Assessment of Victoria's estuaries using the Index of Estuary Condition: Background and Methods 2021

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Front cover photo credit

Mouth of Fitzroy River estuary (Jarred Obst, Glenelg Hopkins Catchment Management Authority).

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Acknowledgment

We acknowledge and respect Victorian Traditional Owners as the original custodians of Victoria's land and waters, their unique ability to care for Country and deep spiritual connection to it. We honour Elders past and present whose knowledge and wisdom has ensured the continuation of culture and traditional practices.

We are committed to genuinely partner, and meaningfully engage, with Victoria's Traditional Owners and Aboriginal communities to support the protection of Country, the maintenance of spiritual and cultural practices and their broader aspirations in the 21st century and beyond.

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1. Introduction

1.1 Context for the IEC

The Victorian Waterway Management Program has well-established methods for monitoring the environmental condition of rivers and wetlands. The Index of Estuary Condition (IEC) framework was developed to address a lack of consistent and systematic measurement of estuarine condition in Victoria (Arundel et al. 2009) and to better align estuarine assessments with the established condition assessment and reporting approaches used for other waterway types. Specifically, the IEC assesses estuary condition for the purposes of:

- Reporting on estuarine condition to communities
- Guiding state policy and regional planning of estuary management
- Providing a benchmark for estuary environmental condition.

State-wide condition assessment programs provide information about the overall environmental condition of Victoria's waterways and guide state policy and regional investment programs. The current approach is to rotate these assessments among estuaries, wetlands, and rivers at the long time-frames (10 years or more) expected for changes in condition at the broad spatial scales assessed, and in response to changes in threats, management regimes or environmental contexts. In the intervening periods, revisions to policy and regional investment decisions will be informed by targeted monitoring of key aquatic values and threats at specific waterway assets, evaluations of the effectiveness of management interventions, and strategic research to fill critical knowledge gaps.

The first state-wide IEC benchmark assessed 101 Victorian estuaries, and the results are reported in DELWP (2021).

1.2 What are estuaries and why they are important?

Estuaries occur where fresh waters meet the sea, usually at the mouths of rivers. They are partially enclosed waterbodies that may be permanently or intermittently open to the sea and, because of the dilution of ocean water with fresh water, have salinities that vary from almost fresh to very saline (Tagliapietra et al. 2009). As many organisms, nutrients and pollutants move between inland rivers and coastal waters through estuaries, the condition of these crucial conduits affects the biota and condition of upstream waterways and downstream marine environments.

Estuaries are highly dynamic and complex environments. They contain diverse habitats and ecosystems, including open water, rocky reefs, intertidal sand and mudflats, mangroves, saltmarshes and seagrass beds. Estuaries fulfil many key ecological functions; for instance, they are major sites of organic matter decomposition and nutrient cycling, provide habitat for waterbirds and are important nursery grounds for many fish species. They also support a range of ecosystem services (Pinto and Marques 2015, Boerema and Meire 2017), including being culturally important for Traditional Owners, supporting industries such as tourism, ports and recreational fisheries, providing storm and flood protection, and maintaining water quality and nutrient cycling.

1.3 Measuring estuary condition

1.3.1 Defining estuary condition

The Victorian Waterway Management Strategy (DEPI 2013a) acknowledges that threatening processes (threats) will influence waterway condition and values. Broadly, the main threats to Victorian estuaries are intensification of catchment development (including urbanisation of coastal regions), modification to flow regimes, and modifications to estuary mouths (Barton et al. 2008). Threat assessment and their mitigation rely on an effective measure of environmental condition in estuaries.

As there is no universally accepted definition of environmental condition, it is important to define what is meant by “condition” in the IEC. The following definition is used:

> *Environmental condition measures the extent to which environmental attributes that characterise an ecosystem in its desired state have been retained (or degraded).*

This definition is consistent with relevant Victorian policies and tools including Habitat Hectares (Parkes et al. 2003), the Index of Wetland Condition (IWC: DSE 2005a, 2009, DELWP 2016, 2018) and the Index of Stream Condition (ISC: DSE 2005b).

In this context, the ‘desired state’ may be characterised in several ways, including:

- supporting complex ecological structures and networks
- supporting maximum diversity of native species
- being free of invasive or exotic species
- having natural ecological, hydrological, and geomorphological processes that continue to operate effectively, including maintaining spatial and functional links with other systems and regions
- being relatively undisturbed by post-European human activity.

1.3.2 Selecting metrics to measure estuary condition

Estuaries are complex and dynamic ecosystems owing to the mixing of fresh and marine waters and the connectivity between estuaries and their catchments. Not only are estuaries inherently complex, they respond to threats in many ways. These responses usually represent changes to how an estuary functions (e.g. cycling nutrients) or reductions in an estuary’s ability to perform certain functions (e.g. supply suitable habitat for fauna, provide particular ecosystem services).

Many complex environmental factors or estuary responses that can contribute to the concept of estuarine condition cannot be feasibly measured within a broad-scale ‘snapshot’ assessment such as the IEC. For example, hydrology will influence many of the components of estuaries such as water quality and biota at a range of different time scales (e.g. daily, seasonally, annually and over decades), but it is infeasible to measure across these time scales for all estuaries across the state.

As it is not feasible in the IEC to directly measure all the complex factors and responses that contribute to estuary condition, parameters (metrics) are carefully selected as proxies for these. Collectively, these proxies provide information on environmental processes or values and/or the pressures, stressors or both (collectively threats) thought to act on these processes or values to influence condition (DEPI 2010, 2013b, Roper et al. 2011, Sinclair and Kohout 2018, Venables and Boon 2016).

The IEC includes metrics that characterise both threats and condition. Threat metrics represent information on stressors and pressures. Stressors are defined as physical, chemical, environmental, and biological attributes or processes that reduce estuary condition whereas pressures are natural and anthropogenic attributes or processes that introduce or aggregate the effect(s) of stressors. Condition metrics represent measurable aspects (or proxies) of estuary condition.

Threats are responsible for changes in estuary condition. Condition metrics provide information on estuary condition, often integrating the influences of multiple interacting threats. An example of a condition metric is pelagic Chlorophyll *a* concentration, which is influenced by threatening processes such as changes to nutrient availability and hydrology (stressors), and point and non-point source nutrients arising from altered catchment land uses (pressures). An example of a threat metric is the Modification of Freshwater Inflows to estuaries, which can influence water quality and the composition of fish fauna assemblages (aspects of condition). Assigning each IEC metric to represent measures of either condition or threat aids interpretation of results and conceptual understanding of observed estuary environmental condition to guide management options.

Multiple metrics are required to adequately assess environmental condition (Sinclair et al. 2015, 2018). Collectively, these metrics aim to distil environmental complexity into simpler forms that remain scientifically valid (Stoddard et al. 2008), based on the assumption that this multimetric approach is more robust and sensitive to a broader range of disturbances than individual metrics alone (Karr 1981). Ideally, each of these metrics should be:

- Conceptually grounded with a valid and defensible scientific basis, supported by robust theoretical cause-and-effect relationships and/or empirical evidence

- Responsive to threats without extremely high levels of variance that make it difficult to identify differences among estuaries or over time (also referred to as having a high signal-to-noise ratio).

These attributes were optimised when developing the metrics for the five sub-indices that comprise the IEC. There are also desirable attributes for the suite of metrics as a whole that are used to calculate aggregated indices. Suites of metrics should ideally:

- Provide information on the extent to which present condition is influenced by threats operating at multiple temporal and spatial scales
- Be non-redundant, with each metric or sub-group of metrics introducing novel information to avoid over-emphasising some desirable or undesirable attributes of the system and unintentionally inflating or deflating the measure of overall estuary condition.

1.4 Development of the Index of Estuary Condition

The IEC framework was developed to address a lack of consistent and systematic measurement of estuarine condition in Victoria (Arundel et al. 2009). The IEC was conceptualized (Arundel et al. 2008), drafted (Arundel et al. 2009), trialled to refine methods (Pope et al. 2015), and reviewed and revised (Woodland and Cook 2015). The aim was to align the IEC with the established state-wide condition assessment and reporting tools used for Victorian rivers, streams and wetlands. In particular, the IEC was developed to be consistent with the Index of Stream Condition (ISC) because estuaries are fed by streams and rivers. Its application is consistent with the ISC approach of assessing assets, threats and condition to support adaptive management of natural resources through the Victorian waterway management framework (Arundel et al. 2009). The IEC was also developed to yield information suitable for informing regional planning of waterway management and support development of Regional Waterway Strategies.

The multi-metric IEC for Victorian estuaries was recommended following expert workshops and interrogation of a range of candidate measures with existing knowledge, evidence, and monitoring and assessment approaches (Arundel et al. 2009). Since its inception in 2008, the IEC has been refined as understanding of Victoria's estuaries has improved, monitoring approaches have been tested, and new approaches to assessing estuarine condition have become available. Criteria used to assess candidate measures throughout the development and refinement of the IEC included:

- cost effectiveness
- measurement variability (i.e. accuracy and precision)
- responsiveness of metrics and measurements to threats (and threat alleviation via management interventions)
- interpretability.

The IEC, like the ISC, required component measures to be transparent, intuitive and to represent an appropriate balance among cost, rapidity of assessment, and scientific rigour (DEPI 2013b).

The methods outlined in this report are built on a suite of work and input from experienced policy makers, natural resource managers, and scientists. The measures (and approaches to their measurement) ultimately included in the IEC were considered suitable for state-wide application and assessment in all types of Victorian estuaries (Woodland and Cook 2015).

1.5 Structure of the Index of Estuary Condition

1.5.1 Estuaries for inclusion in the IEC

Estuaries were included for assessment in the IEC if they are at least 1 km long or have lagoonal lengths of at least 300 m. Watercourses that run into coastal embayments (i.e. Western Port, Port Phillip Bay, Corner Inlet) and into the Gippsland Lakes were also included.

1.5.2 Sub-indices, measures and metrics for the IEC

The IEC integrates information on five key aspects of estuary condition described in separate sub-indices: Physical Form, Hydrology, Water Quality, Flora and Fish. These sub-indices are aggregated to provide an overall 'snapshot' measure of environmental condition at the time of monitoring (Section 1.3).

The rationale for, and development of, the measures and metrics that underpin the five IEC sub-indices are fully described in reports listed in Table 1. Metrics are quantifiable measurements that are used to track the status of one or more variables that collectively describe the status of a measure. The purpose of the current IEC methods report is to consolidate and summarize these methods.

At the time of release, this IEC methods report provided the most current and comprehensive description of all relevant methods and includes updates for some of those methods that had not previously been reported.

Table 1: References describing the rationale and methods applied to various measures for the first state-wide IEC benchmark.

Sub-index	Measure	Reference
Physical Form	Artificial Barriers	Pope et al. (2015)
	Artificial Shorelines	This report
Hydrology	Modification of Marine Exchange	McSweeney (2019)
	Modification of Freshwater Inflows	Jacobs (2019, 2020)
Water Quality	Turbidity	This report
	Chlorophyll *a*	This report
Flora	Fringing Vegetation	Sinclair and Kohout (2018), Sinclair et al. (2020)
	Submerged Vegetation	Woodland et al. (2015), Woodland and Cook (2015)
Fish	Fish Assemblage Structure	Warry and Reich (2013); this report

The five sub-indices are scored from 1 (poorest condition) to 10 (best condition), and are made up of one or more measures. Measures are underpinned by one or more metrics that provide information on threats or condition. Metrics are combined in such a way that each measure within a sub-index contributes equally to the sub-index score (Table 2).

Table 2: Sub-indices, measures and metrics that make up the Index of Estuary Condition (IEC). The direction of the metric response to threats is included (see subsequent sections of this report for further detail and conceptual diagrams), and proportional contributions of metrics to sub-indices are provided to illustrate that metrics are weighted and combined so that each measure within a sub-index contributes equally to the sub-index score (e.g. the Fringing Vegetation and Submerged Vegetation measures contribute equally to the Flora sub-index despite Fringing Vegetation being underpinned by three metrics whereas Submerged Vegetation is underpinned by only one metric).

Sub-Index	Measure	Metric	Metric response to threat	Metric scoring range	Metric type	Proportional contribution to sub-index
Physical Form	Artificial Barriers	Percent of the natural estuary length affected by an artificial instream barrier	Increase	1 – 5	Threat	0.50
	Artificial Shorelines	Proportion of estuary perimeter bounded by built structures	Increase	1 – 5	Threat	0.50
Hydrology	Modification of Marine Exchange	Percent artificial openings (for intermittently open and closed estuaries (IOCE)) *or* Degree of entrance engineering works (for permanently open estuaries)	Increase	1 – 5 *or* 1, 3, 5	Threat	0.50
	Modification of Freshwater Inflows	Percent summer runoff intercepted	Increase	1 – 5	Threat	0.25
		Percent winter runoff intercepted	Increase	1 – 5	Threat	0.25
Water Quality	Turbidity	Turbidity	Increase	1 – 5	Condition	0.50
	Chlorophyll *a*	Chlorophyll *a* concentration	Increase	1 – 5	Condition	0.50
Flora	Fringing Vegetation	Percent of fringe covered by built structures	Increase	0 – 100	Condition	0.17
		Nativeness of fringing vegetation	Decrease	0 – 100	Condition	0.17
		Structural complexity of fringing vegetation	Decrease	0 – 100	Condition	0.17
	Submerged Vegetation	Ratio of macroalgae to total submerged vegetation	Increase	1 – 5	Condition	0.50
Fish	Fish Assemblage Structure	Species that can complete their life cycle within estuaries - richness	Decrease	1, 3, 5	Condition	0.14
		Introduced species – presence/absence	Present	1 – present 5 – absent	Condition	0.14
		Demersal species – richness	Decrease	1, 3, 5	Condition	0.14
		Demersal species – relative abundance	Very high or very low	1, 3, 5	Condition	0.14
		Trophic specialists – richness	Decrease	1, 3, 5	Condition	0.14
		Trophic specialists – relative abundance	Decrease	1, 3, 5	Condition	0.14
		Diadromous species – richness	Decrease	1, 3, 5	Condition	0.14

1.5.3 Reference conditions

Reference conditions are used to provide context for the values of threat and condition metrics. The reference condition is generally accepted to be the condition of an ecosystem prior to intensive development of the immediate and/or surrounding landscape (e.g. condition of estuaries before European settlement in the Australian context). The reference condition is used as a point of comparison for the current ecosystem condition. An ecosystem with characteristics that closely resemble those of its reference condition is assumed to be in better environmental condition than an ecosystem that is very different to its reference condition. The aim of using reference conditions in the IEC is to allow consistent comparisons of the environmental condition of different types of estuaries across the state.

The reference condition of an ecosystem can be determined from a variety of sources, including data from 'pristine' or minimally disturbed sites, historical data, modelling, expert opinion (Stoddard et al. 2006) or a combination of two or more of these. An alternative approach is to use sample data to derive a reference condition by examining the distribution of observed values. The 'best available' values of metrics are used to establish reference condition (Harris and Silveira 1999, Harrison and Whitfield 2006). This approach is useful when prior classification of reference condition is impeded by a lack of pristine or 'good' quality locations, historical records or appropriate data to support predictive models (Harrison and Whitfield 2004; Stoddard et al. 2006). There are few pristine estuaries in Victoria and limited historical records for several condition metrics (e.g. fish). In these cases, the 'best available' values of these metrics based on sampled data were used to define their reference condition.

The IEC uses different approaches to defining reference condition for different measures and their metrics (Table 3). The approaches used were determined by data availability and the strength of conceptual understanding of post-European impacts on estuaries.

Table 3: Approaches to defining reference conditions for measures comprising the IEC.

Sub-index	Measure	Approach to defining reference condition
Physical Form	Artificial Barriers	Undisturbed or unmodified state used as reference – assumed to be the absence of artificial instream structures (e.g. weirs; Pope et al. 2015).
	Artificial Shorelines	Undisturbed or unmodified shorelines used as reference – assumed to be the absence of built structures around the estuary fringe (e.g. rock walls, docks; Woodland and Cook 2015)
Hydrology	Modification of Marine Exchange	Undisturbed or unmodified marine exchange used as reference – assumed to be the absence of built structures, entrance maintenance activities (e.g. dredging, installation of training walls) or artificial opening for intermittently open and closed estuaries (Pope et al. 2015).
	Modification of Freshwater Inflows	Undisturbed or unmodified freshwater inflows used as reference – assumed to be the absence of anthropogenic water storages within an estuary catchment (Jacobs 2020).
Water Quality	Turbidity	Observed conditions in minimally modified estuaries used as reference – the State Environmental Protection Policy (Waters) (SEPP Waters) (2018) objectives for riverine estuaries are used. These were derived from empirical data collected from minimally modified estuaries.
	Chlorophyll *a*	Observed conditions in minimally modified estuaries used as reference – the State Environmental Protection Policy (Waters) (SEPP Waters) (2018) objectives for riverine estuaries are used. These were derived from empirical data collected from minimally modified estuaries.
Flora	Fringing Vegetation	Undisturbed or unmodified conditions used as reference – assumed to be the Ecological Vegetation Class (EVC) benchmarks (https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks) and the absence of built structures within the 'pre-1750 intertidal zone' (Sinclair and Kohout 2018)
	Submerged Vegetation	Observed conditions in minimally modified estuaries used as reference – assumed to be the dominance of seagrass over macroalgae as observed in estuaries with minimally modified catchments (Woodland et al. 2015, Cook et al. 2018).
Fish	Fish Assemblage Structure	'Best available' values from the observed dataset used as reference – assuming that the state-wide assessment captures a range of possible ecological conditions and that using a 'best available' approach to referencing can help avoid a degree of circularity by relying on observed values of metrics rather than assuming which estuaries are of high quality. Used because of the limited historical records of Victorian estuarine fish assemblages or appropriate data to support predictive models (Warry and Reich 2013).

1.6 IEC provides a snapshot not a trend

Trend analysis is concerned with measuring a change over time (a trend) and is often used to detect changes in condition that are outside the natural range of variation over time.

The design of environmental condition assessment programs to measure trends requires intensive data collection using comparable methods over long periods (typically 10 - 20 years). This ensures that, statistically, there are sufficient data to be able to reliably infer a directional (positive or negative) change in condition (or no change at all) and to confidently assess whether any directional change exceeds the expected natural variability. To reliably detect trends, monitoring may need to be designed in ways that are specific to a certain estuary or group of estuaries, potentially restricting the general applicability of indicators and/or monitoring protocols for state-wide assessment.

The IEC is not designed to be used to assess trends. There is an expectation that methods may evolve over time as new technologies (e.g. remote sensing, analyses of environmental DNA) develop; the IEC approach retains the flexibility to change methods during subsequent assessments. This is also the case for Victoria's other state-wide indices of condition: the Index of Stream Condition (ISC) and the Index of Wetland Condition (IWC). Here, the IEC is used to provide a 'snapshot' of condition across the state at the time of monitoring.

1.7 Purpose and structure of this report

The current IEC assessment provides the first benchmark of condition for Victorian estuaries. The results (DELWP 2021) of this state-wide condition 'snapshot' can be used to inform future monitoring and question-driven research or investigations of strategic benefit to the Victorian Waterway Management Program.

The purpose of this report is to document the methods of the first state-wide IEC assessment, by:

- Providing conceptual models illustrating the relationships between threats (pressures and stressors), ecosystem and biotic responses, and IEC measures and metrics
- Detailing the methods for each sub-index, including:
    - data collection
    - metric calculation
    - metric scoring
    - data quality assessment
    - data interpretation
- Describing the approach used to aggregate sub-indices to calculate an overall IEC score.

2. Physical Form

Estuaries are characterised by the movement of plants, animals, nutrients, sediment, and water both longitudinally (i.e. between freshwater, estuarine, and marine environments) and laterally (i.e. into fringing habitats). Anthropogenic changes to estuaries can compromise both longitudinal and lateral movements, which in turn can affect estuary condition.

The Physical Form sub-index of the IEC consists of two measures of the level of disruption to longitudinal and lateral connectivity: Artificial Barriers and Artificial Shorelines.

2.1 Artificial Barriers

The IEC includes a measure of how artificial instream barriers affect the longitudinal movement of biota between the estuary and upstream fluvial environment. Artificial barriers can threaten flora and fauna within estuaries by interrupting their migration or dispersal and/or reducing the extent of estuarine habitats (Figure 1). Common barriers are weirs, causeways and culverts as well as ‘sand slugs’ from human-induced upstream erosion events.

The metric for Artificial Barriers in the IEC is the percentage of the natural length of the estuary that has been affected by an artificial barrier. The nature of the artificial barrier (whether it is a partial or complete barrier to biota) is considered during the scoring of this metric (Pope et al. 2015).

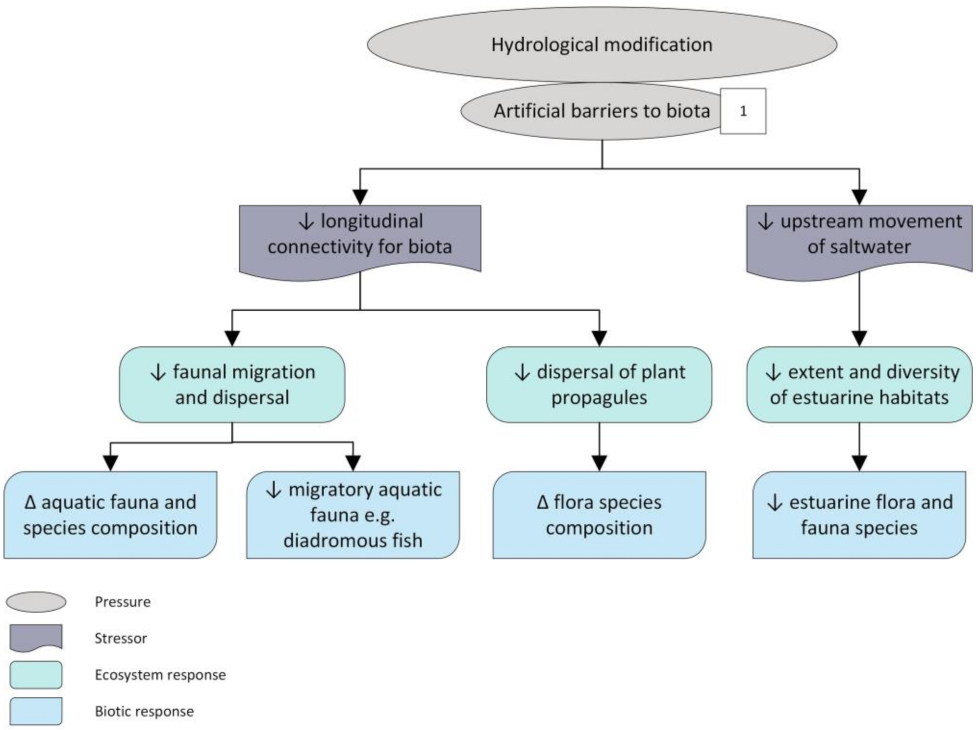

Figure 1: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which the threat of artificial instream barriers (pressure) can link to stressors and ecosystem and biotic responses. The numbered box indicates the pressure addressed by the Artificial Barrier measure in the IEC. Arrows within boxes indicate the direction of change; Δ indicates a possible change in either direction. Note: this diagram is simplified to show impacts of artificial barriers on movement of biota, but these barriers will also affect movement of other things like nutrients or food that will ultimately also affect biota.

2.1.1 Data collection and metric calculation – Artificial Barriers

Ninety-seven estuaries were assessed for the presence of Artificial Barriers. Assessments undertaken by Pope et al. (2015) during the IEC implementation trial were used to report on 64 estuaries. A further 27 estuaries were assessed using the same methods for Artificial Barriers in 2017-2019. Six estuaries within National Parks and wilderness areas were assumed to be free of artificial barriers, despite not being assessed in the field.

A combination of existing data sources and field data collection was used to identify the existence of artificial barriers and whether these structures represented partial or complete barriers (Pope et al. 2015). Partial barriers were those that were considered to allow limited or intermittent movement of biota. Complete barriers were those that represented an absolute and permanent barrier to biota. Where artificial barriers were identified, the most likely location of the natural (pre-1750) inland extent (length) of the estuary was derived from historical documents, field observations, elevation data and geomorphology to provide a 'natural estuary length'.

A subset of Victoria's estuaries originally functioned as freshwater systems. Examples include the small riverine tributaries to the Gippsland Lakes where downstream salinities increased following the construction of a permanent opening to the sea at Lakes Entrance in 1889. Some current estuaries are the product of drainage schemes and did not exist naturally (e.g. Elwood Canal and Patterson River). In these cases, there was no true natural inland extent of the estuary. Therefore, when calculating the Artificial Barriers measure, the endpoints of the 'natural' inland extents for these estuaries were estimated from the most likely locations had there been a marine influence downstream (Pope et al. 2015).

