐	☐	☐	☐	☐	

Appendix 13. Definition of terms used within report.

TERM	DEFINITION
Adaptive capacity	The ability of a species, ecosystem, or system to adjust to climate change, moderate potential damage, or cope with consequences.
Adaptation planning	Process of adjusting to actual or expected climate change and its effects.
AHD	Australian Height Datum — The official national vertical datum used within Australia.
Anomaly	The deviation of a climate variable from a defined baseline period (e.g. temperature, runoff).
Barrage	Structures are designed to regulate water flow between the Lower Lakes and the Coorong, and to limit seawater intrusion.
Baseline	A reference period used to compare future climate projections (e.g. 1976–2005 runoff baseline period).
Biogenic habitat	Habitat is created by living organisms that support other species (e.g. seagrass beds, macrophytes).
Climate change vulnerability assessment (CCVA)	Also, vulnerability assessment. Method that uses a structured approach to evaluate the susceptibility of species or systems to climate change based on sensitivity and exposure.
Confidence weighting	Adjustment of sensitivity or exposure scores according to confidence in the supporting information, reducing the influence of scores based on less certain information.
Connectivity	The degree to which aquatic habitats are physically and functionally linked, enabling movement of organisms, water, and resources.
Coorong	An extensive coastal lagoon system in South Australia, comprising the Murray Estuary, North Lagoon, and South Lagoon.
Coupled Model Intercomparison Project Phase 5 (CMIP5)	The fifth phase of an international climate-modelling framework that provides coordinated climate model simulations used to assess historical and projected climate conditions.
Driver	A key environmental factor (e.g. air temperature) that is altered by a climate pressure (e.g. warming) that influences fish biology or ecology.
Environmental water	Water managed or delivered to maintain or improve ecological conditions and ecosystem processes.
Exposure	The extent to which a species or system experiences climatic or environmental change.
Fecundity	The reproductive output of an individual, commonly expressed as the number of eggs produced by a female during a spawning event or reproductive period.
Fish assemblage	A group of fish species occurring together in a particular habitat or ecosystem.
Fish guild	A classification of fish based on ecological function or habitat use (e.g. freshwater stragglers, diadromous species).
Fishway	A structure designed to facilitate fish movement past an artificial barrier such as a barrage or weir.
Flow regime	The pattern of river discharge over time, including magnitude, timing, and variability of flows.
Fork length (FL)	Length of a fish measured from the tip of the snout to the posterior end of the middle caudal rays.
Freshwater inflow	Water entering the system from rivers or catchments, influencing salinity, connectivity, and habitat conditions.
Gigalitre (GL)	A unit of volume equal to one billion litres (1,000 megalitres).
Gigalitres per year (GL/year)	A unit of annual water volume equal to one billion litres per year.

Head difference	The difference in water level or hydraulic pressure between locations that drives water movement; across the barrages, the difference between Lower Lakes and Coorong water levels influences flow and fishway operation.
High Ocean Exchange	A hypothetical secondary hydrological state characterised by increasing marine influence on waterbody hydrodynamics.
Hydroclimate	The interaction between climate and the hydrological cycle, which determines the availability and movement of water within a catchment.
Hydrodynamics	The movement and behaviour of water within an aquatic system, including processes associated with water levels, flows, currents and exchange between water bodies.
Hydrological Minimum	A hypothetical primary hydrological state characterised by reduced freshwater inflows, increased drying, and elevated salinity.
Hypersalinity	Salinity 40 psu and often extremely high salinity (often > 60 psu), typically resulting from evaporation that exceeds freshwater inputs.
Hypoxic	Low concentrations of dissolved oxygen in water, at levels that may adversely affect aquatic organisms.
Lakes and Coorong Fishery	A multi-species commercial fishery operating across the Lower Lakes, Coorong Estuary and adjacent nearshore marine waters.
Life-history traits	Biological characteristics (e.g. growth, reproduction, migration) that define a species' ecological strategy.
Life-history strategy	A combination of traits describing how a species allocates energy to growth, survival and reproduction; in this report broadly classified as opportunistic, periodic or intermediate.
Life stage	A distinct developmental stage in a fish's life cycle (e.g. egg, larva, juvenile or adult).
Lower Lakes	Lake Alexandrina, Lake Albert and other waters extending upstream from the barrages to the junctions of inflowing tributaries.
Macrophytes	Aquatic plants (submerged or emergent) that provide habitat and influence ecosystem processes.
Marine heatwave	A prolonged discrete anomalously warm water event that can be described by its duration, intensity, rate of evolution, and spatial extent.
Median	The middle value in a dataset, also referred to as the 50th percentile value. Used to represent central tendency in climate projections.
Microsiemens per centimetre (µS/cm)	A unit of electrical conductivity commonly used to measure the capacity of water to conduct an electrical current.
Murray–Darling Basin	The river catchment encompassing the Murray and Darling river systems and their tributaries across south-eastern Australia.
Periodic life-history strategy	A strategy generally characterised by relatively large body size, greater longevity, later maturity and high fecundity, with reproductive success often varying among years.
Projection	An estimate of modelled future environmental conditions based on scenarios about climate drivers to represent possible outcomes.
Opportunistic life-history strategy	A strategy generally characterised by small body size, early maturity, short generation time and frequent reproduction, enabling rapid population responses when conditions are favourable.
Representative Concentration Pathway	A greenhouse-gas concentration trajectory used to represent different possible levels of future radiative forcing and associated climate change.
Recruitment	The addition of new individuals (typically juveniles) to a population.
Practical salinity unit (psu)	A commonly used expression of salinity based on the Practical Salinity Scale.
Resilience	The capacity of a system or population to withstand disturbance and maintain function.

Reverse head	A condition in which Coorong water levels exceed Lower Lakes water levels, reversing the usual hydraulic gradient and potentially allowing saline water to move towards or into the Lower Lakes.
Runoff	The portion of precipitation that flows over land into waterways, influencing freshwater availability.
Salinity gradient	The spatial variation in salinity across an aquatic system, which may range from freshwater to marine conditions in an estuary.
SAM	The minimum fish length reached upon attaining maturity for each sex.
Scenario (climate)	A plausible future condition based on assumptions about climate pressure (e.g. drying, warming, sea-level rise).
Sea-level rise (SLR)	An increase in sea height due to thermal expansion and melting ice, influencing coastal and estuarine systems.
Sensitivity	The degree to which a species or system can be affected by environmental change.
Time horizon (TH)	A future, multi-decade period for which climate or environmental conditions are assessed; in this study, 2050 (2041–2059) and 2090 (2081–2099).
Total length (TL)	Length of a fish measured from the tip of the snout to the posterior tip of the caudal rays.
Trait-based approach	A method that uses species' biological characteristics to assess responses to environmental change.
Vulnerability	The overall susceptibility to climate change, typically a function of exposure and sensitivity.
Water quality	Physical and chemical characteristics used to describe or quantify the quality of water pertaining to ecosystem health.


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