Naturally occurring instream barriers have also been removed from some estuaries (e.g. removal of the Yarra Falls from the Yarra River in 1883). This has increased the current extent of these estuaries compared to natural conditions (Pope et al. 2015). For the purposes of the Artificial Barriers measure, the 'natural' inland extent for these estuaries was estimated based on conditions following the removal of any naturally occurring barriers (Pope et al. 2015).

The quality of the data underpinning estimates of metrics for this measure in each estuary was classified into three categories according to the available evidence (Table 4).

Table 4: Criteria for the three classes of data quality for the Artificial Barriers measure for the IEC.

Data Quality	Location of artificial barriers	Location of natural inland extent of the estuary
High	Measured or estimated	Accurate documentation with clearly defined position of the saltwater or tidal limit, including natural barriers and/or Substantial natural rise in bed of waterway observed upstream of any identified barrier, or a mapped specific geographic feature that would limit upstream extent of the estuary
Moderate	Measured or estimated	Partial documentation with approximate location of the saltwater or tidal limit and/or Rise in bed of waterway observed but uncertainty around the inland extent of saltwater movement
Low	Derived or assumed	No or very limited documentation of location of the saltwater or tidal limit and/or General vicinity of likely inland extent of saltwater identified but details of the bedform unknown

The metric Percent Natural Estuary Length Affected by Artificial Barriers was calculated using the equation:

$$\textit{Percentage natural estuary length affected by artificial barriers} = \frac{\textit{Length above the instream barrier to the natural extent of estuary}}{\textit{Total length of the natural estuary}} \; X\,100$$

2.1.2 Reference conditions - Artificial Barriers

Undisturbed or unmodified conditions were used as reference conditions for assessing the Artificial Barriers measure for the IEC. These were assumed to be the absence of artificial instream barriers (Pope et al. 2015).

2.1.3 Metric scoring - Percent Estuary Length Affected by an Artificial Barriers

The metric Percent Natural Estuary Length Affected by Artificial Barriers was scored using a scale of 1 to 5 and applying the scoring thresholds recommended by Pope et al. (2015). Scoring reflects the distance of any artificial barrier downstream from the natural inland extent of the estuary and the degree to which the barrier restricts movement of biota (Table 5).

Table 5: Scoring thresholds for the metric Percent Natural Estuary Length Affected by an Artificial Barriers (Pope et al. 2015).

% of estuary length affected	Intermittent or selective interference with movement of biota	Completely blocked movement of biota
0	5	5
>0 - 5	5	4
>5 - 25	4	3
>25 - 50	3	2
>50	2	1

2.2 Artificial Shorelines

Artificial shorelines can alter the hydrodynamics of estuaries and limit their migration in response to sea-level rise or reduced inflows induced by climate change (Fujii 2012). They often impede or prevent the movement of plants, animals, sediment, nutrients and water between estuaries and their fringing habitats, affecting biogeochemical processes such as organic matter breakdown. Hardened or armoured shorelines may reduce habitat availability for intertidal seagrasses (Patrick et al. 2014) and many fish and invertebrate communities (Seitz et al. 2006, Bilkovic and Roggero 2008). Artificial or built structures along estuarine shorelines, particularly ones like pontoons and pilings, are often more effectively colonised by introduced species than their native counterparts, enhancing the spread and establishment of introduced species in estuaries (Glasby et al. 2006).

The metric for the Artificial Shorelines measure in the IEC is the percentage of an estuary's perimeter consisting of built structures. This requires defining the perimeter of an estuary and then measuring it consistently.

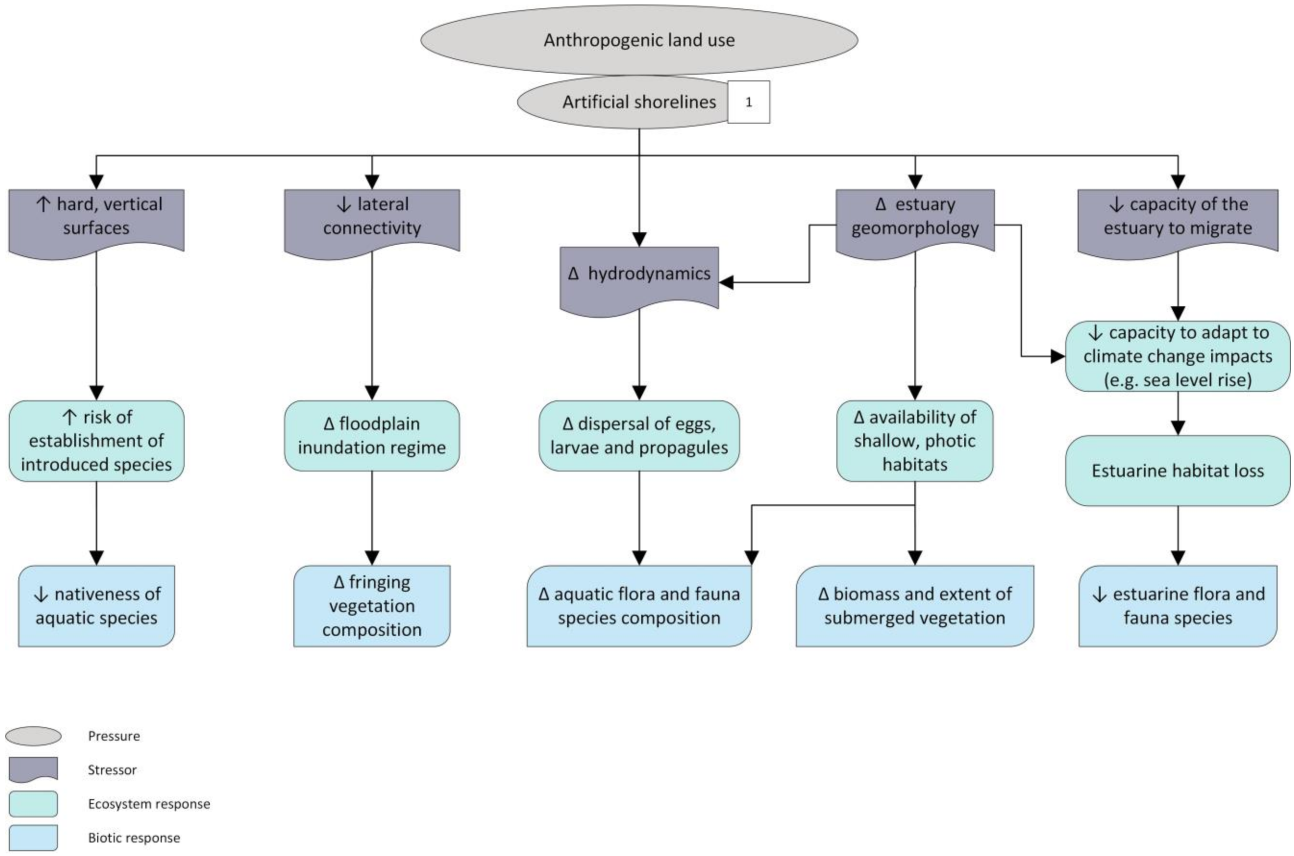

Figure 2: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which the threat of artificial shorelines (pressure) can link to stressors and ecosystem and biotic responses. The numbered box indicates the pressure addressed by the Artificial Shorelines measure in the IEC. Arrows within boxes indicate the direction of change; Δ indicates a possible change in either direction.

2.2.1 Defining the perimeter (shoreline) of an estuary

Defining the perimeter of an estuary is difficult because estuaries change size and shape with tidal movements and with flooding that occurs if the mouth is constrained or closed. The uncertainty caused by this variability affects how much of the estuary perimeter is considered to be artificial in many estuaries. For example, water in an estuary at low tide may not reach any built structures but at high tide, it may be bordered entirely within built earthen bund walls.

For the Artificial Shorelines measure, the perimeter of an estuary was considered to encompass estuary channels and lagoonal areas but not areas that may be inundated at high tide and covered with emergent vegetation including mangroves, saltmarshes or reedbeds.

In cases where prior mapping had defined the extent of open water, those polygons were used. Where prior polygons were not available, new ones were created based on either aerial or satellite imagery.

2.2.2 Quantifying the length of the estuary perimeter

Measuring the perimeter of complex shapes such as estuaries is problematic because the value depends on the resolution with which the shape is represented. For example, if line segments of 1 m are used to define the Australian coastline, the value for the perimeter would be much larger than if line segments of 1 km were used. For this reason, the length of the estuary perimeter is not used directly in the calculation of the Artificial Shorelines metric.

Instead, a narrow band (buffer) was created around the apparent perimeter of the estuary polygon to compensate for the effects of variable resolution. The area of this buffer was treated as a surrogate for the length of the perimeter. A 10 m buffer was used for this measure (Appendix A).

The area of built structures intersected by this buffer was used as a surrogate for the percentage of the perimeter bounded by built structures, using the following equation:

$$\begin{array}{l}\textit{Percentage of perimeter}\\ \textit{bounded by built structures}\end{array} = \frac{\textit{Total area of all built structures within buffer}}{\textit{Total area of buffer}} \; X\ 100$$

2.2.3 Definition of Built Structures

Two scenarios were encountered that were relevant to the definition of built structures:

Scenario 1: Within the footprint of the original estuarine floodplain, areas that met the definition of ‘built structures’ used in Sinclair and Kohout (2018) were included. This definition is: any artificial structure constructed from concrete, wood, brick or formed earth, or permanent open water in artificial impoundments (Figure 3a).

Scenario 2: Often, the original course of the estuary had been artificially modified so that it flowed through areas that were not part of the original estuarine floodplain (e.g. the channels on the Merri and Yarra Rivers; new estuaries created via channels and drains such as Lake Wellington Main Drain and the Patterson River). In these cases, the estuary perimeter is now artificial and so any area surrounding a new flow route or a channel that is not lined with estuarine vegetation was considered as a built structure (Figure 3b).

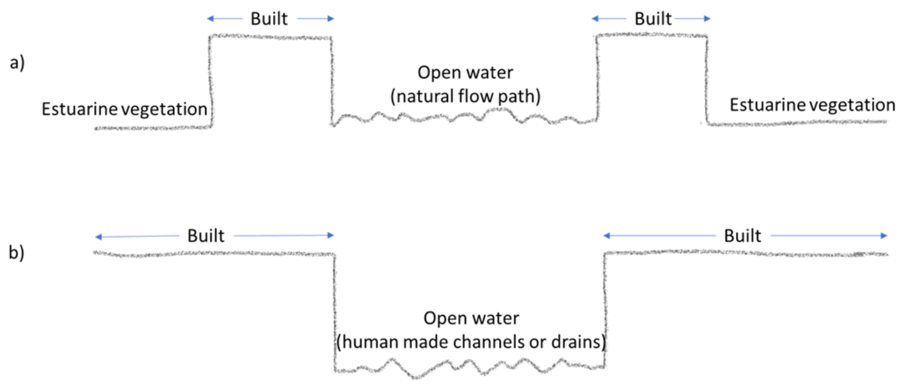

Figure 3: Diagram illustrating two scenarios relevant to the definition of built structures: a) built structures occurring within the footprint of the original estuarine floodplain, and b) a case where the course of the estuary has been modified and now flows through areas that were not part of the original estuarine floodplain.

2.2.4 Reference conditions – Artificial Shorelines

An undisturbed or unmodified state was used as the reference condition for assessing the Artificial Shorelines measure for the IEC. This condition was assumed to be the absence of built structures around the estuary perimeter (Pope et al. 2015, Woodland and Cook 2015).

2.2.5 Metric scoring - Artificial Shorelines

The Artificial Shorelines measure was scored using a scale from 1 to 5 and applying the scoring thresholds in Table 6.

Table 6: Scoring thresholds for the Artificial Shorelines measure for the IEC.

Percentage of perimeter bounded by artificial structures (%)	Score
0 – 0.1	5
>0.1 – 1	4
>1 – 10	3
>10 – 50	2
>50	1

2.2.6 Data quality – Artificial Shorelines

Data for the Artificial Shorelines measure for most estuaries were assigned a 'high' data quality rating because built structures could be confidently delineated during the assessments for the measure Fringing Vegetation (Section 6.1) from aerial imagery.

The data for three estuaries (i.e. Elwood Canal, Shipwreck Creek and Sealers Creek) were assigned a 'moderate' data quality rating because the values of the metrics for the Artificial Shorelines measure were assumed for these estuaries based on knowledge of the systems and brief inspection of aerial and satellite imagery. Shipwreck Creek and Sealers Creek lie within National Parks and were assessed as having no artificial shorelines. Elwood Canal is a completely constructed estuary that could not be scored for Fringing Vegetation (Section 5.1), and was assessed as having a 100% artificial shoreline.

2.3 IEC Physical Form sub-index score

The Physical Form sub-index score was calculated by combining the Artificial Barriers and Artificial Shorelines measures. The scores for each measure were added together and converted to a scale of 1 to 10 using the equation below. As the two measures are scored on a scale of 1 to 5, the resulting sub-index would span a scale of 2 to 10 if the measures had been merely added together.

$$\text{Physical Form score} = \frac{((\text{Artificial Barriers score} + \text{Artificial Shorelines score}) - 2) \times 9}{8} + 1$$

The Physical Form sub-index score ranges from 1 to 10. A value of 1 is assigned to estuaries under the most pressure from modified physical form (i.e. inferred poor condition) whereas a value of 10 is assigned to estuaries under the least pressure from modified physical form (i.e. inferred good condition).

2.4 Notes on interpreting Physical Form results

- The metrics within the Artificial Shorelines and Artificial Barriers measures of the IEC are proxies for the modification of the physical form in estuaries. Lateral and longitudinal connectivity in an estuary will also be influenced by other factors such as hydrology, bathymetry and geomorphology.
- The ecological effects of modified physical form will depend on the magnitude, type and duration of any changes as well as the sensitivity of biota and ecosystem processes (e.g. nutrient cycling, production) to these changes.

3. Hydrology

Estuaries are conduits through which nutrients, particulates and biota can move between freshwater catchments and the marine environment. Hydrological connectivity is therefore a key feature of estuaries, with freshwater from rivers mixing with saline water from the ocean. Disruptions to this connectivity, via changes to either freshwater or marine inputs, can alter the fundamental nature of an estuary.

The Hydrology sub-index of the IEC consists of two measures: Modification of Marine Exchange and Modification of Freshwater Inflows.

3.1 Modification of Marine Exchange

Marine exchange represents the nature of the connection between the estuary and the ocean, and is a function of the channel's cross-sectional area and its capacity to route flow seaward. This is influenced by the relative balance between onshore sediment transport and ebb-tidal currents at the mouth (McSweeney 2019). The cross-sectional area of an estuary mouth is often altered by human activities. These activities include artificially opening intermittently open and closed estuaries (IOCEs) and undertaking engineering works such as dredging and construction of training walls that modify the entrances of permanently open estuaries (Pope et al. 2015). In some cases, the entrances of estuaries that naturally operated as IOCEs have been engineered to remain permanently open (Boon et al. 2008). Changes in freshwater flows are also likely to alter the connectivity between estuaries and marine environments (Gillanders and Kingsford 2002). The modification of hydrological exchange between estuaries and the marine environment can influence estuarine water quality, physical processes, geomorphology and floodplain inundation regimes, with repercussions for ecological processes (e.g. organic matter breakdown, nutrient cycling) and biota (Becker et al. 2009, Conde et al. 2015, Swift et al. 2018, Whitfield et al. 2012) (Figure 4). The Modification of Marine Exchange with an estuary has been included as a measure of threat in the IEC. This measure uses different metrics for IOCEs and permanently open estuaries.

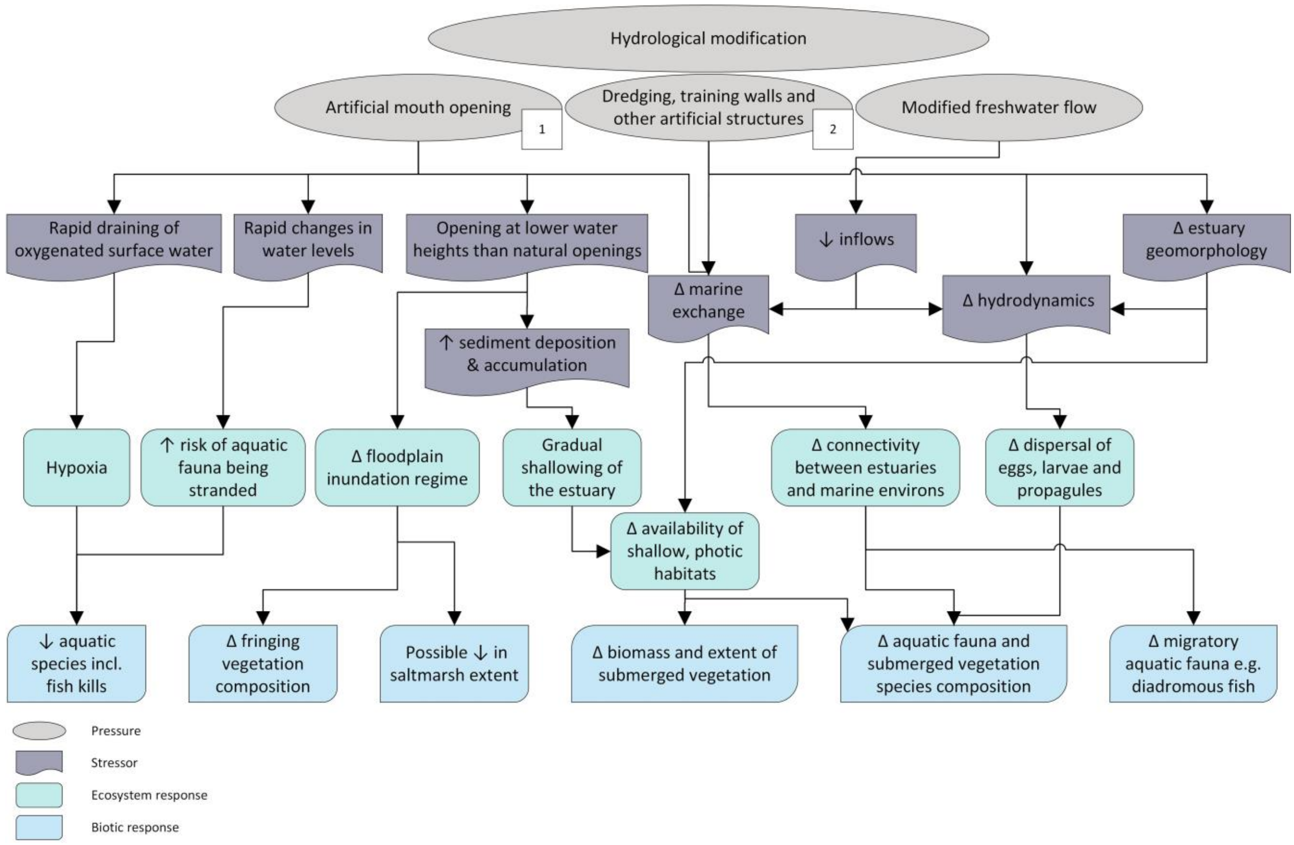

Figure 4: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which the threat (pressure) of modified marine exchange caused by artificial mouth opening, engineering activities (e.g. dredging, training walls, other artificial structures at estuary entrances) and modified freshwater flows can link to stressors and ecosystem and biotic responses. The numbered boxes indicate the pressures addressed by the metrics used in the Modification of Marine Exchange measure in the IEC: 1. Artificial mouth openings in intermittently open and closed estuaries (IOCEs), 2. Occurrence and intensity of dredging, training walls and other artificial structures (i.e. estuary entrance engineering works) at permanently open estuaries. Arrows within boxes indicate the direction of change; Δ indicates a possible change in either direction. Note that the parameter 'Modified freshwater flow' is included in this model for completeness but also has its own specific model below (Figure 5).

3.1.1 Modification of Marine Exchange in IOCEs – the Percent Artificial Openings metric

The text in this section is from McSweeney (2019) which details the rationale, data sources, methods and results for the Percent Artificial Openings metric of the Modification of Marine Exchange. Refer to McSweeney (2019) for additional information.

IOCEs are found in wave-dominated, micro-tidal settings on rivers with seasonally or inter-annually variable flow regimes (Cooper 2001). In Victoria, ~90% of open coastal estuaries intermittently close to the ocean (McSweeney et al. 2017). The entrance state (i.e. open or closed) is a function of onshore vs offshore energy and sediment transport acting within the entrance channel (Duong et al. 2016).

Entrance closure occurs during periods of low river flow where waves deliver sediment onshore to fill in the entrance channel. When closed, freshwater inflows are a more important influence on environmental conditions within the estuary than wave processes. IOCEs open naturally when the water level in the backing lagoon increases to overtop the berm and incise a channel to the ocean. This typically occurs in response to high river flow (Morris and Turner 2010) or when ocean waves repeatedly wash over the berm (Hart 2009).

Closed estuaries are artificially opened when low-lying land and infrastructure becomes inundated for long periods of time (Mondon et al. 2003, O'Toole et al. 2013). In Victoria, the Estuary Entrance Management Support System (EEMSS) was developed as a decision-support tool to assess the risks and benefits of artificially opening estuaries (Arundel 2006).

Fifty-six IOCEs were identified in Victoria by McSweeney (2019). However, three of these (i.e. Cumberland River, Wild Dog Creek and Skenes Creek) were considered too small for inclusion in the state-wide assessment of the IEC (Pope et al. 2015). Of Victoria's 56 IOCEs, the entrances of 30 (54%) are artificially opened.

3.1.2 Metric calculation - Percent Artificial Openings

Primary and secondary data sources for Percent Artificial Openings, and how these data were collected, are described in McSweeney (2019). Primary sources include high-quality data (see below and Table 8) that were reviewed by estuary entrance managers, and include observations of natural and artificial openings. Secondary sources include data which were used for supporting observations of entrance condition but do not contain records of the mode of opening (e.g. remotely sensed data, photo-point monitoring). Data from all available sources were collated for each IOCE that was known to be artificially opened. A time-series of estuary entrance condition was created for each estuary dating back to the earliest reliable record of entrance opening and continuing to 10 May 2019.

To quantify the degree of modification to the natural opening regime of IOCEs, the number of artificial openings was divided by the total number of known openings and converted to a percentage using the equation below. Openings that occurred but whose mode was unknown were excluded from the calculation.

$$Percent\ artificial\ openings = \frac{number\ of\ artificial\ openings}{number\ of\ known\ openings} \times 100$$

3.1.3 Reference conditions – Percent Artificial Openings

Undisturbed or unmodified conditions were used as the reference conditions. These were assumed to be the absence of artificial openings (Pope et al. 2015, McSweeney 2019).

3.1.4 Metric scoring - Percent Artificial Openings

For IEC, a new scoring method (Table 7) was adopted for the Percent Artificial Openings metric and applied to all IOCEs. This method expands upon the approach developed by Pope et al. (2015), further discriminating among estuaries whose percentage of artificial relative to natural openings exceeded 50%. This increased discrimination was warranted because there is increased potential for more highly modified artificial opening regimes (i.e. >75% modification to the opening regime) to impact on estuarine ecological processes and biota and to drive shifts in mouth morphology (Stephens and Murtagh 2012), above the potential at the lower threshold of 50% modification of the opening regime. The revised scoring system is also more sensitive for detecting future shifts in the number of artificial openings as more data are collected. All IOCEs that were not artificially opened received a score of 5.

Table 7: Scoring thresholds for the Percent Artificial Openings metric applied to IOCEs.

Score	% artificial openings	Description
5	0	Completely unmodified condition; no alteration to natural opening regime
4	1 - 50	Minor modification to opening regime – most openings natural
3	51 - 74	Moderate modification to opening regime
2	75 - 99	High modification to opening regime
1	100	Complete modification to entrance opening regime
NA	unknown	Insufficient data to calculate the Percent artificial openings metric

3.1.5 Data quality – Percent Artificial Openings

For quality assurance, a data quality score was calculated for each estuary. This scoring system differs from that used in the previous IEC implementation trial (Pope et al. 2015) and reflects the increased diversity of accessed data over the past half of a decade.

Data quality was assigned using a weighted multi-criteria analysis. For each artificially opened IOCE, four criteria were applied:

C1: Certainty of opening mode: for how many openings is the mode of opening certain?

C2: Sources of data: how reliable are the sources (i.e. primary vs secondary sources) of data?

C3: Length of record: how long is the record of entrance condition observations (in years)?

C4: Interval between observations: how often is the entrance condition recorded?

The scoring system and data quality assignment approach are shown in Table 8. Criteria 1 and 2 (both scored out of 10) were weighted higher than Criteria 3 and 4 (both scored out of 5). This differential weighting was chosen because the certainty of the mode of opening and the data quality both determine the ability of the metric to accurately represent entrance modification. Criteria 3 and 4 relate to the quality of the record, and this governs the representativeness of each estuary's dataset in the long term. Criterion 4 also indicates the certainty that all openings are captured within the dataset which was relevant for estuaries where primary sources of data were limited.

The scores from each criterion were added together, with a total of 30 points possible. A final data quality score, expressed as percentage, was calculated using the following equation:

$$Data\ quality\ (final\ percentage) = \left(\frac{score\ C1 + score\ C2 + score\ C3 + score\ C4}{30}\right) x\ 100$$

Table 8: System for weighting the four criteria used to assess the quality of the data record; RS – remotely sensed data; EW – EstuaryWatch data; EGCMA – East Gippsland Catchment Management Authority records; EEMSS: Estuary Entrance Management Support System.

Points	Criterion 1 - Certainty of mode	Points	Criterion 2 - Sources of data
10	100% of opening modes known	10	EEMSS/EGCMA + EW + secondary sources
8	99 - 75% of opening modes known	8	EW + secondary sources
6	74 - 50% of opening modes known	6	Literature + secondary sources + RS data
4	25 - 49% of opening modes known	4	At least one secondary source + RS data
2	1 - 24% of opening modes known	2	RS data only
0	0% of opening modes known	0	No data
Points	**Criterion 3 - Length of record (years)**	**Points**	**Criterion 4 - Interval between observations (days)**
5	>10	5	<5
4	>8 - 10	4	5 - 9
3	>5 - 8	3	10 - 14
2	>2 - 5	2	15 - 20
1	0 - 2	1	>20
0	No record	0	No data

The percentage scores of data quality (as per the equation above) were assigned to categories of data quality according to the thresholds in Table 9.

Table 9: Data quality categories assigned to the percentage scores of data quality derived from the combination of the four weighted criteria.

Data quality (%)	Data quality category
100	Perfect data quality - or all natural openings
75 - 99	Very high data quality
51 - 74	High data quality
50 - 25	Moderate data quality
24 - 1	Poor data quality
0	Data unsuitable to support metric calculation

3.1.6 Notes on interpreting results – Percent Artificial Openings

- In the current assessment of estuarine condition, the Percent Artificial Openings metric applies equally to authorised and illegal artificial opening of closed IOCEs. However, illegal openings are likely to present a greater risk to estuarine values than authorised openings which are preceded by suitable risk assessment.
- The Percent Artificial Opening metric is a proxy for the Modification of Marine Exchange in IOCEs. Actual measurement of marine exchange modification would require currently unavailable data such as pre-European opening/closing regimes for artificially opened IOCEs.
- Reductions in freshwater inflows caused by anthropogenic activities (e.g. water interception and extraction within the catchment) can decrease the frequency of natural estuary openings. Information on freshwater inflows is not incorporated in the current measure of Modification of Marine Exchange in the IEC. However, anthropogenic modification to freshwater inflows is addressed in the measure Modification of Freshwater Inflows (Section 3.2).

- The metric does not capture artificial entrance closures undertaken by estuary managers. This occurred at only one Victorian IOCE (Merricks Creek).

3.1.6 Modification of Marine Exchange in permanently open estuaries – the Entrance Engineering Works metric

Engineering structures and processes intended to modify the morphology of estuary entrances underpin the metric used to assess the Modification of Marine Exchange in permanently open estuaries (Pope et al. 2015). These structures and processes are typically aimed at increasing the cross-sectional area of estuary entrances to facilitate boat passage, and include dredging and the construction of training walls (Pope et al. 2015).

Assessments of engineering structures by Pope et al. (2015) during the IEC implementation trial were used for state-wide IEC reporting in 2021 (DELWP 2021). This metric was calculated for 53 permanently open estuaries.

3.1.7 Metric calculation – Entrance Engineering Works

The calculation of Entrance Engineering Works is based on the occurrence and degree of constructed structures (i.e. training walls) and dredging at estuary entrances. Data on dredging, training walls, or both, at the entrance of each estuary were collated from direct field observations and information gathered from Port Authority documents, satellite imagery, Vicmap hydrologic structures and elevation morphology layers, and interviews with waterway managers (Pope et al. 2015).

Of the estuaries that enter major embayments and estuarine lake systems (i.e. Port Phillip Bay, Westernport Bay, Corner Inlet/Nooramunga and the Gippsland Lakes), only the Gippsland Lakes have had a major increase in marine exchange due to engineering works at the entrance. However, Port Phillip Bay and Corner Inlet have had alterations to their entrances (Pope et al. 2015).

3.1.8 Reference conditions – Entrance Engineering Works

Undisturbed or unmodified conditions were used as the reference conditions for Entrance Engineering Works. This absence of entrance engineering works is assumed to represent the degree of marine exchange that has not been modified by humans. However, the metric does not explicitly include changes to entrance morphology caused by fluvial processes (e.g. increased sedimentation or a decrease in flows large enough to affect the depth or width of entrances (Pope et al. 2015)).

3.1.9 Metric scoring – Entrance Engineering Works

The metric used to estimate Modification of Marine Exchange in permanently open estuaries was scored on a three-point scale (1, 3, 5) developed by Arundel et al. (2009) and applied in the IEC implementation trial by Pope et al. (2015). The scoring criteria are described in Table 10.

Numerous Victorian estuaries enter coastal embayments or estuarine lake systems. Metric scoring accounts for whether there have been major modifications to the marine exchange of these embayments or lake ecosystems.

Table 10: Scoring criteria for the Entrance Engineering Works metric applied to permanently open estuaries.

Score	Criteria
5	Essentially natural marine exchange: • no dredging of entrance and no training walls or other built structures present, *and* • entrance not artificially constructed, *and* • no major modification to marine exchange of the embayment or estuarine lake system into which the estuary enters
3	Some modification to marine exchange: • no dredging of entrance, *but* minor engineered structures present at entrance, *or* • artificially constructed entrance, *or* • major modification to marine exchange of the embayment or estuarine lake system into which the estuary enters
1	Considerable modification to marine exchange: • entrance dredged, *or* • training walls present

3.1.10 Data quality – Entrance Engineering Works

The data used to derive scores for the Entrance Engineering Works metric were assigned to a category of low, medium or high data quality (Pope et al. 2015). These categories represent the level of knowledge about the presence or occurrence of estuary entrance engineering works and their influence on hydrological exchange with the marine environment (Table 11).

Table 11: Criteria used to categorise the quality of data underpinning the Entrance Engineering Works metric applied to permanently open estuaries.

Data quality	Criteria
High	Presence of structures and occurrence of dredging documented and effective in modifying marine exchange. The absence of minor structures and dredging documented
Medium	Presence of structures derived from maps
Low	Lacking documentation of possible dredging. Identified structures known to be ineffective in modifying marine exchange
Unknown	Unable to establish whether data exist to support derivation of the metric

3.1.11 Notes on interpreting results – Entrance Engineering Works

- The Entrance Engineering Works metric is a proxy for the Modification of Marine Exchange in permanently open estuaries. The influence of dredging and built structures on marine exchange may vary among estuaries.
- The ecological effects of increased marine exchange in permanently open estuaries will depend on the severity, frequency and duration of any hydrological changes as well as the sensitivity of ecological processes (e.g. organic matter decomposition) and biota to these changes.
- Reductions in freshwater inflows to estuaries from anthropogenic activities such as water interception and extraction within the catchment can influence the geomorphology of estuary entrances. This information is not incorporated into the Entrance Engineering Works metric, which only addresses modifications to estuary entrances. However, anthropogenic modification to freshwater inflows is addressed in the measure Modification of Freshwater Inflows (Section 3.2).

3.2 Modification of Freshwater Inflows

The text in this section is drawn from Jacobs (2019) and Jacobs (2020). These reports detail the rationale, data sources, methods and results for the measure Modification of Freshwater Inflows and its metrics in the IEC. Refer to these reports for additional information.

Freshwater inflow regimes strongly influence estuarine structure and function via multiple pathways (Figure 5). The volumes, frequency and timing of freshwater inflows affect a range of physical and biological processes within estuaries (Pierson et al. 2002). Modification of freshwater inflows may alter the nature and magnitude of delivery of sediments and nutrients to estuaries, with repercussions for estuarine productivity (Woodland et al. 2015). In IOCEs, freshwater inflows often govern when the entrance opens and can affect connectivity between estuaries and the marine environment (McSweeney 2019), with impacts for aquatic biota.

Many estuarine organisms rely on freshwater inflows for survival. Freshwater inputs may cue significant life-history stages, such as reproduction or migration (Hancock and Bunn 1999, Montgomery et al. 2000). Consequently, modification of the timing or seasonality of these inflows (particularly “freshes”) can interrupt spawning and migration, and ultimately recruitment, of aquatic species such as diadromous fish that use estuaries. Reductions in freshwater inflows reduce mixing between freshwater and saltwater layers within estuaries (CSIRO et al. 2009). A lack of mixing between these layers can cause the denser saline layer to become hypoxic (and possibly anoxic) which may kill estuary organisms (Barton and Sherwood 2004) or interrupt their life cycles (e.g. spawning, Nicholson et al. 2008).

Any changes to the natural flow regime that result in increases in the durations of low- or cease-to-flow periods, reductions in the frequency and duration of freshes, high flows and floods, and changes to the timing of flow components beyond the range of natural variability are likely to represent a threat to estuary structure and function. The magnitudes of these impacts will vary from estuary to estuary depending on characteristics such as estuary shape, size and entrance type.

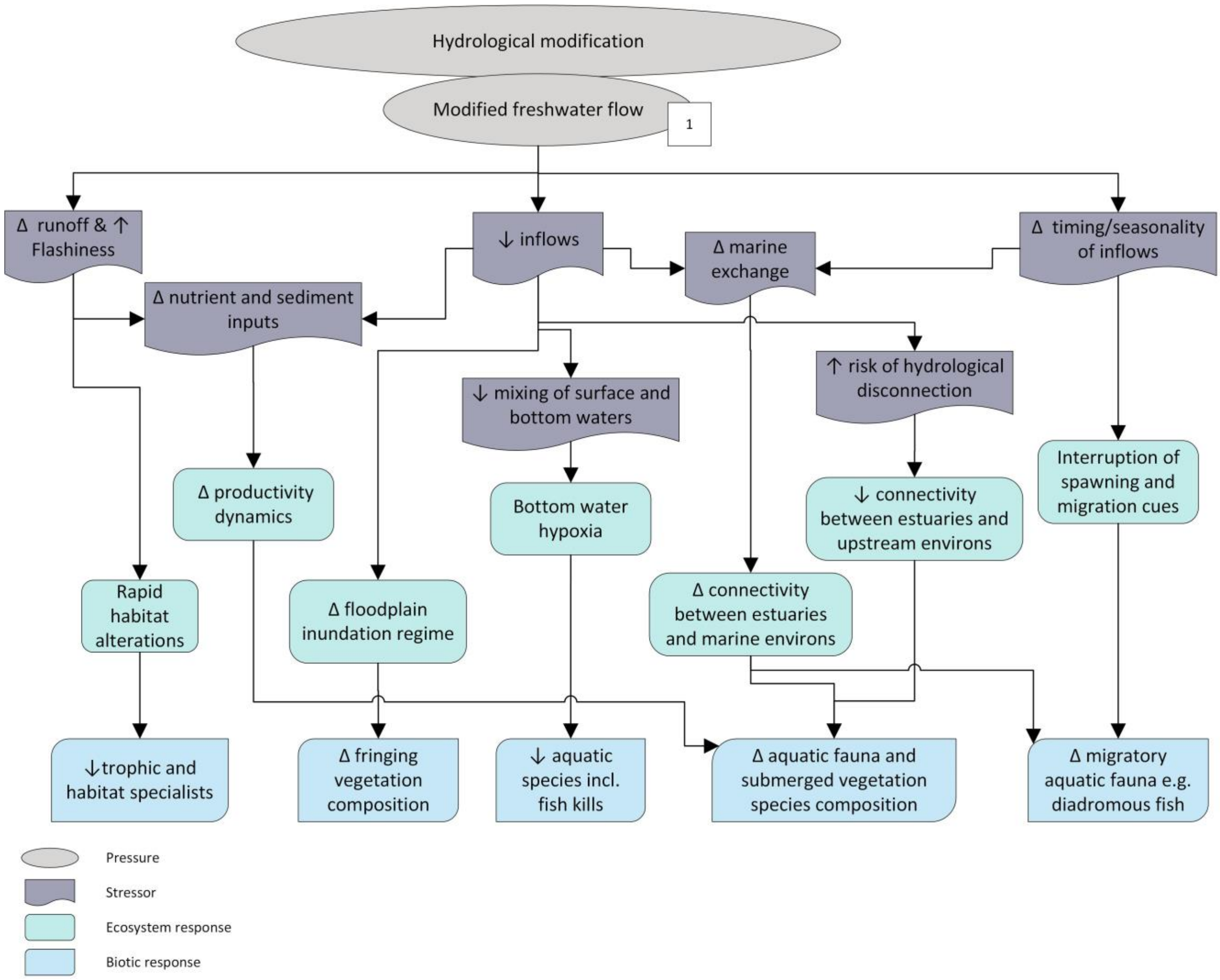

Figure 5: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which the threat (pressure) of modified freshwater inflows can link to stressors and ecosystem and biotic responses. The numbered box indicates the pressure addressed by the Modification of Freshwater Inflows measure in the IEC. Arrows within boxes indicate the direction of change; Δ indicates a possible change in either direction.

3.2.1 Modification of Freshwater Inflows – approaches for Victorian estuaries

The Modification of Freshwater Inflows measure developed in the IEC implementation trial was based on the Index of Stream Condition (ISC) hydrological metric (for estuaries whose inflowing rivers had been assessed using the ISC – 59 estuaries) or an assessment of the volume of farm dams per km^2 of upstream catchment (for the mostly small estuaries whose inflowing rivers had no ISC assessment – 42 estuaries) (Arundel et al. 2009). On reviewing the IEC implementation trial, Woodland and Cook (2015) endorsed this approach. However, they recommended exploring alternative approaches that were better tailored to reflect hydrological stresses between rivers and their estuaries.

An investigation of alternative approaches led to the development of a revised metric that characterises modification to freshwater inflows based on an estimate of water usage (storage) relative to its estimated availability (runoff) in a catchment (Jacobs 2020). This metric is underpinned by data that are available across the state, and can therefore be applied consistently to all Victorian estuaries. There is also a good correlation between scores from this revised metric and those based on the ISC metric (where they were able to be calculated).

3.2.2 Metric calculation - Modification of Freshwater Inflows

The Modification of Freshwater Inflows measure is based on the volume of water that is stored (a proxy for interception or usage) within the estuary's catchment relative to the total volume of available runoff (a proxy for water availability). The proxy metric Total Catchment Storage Volume was considered to be capable of distinguishing coarse differences in freshwater inflow modifications among Victorian estuaries, and is estimated using the volume of water storages within an estuary catchment. The proxy metric Water Availability is estimated using modelled seasonal runoff across the catchment.

Two variants of the metric using (i) summer (December to May) and (ii) winter (June – November) runoff were calculated separately and then integrated for reporting the Modification of Freshwater Inflows measure. These two variants were used because differences in seasonal runoff potentially impact on estuaries in different ways. For example, reduced inflows in winter can reduce the frequency of high-flow events capable of opening closed IOCEs whereas reduced inflows in summer can increase the risk of eutrophication or lead to loss of hydrological connectivity.

The equations used to determine the two variants are:

$$Percent\ runoff\ intercepted\ (\text{summer}) = \frac{total\ catchment\ storage\ volume\ (ML)}{summer\ (Dec - May)\ runoff(ML)} \times 100\%$$

$$Percent\ runoff\ intercepted\ (\text{winter}) = \frac{total\ catchment\ storage\ volume\ (ML)}{winter\ (Jun - Nov)\ runoff(ML)} \times 100\%$$

Total Catchment Storage Volume

For each estuary's catchment, the metric Total Catchment Storage Volume was obtained from the Farm Dam Boundaries Spatial Layer publicly available on the DataVic website (https://www.data.vic.gov.au/). All storages were included except for wastewater lagoons and aquaculture and industrial storages which were excluded because they can store water originating from outside the catchment of interest and were also rare. Total catchment storage was used as the numerator (rather than just farm dam volume) because it usually better represented threats in catchments with significant irrigation or urban water use. Estimated volumes of storages in the NSW portion of the Snowy River's catchment were added to the Victorian volumes to better represent total storage volume in this catchment.

Water Availability

Ideally, the metric Water Availability would be estimated using data collected from flow gauges upstream of each estuary. However, many streams flowing into estuaries are not gauged or do not have modelled inflow, so measured stream flow volumes were not consistently available across the state (for example, only 58% of estuaries considered for the Modification of Freshwater Inflows measure had Flow Stress Ranking metrics calculated (SKM 2011)). Furthermore, there were no reliable model estimates of end-of-catchment stream flows available for all the estuaries of interest. Consequently, the Bureau of Meteorology's Australian Water Resource Assessment Landscape model (AWRA-L model) was used to derive runoff estimates for each estuary. Despite some limitations (acknowledged below), this model provided runoff estimates for all IEC estuaries in a consistent way.

AWRA-L is a daily grid-based (5 km) water balance model that is conceptualised as a small un-impacted catchment. Runoff is simulated through the landscape from rainfall entering each grid cell and then passing through vegetation and soil before leaving via evapotranspiration, runoff or deep drainage to groundwater (Frost et al. 2018). Gridded runoff data are publicly available from http://www.bom.gov.au/water/landscape/ or upon request for the period 1990 to present. To calculate the Water availability metric, winter (June-November) and summer (December – May) runoff were used for the 2010 - 2019 period, inclusive.

Overall, there were good correlations between observed flows and the runoff predicted by AWRA-L for most catchments (Jacobs 2019). In all cases, AWRA-L outputs also closely matched the observed seasonal and annual patterns of variability. The AWRA-L model tended to overestimate runoff in some instances, attributed to river regulation and land-use impacts on flow. Hence, the metric Water Availability is based on an estimate of the un-impacted runoff rather than the actual observed flow. Uncertainty in the model outputs is reflected in the data quality ratings for this metric (Section 3.2.5).

3.2.3 Reference condition - Modification of Freshwater Inflows

Undisturbed or unmodified conditions were used as the reference condition. This was assumed to be the absence of water storages within an estuary catchment (Pope et al. 2015; Jacobs 2020).

3.2.4 Metric scoring - Modification of Freshwater Inflows

The Modification of Freshwater Inflows measure was scored on a scale of 1 to 5, applying the scoring thresholds in Table 12. Estuaries without modification (i.e. no catchment storage) received a score of 5. Thresholds for the remaining categories of score were informed by the distribution of observed values.

Table 12: Scoring thresholds for the Modification of Freshwater Inflows measure for the IEC. Both the summer runoff and winter runoff variants are scored using these thresholds.

Score	Percent runoff intercepted (%)	Description
5	0 - 0.0001	No modification (<0.0001% of runoff intercepted by catchment storages)
4	>0.0001 - 5	Minor modification (>0.0001 - 5% of runoff intercepted by catchment storages)
3	>5 - 20	Moderate modification (>5 - 20% of runoff intercepted by catchment storages)
2	>20 - 70	High modification (>20 - 70% of runoff intercepted by catchment storages)
1	>70	Very high modification (> 70% of runoff intercepted by catchment storages)

The scores for the two variants of the metric were added together and converted to a 1 – 10 scale using the formula below. This formula was used because simply adding the two variants (each scored 1 – 5) would have resulted in the scores for the Modification of Freshwater Inflows measure ranging from 2 – 10.

$$\textit{Modification of Freshwater Inflows score} = \frac{((\textit{Percent streamflow intercepted (summer) score} + ((\textit{Percent streamflow intercepted (winter) score}) - 2) \times 9}{8} + 1$$

3.2.5 Data quality - Modification of Freshwater Inflows

The quality of Modification of Freshwater Inflows data for each estuary was assigned to one of three categories based on an assessment of sources of variability and uncertainty in AWRA-L runoff estimates, and in catchment storage volume estimates based on the extent of waterway regulation.

Confidence in estimates of catchment storage volume is highest for catchments with high levels of regulation and where usage is associated with large storages (e.g. irrigation areas and large water supply storages) such as the Latrobe, Snowy and Glenelg catchments. This is because there is likely to be a more consistent diversion from these storages from year to year, even in wet years. In comparison, data quality is lowest in unregulated catchments where farm dams make up most of the total storage, farm dam volume is only a surrogate for usage, and actual usage depends on multiple factors such as climate conditions and land use that are not explicitly captured in calculation of the measure.

There is also a high confidence in scores for forested catchments with low storage volumes regardless of catchment area. This is because even if there is uncertainty in runoff estimates from small catchments, the low storage volume relative to runoff means that it is possible to confidently assign high scores for Modification of Freshwater Inflows. Examples include the large forested catchments of East Gippsland and the small forested catchments of the Otway Coast.

Data quality is low for estuaries where most of the catchment is urbanised (e.g. Elwood Canal, Laverton Creek, Skeleton Creek, Patterson River). This is because AWRA-L runoff estimates do not account for stormwater runoff, which elevates flows in urban catchments. For larger urban catchments, such as the Yarra and Werribee, confidence in AWRA-L estimates is higher because there is a relatively small proportion of the catchment that is influenced by urban runoff compared to the whole catchment area.

A summary of data quality categories associated with different catchment types and contexts is provided in Table 13.

Table 13: Data quality categories for the Modification of Freshwater Inflows measure.

Data quality category	Catchment types
High	▪ Forested catchments with no regulation and where catchment storage is very low or absent, irrespective of catchment area. ▪ Large (>500 km^2) rural and urban catchments with high levels of regulation (i.e. diversions for irrigation or water supply) and large storages relative to farm-dam volumes.
Medium	▪ Medium-sized rural catchments (150 - 500 km^2) with low levels of regulation and where water usage is mainly associated with small farm dams and private diversion (i.e. no large diversions for extensive irrigation or water supply). Although farm dams are included in the measure of usage, the actual usage from farm dams can vary substantially. The measure does not capture private diversions.
Low	▪ Small rural catchments (<150 km^2) where usage is predominantly associated with small farm dams and private diversions. ▪ Catchments that are mainly urbanised.

3.2.6 Notes on interpreting results - Modification of Freshwater Inflows

- The Total Catchment Storage Volume used as a proxy for water usage or interception were considered adequate to distinguish coarse differences for Modification of Freshwater Inflows among Victorian estuaries. However, it should be noted that Total Catchment Storage Volume used in the calculation of the Modification of Freshwater Inflows measure does not account for all sources of water extraction within catchments. Extraction sources that are not accounted for include, for example, direct pumping for irrigation, stock and domestic as well as groundwater extraction which can also influence streamflow.
- Uncertainties are associated with AWRA-L's runoff estimates, as this model is calibrated on a national scale and does not route flow through the catchment. The model tends to perform well in coastal Victorian catchments but it performs poorly for catchments along the Great Dividing Range and ephemeral systems (Frost and Wright 2018). It also does not consider impacts associated with changed runoff characteristics in urban areas. Furthermore, AWRA-L's runoff estimates tend to be more unreliable for smaller catchments. This is because the grid-square for the runoff estimate is 25 km^2 (5 km x 5 km), so smaller catchments (<150 km^2) have fewer runoff estimates per catchment than larger ones which increases uncertainty in the total runoff estimates for smaller catchments.
- The measure does not incorporate metrics of water recovery or the delivery of environmental water. Some coastal rivers have environmental water entitlements which are managed in ways that include consideration of estuarine outcomes (e.g. Werribee River).

3.3 IEC Hydrology sub-index score

The Hydrology sub-index score for the IEC was calculated by combining the scores of the Modification of Marine Exchange and Modification of Freshwater Inflows measures. The score for the Modification of Marine Exchange measure (scored from 1 to 5) was multiplied by 2 prior to being added to the score for the Modification of Freshwater Inflows measure (scored from 1 to 10). The resulting number (with a possible range of 3 – 20) was then converted to a 1 – 10 scale using the following equation.

$$Hydrology\ score = \frac{\big((Modification\ of\ Marine\ Exchange\ score\ x\ 2) + (Modification\ of\ Freshwater\ Inflows\ score) - 3\big) \times 9}{17} + 1$$

Scores for the Hydrology sub-index ranged from 1 to 10. A score of 1 was assigned to estuaries under the most pressure from modified hydrology (i.e. inferred poor condition) whereas a score of 10 was assigned to estuaries under the least pressure from modified hydrology (i.e. inferred good condition).

4. Water Quality

Water quality influences most ecological and biogeochemical processes that support estuarine ecosystem structure and function. Many estuarine organisms, such as submerged plants and fish, are also directly affected by water quality. Consequently, many programs monitoring estuarine condition include measurements of water quality (Lacouture et al. 2006, Roper et al. 2011).

The Water Quality sub-index of the IEC consists of two measures: Turbidity and Chlorophyll *a*.

Anthropogenic land use usually increases inputs of toxicants, sediments and nutrients to estuaries, and these inputs can be influenced by modified hydrology (Figure 6). Elevated nutrient inputs to estuaries can disrupt ecological processes such as primary production, nutrient cycling, energy transfers across trophic levels, and dynamics of consumer species (Valiela et al. 1992; Nixon and Buckley 2002). Pelagic primary producers (e.g. phytoplankton) and benthic plants capable of rapid nutrient uptake (e.g. macroalgae) respond quickly to nutrient enrichment. Chlorophyll *a* is the critical pigment used during photosynthesis by phytoplankton. It is a common proxy metric for representing primary production in estuaries and other water bodies, often assessed in monitoring programs as a measure of condition representing anthropogenic eutrophication (Lacouture et al. 2006, Roper et al. 2011).

Water clarity is another important characteristic of the water column in estuaries because the depth of light penetration is a critical limiting factor in the type and extent of benthic vegetation. Elevated sediment inputs to estuaries decrease water clarity and increase turbidity, with repercussions for primary and secondary production (Figure 6). Turbidity is readily measured, provides a reasonable proxy for assessing the availability of light at depth, and can be considered an aspect of the physical condition of estuary waters (Woodland and Cook 2015).

The availability of dissolved oxygen often determines the suitability of an area for pelagic, demersal and benthic organisms as well as influencing the identity and rate of biogeochemical processes (Woodland and Cook 2015). Hypoxia of estuary waters (particularly bottom waters) can stem from a lack of mixing between surface and bottom waters during periods of low inflows and through eutrophication. Hypoxia can lead to mortality of species inhabiting the estuary (Barton and Sherwood 2004) or interrupt crucial life cycle stages such as spawning (Nicholson et al. 2008; Figure 6).

Dissolved oxygen is lost from estuarine waters through respiration by microbes, plants and animals, and during physical processes such as passive atmospheric exchange and tidal flux. It is replenished by other physical processes (e.g. wind-driven mixing, passive atmospheric exchange, fluvial water inputs) and during photosynthesis by aquatic plants. Consequently, concentrations are highly variable over tidal, diurnal and seasonal time scales (Pope et al. 2015; Woodland and Cook 2015). Furthermore, extremes in dissolved oxygen conditions (e.g. periods of anoxia) are often event-based phenomena and can be difficult to capture without continuous measurements (Pope et al. 2015). Thus, even though dissolved oxygen affects estuarine condition, the huge inherent spatial and temporal variability of this parameter hinders its utility for state-wide programs assessing condition, and dissolved oxygen was not included as a metric in the IEC for reporting in 2021. Dissolved oxygen is also not included in the New South Wales Estuary Monitoring, Evaluation and Reporting program (Roper et al. 2011).

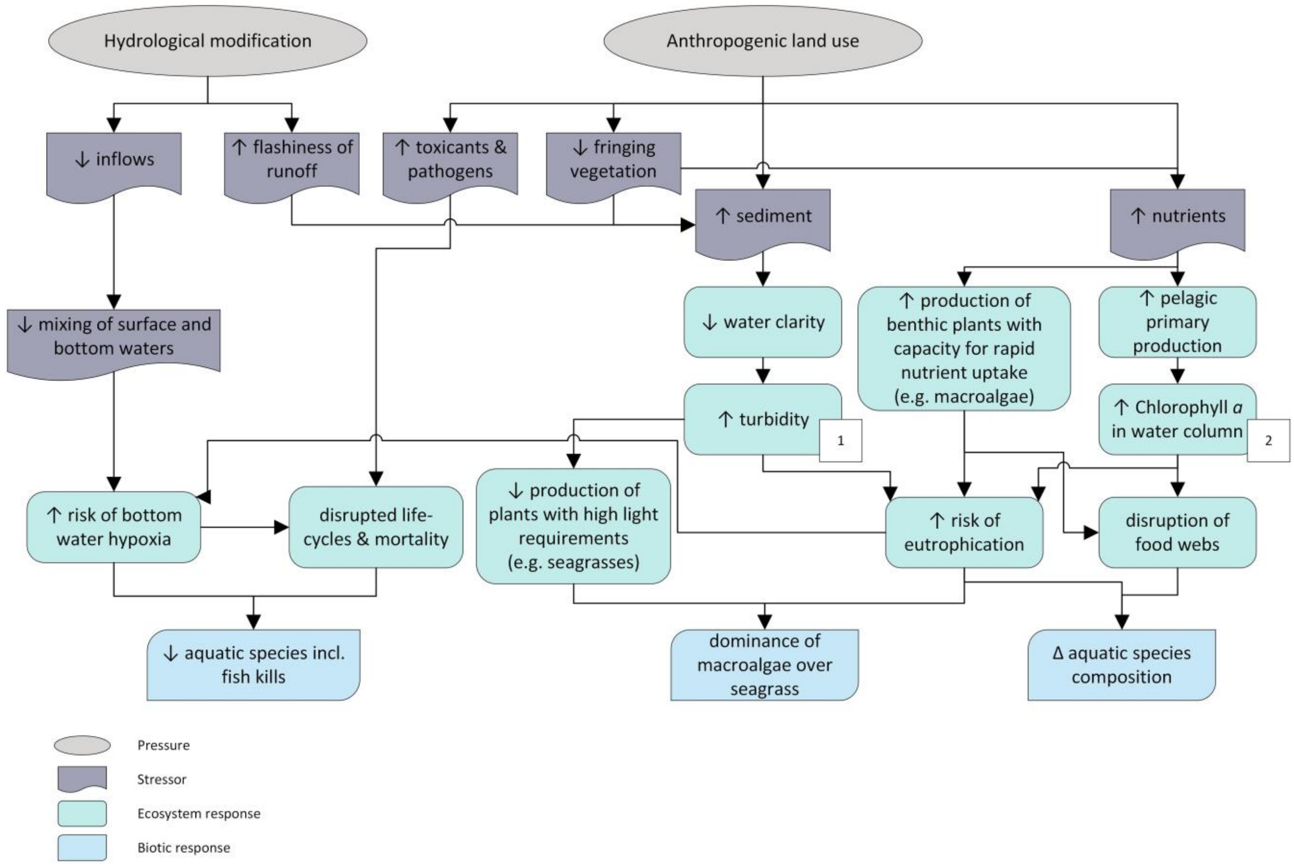

Figure 6: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which pressures and stressors can link to water quality and ecosystem and biotic responses. The numbered boxes indicate the water quality responses that are addressed by the two measures in the Water Quality sub-index of the IEC: 1. Turbidity; 2. Chlorophyll *a* concentrations in the water column. Arrows within boxes indicate the direction of change; Δ indicates a possible change in either direction.

4.1 Water Quality data collection

Water quality data were collected from 92 estuaries for the IEC assessment. Five of these estuaries were sampled by Monash University researchers as part of a complementary project in 2013 - 2014. The remaining 87 estuaries were sampled during 2017 - 2019 by professional scientists, Catchment Management Authority personnel and EstuaryWatch citizen scientists.

Sampling was undertaken in spring, summer and autumn as this timing corresponds to: 1) critical growth periods for submerged vegetation, 2) periods of elevated precipitation and potential runoff that can affect water clarity and primary production, and 3) high pelagic phytoplankton concentrations (a contributor to turbidity). Other estuary monitoring programs (Lacouture et al. 2006, Roper et al. 2011) use a similar approach for targeted sampling of turbidity and water clarity during periods of high demersal and pelagic productivity.

Estuaries were sampled on 3 - 10 occasions during each sampling period. The number of samples obtained from an estuary is reflected in its data quality rating (Section 4.1.6).

Estuaries were divided into three zones (lower, middle and upper) to ensure adequate representation of the longitudinal differences in habitat types and water quality that typically occur in estuaries. Delineation of zones was based on the likely long-term salinity regime in each estuary inferred from a combination of geomorphological characteristics and littoral and in-stream vegetation composition (Arundel et al. 2009). The lower zone encompassed the mouth of each estuary; this is the landward side of the berm (if the estuary is closed) or upstream of the entrance to an embayment or the ocean (if permanently open). The upper region was delineated as the estimated maximum upstream extent of consistent saline intrusion.

Data from the middle and upper zones were used in the calculation of the Turbidity measure as these data are more likely to reflect impacts of catchment-based threats than data from the lower zone. Using turbidity data from the lower zone for comparisons among estuaries is perilous because tidal amplitude will greatly influence turbidity values. Although tidal amplitude strongly influences estuary structure and function, it is an intrinsic characteristic of each estuary, whereas the focus of the IEC is on capturing impacts of anthropogenic pressures and stressors on estuary condition. In the few instances where the middle and upper zones of an estuary were not accessible, data from the lower zone were used in the calculation of the Turbidity measure, and this is reflected in the data quality ratings (Section 4.1.6).

Chlorophyll *a* samples were taken from the upper zone only. The upper estuary is where the signal of catchment-derived nutrient enrichment is most likely to be present, maximising discrimination among estuaries for detecting this threat.

4.1.1 Turbidity

Turbidity was measured using calibrated fluorescence sensors, and expressed in Nephelometric Turbidity Units (NTU). Turbidity is the degree to which light is scattered by particles suspended in a liquid, and the units of its measurement reflect the wavelength of light and the angle at which the detector is positioned. Turbidity expressed in NTU has been measured by a single detector at 90 degrees to an incident beam of white light.

At each site within an estuary, measurements were taken just below the surface, 1 m below the surface, 1 m above the bottom, and just above the bottom. If the depth at a site was ≤2 m, then measurements were taken just below the surface and just above the bottom. Turbidity was measured at replicate sites within the upper and middle zones of an estuary, where possible. The number of samples obtained from an estuary is reflected in its data quality rating (Section 4.1.6).

If there was an apparent channel, sampling was conducted in the centre of the channel, where possible. If there was no obvious channel, sampling was conducted at the approximate centre of the water body, where possible. If a boat was not available or if measurement sites were inaccessible by boat, sensors were lowered from a structure spanning or projecting into the estuary (e.g. pier, pontoon, bridge). If an appropriate structure was not available, sampling was conducted by wading into the estuary up to a maximum safe depth.

Quality assurance and quality control (QAQC) checks were performed on the sample data. This included screening for extreme values in bottom measurements which may have indicated disturbance of the substratum by the sensors.

4.1.2 Chlorophyll *a*

Triplicate samples of surface water were taken from a single site in the upper estuary and were filtered onto glass-fibre filters (Whatman GF/C), 0.7 µm pore size, using a syringe filter or hand-held vacuum pump. Water was filtered until the glass fibre filter clogged and water no longer passed through. Filters were wrapped in aluminium foil and then placed on ice before freezing and subsequent transport to the laboratory. Once samples reached the laboratory, they were frozen at -20°C until analysis.

Chlorophyll *a* was chemically extracted from the filters prior to spectrophotometric analyses, according to the standard methods described by Lorenzen (1967). Briefly, samples were extracted in 90% acetone overnight at 4°C. Samples were then centrifuged and the absorbance of the supernatant measured at 665 nm before and after acidification with two drops of 1M hydrochloric acid. Chlorophyll *a* concentration was calculated using the equations in Lorenzen (1967).

Quality assurance and quality control (QAQC) checks were performed on the sample data.

4.1.3 Metric calculation – Turbidity and Chlorophyll *a*

As the two measures Turbidity and Chlorophyll *a* are to be used for assessing estuarine condition, the data for these two parameters must be set in the context of compliance with an appropriate standard. Thus, the proportion of values for either measure within an estuary that exceeds this standard (i.e. non-compliance) provides a measure of condition based on water quality. However, these standards are being used to contextualise observed values in terms of ecological relevance not to report on compliance per se.

The IEC uses the objectives for riverine estuaries set out in the State of the Environment Protection Policy (Waters) (2018) as the defined compliance value for the calculations of these two measures. These compliance values are based on the 75th percentile of observed values from a long-term dataset held by EPA Victoria, and are 10 NTU for turbidity, and 3 µg/L for chlorophyll *a*. For the two measures in the IEC, the 'worst-expected value' (see below) was calculated as the 98^{th} percentile of the observed NTU and chlorophyll *a* values, which is consistent with the approach used by OEH NSW (2013).

The overall approach to metric calculation mirrors the methods used by the NSW Office of Environment and Heritage (OEH NSW 2013) and was as follows:

1. **Calculating the non-compliance value**. The proportion of observed values that were non-compliant with a specified compliance value was calculated to derive a non-compliance value. In this case, 'observed values' refers to the mean of measurements taken on a given sampling trip (see below). The non-compliance values can range from 0 (i.e. all observed values are compliant) to 1 (i.e. all observed values are non-compliant). A non-compliance value of 0.4, for example, indicates that 40% of observed values were non-compliant and 60% compliant.
2. **Calculating the worst-expected value**. The worst-expected value was calculated as the 98^{th} percentile of observed values from all estuaries assessed for that parameter in the IEC. Water Quality data were collected from 94 Victorian estuaries during the IEC assessment. Therefore, Water Quality metric results are contextualised with the worst conditions observed in Victorian estuaries during IEC sampling.
3. **Calculating the distance value**. For estuaries where non-compliance was detected (i.e. the non-compliance value exceeded zero), the distance value describes the distance of an observed value that lies between the defined compliance value and the worst-expected value, and is a measure of the magnitude of non-compliance. The distance value ranges from 0 when values are compliant (i.e. have a non-compliance score of 0) and 1 when worst-expected values are obtained.
4. **Calculating the metric value**. The metric value is the geometric mean of the non-compliance value and the distance value, calculated using the following equation:

$$Metric\ value = \sqrt(non - compliance\ value\ \times distance\ value)$$

Turbidity was sampled at up to four depths in each estuary for the IEC (Section 0). There is a single turbidity objective for riverine estuaries included in SEPP Waters because analyses conducted by EPA Victoria to inform the SEPP Waters revision found no systematic depth stratification in turbidity. Consequently, the average turbidity across all depths sampled was used in calculations of the Turbidity measure for the IEC.

For a given sampling trip, the average of all samples from an estuary was calculated. For example, if the estuary was sampled on a given day at six sites, the average of these six measurements was taken to calculate the trip mean. This was done to provide an integrated picture of Water Quality within an estuary as the spatial scale of interest for the IEC is the 'whole-of-estuary'.

For each estuary, the 75^{th} percentile of trip means was calculated, and this was used as the observed value for metric calculations. This is consistent with the approach included in SEPP Waters of using the 75^{th} percentile of observed data to assess compliance against SEPP Waters objectives. This means that some water quality samples from within an estuary may exceed the SEPP objective on some occasions but the estuary will not be assessed as being non-compliant unless the 75^{th} percentile value is non-compliant.

However, it is important to note that this use of the 75^{th} percentile to provide a suitably robust assessment of compliance with SEPP objectives is predicated on there being a minimum of 11 data points collected from monitoring over one year. The IEC is *not* intended to assess compliance against SEPP Waters obligations (see Section 4.3 for notes on interpreting results).

4.1.4 Reference conditions – Turbidity and Chlorophyll *a*

Conditions observed in minimally modified estuaries were used as reference conditions for the two IEC Water Quality measures. These conditions were the SEPP Waters objectives for riverine estuaries (Section 5.1), and were derived from long-term data collected from minimally modified estuaries (EPA 2011).

4.1.5 Metric scoring – Turbidity and Chlorophyll *a*

The Turbidity and Chlorophyll *a* measures were scored using a scale of 1 to 5 and applying the scoring thresholds in Table 14. The scoring approach highlighted estuaries without impact (i.e. observed water quality does not exceed SEPP Waters objectives and appears adequate to protect beneficial uses, score = 5).

Thresholds for the remaining scoring categories were informed by the distribution of observed values by using inter-quartile ranges.

Table 14: Scoring thresholds for the Turbidity and Chlorophyll *a* (Chl a) measures for the IEC.

Score	Metric values Turbidity	Metric values Chlorophyll *a*
5	0	0
4	>0 – 0.18	>0 – 0.11
3	>0.18 – 0.31	>0.11 – 0.18
2	>0.31 – 0.6	>0.18 – 0.36
1	>0.6	>0.36

4.1.6 Data quality – Turbidity and Chlorophyll *a*

Categories of data quality were assigned to the scores for the Turbidity and Chlorophyll *a* measures (Table 15 and Table 16). These categories were based on the spatial distribution of sample locations within an estuary and the temporal replication of sampled data used for the calculations (Section 4.1.3).

Table 15: Data quality categories for the Turbidity measure.

Spatial distribution of sample locations		Sampling frequency			
		7 - 10 trips	**5 - 6 trips**	**3 - 4 trips**	**<3 trips**
Zones	**Middle + Upper**	Very high	High	Medium	Medium
	Upper only	Very high	High	Medium	Low
	Middle only	Medium	Low	Low	Low
	Lower only	Low	Low	Low	Low

Table 16: Data quality categories for the Chlorophyll *a* measure.

Spatial distribution of samples	Sampling frequency		
	>5 trips	**3 – 5 trips**	**<3 trips**
Upper	High	Medium	Low

4.2 IEC Water Quality sub-index score

The Water Quality sub-index score was calculated by combining the scores of the Turbidity and Chlorophyll *a* measures. As the Turbidity and Chlorophyll *a* measures were each scored from 1 to 5, the resulting sum (with a possible range of 2 to 10) was converted to a scale of 1 to 10 using the following formula:

$$Water\ Quality\ score = \frac{\left((Turbidity\ score + Chlorophyll\ a\ score) - 2\right) \times 9}{8} + 1$$

The Water Quality sub-index ranged from 1 to 10, with a value of 1 assigned to estuaries with the worst Water Quality and a value of 10 assigned to estuaries with the best Water Quality.

4.3 Notes on interpreting Water Quality results

- The sampling of Water Quality for this round of assessment of the IEC probably did not capture the full range of Water Quality conditions within the sampled estuaries. For example, sampling directly following high-flow events was avoided in an attempt to capture 'baseline' Water Quality in each estuary. High-flow events can influence many water quality parameters, including turbidity and chlorophyll *a* concentrations, with repercussions for estuarine processes and biota. It is possible that pressures and stressors have impacts on water quality that are only apparent during high-flow or pulsed events which were not captured by IEC sampling. Therefore, some estuaries that scored well according to the IEC Water Quality sub-index may have experienced episodic water quality issues that were not detected in the current assessment.
- Water Quality data collected over different years was used in the IEC assessment. Effects of interannual and climatic variability (particularly rainfall and temperature) on Water Quality observations have not been analysed or accounted for in IEC metric calculations. Rainfall influences the delivery of sediments and nutrients to estuaries. Temperature influences primary productivity within estuaries. Further investigation and analyses are required to better understand patterns of variability in estuarine chlorophyll *a* and turbidity in relation to climate variability.
- The IEC is not intended for assessment of compliance against SEPP Waters obligations. Instead, this sub-index of the IEC aims to convey Water Quality information on estuary condition that is ecologically meaningful. That is why the IEC uses an approach that explicitly sets the observed turbidity and chlorophyll *a* values in the context of the SEPP Waters objectives for riverine estuaries which are intended to protect the beneficial uses of Victoria's estuaries.

5. Flora

The Flora sub-index of the IEC consists of two measures: Fringing Vegetation and Submerged Vegetation.

5.1 Fringing Vegetation

The text in this section is from Sinclair and Kohout (2018) which details the rationale, development and methods for the assessment of the Fringing Vegetation measure. See Sinclair and Kohout (2018) and Sinclair et al. (2020) for additional details.

In the context of the IEC, fringing vegetation refers to the vegetation which grows above the permanently inundated portion of the estuary but within the zone of influence of the more-or-less saline estuarine waters. It includes vegetation in intertidal areas and riparian areas in the estuary but not subtidal vegetation (see Sinclair and Kohout 2018 for more precise working definitions of vegetation zones). Five broad plant communities make up most of vegetation in the estuarine fringe in Victoria: mangroves, coastal saltmarsh, marshlands, ephemeral pools and swamp scrub (Duke 2006; Boon et al. 2015; 2016).

Fringing vegetation is a vital consideration in the assessment of estuary condition. First, fringing vegetation itself is an asset with its own inherent values. These include the organisms that largely inhabit the fringe, some of which may be rare or threatened such as Salt Lawrencia (*Lawrencia spicata*). Aesthetic values are also important inherent values of fringing vegetation, and provide a basis for tourism, recreation, education and research (Barbier et al. 2011).

Second, the condition of the fringing vegetation influences that of the rest of the estuary. Degradation or loss of fringing vegetation impacts estuarine biological function (Roper et al. 2011). For example, fringing vegetation filters flows of water, chemicals (e.g. nutrients, toxins) and organisms that come from the catchment (Mondon et al. 2009). It also can intercept some stormwater runoff and reduce lateral erosion and littoral water velocities during flooding (Adams and Riddin 2007). Fringing vegetation contributes to the role that estuaries play in naturally protecting the quality of coastal waters by diluting, filtering and settling out sediments and excess nutrients (Tagaza 1995). Fringing vegetation may also supply energy and material to adjacent systems, both from primary production when vascular plant detritus is exported and enters offshore food webs (Boon et al. 2011) and secondary production when herbivores supported by fringing vegetation and/or their spawn enter those food webs (Mazumder et al. 2006).

Vegetation of the fringe provides habitat (e.g. snags, roots, branches for perching) for estuarine fauna, some of which are listed as threatened (Saintilan and Rogers 2013). Many fish species that live as adults in the open water of the estuary or the sea live as larvae in the shallows amongst the vegetation that fringes estuaries (Hindell and Jenkins 2004, Nagelkerken et al. 2008). As the quality of larval habitat varies depending on vegetation cover, type, structure and structural complexity, the condition of the fringing vegetation directly affects larval habitat (e.g. Payne and Gillanders 2009).

Many birds also use the fringing vegetation. Small passerines roost in dense riparian vegetation (Boon et al. 2011, Pope et al. 2015). In Victoria, saltmarshes provide an important food source for the critically endangered Orange-bellied Parrot (*Neophema chrysogaster*) (Mondon et al. 2009), vital habitat for migratory birds, and nest sites for colonial-nesting waterbird species (Spencer et al. 2009).

Because of its position along the water-land interface, fringing vegetation is exposed to pressures and stressors from both aquatic and terrestrial sources (Figure 7). Sinclair and Kohout (2018) present more detailed descriptions of the pathways depicted in Figure 7.

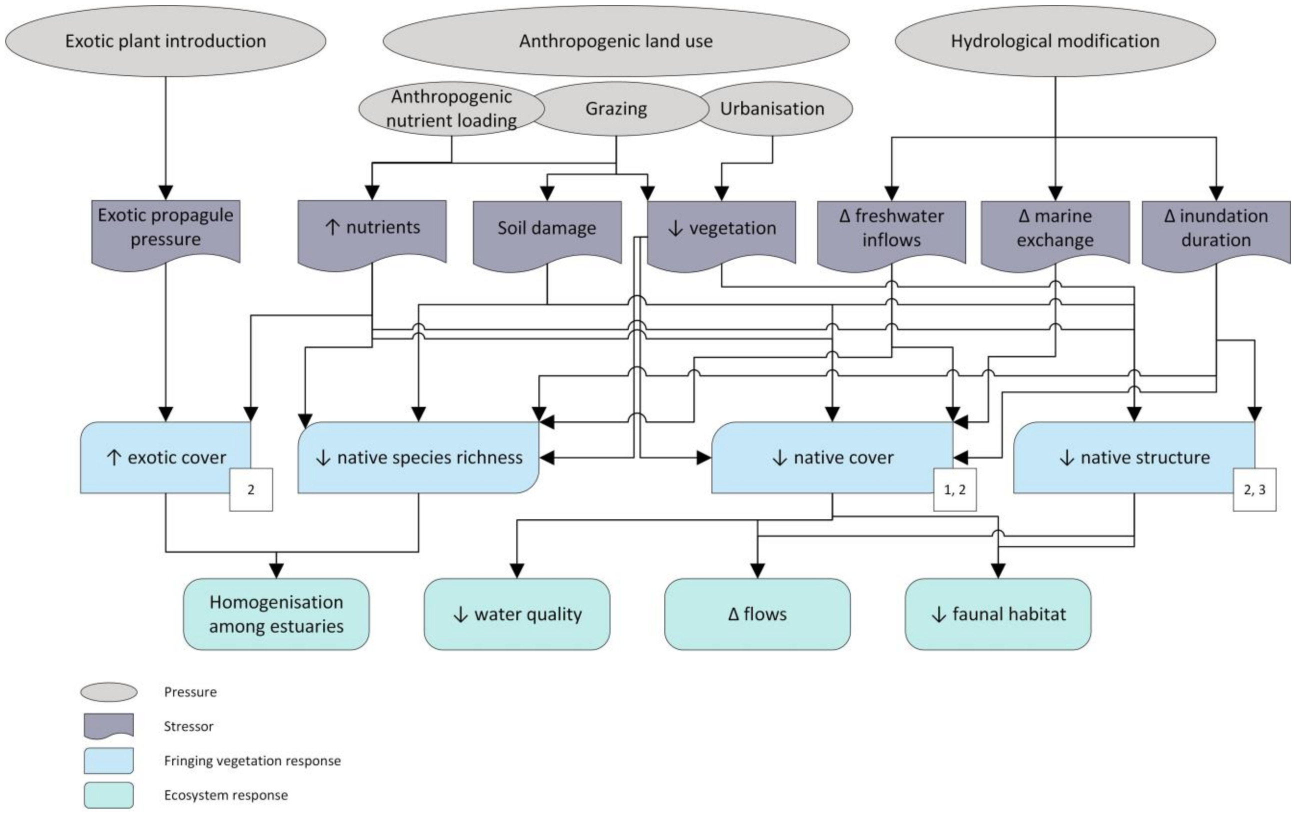

Figure 7: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which pressures and stressors can link to fringing vegetation responses and ecosystem responses. The numbered boxes indicate which fringing vegetation responses are addressed by the three metrics underpinning the Fringing Vegetation measure in the IEC: 1. Percentage of the fringe area that is covered by built structures, 2. Nativeness of the fringing vegetation, 3. Structural complexity of the fringing vegetation. Arrows within boxes indicate the direction of change; ∆ indicates a possible change in either direction.

5.1.1 Data collection – Fringing Vegetation

The IEC assessment of Fringing Vegetation produced a single score (0 – 100) which was intended to provide a measure of an estuary's fringing vegetation condition relative to other estuaries and to accord with expert views on estuary condition.

This single score for the Fringing Vegetation measure was based on integrating three distinct metrics, each scored on a 0 – 100 scale:

1. Percentage of fringe area that is covered by built structures.
2. Nativeness of the fringing vegetation.
3. Structural complexity of the fringing vegetation.

The rapid assessment approach required observers who are knowledgeable in estuarine ecology and estuarine plants (to a similar level as required for Habitat Hectares (DSE 2004) or the Index of Wetland Condition (DELWP 2018)).

The assessment process involved the following steps at each estuary:

1. In the field, the extent of the fringing vegetation and any built structures that impinge on it were mapped (by annotating an aerial photograph).
2. Fringing vegetation was mapped according to Ecological Vegetation Classes (EVC) (https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks) and subjectively classified into areas of different condition with reference to vegetation structure, weediness and land use. In practice, this step often involved using pre-existing vegetation maps.
3. Each patch of vegetation that could be accessed was scored for its degree of invasion by perennial weeds and its structural resemblance to a relevant benchmark

(https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks). The assessment is rapid, not exhaustive.

4. Additional observations were recorded on the score sheet, relating to 'Health of the dominant plant species' and 'Extent of engineered hydrological modifications'.
5. Back in the office, scores were estimated for the areas that could not be observed in the field. These estimated scores were based on the areas that were observed.
6. The areas of all relevant polygons were calculated, and a score produced for each metric using the formulae described in Sections 5.1.3 to 5.1.5, below.
7. The uncertainty attributable to incomplete observation was quantified by producing an upper and lower score (assuming the best and worst possible scores, respectively) for the un-observed areas.

5.1.2 Reference conditions – Fringing Vegetation

Undisturbed or unmodified conditions were used as the reference conditions for assessing the Fringing Vegetation measure for the IEC. Reference conditions were assumed to be the Ecological Vegetation Class (EVC) benchmarks and the absence of built structures within the 'pre-1750 intertidal zone' (Sinclair and Kohout 2018). Estuaries that did not exist before 1750 (e.g. artificially constructed drains or waterways) cannot be scored using the IEC approach.

In practice, delineation of the fringing vegetation was challenging for several reasons:

- The lower boundary of the fringing vegetation may be ambiguous because the extent of the "permanently inundated" portion of the estuary may not be obvious at the time of assessment. For estuaries or portions of estuaries that are fringed by mangroves, the seaward margin of the mangroves was considered the lower boundary of the fringing vegetation (i.e. the mangroves were included). For estuaries lacking mangroves, the boundary of the fringing vegetation was defined by the estuary layer in the DELWP Victorian Spatial Data Library as the edge of this layer approximates the lower edge of the intertidal zone. Seagrass vegetation (vegetation dominated by species of *Zostera*, with little or no mangrove cover) was always excluded from the fringing vegetation and treated as part of the "permanently inundated" portion of the estuary, even if it was exposed at the time of assessment, as occasionally occurs. Water-Mats (*Lepilaena* species) and Sea Tassels (*Ruppia* species) were only included when growing in pools surrounded by intertidal vegetation.
- The landward boundary of the fringing vegetation may be ambiguous because the inland extent of the hydrological influence of the estuary is not obvious. For the purposes of the IEC, the fringing vegetation includes:
  - all of the estuarine portions of the 'pre-1750 intertidal zone' defined by Boon et al. (2011) and further described in Sinclair and Boon (2012), and
  - all wetlands or damplands which showed a brackish influence in their species composition (as determined by a botanist, based on the occurrence of salt-tolerant species such as *Juncus kraussii*, *Selliera radicans* and *Samolus repens*), and which were contiguous with the flats of the estuary (i.e. excluding any nearby wetlands of this type which were separated by raised ground, dunes, etc.).
- Portions of the fringing vegetation may be hydrologically disconnected from the estuary by human impacts such as sea walls and roads. The full pre-impact extent of the fringing vegetation must be assessed for IEC, requiring that the 'pre-1750' boundary of the fringing vegetation be delineated. For intertidal areas, these areas have been defined by Boon et al. (2011) and further described in Sinclair and Boon (2012). The areas delineated by Sinclair and Boon (2012) were used unless field examination showed them to be incorrect.
- The extent of the fringe may be ambiguous for estuaries that meet low-energy coasts fringed by a continuous intertidal zone (e.g. Western Port, Corner Inlet). In these cases, multiple estuaries may be joined by a continuous intertidal zone. In theory, only that portion of the intertidal zone with some influence from each stream is relevant. However, this is virtually impossible to determine in practice. For this assessment, the fringing vegetation relevant to each estuary was divided from the coastal intertidal zone. This was done by delineating a buffer around the "permanently inundated" portion of the estuary, with the buffer width determined by the flow in each stream (assuming that large streams exert more influence on the coastal zone than small streams). The resulting 'flow-based buffer' only applied to

estuaries with fringe boundaries that were ambiguous because they meet a coastal intertidal zone. See Sinclair and Kohout (2018) for details.

- Estuaries may be joined to very extensive lagoon systems that run parallel to the coast and extend far from the estuary mouth. For the purposes of IEC, the entirety of such systems was included as long as they met the other criteria (e.g. they had a brackish influence associated with the estuary). In rare cases, two estuaries may be linked via a continuous marshland strip. For example, Merriman Creek, a minor stream at Seaspray in Gippsland, is connected via marshland to the extensive Lake Reeve system (over 30 km long), which eventually meets the Gippsland Lakes. In this case, an arbitrary boundary was drawn between the two systems. The same approach was used for estuaries that were connected by coastal intertidal marshes.

To avoid inconsistency in application, the extent of fringing vegetation was defined for each estuary during the first IEC assessment period (Sinclair and Kohout 2018) and should remain unchanged for future assessments.

5.1.3 Fringing Vegetation Metric 1: Percentage of the Fringe Area That is Covered by Built Structures

This metric assessed the Percentage of the Fringe Area That is Covered by Built Structures. Built structures are often detrimental because they remove fringing vegetation, which can reduce their buffering role (Section 5.1) and habitat complexity. The higher the Percentage of the Fringe Area That is Covered by Built Structures, the lower the score for this metric. This metric was assessed against a single baseline (i.e. 0% cover by built structures) that applies to all estuaries, allowing comparisons among estuaries and longitudinal comparisons down a single estuary.

The assessment required the following steps:

1. A map of the estuary fringe was produced, showing all portions that are built structures (Step 1 in Section 5.1.1). The spatial resolution of this map permitted structures larger than 10 x 10 m to be distinguished.
2. The total area of the fringe was calculated and recorded (*F*).
3. The area covered by built structures was calculated and recorded (*BS*).

A score (0 - 100) was derived, using the following formula:

$$Fringing\ Vegetation\ Metric\ 1\ Score = 100 - \left(100 \times \frac{BS}{F}\right)$$

The score for this metric was recorded to the nearest integer. The score sheet required the map and the raw areas to be recorded. No uncertainty was recorded for this indicator, which may be assessed from aerial imagery with pixel sizes <1 m. See Sinclair and Kohout (2018) for a detailed definition of built structures.

5.1.4 Fringing Vegetation Metric 2: Nativeness of the Fringing Vegetation

This metric assessed the extent of cover by exotic plant species that have invaded the estuarine fringe. Exotic species are typically detrimental because they occupy niches that would otherwise be occupied by native species and are likely to alter the structure of the vegetation, reducing its habitat value and/or impairing its natural ecological functions. A lack of exotic plants (weeds) confers a high score; an abundance of weeds confers a low score. This metric was assessed against a common baseline (i.e. 0% cover by exotic plants) that applies to all estuaries, allowing comparisons among estuaries and longitudinal comparisons down a single estuary.

The scoring sheet for this metric required estimates of cover, along with records of uncertainty (i.e. plausible bounds, Section 5.1.6).

The assessment involved the following steps:

1. A map was produced, showing all areas that are fringing vegetation (F) (i.e. all areas of the fringe that are not built structures, derived from the same map produced for Percentage of the fringe area that is covered by built structures).

2. This area of fringing vegetation was divided into *n* patches (where *n* is the total number of patches in the area), with each patch representing an area that is a single EVC in a single condition state. The area of each patch was recorded (A_1, $A_2 \ldots A_n$).

3. The percentage cover of all perennial vegetation was estimated for each patch (CP_1, $CP_2 \ldots CP_n$).

4. The percentage cover of all exotic perennial species was estimated for each patch (CE_1, $CE_2 \ldots CE_n$)

5. A score (0 - 100) was derived, using the following formula:

$$Fringing\ Vegetation\ Metric\ Score\ 2 =$$

$$100 - ((100 \text{ x } CE_1/CP_1 \text{ x } A_1/F) + (100 \text{ x } CE_2/CP_2 \text{ x } A_2/F) \ldots.. + (100 \text{ x } CE_n/CP_n \text{ X } A_n/F))$$

The score for this metric was recorded to the nearest integer. The score sheet also required inclusion of the map and all raw estimates. Areas of bare ground were included, and their cover per patch (CE/CP) was assigned a value of 0 (meaning that bare ground does not cause a score reduction).

This metric could not be assessed accurately from aerial imagery, and field assessment was recommended. Where patches could not be directly observed, the process described in Section 5.1.6 ('Expressing uncertainty for estuaries with partial assessment coverage') was used.

5.1.5 Fringing Vegetation Metric 3: Structural Complexity of the Fringing Vegetation

This metric assessed whether the fringing vegetation possessed the mix and cover of life-forms that would be expected to be prominent, given the vegetation types (i.e. EVCs) that were present. This metric produced a score that was calculated with reference to benchmarks specific to each EVC, enabling each estuary to be compared with its condition on previous dates (longitudinal studies). The benchmarks (for IWC and Habitat Hectares) have been developed so that their scores are reasonably consistent with each other. Coarse among-estuary comparison is possible if the results are interpreted cautiously.

The scoring sheet for this metric required raw estimates of cover, along with their uncertainty (i.e. plausible bounds).

The assessment required the following steps:

1. A map was produced showing all areas that are fringing vegetation (F) (i.e. all areas of the fringe that are not built structures, derived from the same map produced for the other two metrics in the Fringing Vegetation measure).

2. The total area of fringing vegetation was divided up into *n* patches (where *n* is the total number of patches in the area), with each patch representing an area that is a single EVC in a single condition state. The area of each patch was recorded (A_1, $A_2 \ldots A_n$). These were the same patches required for the metric Nativeness of the Fringing Vegetation.

3. For each EVC, the appropriate benchmark was consulted (see above; DELWP 2016), which specified the life-forms that are expected to be present (listed under "Critical Life-form Groups" in IWC), and the cover that the species within that life-form are expected to attain ($E_1 – E_n$, expected covers for life-forms in an EVC with *n* life-forms).

4. For each patch, the actual cover of native species attributable to each of the benchmark life-forms was estimated ($O_1 – O_n$, observed covers for life-forms in an EVC with *n* life-forms).

5. For each patch, a score (0 – 100) was derived using the following formula:

$$Patch\ score = 100 \times (\mathrm{O}_1\ +\ \mathrm{O}_2 + \ldots \mathrm{O}_n)/(\mathrm{E}_1\ +\ \mathrm{E}_2 + \ldots \mathrm{E}_n)$$

If O exceeded E, O was assumed to be equal to E for the purposes of scoring. In polygons of bare ground, the patch score was set to 100.

6. A score (rounded to the nearest integer) was then derived for the metric, across all patches, using the following formula:

$$Fringing\ Vegetation\ Metric\ Score\ 3 =$$

$$((Patch\ score\ 1)\ x\ (A_1/F))\ + ((Patch\ score\ 2)\ x\ (A_2/F)).....+((Patch\ score\ n)\ \mathrm{x}\ (A_n/F))$$

This metric could not be assessed accurately from aerial imagery, and field assessment was recommended. If patches could not be directly observed, the process described in Section 5.1.6 (‘Expressing uncertainty for estuaries with partial assessment coverage’) was applied.

5.1.6 Expressing uncertainty for estuaries with partial assessment coverage

Given the inevitable constraints on field effort, it was necessary that scores could be derived for estuaries that could be only partially assessed. This was done by reporting scores as best estimates with upper and lower bounds, that represented the degree of uncertainty introduced to the score owing to the incompleteness of the assessment (noting that there was additional uncertainty due to other factors). The upper bound represented the highest score possible based on what was possible in the areas that were not seen. Similarly, the lower bound represented the lowest score possible. The more of the estuary that was observed, the narrower these bounds.

For Metric 1 (Percentage of the Fringe Area That is Covered by Built Structures), recent aerial imagery was used to map built structures in all areas, including those which were not visited, and a single score was derived in the absence of a site visit. Therefore, there was no need to express uncertainty due to partial site coverage for this metric. However, for Metrics 2 and 3 (Nativeness of the Fringing Vegetation and Structural Complexity of the Fringing Vegetation) that needed field assessment, bounds of uncertainty were required.

Determination of the bounds for uncertainty required the following steps:

1. For Metrics 2 and 3, all polygons on the map were annotated as to whether they were “directly observed” or “not observed”. For many polygons, a partial or imperfect observation was made (e.g. from a distance through binoculars, or of only one small portion of a larger polygon). In these cases, the observer selected the most appropriate category. Some EVCs hardly vary across large areas and may be adequately observed with only cursory effort (e.g. Mangrove Shrubland). However, other EVCs vary substantially with regard to species composition, degree of exotic invasion and structure, and these EVCs require more intensive survey effort to be designated “directly observed”. Regardless of its vegetation, any area that was more than 250 m from a point that was actually observed was designated as “not observed”.

2. The polygons that were “not observed” were assigned data by extrapolating scores from “directly observed” polygons with the same EVC and context. The overall score for each metric was derived from this extrapolation and reported as the ‘best estimate’.

3. The polygons that were “not observed” were then re-assigned values that produced a zero score (100% exotic cover; none of the benchmark life-form groups). A new overall score for each metric was produced using these scores, representing the lower bound. A similar process was repeated with the highest score possible for each “not observed” polygon to derive the upper bound. This process may lead to upper and lower bounds of different sizes.

The overall estuary score combined the ‘best estimates’ for all three metrics, reported with the most extreme combination of upper and lower bounds. This expression of uncertainty in metrics was used for the Fringing Vegetation measure rather than categorical data quality rating applied to other IEC measures. See Sinclair and Kohout (2018) and Sinclair et al. (2020) for additional information on the Fringing Vegetation assessment method.

5.1.7 Combining metrics to produce the Fringing Vegetation measure

Each of the three metrics described above produced a score ranging between 0 and 100. To combine them, their arithmetic mean (simple average) was calculated to produce the Fringing Vegetation measure (0 – 100).

5.1.8 Notes on interpreting Fringing Vegetation results

- All condition metrics are subjective, including those used to assess Fringing Vegetation in the IEC. There will always be some variables that are represented at low resolution or not captured. Therefore, further observation will often be necessary to more fully understand the ecology of the estuarine fringe.
- For state-wide reporting of estuary condition, visual estimates such as those used in metrics for the IEC Fringing Vegetation measure are suitable and appropriate. However, if subtle changes within an estuary must be monitored, a different strategy is recommended. This strategy would likely include quantitative sampling of specific variables at appropriate spatial and temporal scales. In such cases, the primary vegetation data are probably more useful for understanding changes and drawing inferences rather than the metrics which are best treated as secondary means of simplifying and interpreting the primary data.
- Fifteen of Victoria's estuaries had no obvious edge to their fringing vegetation which merged into the nearby coastal marsh vegetation. For these estuaries, an arbitrary buffer was used to define the extent of their fringe. This buffer was scaled according to stream flow, on the assumption that higher-flow streams exert a wider influence. The size of this arbitrary buffer influences the score because the area of the fringe is used as a denominator in the score calculations for all components. This is not a problem for longitudinal comparisons within an estuary, but it does mean that comparison of the score of one estuary to another is dependent on the arbitrary choice of buffer size for these estuaries (see Sinclair and Kohout 2018, Sinclair et al. 2020).

5.2 Submerged Vegetation

In the context of the IEC, Submerged Vegetation refers to aquatic plants attached to bottom sediments that are generally entirely submerged but may be exposed during very low tides.

In Victorian estuaries, submerged vegetation consists primarily of macroalgae and seagrass. Seagrass species recorded from Victorian riverine estuaries include *Zostera muelleri, Zostera nigracaulis* and *Ruppia* spp. Seagrasses are rooted macrophytes that have higher light requirements and slower uptake of nutrients than macroalgae. When nutrient inputs to estuaries are elevated, macroalgae can proliferate due to their capacity for rapid nutrient uptake, and this leads to an increase in the overall biomass and extent of submerged vegetation. Their low light requirements mean that macroalgae can also tolerate higher turbidity associated with the increased sediment inputs that often accompany elevated nutrient inputs. Under these conditions, the proliferation of macroalgae shades seagrasses which further reduces the light needed for seagrasses to grow. Consequently, nutrient enrichment and reduced light availability often lead to the dominance of macroalgae over seagrasses in submerged vegetation communities (Figure 8).

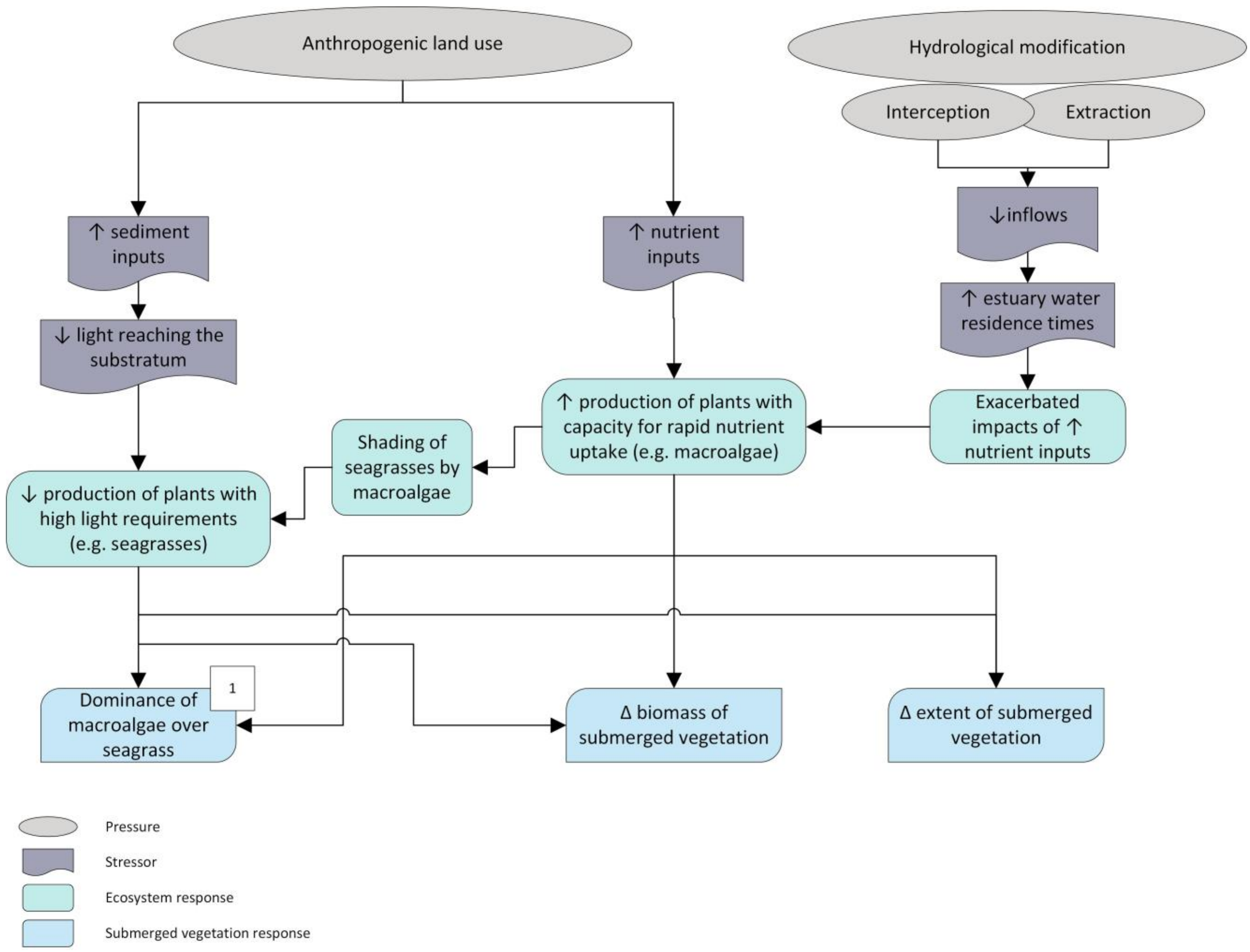

Figure 8: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which pressures and stressors can link to ecosystem and submerged vegetation responses. The numbered box indicates the submerged vegetation response that is addressed by the submerged vegetation metric (i.e. the ratio of the areal coverage of macroalgae to the total area of submerged vegetation. Arrows within boxes indicate the direction of change; Δ indicates a possible change in either direction.

The dominance of macroalgae over seagrasses in response to elevated nutrient loads in estuaries is well-established (Valiela et al. 1997; Hauxwell and Valiela 2004; Woodland et al. 2015). This dominance can be measured as the ratio of the coverage (m^2) of macroalgae (MA) to the total area of submerged vegetation (TV) within an estuary (MA:TV). MA:TV has been shown to increase (i.e. increased dominance of macroalgae) with increasing nitrogen loading to Victorian estuaries (Woodland et al. 2015) as well as increasing proportions of an estuary's catchment subject to land uses that generate or apply fertilisers (Cook et al. 2018). Cook et al.

(2018) reported a transition to a dominance of macroalgae once the proportion of fertilized land in the catchment exceeded 24%, highlighting the sensitivity of estuaries to catchment land use.

For the IEC, derivation of the Submerged Vegetation metric required field-mapping of each estuary's submerged vegetation (Section 5.2.1), followed by calculation of the ratio of macroalgae to total submerged vegetation (MA:TV, Section 5.2.2).

5.2.1 Data collection – Submerged Vegetation

Submerged Vegetation was measured in 80 estuaries. Twenty of these estuaries were sampled by scientists from Monash University as part of complementary projects in 2013 - 2015. The remaining 60 estuaries were sampled during 2017 - 2019 by scientists from DELWP's Arthur Rylah Institute for Environmental Research (ARI).

This mapping required data on the types and coverage of benthic vegetation in each estuary. Field data were collected at various locations within the estuary to ground-truth the mapping of full coverage derived from available aerial imagery. These field data were collected in late spring, summer and early autumn to correspond with warmer water temperatures and longer photoperiods, and to avoid winter periods of submerged vegetation dieback.

Before ground-truthing, aerial images of each estuary were investigated to identify potential areas of habitat or ambiguity. On some occasions, a Remotely Piloted Aircraft System (RPAS) was flown to scope areas and obtain imagery for later mapping. In the field, ground-truthing by taking photos of the benthic environment largely followed the protocols outlined in Woodland and Cook (2015). A GoPro camera (Version 5) on a 2.5 m pole was submerged to record the benthic environment. A boat-mounted side-scan sonar was also used to identify changes in benthic coverage. Sampling points were at points of interest that were identified from scoping aerial imagery, visually from a boat or when using the side-scan sonar. The GoPro camera was set to take a photo every two seconds. Several photos in each set were taken above the water line to record the Global Navigation Satellite System (GNSS) location. During post-processing, each set of images was matched to its spatial location.

If the estuarine bed was visible, photographic samples were collected of the different vegetation types present. Coverages of these different vegetation types (e.g. dense, sparse) were also photographed as well as bare ground. If visibility was limited or the benthic environment was obviously largely bare, photographic samples were only taken at low spatial resolution or when the estuarine environment changed noticeably.

During the field survey, the mouth of each estuary was noted to be either closed, partially open (usually a small outflow of estuarine water) or fully open (subject to the inflow and outflow of tidal waters).

5.2.2 Data-processing and metric calculation – Submerged Vegetation

Ground-truthed benthic images were linked to their GNSS coordinates. These images were then mapped over high-resolution (<20 cm) aerial imagery sourced from the DELWP imagery archive. Only imagery later than 2010 was considered, with most of the imagery used captured later than 2015. RPAS imagery collected for some estuaries was geo-rectified and mosaicked to provide very high-resolution imagery for mapping. Mapping using the RPAS imagery conformed more accurately with ground-truthed images because it was generally collected on the same day or the day before.

Before mapping submerged vegetation, the banks of the estuaries were delineated. The banks were re-digitised in cases where the standard DELWP estuary-feature layer differed greatly from recent imagery of the estuary. Using aerial imagery (archival and drone), broad areas of contrasting appearance were mapped. These mapped areas were then compared to the geo-referenced ground-truthed benthic images and assigned a vegetation-coverage class and an indication of classification confidence (High, Medium or Low). Once areas had been classified, total areas per estuary were generated for each of the classification types (Table 17).

For each estuary, MA:TV was then derived from the ratio of total macroalgae area to total vegetated area (i.e. seagrass and macroalgae) as per Woodland and Cook (2015). Briefly, vegetated habitat areas were weighted by their coverage classes such that polygons assigned 'sparse–medium' coverage were considered to contain 50% vegetation, and those assigned as 'dense' coverage were considered as being ~100% vegetated. For example, a 10 m^2 patch of 'medium' seagrass would be designated as having 5 m^2 of seagrass and 5 m^2 of bare sediment. Total areas of each estuary and each coverage-weighted habitat class were calculated.

The ratio of macroalgae to total vegetation (MA:TV) was calculated as the sum of the weighted macroalgae areas divided by the sum of the weighted seagrass and macroalgae areas. MA:TV ranges from 0 to 1. For 15 of the 80 assessed estuaries, MA:TV could not be calculated because no submerged vegetation was detected in these estuaries.

Table 17: Categories of submerged vegetation and their percent coverage classes used for the Submerged Vegetation metric (Macroalgae to Total Vegetation, MA:TV) of the IEC.

Submerged vegetation category	Percent coverage
Seagrass dense	50 - 100%
Seagrass sparse	<50%
Macroalgae dense	50 - 100%
Macroalgae sparse	<50%
Bare sediment/unvegetated rocky reef	No vegetation coverage
Channel habitat >2 m depth	No vegetation coverage

5.2.3 Reference conditions – Submerged Vegetation

Conditions observed in minimally modified estuaries were used to estimate reference conditions for assessing the Submerged Vegetation measure for the IEC. Dominance of seagrass in estuaries was assumed to represent reference conditions because this was observed in estuaries with minimally modified catchments (Woodland et al. 2015, Cook et al. 2018). Therefore, MA:TV values approaching zero represented reference conditions.

5.2.4 Metric scoring – Submerged Vegetation

The MA:TV metric was scored on a scale of 1 to 5 by applying the scoring thresholds in Table 18.

Table 18: Scoring thresholds for the Submerged Vegetation metric (Macroalgae to Total Vegetation, MA:TV) of the IEC.

Score	MA:TV	Description
5	0 - 0.2	Submerged vegetation dominated by seagrass, minimal (or no) macroalgae present
4	>0.2 – 0.4	Submerged vegetation mostly seagrass with some macroalgae present
3	>0.4 – 0.6	Submerged vegetation represented by approximately equal amounts of seagrass and macroalgae
2	>0.6 – 0.8	Submerged vegetation mostly macroalgae with some seagrass present
1	>0.8	Submerged vegetation dominated by macroalgae, minimal (or no) seagrass present

5.2.5 Notes on interpreting Submerged Vegetation results

- Results represent a single 'snapshot' in time, and the area of submerged vegetation cover can vary within and among years in an estuary. However, by standardizing macroalgal area as a proportion of total submerged vegetation cover, our analysis reduces the influence of physical factors such as sediment movement and hydrodynamics that often limit the growth of benthic vegetation (Cook et al. 2018).
- The MA:TV metric also accounts for estuaries with different bathymetric profiles because MA:TV is functionally constrained to those areas where light penetration can support benthic vegetation (Woodland et al. 2015; Cook et al. 2018).
- The metric cannot be calculated for estuaries where submerged vegetation is not detected. Estuaries without submerged vegetation tend to be estuaries that (i) are deep and channelized (e.g. the Latrobe and Avon rivers), (ii) have high tidal amplitudes with almost complete exposure at low tide (e.g. Stockyard Creek), or (iii) have waters high in tannins that limit light penetration (e.g. Thurra and Mueller rivers).

5.3 IEC Flora sub-index score

The Flora sub-index score was calculated by combining the scores of the integrated Fringing Vegetation measure (scored 1 – 10; Section 5.1.7) and the Submerged Vegetation measure (MA:TV metric, scored 1 – 5; Section 5.2.4) using the equation below. The score for the Submerged Vegetation measure was multiplied by 2 before being added to the score for the Fringing Vegetation measure (scored 1 – 10). This number (with a possible range of 3 to 20) was then converted to a scale of 1 to 10 using the equation below.

$$Flora\ score = \frac{((Fringing\ Vegetation\ score\ +\ (MA\!:\!TV\ score\ \times\ 2))\ -3)\times 9}{17} +\ 1$$

The Flora sub-index ranged from 1 to 10, with a value of 1 assigned to estuaries with vegetation in the poorest condition and a value of 10 assigned to estuaries with vegetation in the best condition.

More estuaries were assessed for the Fringing Vegetation measure than the Submerged Vegetation measure. Where an estuary was only assessed for Fringing Vegetation, the score for the integrated Fringing Vegetation measure (scored 1 – 10; Section 5.1.7) alone was used for the Flora sub-Index. Where this occurred, it is noted in data files and results tables (see DELWP 2021).

6. Fish

Measures of biological integrity are an important part of environmental condition assessments (Karr 1981; Yoder and Rankin 1998; Barbour et al. 2000) and have been incorporated into the Victorian IEC framework in the Flora and Fish sub-indices. The integrity of biological communities is influenced by the effects of multiple impacts on physical, chemical and biological components of an ecosystem, and thereby monitoring such communities provides an integrated picture of ecological condition (Barbour et al. 2000).

As it is rarely feasible to measure all aspects of all biological communities, certain groups of organisms have been identified as useful indicators of biological integrity within estuaries, including fish (Deegan et al. 1997; Harrison and Whitfield 2004; Arundel et al. 2009). Fish are considered to be potentially useful indicators of estuary condition as they occupy relatively high trophic levels and therefore require a diverse range of intact ecosystem processes for them to survive, grow and reproduce (Deegan et al. 1997). In addition, they are taxonomically well known and popular with the public, and estuaries often have diverse assemblages of them. Fish are hypothesised to respond to chemical, physical and ecological disturbances prompted by major threats to estuaries, including anthropogenic land use, hydrological modification and geomorphological modification, with repercussions for assemblage composition. Estuarine fish assemblages include species with different trophic ecology (herbivores to piscivores), habitat associations (e.g. benthic, demersal or pelagic habitats) and occupancy patterns (e.g. opportunistic or resident; Elliott et al. 2007). Therefore, there are multiple pathways by which human pressures can influence fish assemblages, trophic structure and habitat use (Figure 9; Deegan et al. 1997).

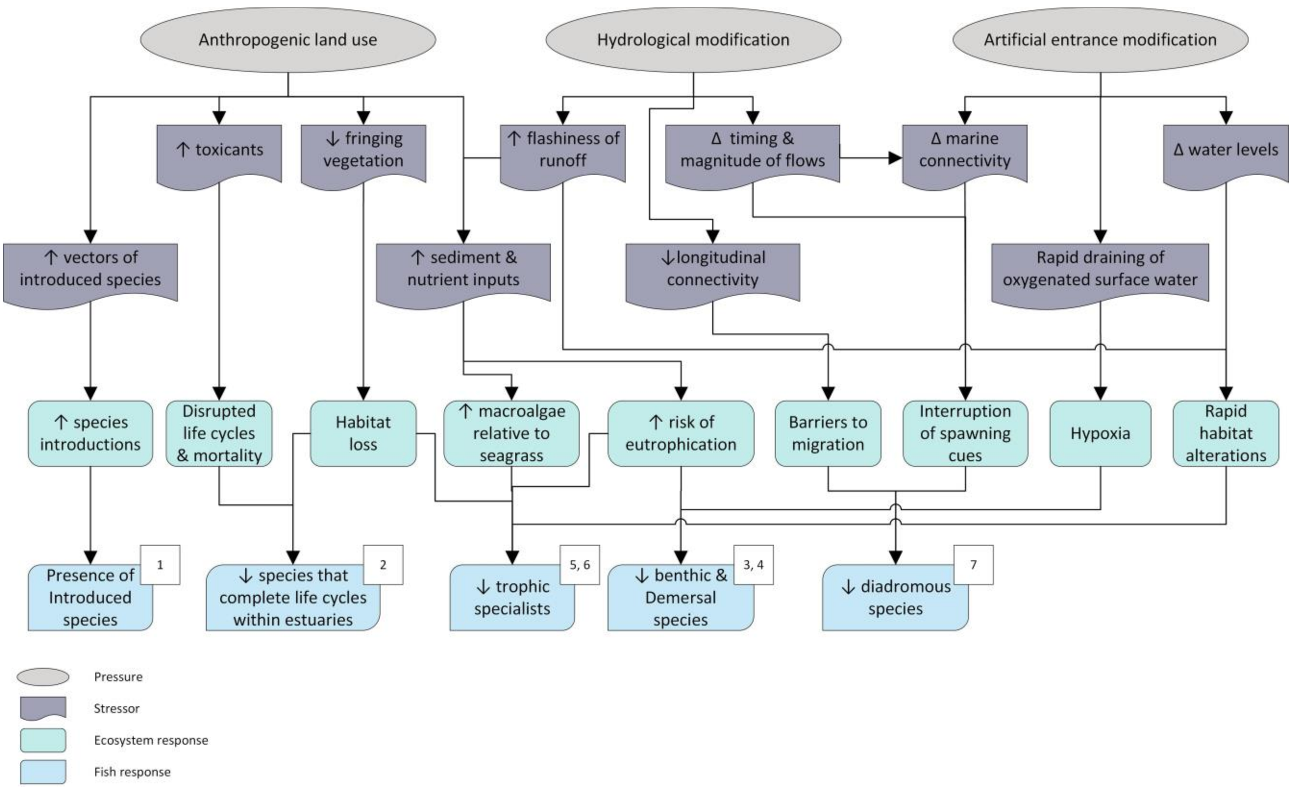

Figure 9: Conceptual diagram depicting the main pathways (black arrows linking symbols) by which pressures and stressors can link to ecosystem and fish responses. The numbered boxes indicate which fish responses are addressed by the metrics included in the Fish sub-index of the IEC (see Table 20). Arrows within boxes indicate the direction of change; Δ indicates a possible change in either direction.

Fish-based multi-metric indices have been developed for estuarine condition assessments worldwide, including in Western Australia (Hallett et al. 2012), South Africa (Harrison and Whitfield 2004; 2006), Europe (Borja et al. 2004; Cabral et al. 2012) and the United States (Deegan et al. 1997). These indices aim to distil environmental complexity into simple forms that remain scientifically valid (Stoddard et al. 2008). They are typically used in complex systems, such as estuaries, where the underlying causal processes determining ecological structure are poorly understood (Schoolmaster et al. 2012). Multi-metric indices derived from fish assemblage data often correlate with measures of human disturbances (Harrison and Kelly 2013; Martinho et al. 2015). However, relationships between single metrics of fish assemblages and human disturbances are often unclear (Martinho et al. 2015; Warry et al. 2018). This may be due to the often coarse but cost-effective methods used to sample fish for broad-scale condition assessment programs (Cabral et al. 2012) and/or the highly variable nature of estuarine fish assemblages (Sheaves 2016). Biogeographic, physical and ecological drivers of fish recruitment to estuaries, along with many other estuary-specific characteristics, all contribute to the inherently high background variation in estuarine fish assemblages that make it difficult to disentangle human impacts from natural variability (Sheaves 2016).

6.1 Data collection - Fish

Multi-metric fish indices rely on robust, consistent and repeatable sampling methods. An appropriate protocol for rapid assessment of estuarine fish assemblages was developed for the Victorian IEC during the IEC implementation trial (Warry and Reich 2013).

6.1.1 Sampling protocol - Fish

Fish data were collected from 88 estuaries for inclusion in the IEC assessment (Table B1). Thirty-one of these estuaries were sampled during the IEC implementation trial (2010 - 2012). An additional 12 estuaries in the Port Phillip and Western Port region were sampled in 2014.

The fish sampling protocol was designed to characterise the fish assemblage rather than yield accurate estimates of abundance or biomass. With the resources available, sampling could be completed in a single day for a typical estuary but needed two days for larger estuaries.

Sampling was undertaken in autumn (February – May). In Victorian estuaries during autumn, freshwater inputs are generally lower and the salt-wedge is usually further upstream than at other times of the year (Jenkins et al. 2010 and references therein). At this time, there is greatest use of estuaries by species with marine associations, and juveniles of species that have recruited in spring and summer are present (e.g. Black Bream *Acanthopagrus butcheri*; Williams et al. 2012; Jenkins et al. 2018). Catadromous species such as Tupong *Pseudaphritis urvilli* and Shortfinned Eels *Anguilla australis* are also most likely to be present in estuaries during this period (Crook et al. 2010; Crook et al. 2014).

Assessment of the temporal variation in metrics of the Fish sub-index during the IEC implementation trial indicated that the values derived using autumn data were generally less variable than those derived using data from autumn and spring (Warry and Reich 2013). Estuaries are characteristically dynamic systems subject to large natural cycles of physical and chemical conditions, including freshwater inflows and associated impacts on salinity and temperature profiles. This variability can prompt frequent temporal shifts in the distributions of different fish taxa into or out of estuarine environments. Using metrics with the least temporal variability helps maximise the potential to detect signals from broad catchment-based threats to estuaries above the noise of natural variability. Complementary data collected at finer scales, including water quality and habitat data, help explain temporal dynamics in estuarine fish fauna that may be related to natural cycles of variability rather than shifts in threats and their associated impacts. Consequently, sampling during autumn only was adopted for the state-wide assessment of the IEC.

Estuaries were divided into three zones (lower, middle and upper) as per Arundel et al. (2009) to ensure adequate representation of the longitudinal differences in habitat types that typically occur in estuaries. Zones were delineated using information on the long-term salinity regime inferred from a combination of geomorphological characteristics and the composition of littoral and aquatic vegetation (Arundel et al. 2009). The lower zone encompassed the mouth of each estuary. The upper zone was delineated by the estimated maximum upstream extent of consistent saline intrusion.

The sampling protocol aimed to maximise the detection of species from a variety of ecological guilds. The principal sampling protocol consisted of fyke and mesh netting. This was augmented with seine netting where possible. Single-wing fyke nets were replicated ($n \geq 3$) in each zone in all estuaries. Multi-panel mesh nets were replicated at the whole-of-estuary scale with $n \geq 3$ set in all estuaries. Seine netting was conducted opportunistically in the lower zone where the substrate, tidal currents and estuarine geomorphology permitted effective use of the gear. The replication of fyke nets was considered adequate for estimating species diversity (Warry and Reich 2011) but replication of other gear types was constrained by the rapid assessment approach.

Ultimately, replication of the different gears (Section 7.1.2) was a realistic maximum of the effort that a team of two people could achieve in a single day and was repeatable.

The use of electrofishing technology capable of operation in estuarine salinities for systematic estuarine fish assessment was investigated during the IEC implementation trial. Like netting, electrofishing was biased towards particular species and size-classes (see Warry et al. 2013). As netting protocols offered greater flexibility and consistency than estuarine electrofishing for rapid assessment of estuarine fish (Warry et al. 2013), a net-based sampling protocol was adopted for the IEC.

6.1.2 Sampling gears - Fish

Experimental mesh nets were used to target larger (>15 cm) mobile fishes. Nets were 1.5 m deep, 50 m long and composed of five panels, each of which was 10 m long and of a different mesh size. Each panel was sewn together to form a single continuous net. The five mesh sizes (25 mm, 38 mm, 50 mm, 63 mm and 76 mm stretch polypropylene mesh) were intended to sample fish from a variety of size classes and body shapes. Mesh nets were set parallel to the banks to minimise interference from boat traffic and tidal currents, and typically deployed on a flood tide for at least two hours.

Fyke nets were used to target small and juvenile fish and eels. Each fyke net had a single 5 m long wing and four aluminium rings with a 50 cm high D-shaped entrance ring. Nets were constructed of 5 mm knotless mesh. Fyke nets were typically set at a 45º angle to the banks. The direction of the net relative to flow and whether the wing or cod-end was positioned on the bank was haphazardly selected. Nets were typically deployed for 4 hours. However, sometimes sink times had to be less where tidal ranges determined boat access and meant that the nets would be exposed at low tide.

A small beach-seine net was used to target fish from a range of sizes, species and life-stages. This seine was 2.5 m deep and 15 m long with 2 mm knotless mesh. It was deployed in a broad arc and retrieved by two operators using a pursing technique (Jenkins and Sutherland 1997). Seining could only be done at sites where the substrate and morphology of the estuary permitted personnel to wade or manoeuvre a boat. At some sites, strong tidal currents also prohibited effective use of seine nets.

6.1.3 Taxonomic identification - Fish

Upon capture, fish were identified, counted and measured. Where possible, fish were identified in the field by researchers with experience in estuarine fish identification. Taxonomic keys from Gomon et al. (2008) were used in the field where morphological traits of specimens could be viewed clearly with the naked eye. Fish that could be confidently identified in the field were returned to the water alive, where appropriate.

Specimens that could not be identified in the field were preserved in 70% alcohol for later identification in the laboratory, often under a dissecting microscope. Multiple individuals of species for identification were retained where possible. In the laboratory, taxonomic keys from Gomon et al. (2008) were used to confirm fish identifications.

6.2 Assignment of fish species to ecological guilds

Partitioning species into ecological guilds is a common feature of multi-metric indices based on fish. These guilds include how fish use the estuary, occupy the water column or feed (Potter and Hyndes 1999, Elliott et al. 2007). The guild approach is particularly useful for comparing fish assemblages across biogeographic boundaries because it overcomes issues of spatially restricted species' distributions influencing comparisons that are based on taxonomic identities (Elliott et al. 2007). This is particularly relevant for the Victorian IEC which aims to provide state-wide benchmarking of estuarine condition (Arundel et al. 2009). Using multiple functional guilds also helps to relate the structure of estuarine fish assemblages to ecological function, and to detect where human activities have altered these functions (Harrison and Whitfield 2004).

For the IEC, each fish species was assigned to three guilds (based on their estuary use, habitat and trophic group; categories listed in Table 19) based on their ecology or that of congeneric species as described in the primary literature and an online ecological database (www.fishbase.org/ [accessed December 2015]) (Table 19). These three ecological guilds underpinned the seven metrics of the Fish sub-index of the IEC.

Table 19: Guilds into which fish were assigned based on estuary use, habitat and trophic group for the IEC

Guild	Guild category	Description of guild category
Estuary use	Diadromous	Species that migrate between marine and fresh waters – e.g. individuals live in fresh waters and migrate to marine waters to spawn or *vice versa*.
	Estuarine and freshwater	Species can complete their life cycle within estuaries, but populations can occur in estuaries and fresh waters.
	Estuarine and marine	Species can complete their life cycle within estuaries, but populations can occur in estuaries and marine waters
	Freshwater estuarine-opportunist	Species that spawn in fresh water, but individuals regularly enter estuaries in substantial numbers
	Freshwater straggler	Species that spawn in fresh water, but individuals enter the upper reaches of estuaries sporadically
	Marine estuarine-opportunist	Species that spawn in marine waters, but individuals regularly enter estuaries in substantial numbers, particularly as juveniles
	Marine straggler	Species that spawn in marine waters but enter lower reaches of estuaries sporadically
	Solely estuarine	Species that complete their life cycle within estuaries and populations occur only in estuarine waters
Habitat	Benthic	Species that live and feed on the substratum
	Demersal	Species that live in the lower portions of the water column but are associated with the substratum through feeding or structural habitat
	Pelagic	Species that live and feed in the water column
Trophic group	Detritivore	Feed predominantly on detritus and microphytobenthos
	Herbivore	Grazing predominantly on living macroalgae, macrophytes and phytoplankton
	Omnivore	Feed on a mixture of algae, macrophytes, detritus, epifauna and infauna
	Opportunist	Opportunistically feed on a diverse range of food sources including finfish, infauna, epifauna and zooplankton
	Piscivore	Feed predominantly on finfish
	Zoobenthivore	Feed on invertebrates closely associated with the substratum
	Zooplanktivore	Feed predominantly on zooplankton

6.3 Fish metric selection

There are various approaches to selecting metrics associated with fish. These approaches include using best professional judgements or expert opinions based on a conceptual understanding of the relationships between fish responses and threats (Karr 1981; Deegan et al. 1997). Quantitative analytical approaches have also been used that interrogate the relationships between individual metrics or the aggregated index and predictors of threats (Stoddard et al. 2008; Esselman et al. 2013). In other cases, a combination of professional judgements and data-driven approaches has been used (Cabral et al. 2012).

Warry and Reich (2013) used a data-driven approach to recommend a suite of guild-based metrics for inclusion in the IEC Fish sub-index based on responses of individual metrics to various threats (e.g. modified catchment land use, altered hydrology). Metrics that did not demonstrate relationships with specific threats were discarded. Re-analysis of the fish data from the samples collected during the IEC implementation trial by Warry et al. (2018) showed that some of the metrics previously recommended by Warry and Reich (2013) were associated with environmental characteristics (e.g. tidal exchange) rather than threats.

The approach used to select metrics will have a strong influence on the resultant multi-metric index (such as the Fish sub-index in the IEC). Adopting data-driven approaches that assess relationships between metrics and threats generally leads to multi-metric indices that are more strongly correlated with available data on human pressures. However, this may come at the expense of interpretability and can overlook metrics with a strong conceptual basis that may be responding to threats that have been poorly parameterised due to data limitations or the use of proxies (Zucchetta et al. 2020). For this reason, a combination of data-driven approaches, conceptual understanding of fish responses to threats, and information from the published literature was used to select metrics for the IEC Fish sub-index.

Using this combination resulted in the reintroduction of some metrics that were earlier discarded, during the IEC implementation trial. These metrics were included because it was recognised that the parameters used to characterise threats strongly influence the construction of a multi-metric index, they may not have been parameterised particularly well (e.g. using land use as a proxy for nutrient loading) due to data limitations, and they may have overlooked complex or interacting threats. The selected metrics represent a range of fish assemblage characteristics, including measures of species diversity and composition, habitat use and trophic structure, with strong conceptual links to stressors and pressures (Figure 9; Table 20).

Table 20: The seven selected metrics for the Fish sub-index, together with the direction of their predicted response to pressures and stressors, a description and conceptual rationale for each metric, and a list of some of the literature supporting their use; MMI = multi-metric index.

Metric	Description and rationale	Supporting literature
Species That Can Complete Their Life Cycle Within Estuaries – Richness [Response to threats – Decrease]	The richness of species that have populations that can complete their life cycle within estuaries (i.e. estuarine residents). An estuary in good condition should support a high richness of species that rely on a suite of ecological functions to complete their life cycles. For example, appropriate water quality, biogenic habitat, primary and secondary productivity, and hydrological connectivity are required to support spawning, recruitment, growth and survival of estuarine species. The number of species that complete their life cycles within estuaries measures a group of fishes that are probably most susceptible to estuarine degradation because of their strong dependence or continuous association with these environments (Harrison and Whitfield 2004). Restricting a measure of species richness to only those species that can complete their life cycles within estuaries helps to mitigate noise introduced to measures of absolute species richness or diversity by differences in estuary typology (e.g. the magnitude of tidal exchange, see Warry et al. 2018) and biogeography (e.g. fluctuations of the influence of the East Australian Current in East Gippsland that leads to detection of marine-associated fish species outside their typical core ranges) as well as opportunistic or accidental movements of transient species (e.g. freshwater and marine stragglers). Consequently, this metric facilitates comparisons across Victorian estuaries by better contextualising a measure of species richness within a suite of species that are reasonably expected to occur in all Victorian estuaries that are in good condition. Examples of species that can complete their life cycles within estuaries include: Black Bream (*Acanthopagrus butcheri*), Tamar River Goby (*Afurcagobius tamarensis*), Yelloweye Mullet (*Aldrichetta forsteri*), Estuary Perch (*Macquaria colonorum*)	Metrics of the richness of estuarine resident fish species are found in several MMIs including the: • Estuarine Biotic Integrity Index (EBI) – USA (Deegan et al. 1997) • Estuarine Fish Community Index (EFCI) – South Africa (Harrison and Whitfield 2004; 2006) • Estuarine Fish Community Index (EFCI) – NSW, Australia (Roper et al. 2011) • Estuarine Demersal Indicators (EDI) – Spain, used to assess fish quality within the European Water Framework Directive (Borja et al. 2004; Uriarte and Borja 2009) • Transitional Fish Classification Index (TFCI) – United Kingdom (Coates et al. 2007). • Estuarine Fish Assessment Index (EFAI) – Portugal (Cabral et al. 2012) A study of fish assemblages from 31 Victorian estuaries found that overall Shannon diversity and juvenile species richness were positively associated with tidal exchange but were not associated with measures of catchment or floodplain modification (Warry et al. 2018). This suggested that such metrics of species diversity are unlikely to be useful for state-wide benchmarking of estuary condition.

Metric	Description and rationale	Supporting literature
Introduced Species – Presence [Response to pressures and stressors – Present]	The presence of introduced species is a direct measure of anthropogenic modification. Introduced species are often invasive and compete with or predate on native fish and other fauna. Some invasive species can also alter habitat for native species. Examples of species introduced to Victorian estuaries include: Yellowfin Goby (*Acanthogobius flavimanus*), Trident Goby (*Tridentiger trigonocephalus*), Eastern Gambusia (*Gambusia holbrooki*), Barramundi (*Lates calcarifer*) Freshwater associated introduced species were removed from calculations, based on the rationale that in cases where these species were detected the upper reaches of the estuaries were likely under the influence of freshwaters and it will be more conservative to restrict comparisons among estuaries to truly estuarine invasive species. There may have been many more catchments where these species are present in freshwater reaches and have the potential to enter estuaries under favourable freshwater inflow conditions, but not detected on the day of sampling for the IEC. The freshwater introduced species that were excluded from calculation of the invasive species metric are: a. *Cyprinus carpio* – European Carp b. *Macquaria ambigua* - Golden Perch c. *Salmo trutta* – Brown Trout	Metrics of introduced species occur in several estuarine fish MMIs including the: • Estuarine Fish Community Index (EFCI) – South Africa (Harrison and Whitfield 2004; 2006) • Estuarine Demersal Indicators (EDI) – Spain, used to assess fish quality within the European Water Framework Directive (Borja et al. 2004; Uriarte and Borja 2009) • Estuarine Fish Assessment Index (EFAI) – Portugal (Cabral et al. 2012) • Estuarine Fish Community Index (EFCI) – NSW, Australia (Roper et al. 2011)

Metric	Description and rationale	Supporting literature
Demersal Species – Richness [Response to pressures and stressors – Decrease]	The richness of demersal species that live in the lower portion of the water column but feed in the benthic zone and/or use it as structural habitat. Demersal species richness can be negatively impacted by anthropogenic land use and subsequent elevated nutrient loads that can cause: (1) changes in demersal habitat structure such as loss of seagrass and dominance of macroalgae (Hauxwell et al. 2003; Woodland et al. 2015); (2) shifts in primary production supporting estuarine food webs (Nixon et al. 2001) such as greater contributions of macroalgae and phytoplankton; and/or (3) hypoxia of bottom waters (Paerl et al. 2014). Physical and chemical alterations of demersal environments by dredging, sedimentation and accumulation of toxins may have further impacts on demersal species. Juveniles tend to be more susceptible to these effects than adults because of their lower physiological tolerance to low concentrations of oxygen and their generally lower mobility which limits their capacity to escape unfavourable habitats (Breitburg et al. 2002). Examples of demersal species include: Black Bream (*Acanthopagrus butcheri*), Port Jackson Glassfish (*Ambassis jacksonianus*), Luderick (*Girella tricuspidata*), King George Whiting (*Sillaginodes punctatus*).	A study of fish assemblages from 31 Victorian estuaries found that demersal species richness was negatively associated with catchment land use modification, and that these patterns were driven by juvenile life forms (Warry et al. 2018). Metrics of the richness of species associated with the substratum occur in several estuarine fish MMIs including: • The Estuarine Biotic Integrity Index (EBI) – USA (Deegan et al. 1997) • The Estuarine Biotic Index (EBI) – Belgium (Breine et al. 2007) • A fish-based index to assess ecological quality of transitional waters – France (Delpech et al. 2010). These three metrics were predominantly focused on benthic rather than demersal species. However, the richness of benthic species was not found to be associated with catchment land use modification in Victoria (Warry et al. 2018). In Victorian estuaries, the benthic guild is dominated by species from the family Gobiidae. Many genera of gobies can tolerate hypoxia for extended periods (Congleton 1974; Takegaki and Nakazono 1999; Nilsson et al. 2004) suggesting that tolerance of eutrophic and hypoxic conditions may be an ancestral trait (Nilsson et al. 2004) and accounts for the lack of observed associations between benthic species richness and indicators of threats to estuaries.
Demersal Species – Relative Abundance [Response to pressures and stressors – very high or very low]	The relative abundance of demersal species that live in the lower portion of the water column but feed in the benthic zone and/or use it as structural habitat. The effects of anthropogenic land use and subsequent elevated nutrient loads on demersal habitats described above will reduce the abundance of demersal species relative to those of pelagic species (that do not rely on benthic or demersal habitats) or benthic species (which may be tolerant to some degree of benthic or demersal habitat degradation). The relationship between this metric and estuarine condition was assumed to be non-linear, with either very low or very high relative abundances indicating impacts on estuarine condition. A fish assemblage completely dominated by demersal species was considered to be as undesirable as one devoid of demersal species.	The abundance of demersal fish was negatively associated with the dominance of macroalgae over seagrass in a study of three small Victorian estuaries (Warry 2017). Macroalgal dominance and concomitant seagrass decline stemming from nutrient enrichment have had negative effects on fish abundance elsewhere (Deegan et al. 2002; Wyda et al. 2002). Metrics of the relative abundance of species associated with the substratum have been included in several estuarine fish MMIs (e.g. Deegan et al. 1997; Breine et al. 2007; Delpech et al. 2010). In these studies, both very high and very low relative abundances were considered to indicate impacts on estuarine condition.

Metric	Description and rationale	Supporting literature
Trophic Specialists – Richness [Response to pressures and stressors – Decrease]	The richness of species that are trophic specialists rather than generalists. Trophic specialists include herbivores, piscivores, zoobenthivores and zooplanktivores. Anthropogenic land use, hydrological modification and artificial estuary openings can prompt habitat loss, cause rapid habitat alterations and changes in water levels, and homogenise primary productivity towards domination by producers capable of rapid nutrient uptake. These impacts reduce the availability of niches for trophic specialists. A diversity of intact habitats and trophic pathways is needed to support a high richness of trophic specialists. For example, the presence of herbivores indicates that suitable submerged vegetation is present and the water quality within these habitats is suitable to support grazing. If piscivores are present, it is assumed that the productivity of smaller fish is sufficient to support larger predators and that the habitats required by these species are available (e.g. sufficient biogenic structure, geomorphological or bathymetric features). If zoobenthivores are present, it is assumed that secondary productivity is sufficient and suitable habitats are available to support benthic secondary consumers. Instead of metrics based on individual feeding guilds (e.g. piscivores or zoobenthivores), an aggregated metric of trophic specialisation captures information on the availability of multiple trophic niches and provides a more holistic picture of the trophic structure of fish assemblages.	The richness of trophic specialists has been included in: • Fish assemblage-based estuarine health indices for the Swan-Canning Estuary, Western Australia (Hallett et al. 2012)). The richness of individual trophic specialist guilds has been used in other estuarine fish MMIs, particularly the richness of piscivores (Borja et al. 2004; Harrison and Whitfield 2004, 2006; Cabral et al. 2012; Coates et al. 2007; Delpech et al. 2010). However, piscivores were only detected in 57% of estuaries sampled for the IEC, which limited the potential for a piscivore-based metric to discriminate among Victorian estuaries for state-wide benchmarking.
Trophic Specialists – Relative Abundance [Response to pressures and stressors – Decrease]	The relative abundance of species that are trophic specialists rather than generalists. Trophic specialists include herbivores, piscivores, zoobenthiviores and zooplanktivores. As described above, anthropogenic land use, hydrological modification and artificial estuary openings can prompt habitat loss, rapidly alter habitats and water levels, and lead to primary productivity dominated by producers capable of rapid nutrient uptake. All these impacts reduce the availability of niches for trophic specialists. A diversity of intact habitats and productivity pathways is needed to support a greater abundance of trophic specialists relative to trophic generalists.	The relative abundance of trophic specialists has been included in: • Fish assemblage-based estuarine health indices for the Swan-Canning Estuary, Western Australia (Hallett et al. 2012). The relative abundance of individual trophic specialist guilds has been used in other estuary fish MMIs. Although the relative abundance of piscivores is often included (Borja et al. 2004; Harrison and Whitfield 2004, 2006; Cabral et al. 2012; Coates et al. 2007; Delpech et al. 2010), piscivores were only detected in 57% of estuaries sampled for the IEC. Therefore, the potential of this metric to discriminate among Victorian estuaries for state-wide benchmarking was limited.

Metric	Description and rationale	Supporting literature
Diadromous Species – Richness [Response to pressures and stressors – Decrease]	The richness of diadromous species that migrate between marine and freshwaters. Information on diadromous species indicates the extent of connectivity in and through the estuary. Modification of hydrology in estuarine catchments can negatively impact diadromous species by altering the timing and magnitude of freshwater inflows and therefore disrupting hydrological cues for spawning and migration. Reductions in longitudinal connectivity through either instream artificial barriers (e.g. weirs) or reduced flows that disconnect upstream habitats from the estuary will interrupt migration of individuals through the estuary. Consequently, fewer diadromous species are expected in estuaries with instream barriers and/or modified freshwater inflows. Examples of diadromous species include: Shortfinned Eel (*Anguilla australis*), Longfinned Eel (*Anguilla reinhardtii*), Australian Grayling (*Prototroctes maraena*), Tupong (*Pseudaphritis urvillii*)	The richness of diadromous species has been included in: • A zone-specific index of biotic integrity (Z-EBI) – Belgium (Breine et al. 2010) • The Estuarine Fish Assessment Index (EFAI) – Portugal (Cabral et al. 2012).

6.3.1 Metric calculation - Fish

For the metric Species That Can Complete Their Life Cycle Within Estuaries - Richness, data on species in the 'solely estuarine', 'estuarine and marine' and 'estuarine and freshwater' estuary-use guilds were used. Introduced species were excluded.

The Introduced Species metric was calculated as the presence of exotic species or translocated native species.

The two metrics based on demersal species (richness and relative abundance) used data on fish in the demersal habitat guild. Introduced species were excluded.

The two trophic-specialist metrics (richness and relative abundance) used data on fish identified as trophic specialists, including the herbivore, piscivore, zoobenthivore and zooplanktivore trophic guilds. Introduced species were excluded. Species assigned to the 'marine straggler' and 'freshwater straggler' estuary-use guilds were excluded from the calculations of trophic guilds because these species do not depend on food resources within estuaries (Elliott et al. 2007).

6.3.2 Reference conditions - Fish

Several approaches to developing reference conditions exist (see Stoddard et al. 2006). As comparable reference or 'pristine' sites are rare, researchers have defined reference conditions using, for example, historical records, expert judgement, predictive models and information from sites deemed to be least impacted as determined by physical and chemical data (Harrison and Whitfield 2004; Stoddard et al. 2006). An alternative approach that was used by Harrison and Whitfield (2006) and employed for IEC is to use sample data to derive reference conditions by examining the distribution of values for each metric. The 'best available' values of candidate metrics are then used to establish reference conditions (Harris and Silveira 1999; Harrison and Whitfield 2004).

This approach is useful when prior classification of reference conditions is impeded by a lack of pristine or 'good-quality' sites, historical records or appropriate data to support predictive models (Harrison and Whitfield 2004; Stoddard et al. 2006). There are very few pristine estuaries in Victoria and only limited historical records of fish fauna that could be used to develop referential benchmarks of fish condition (e.g. Pre-European Reference Condition for fish (PERCH) lists, as developed for inland waters (Lieschke et al. 2013)). Although using 'best available' values for defining reference conditions has limitations (e.g. the potential for shifting baselines), it was considered valid for the first assessment of the IEC in Victoria, which had a primary aim of benchmarking condition across the state at the time of monitoring.

If assessment captures the range of possible ecological conditions, using a 'best available' approach to set reference conditions can help avoid circular reasoning because the approach relies on observed values of metrics rather than assumptions about which estuaries are of 'good quality' (Harrison and Whitfield 2004). The current IEC assessment included the most and least impacted estuaries in Victoria. Further, using sample data to derive reference conditions also inherently accounts for biases of the IEC sampling methods in their 'catchability' of certain fish species.

For the current assessment, reference conditions were defined as the mean of the upper quintile of metric values for most metrics based on absolute richness. The single exception was the metric Diadromous Species – Richness. The referential species lists derived for this guild were based on their historical distributions. This was feasible because few diadromous species are expected to occur in Victorian estuaries during autumn and the historical distributions of these species are relatively well understood. All but one diadromous species – Longfinned Eel (*Anguilla reinhardtii*) - were expected to have state-wide distributions. Reference conditions were not specified for metrics based on relative abundance.

6.3.3 Metric scoring - Fish

Once reference conditions were established, the deviation of each metric from its reference state was assessed based on the distribution of the metric's values. Each metric was given a score of 1, 3 or 5 according to the degree of deviation from the reference (1 = worst condition; 5 = best condition). The percentage deviation that delineated the scoring thresholds varied for each metric and was based on the distribution of the observed metric values, alignment with published thresholds and best professional judgement. This was done to maximise the potential for discrimination among estuaries once metric scores were aggregated to produce the IEC Fish sub-index value.

Metric values were divided into three scoring categories (1, 3 and 5) rather than five (1 to 5) because uncertainty exists around the threshold values, and greater resolution of scoring categories may cause estuaries with similar condition to be classified incorrectly as being different. Additional scoring categories could be added once greater conceptual understanding of metric responses to disturbance is developed. Similar scoring systems have been used successfully in other estuarine fish-based multi-metric indices (e.g. Borja et al. 2004; Harrison and Whitfield 2006; Uriarte and Borja 2009).

Scoring thresholds are outlined in Table 21. The distributions of observed values for the seven metrics are presented in Appendix B, Figure B1.

Table 21: Reference and scoring approaches, reference values and scoring thresholds for each IEC Fish metric; NA – not applicable.

Metric	Reference & scoring approach	Reference value	Scoring thresholds		
			5	3	1
Species That Can Complete Their Life Cycle Within Estuaries - Richness	Reference – mean of upper quintile Thresholds – 5 = ≥80% reference, 3 = ≤40 - <80% reference, 1 = <40% reference	13	≥10	5-9	<5
Introduced Species - Presence	Presence/Absence	Absent	Absent	NA	Present
Demersal Species – Richness	Reference – mean of upper quintile Thresholds – 5 = ≥80% reference, 3 = ≤40 - <80% reference, 1 = <40% reference	12	≥10	5-9	<5
Demersal Species – Relative Abundance	Two tailed, based on distribution of the data	NA	25 - 75%	≥10% and <25% *or* >75% and ≤90%	<10% *or* >90%
Trophic Specialists - Richness	Reference – mean of upper quintile Thresholds – 5 = ≥80% reference, 3 = ≤40 - <80% reference, 1 =<40% reference	14	≥11	6-10	<6
Trophic Specialists – Relative Abundance	Single tailed, based on distribution of the data	NA	≥75%	≥25% - 75%	<25%
Diadromous Species – Richness	Reference – expected richness based on species distributions East of, and including, Melbourne	5	≥3	1-2	0
	Reference – expected richness based on species distributions West of Melbourne	4	≥2	1	0

6.3.4 Data quality - Fish

Categories of data quality were assigned to the metric scores for the Fish sub-index. These categories were based on the sampling effort within an estuary – the extent to which the recommended sampling protocol was achieved. In some cases, it was not possible to deploy all the nets recommended by Warry and Reich (2013) due to difficulties associated with access, depth or tidal currents. The data quality categories are described in Table 22.

Table 22: Data quality categories for the seven metrics of the Fish sub-index of the IEC.

Data quality	Sampling effort
Very high	Recommended sampling protocol of n = 9 fyke nets and n = 3 mesh nets achieved
High	n ≥ 8 fyke nets, n = 2 mesh nets, seine nets deployed
Moderate	n < 8 fyke nets, n = 1 - 2 mesh nets, seine nets deployed *or* n = 9 fyke nets, either mesh or seine nets deployed
Low	Only fyke nets deployed

6.4 IEC Fish sub-index score

Once thresholds were defined and metrics scored, the IEC Fish sub-index was calculated by summing the scores of the seven metrics. This approach assumes that each metric has an equal contribution to ecological condition. Most estuarine fish-based multi-metric indices adopt an unweighted summation of metric scores to calculate ecological condition (e.g. Harrison and Whitfield 2006; Breine et al. 2007; Uriarte and Borja 2009). Unless there is a clear rationale for rating particular metrics as more important than others, weighting can introduce bias and uncertainty.

The values of the summed (raw) IEC Fish sub-index can range from 7 to 35. To ease interpretation, these values were converted to a scale of 1 to 10 using the following formula:

$$Fish\ sub-index\ score = \frac{(Observed\ raw\ IEC\ Fish\ subindex\ value - 7) \times 9}{28} + 1$$

6.5 Notes on interpreting Fish results

- Results represent a single 'snapshot' in time for the purposes of state-wide spatial benchmarking. Estuaries are characteristically dynamic systems that are subject to large natural cycles of physical and chemical conditions, including variability in freshwater discharge and associated impacts on salinity and temperature profiles. This variability can prompt frequent temporal shifts in distributions of fish taxa into or out of estuarine environments. More targeted longitudinal sampling over time and better complementary data at the site scale are needed to capture and interpret temporal dynamics in estuarine fish fauna that may be caused by natural cycles of variability rather than shifts in threats and associated impacts.
- Estuaries are characteristically dynamic environments typically exposed to high anthropogenic disturbance due to their location at the bottom of the catchment (Pérez-Domínguez et al. 2012). Therefore, distinguishing natural environmental variability from anthropogenic disturbance can be challenging (Elliott and Quintino 2007). Disturbance gradients among estuaries may co-vary with gradients of natural variability, such as estuary geomorphology, tidal regime or hydrological characteristics.
- Estuaries spanning a range of natural characteristics were scored in the top 20% of the Fish sub-index (DELWP 2021). These included estuaries that are permanently open, intermittently open, flow into embayments or the open sea, and were small (<2 km long) or large (>5 km long). This diversity of

estuaries in the top 20% suggests that the multi-metric Fish sub-index of the ISC is not systematically biased to particular types of estuaries and is suitable for state-wide condition assessment, despite the effects of other natural gradients (e.g. estuary type, size, hydrology).

- Although the Fish sub-index used in the IEC is useful for benchmarking estuarine ecological condition at the Victoria-wide scale, it is like other estuarine fish-based multi-metric indices that lack information about the mechanisms that lead to good or poor condition (Harrison and Whitfield 2006). These multi-metric index approaches will be of most use when they are complemented with other measures of environmental quality (e.g. habitat and water quality data) and targeted approaches to test hypotheses about the mechanisms underpinning estuarine condition.

7. IEC Score Calculation

7.1 IEC Scoring

The sub-indices of the IEC were combined to provide an overall IEC score by using the Inverse Ranking Transformation approach. This approach was also used for the Index of Stream Condition (DEPI 2014). The transformation is valuable because it recognises that a particularly low score for one sub-index is likely to involve one or more pressures and stressors that constrain an estuary's true ecological condition even if the other sub-indices score highly. In such cases, the inverse ranking transformation results in a lowered IEC score.

Ideally, all five sub-indices must have a score to use this approach for calculating the overall IEC. For a number of estuaries, this was not possible as not all sub-indices were assessed. Where one or two sub-indices lacked a score, these were estimated based on the means of the existing sub-index scores. However, if an estuary had fewer than three sub-indices with a score, it was not possible to calculate the overall IEC score.

To calculate the inverse ranking score, the five sub-index scores were placed in ascending order. The lowest score was multiplied by 5, the next lowest score by 4, and so on until reaching the highest score which was multiplied by 1. The totals were then summed to produce a score out of 150. The final step was to divide this score by 3 which yielded a final IEC score out of 50 (ranging from 5 to 50). If necessary, this score was rounded to the nearest whole number. An example of the application of the inverse ranking transformation to calculating an overall IEC score is provided in Table 23.

Table 23: Example of the application of the inverse ranking transformation to calculating an overall IEC score. Scoring follows DEPI (2013b).

Sub-index score (out of 10) ranked lowest to highest among the five sub-index scores for a given estuary	Multiplied by	New Score
6	5	30
7	4	28
8	3	24
8	2	16
10	1	10
Total (score out of 150)		**108**
Divide by 3 (to provide score out of 50)		**36**

7.2 IEC Condition classes

Once the inverse ranking transformation was applied and an overall IEC score out of 50 calculated, estuaries were assigned to one of five condition classes (

Table 24). The condition class is used to communicate the overall condition of the estuary. Although this condition class is useful to convey an overview of estuary condition, the scores of the sub-indices and their underlying metrics capture key information and should also be examined.

Table 24: The thresholds of the overall IEC scores and their corresponding condition classes for the IEC.

Overall IEC score	IEC condition class
41 – 50	Excellent
34 – 40	Good
27 – 33	Moderate
20 – 26	Poor
5 – 19	Very Poor

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Appendix A: Artificial Shoreline Supplementary Information

Different buffer widths could have been used in the calculation of the Artificial Shorelines metric: the proportion of built structures within the estuary perimeter buffer, and it is possible that results may vary based on the particular width chosen. To explore the degree to which this was an issue, two different buffer widths (10 m and 20 m) were applied and the results were found to be highly correlated (r^2 = 0.97; Figure A1). As the discrepancies in metric values were marginal between calculations applying a 10 m versus a 20 m buffer, the narrower 10 m buffer was used for this metric in the IEC.

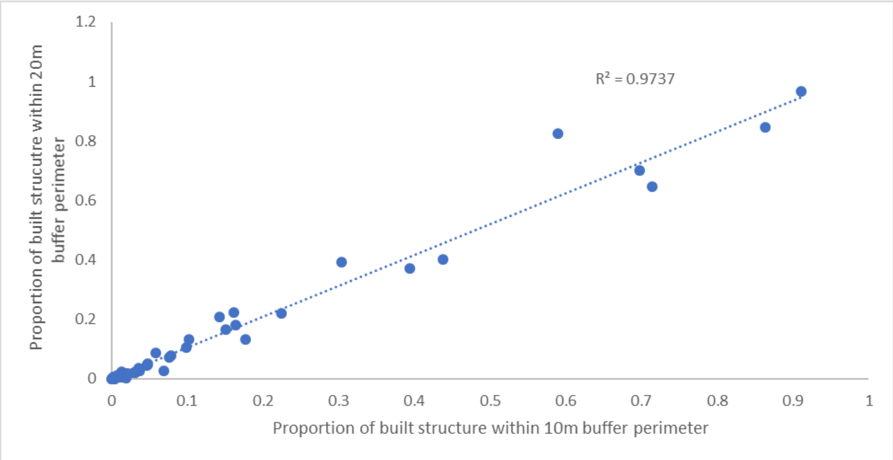

Figure A1: Relationship between the proportion of the perimeter bounded by built structures within a 20 m wide buffer perimeter versus those within a 10 m wide buffer perimeter.

Appendix B: Fish Supplementary Information

Table B1: Estuaries sampled for fish for the IEC, including the year that data were collected, gear replication and a data-quality rating (details in Section 7.7) to indicate instances where the full recommended sampling protocol was not implemented due to, for example, strong tidal currents, depth or geomorphology limiting effective gear deployment; FYN = single-winged fyke net, MSH = multi-panel experimental mesh net, SNE = seine net.

Estuary	Year sampled	FYN	MSH	SNE	Whether the core protocol (n=9 FYN, n=3 MSH) was met	Data-quality rating
Agnes River	2017	9	1		N	MODERATE
Aire River	2011	9	2	3	N	HIGH
Albert River	2018	9	3	3	Y	VERY HIGH
Anglesea River	2019	9	3	2	Y	VERY HIGH
Avon River	2012	9	3		Y	VERY HIGH
Balcombe Creek	2012	9	3	3	Y	VERY HIGH
Barham River	2017	9	2	2	N	HIGH
Barwon River	2018	8	3	3	N	HIGH
Bass River	2012	9	3	3	Y	VERY HIGH
Bennison River	2012	9	3		Y	VERY HIGH
Betka River	2018	9	3	3	Y	VERY HIGH
Bourne Creek	2017	9		2	N	MODERATE
Bruthen Creek	2019	6	1	3	N	MODERATE
Bunga Inlet	2012	9	3	3	Y	VERY HIGH
Bunyip River	2012	9	3		Y	VERY HIGH
Campbell Creek	2019	9	3	2	Y	VERY HIGH
Cann River - Tamboon Inlet	2018	9	3	5	Y	VERY HIGH
Cardinia Creek	2012	9	3		Y	VERY HIGH
Chinamans Creek	2012	9	3		Y	VERY HIGH
Curdies Inlet	2011	9	3	2	Y	VERY HIGH
Darby River	2018	9	3	3	Y	VERY HIGH
Davis Creek	2012	9	3	3	Y	VERY HIGH
Deep Creek	2014	9	2		N	MODERATE
Elwood Canal	2014	9		3	N	MODERATE
Eumerella River	2011	9	3	2	Y	VERY HIGH
Fitzroy River	2011	9	3	2	Y	VERY HIGH
Franklin River	2012	9	3		Y	VERY HIGH
Gellibrand River	2011	9	3	2	Y	VERY HIGH
Glenelg River	2018	9	3	4	Y	VERY HIGH
Hopkins River	2011	9	2	2	N	HIGH
Hovells Creek	2018	9		3	N	MODERATE
Johanna Creek	2019	6	2	2	N	MODERATE
Kananook Creek	2014	9		1	N	MODERATE

Estuary	Year sampled	FYN	MSH	SNE	Whether the core protocol (n=9 FYN, n=3 MSH) was met	Data-quality rating
Kennett River	2011	9	3	2	Y	VERY HIGH
Kings Creek	2014	9	2		N	MODERATE
Kororoit Creek	2012	9	3	2	Y	VERY HIGH
Lake Tyers	2018	10	3	3	Y	VERY HIGH
Lake Wellington Main Drain	2018	9			N	LOW
Lang Lang River	2014	9	3		Y	VERY HIGH
Latrobe River	2018	9	3		Y	VERY HIGH
Laverton Creek	2014	9		2	N	MODERATE
Little River	2012	9	3	2	Y	VERY HIGH
Mallacoota Inlet	2018	9	3	3	Y	VERY HIGH
Maringa Creek	2017	9		1	N	MODERATE
Merri River	2017	9	2	2	N	HIGH
Merri River The Cut	2017	3	1	3	N	LOW
Merricks Creek	2012	9	1	3	N	MODERATE
Merriman River	2012	9	3	3	Y	VERY HIGH
Mississippi Creek	2019	9	3	2	Y	VERY HIGH
Mitchell River	2017	9	3	2	Y	VERY HIGH
Mordialloc Creek	2014	9	2		N	MODERATE
Moyne River	2018	9	2		N	MODERATE
Mueller River	2019	9	3	1	Y	VERY HIGH
Neils Creek	2019	6		1	N	MODERATE
Newlands Arm	2018	9	3	3	Y	VERY HIGH
Nicholson River	2017	9	3	2	Y	VERY HIGH
Nine Mile Creek	2018	9		3	N	MODERATE
Old Hat Creek	2018	9	2	3	N	HIGH
Olivers Creek	2014	9	3		Y	VERY HIGH
Patterson River	2014	9	3		Y	VERY HIGH
Powlett River	2017	9	3	2	Y	VERY HIGH
Screw Creek	2019	9	1		N	MODERATE
Shady Creek	2018	9	1	3	N	MODERATE
Shallow Inlet	2018	9	2	4	N	HIGH
Sherbrook River	2017	9	3	2	Y	VERY HIGH
Shipwreck Creek	2012	9	3	2	Y	VERY HIGH
Skeleton Creek	2014	9	3		Y	VERY HIGH
Slaughterhouse Creek	2017	9	3	2	Y	VERY HIGH
Snowy River	2018	9	3	3	Y	VERY HIGH
Spring Creek	2011	9	3	2	Y	VERY HIGH
St Georges River	2018	9	1	3	N	MODERATE
Stockyard Creek	2018	9	2	2	N	HIGH
Surrey River	2018	9	3	2	Y	VERY HIGH

Estuary	Year sampled	FYN	MSH	SNE	Whether the core protocol (n=9 FYN, n=3 MSH) was met	Data-quality rating
Sydenham Inlet	2017	9	3	3	Y	VERY HIGH
Tambo River	2019	9	3	2	Y	VERY HIGH
Tarra River	2012	9	3	2	Y	VERY HIGH
Tarwin River	2019	9	2	1	N	HIGH
Thompson River	2011	9	1	2	N	MODERATE
Thurra River	2019	6	2	2	N	MODERATE
Tidal River	2018	9	2	3	N	HIGH
Tom Creek	2018	9	3		Y	VERY HIGH
Tom Roberts Creek	2017	7	1	1	N	MODERATE
Tooradin Inlet	2014	9	3		Y	VERY HIGH
Warringine Creek	2012	9	3		Y	VERY HIGH
Watsons Creek	2012	9	3		Y	VERY HIGH
Werribee River	2012	9	3	2	Y	VERY HIGH
Wingan Inlet	2012	9	3	3	Y	VERY HIGH
Wreck Creek	2017	9			N	LOW
Yallock Drain	2014	9	3		Y	VERY HIGH
Yarra River	2012	9	6	3	Y	VERY HIGH
Yeerung River	2018	9	2	3	N	HIGH

Table B2: Fish taxa (scientific and common names, Family) detected during IEC fish sampling, along with their guilds of estuary use, habitat and trophic group (see Table 19); whether the species are trophic specialists or generalists; and whether they are native or introduced species. Translocated native species were classified as introduced.

Species name	Common name	Family	Estuary use guild	Habitat guild	Trophic group guild	Specialist v Generalist	Introduced v Native
Acanthaluteres spilomelanurus	Bridled Leatherjacket	Monacanthidae	Estuarine and marine	Demersal	Omnivore	Generalist	Native
Acanthogobius flavimanus	Yellowfin Goby	Gobiidae	Amphidromous	Benthic	Zoobenthivore	Specialist	Introduced
Acanthopagrus australis	Yellowfin Bream	Sparidae	Solely estuarine	Demersal	Opportunist	Generalist	Native
Acanthopagrus butcheri	Black Bream	Sparidae	Solely estuarine	Demersal	Opportunist	Generalist	Native
Acentrogobius pflaumii	Striped Sandgoby	Gobiidae	Marine estuarine-opportunist	Benthic	Zoobenthivore	Specialist	Introduced
Achoerodus viridis	Eastern Blue Groper	Labridae	Marine straggler	Demersal	Zoobenthivore	Specialist	Native
Afurcagobius tamarensis	Tamar River Goby	Gobiidae	Solely estuarine	Benthic	Zoobenthivore	Specialist	Native
Aldrichetta forsteri	Yelloweye Mullet	Mugilidae	Estuarine and marine	Demersal	Omnivore	Generalist	Native
Ambassis jacksoniensis	Port Jackson Glassfish	Ambassidae	Estuarine and marine	Demersal	Zooplanktivore	Specialist	Native
Ammotretis rostratus	Longsnout Flounder	Pleuronectidae	Marine estuarine-opportunist	Benthic	Zoobenthivore	Specialist	Native
Anguilla australis	Shortfinned Eel	Anguillidae	Catadromous	Benthic	Opportunist	Generalist	Native
Anguilla reinhardtii	Longfinned Eel	Anguillidae	Catadromous	Benthic	Opportunist	Generalist	Native
Aplodactylus arctidens	Marble Fish	Aplodactylidae	Marine straggler	Benthic	Herbivore	Specialist	Native
Arenigobius bifrenatus	Bridled Goby	Gobiidae	Estuarine and marine	Benthic	Zoobenthivore	Specialist	Native
Arenigobius frenatus	Half Bridled Goby	Gobiidae	Estuarine and marine	Benthic	Zoobenthivore	Specialist	Native
Argyrosomus japonicus	Mulloway	Sciaenidae	Marine estuarine-opportunist	Demersal	Piscivore	Specialist	Native
Arripis georgianus	Tommy Ruff	Arripidae	Marine estuarine-opportunist	Pelagic	Piscivore	Specialist	Native
Arripis trutta	Eastern Australian Salmon	Arripidae	Marine estuarine-opportunist	Pelagic	Piscivore	Specialist	Native
Arripis truttaceus	Western Australian Salmon	Arripidae	Marine estuarine-opportunist	Pelagic	Piscivore	Specialist	Native
Atherinosoma microstoma	Smallmouth Hardyhead	Atherinidae	Estuarine and marine	Pelagic	Zooplanktivore	Specialist	Native
Brachaluteres jacksonianus	Southern Pygmy Leatherjacket	Monacanthidae	Marine straggler	Demersal	Omnivore	Generalist	Native
Brachirus nigra	Black Sole	Soleidae	Marine estuarine-opportunist	Benthic	Opportunist	Generalist	Native
Callorhinchus milii	Elephantfish	Callorhinchidae	Marine straggler	Demersal	Zoobenthivore	Specialist	Native
Centropogon australis	Eastern Fortescue	Tetrarogidae	Estuarine and marine	Demersal	Opportunist	Generalist	Native
Chrysophrys auratus	Snapper	Sparidae	Marine estuarine-opportunist	Demersal	Opportunist	Generalist	Native

Species name	Common name	Family	Estuary use guild	Habitat guild	Trophic group guild	Specialist v Generalist	Introduced v Native
Contusus brevicaudus	Prickly Toadfish	Tetraodontidae	Marine straggler	Demersal	Opportunist	Generalist	Native
Cristiceps australis	Southern Crested Weedfish	Clinidae	Marine straggler	Benthic	Zoobenthivore	Specialist	Native
Cyprinus carpio	European Carp	Cyprinidae	Freshwater straggler	Demersal	Opportunist	Generalist	Introduced
Dactylophora nigricans	Dusky Morwong	Cheilodactylidae	Marine straggler	Demersal	Opportunist	Generalist	Native
Dotalabrus aurantiacus	Castelnau's Wrasse	Labridae	Marine straggler	Demersal	Zoobenthivore	Specialist	Native
Engraulis australis	Australian Anchovy	Engraulidae	Marine estuarine-opportunist	Pelagic	Zooplanktivore	Specialist	Native
Enoplosus armatus	Old Wife	Enoplosidae	Marine straggler	Demersal	Zoobenthivore	Specialist	Native
Favonigobius lateralis	Southern Longfin Goby	Gobiidae	Estuarine and marine	Benthic	Zoobenthivore	Specialist	Native
Favonigobius lentiginosus	Eastern Longfin Goby	Gobiidae	Estuarine and marine	Benthic	Zoobenthivore	Specialist	Native
Galaxias maculatus	Common Galaxias	Galaxiidae	Catadromous	Demersal	Zooplanktivore	Specialist	Native
Galaxias truttaceus	Spotted Galaxias	Galaxiidae	Amphidromous	Demersal	Zooplanktivore	Specialist	Native
Gambusia holbrooki	Eastern Gambusia	Poeciliidae	Freshwater estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Introduced
Gerres subfasciatus	Common Silverbiddy	Gerreidae	Marine estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Native
Girella tricuspidata	Luderick	Kyphosidae	Marine estuarine-opportunist	Demersal	Herbivore	Specialist	Native
Gobiopterus semivestitus	Glass Goby	Gobiidae	Solely estuarine	Demersal	Zooplanktivore	Specialist	Native
Gymnapistes marmoratus	Cobbler	Tetrarogidae	Estuarine and marine	Demersal	Zoobenthivore	Specialist	Native
Haletta semifasciata	Blue Weed Whiting	Odacidae	Marine straggler	Demersal	Omnivore	Generalist	Native
Heteroclinus adelaidae	Adelaide Weedfish	Clinidae	Estuarine and marine	Pelagic	Zoobenthivore	Specialist	Native
Heteroclinus wilsoni	Wilson's Weedfish	Clinidae	Marine straggler	Pelagic	Zoobenthivore	Specialist	Native
Hippocampus abdominalis	Big-belly Seahorse	Syngnathidae	Marine straggler	Demersal	Zooplanktivore	Specialist	Native
Hippocampus bleekeri	Pot-belly Seahorse	Syngnathidae	Marine straggler	Demersal	Zooplanktivore	Specialist	Native
Hyperlophus vittatus	Sandy Sprat	Clupeidae	Marine estuarine-opportunist	Pelagic	Zooplanktivore	Specialist	Native
Hyporhamphus melanochir	Southern Sea Garfish	Hemiramphidae	Estuarine and marine	Pelagic	Herbivore	Specialist	Native
Hyporhamphus regularis	River Garfish	Hemiramphidae	Solely estuarine	Pelagic	Herbivore	Specialist	Native
Kestratherina esox	Pikehead Hardyhead	Atherinidae	Marine estuarine-opportunist	Pelagic	Piscivore	Specialist	Native
Lates calcarifer	Barramundi	Latidae	Catadromous	Demersal	Piscivore	Specialist	Introduced
Leptatherina presbyteroides	Silver Fish	Atherinidae	Marine straggler	Pelagic	Zooplanktivore	Specialist	Native
Liza argentea	Goldspot Mullet	Mugilidae	Estuarine and marine	Demersal	Omnivore	Generalist	Native

Species name	Common name	Family	Estuary use guild	Habitat guild	Trophic group guild	Specialist v Generalist	Introduced v Native
Macquaria ambigua	Golden Perch	Percichthyidae	Freshwater straggler	Demersal	Piscivore	Specialist	Introduced
Macquaria colonorum	Estuary Perch	Percichthyidae	Estuarine and freshwater	Demersal	Opportunist	Generalist	Native
Meuschenia freycineti	Sixspine Leatherjacket	Monacanthidae	Marine estuarine-opportunist	Demersal	Omnivore	Generalist	Native
Meuschenia trachylepis	Yellowfin Leatherjacket	Monacanthidae	Marine straggler	Demersal	Omnivore	Generalist	Native
Meuschenia venusta	Stars and Stripes Leatherjacket	Monacanthidae	Marine straggler	Demersal	Omnivore	Generalist	Native
Monacanthidae	Leatherjacket sp.	Monacanthidae	Marine straggler	Demersal	Omnivore	Generalist	Native
Mugil cephalus	Sea Mullet	Mugilidae	Marine estuarine-opportunist	Demersal	Detritivore	Detritivore	Native
Mugilogobius platynotus	Pale Mangrove Goby	Gobiidae	Estuarine and marine	Benthic	Zoobenthivore	Specialist	Native
Mustelus antarcticus	Gummy Shark	Triakidae	Marine straggler	Demersal	Piscivore	Specialist	Native
Myxus elongatus	Sand Mullet	Mugilidae	Catadromous	Demersal	Omnivore	Generalist	Native
Nannoperca australis	Southern Pygmy Perch	Percichthyidae	Freshwater straggler	Demersal	Zooplanktivore	Specialist	Native
Nannoperca obscura	Yarra Pygmy Perch	Percichthyidae	Freshwater straggler	Demersal	Zooplanktivore	Specialist	Native
Neoodax balteatus	Little Weed Whiting	Odacidae	Marine straggler	Demersal	Omnivore	Generalist	Native
Nesogobius maccullochi	Girdled Goby	Gobiidae	Marine straggler	Benthic	Zoobenthivore	Specialist	Native
Omobranchus anolius	Oyster Blenny	Blenniidae	Marine estuarine-opportunist	Benthic	Zoobenthivore	Specialist	Native
Philypnodon grandiceps	Flathead Gudgeon	Eleotridae	Estuarine and freshwater	Benthic	Zoobenthivore	Specialist	Native
Philypnodon macrostomus	Dwarf Flathead Gudgeon	Eleotridae	Estuarine and freshwater	Benthic	Zoobenthivore	Specialist	Native
Platycephalus bassensis	Southern Sand Flathead	Platycephalidae	Marine estuarine-opportunist	Benthic	Piscivore	Specialist	Native
Platycephalus fuscus	Dusky Flathead	Platycephalidae	Estuarine and marine	Benthic	Piscivore	Specialist	Native
Platycephalus laevigatus	Rock Flathead	Platycephalidae	Marine straggler	Benthic	Piscivore	Specialist	Native
Platycephalus speculator	Yank Flathead	Platycephalidae	Marine estuarine-opportunist	Benthic	Piscivore	Specialist	Native
Pomatomus saltatrix	Tailor	Pomatomidae	Marine estuarine-opportunist	Pelagic	Piscivore	Specialist	Native
Prototroctes maraena	Australian Grayling	Retropinnidae	Anadromous	Demersal	Omnivore	Generalist	Native
Pseudaphritis urvillii	Tupong	Pseudaphritidae	Catadromous	Demersal	Zoobenthivore	Specialist	Native
Pseudocaranx georgianus	Silver Trevally	Carangidae	Marine estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Native
Pseudogobius sp.	Eastern Bluespot Goby	Gobiidae	Solely estuarine	Benthic	Omnivore	Generalist	Native
Pseudorhombus jenynsii	Smalltooth Flounder	Paralichthyidae	Estuarine and marine	Benthic	Zoobenthivore	Specialist	Native
Pugnaso curtirostris	Pugnose pipefish	Syngnathidae	Marine straggler	Demersal	Zooplanktivore	Specialist	Native

Species name	Common name	Family	Estuary use guild	Habitat guild	Trophic group guild	Specialist v Generalist	Introduced v Native
Redigobius macrostoma	Largemouth Goby	Gobiidae	Amphidromous	Benthic	Zoobenthivore	Specialist	Native
Retropinna semoni	Australian Smelt	Retropinnidae	Freshwater estuarine-opportunist	Pelagic	Zooplanktivore	Specialist	Native
Rhombosolea tapirina	Greenback Flounder	Pleuronectidae	Marine estuarine-opportunist	Benthic	Zoobenthivore	Specialist	Native
Salmo trutta	Brown Trout	Salmonidae	Anadromous	Pelagic	Opportunist	Generalist	Introduced
Sardinops sagax	Australian Pilchard	Clupeidae	Marine straggler	Pelagic	Zooplanktivore	Specialist	Native
Scobinichthys granulatus	Rough Leatherjacket	Monacanthidae	Marine straggler	Demersal	Omnivore	Generalist	Native
Scorpis aequipinnis	Sea Sweep	Scorpididae	Marine straggler	Demersal	Zooplanktivore	Specialist	Native
Sillaginodes punctatus	King George Whiting	Sillaginidae	Marine estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Native
Sillago ciliata	Sand Whiting	Sillaginidae	Marine estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Native
Sillago flindersi	Eastern School Whiting	Sillaginidae	Marine estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Native
Siphamia cephalotes	Wood's Siphonfish	Apogonidae	Marine straggler	Benthic	Zoobenthivore	Specialist	Native
Sphyraena novaehollandiae	Snook	Sphyraenidae	Marine straggler	Pelagic	Piscivore	Specialist	Native
Spratelloides robustus	Blue Sprat	Clupeidae	Marine estuarine-opportunist	Pelagic	Zooplanktivore	Specialist	Native
Stigmatopora argus	Spotted Pipefish	Syngnathidae	Estuarine and marine	Demersal	Zooplanktivore	Specialist	Native
Stigmatopora nigra	Widebody Pipefish	Syngnathidae	Estuarine and marine	Demersal	Zooplanktivore	Specialist	Native
Syngnathidae	Pipefish sp.	Syngnathidae	Estuarine and marine	Demersal	Zooplanktivore	Specialist	Native
Tasmanogobius lasti	Lagoon Goby	Gobiidae	Estuarine and marine	Benthic	Zoobenthivore	Specialist	Native
Tetractenos glaber	Smooth Toadfish	Tetraodontidae	Estuarine and marine	Demersal	Opportunist	Generalist	Native
Tetraodontidae	Toadfish sp.	Tetraodontidae	Estuarine and marine	Demersal	Opportunist	Generalist	Native
Thalasseleotris adela	Cryptic Sea Gudgeon	Thalasseleotrididae	Marine estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Native
Tilodon sexfasciatus	Moonlighter	Microcanthidae	Marine estuarine-opportunist	Demersal	Zoobenthivore	Specialist	Native
Tridentiger trigonocephalus	Trident Goby	Gobiidae	Marine estuarine-opportunist	Benthic	Zoobenthivore	Specialist	Introduced
Upeneichthys vlamingii	Bluespotted Goatfish	Mullidae	Marine straggler	Demersal	Zoobenthivore	Specialist	Native
Vanacampus phillipi	Port Phillip Pipefish	Syngnathidae	Marine straggler	Demersal	Zoobenthivore	Specialist	Native

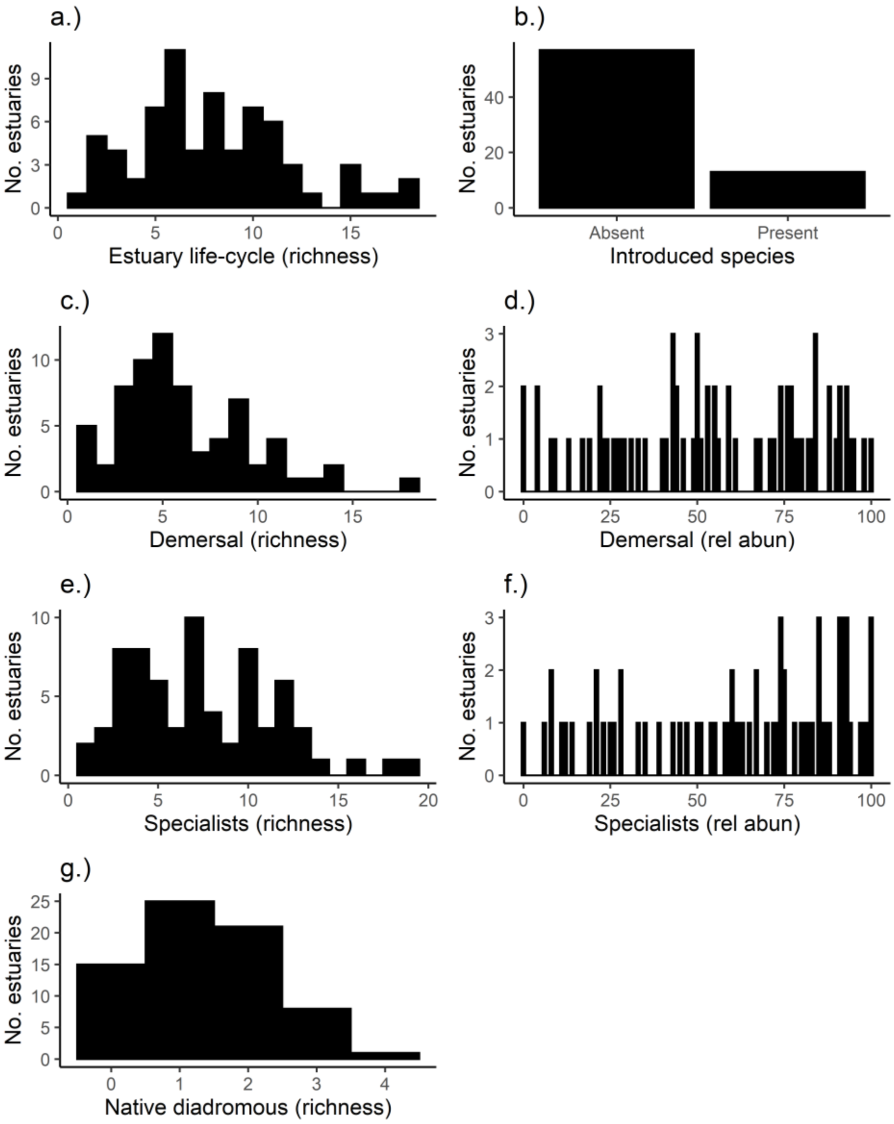

Figure B1: The distributions of observed values for the seven metrics used to calculate the Fish sub-index in the Victorian IEC: a.) richness of species that can complete their life cycle within estuaries, b.) presence-absence of introduced estuarine species, c.) richness of demersal species, d.) relative abundance (rel abun) of demersal species, e.) richness of trophic specialist species, f.) relative abundance (rel abun) of trophic specialist species, and g.) richness of diadromous species.