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Development and validation of fish-based, multimetric indices for assessing the ecological health of Western Australian estuaries

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Abstract

We describe the development of the first fish-based, multimetric indices for assessing and monitoring the health of Australian estuaries, and their application to the nearshore (< 2 m depth) and offshore (> 2 m depth) waters of the Swan Estuary, Western Australia. Suites of fish community metrics, including measures of species composition, diversity and abundance, trophic structure and life history function, were selected via a novel weight of evidence approach on the basis of their sensitivity to detect inter-annual change in estuarine condition. For each selected metric, seasonally-adjusted reference conditions were established for each spatial management zone of the Swan Estuary using 30 years of standardised historical fish assemblage data. This extensive data set provided a sound basis for determining the 'best available' standard of biotic integrity recorded over that time period and thus a reliable benchmark against which the current and future health of the estuary may be assessed and compared. The nearshore and offshore indices were robust to the effects of natural, intra-seasonal variability in environmental conditions, and so provide reliable tools for quantifying and classifying the ecological health of the Swan Estuary and its constituent management zones. The response of the nearshore index to an algal bloom confirmed that it is sufficiently sensitive to quantify ecological health responses to local-scale environmental perturbations and to track the subsequent recovery of the system following their removal. The indices provide managers with a reliable, quantitative method for assessing and communicating the health of the Swan Estuary and, similarly, of other estuaries across south-western Australia.

Keywords: Algal bloom, fish, health, indicator, metric, reference condition.

Regional index terms: Australia, Western Australia, Perth, Swan Estuary.

Running head: Fish-based indices of Australian estuary health.

1. Introduction

In response to increasing anthropogenic degradation of aquatic environments throughout the world, international accords and national legislation have progressively focused on increased environmental reporting and accountability in the management of these ecosystems. Requirements for the monitoring and management of estuaries and other waters have become a foundation of environmental policy in the United States, South Africa and Europe (Karr, 1991; DWAF, 1998; European Community, 2000; Ferreira et al., 2007). For example, the European Union’s Water Framework Directive (WFD) stipulates the use of biological indicators to assess the ecological status of rivers, lakes and transitional waters including estuaries (Ferreira et al., 2007).

In contrast, little has been done throughout Australia since Norris and Norris (1995) highlighted the dearth of schemes employing biological indicators to assess the integrity of its aquatic systems. Indeed, a global review by Borja et al. (2008) drew attention to the alarming lack of direction and consistency among Australian approaches to ecological health assessment, compounded by confusion over state and federal responsibilities. Fish- and macroinvertebrate-based indices are employed to assess the health of Australian rivers (Kennard et al., 2006b; EHMP, 2007), yet relatively few biotic indicators have been developed for assessing the condition of its estuaries (Deeley and Paling, 1998; Scheltinga and Moss, 2007). There is thus a clear and recognised need to develop integrated health assessment schemes for Australian estuaries, embracing indicators of pressures/stressors and of the condition of estuarine biota including fishes (Moss et al., 2006). This need is particularly evident in the state of Western Australia (WA).

Estuaries in south-western Australia are increasingly subject to anthropogenic pressures, with several of these systems being extensively modified by human activities and only one, Broke Inlet, having been assessed as near-pristine during broad-scale national assessments of estuarine status (NLWRA, 2002; 2008). Many of the stressors affecting these estuaries are exemplified in arguably the most intensively impacted and best-studied estuary of south-western Australia, the permanently open Swan-Canning Estuary (hereafter referred to as the Swan Estuary; Fig. 1). This extensively modified system (NLWRA, 2002; 2008) is displaying signs of a general decline in ecosystem health, particularly in its upper reaches (Swan River Trust, 1999; 2003), and therefore represents an ideal example through which to illustrate the development of ecosystem health monitoring tools to facilitate the management of Western Australian estuaries.

The Swan Estuary is approximately 50 km long and covers a surface area of ca 55 km^2, with a catchment extending to 121 000 km^2 (Swan River Trust, 2000). Highly seasonal flows in the two main tributaries of this microtidal estuary, the Swan and Canning rivers, reflect the pronounced seasonality of rainfall in this region. Extensive land clearance within its catchment for agricultural and urban development has greatly increased the magnitude of stressors acting upon the Swan Estuary since European settlement during the early to mid-1800s. These stressors include increased delivery of sediments and nutrients to estuarine waters, leading to persistent eutrophication (Hamilton and Turner, 2001; Swan River Trust, 2009). Mounting salinisation and declining freshwater flows have also extended the spatial and temporal persistence of vertical stratification and hypoxic conditions within the upper estuary (Hamilton et al., 2001; Swan River Trust, 2009). In response to these anthropogenic stressors, the Swan Estuary regularly suffers from periods of severe anoxia (Douglas et al., 1997) and phytoplankton blooms, including those of toxic species (Hosja and Deeley, 1994). The most visible consequences for biota of this environmental decline are the large fish mortality events that have occurred regularly in this system during recent decades (Valesini et al., 2005, unpublished report).

Despite these problems, resource managers of estuaries in WA currently lack a reliable, rapid and affordable method for quantifying the ecological health of estuaries relative to appropriate reference conditions, monitoring temporal changes in estuarine health to detect deterioration beyond critical thresholds, and identifying those zones of individual estuaries at greatest risk of environmental decline. We aimed to address these needs by developing an approach for constructing fish-based, multimetric indices to assess the ecological health of estuaries in south-western Australia. This approach is exemplified via its application to the extensively-modified Swan Estuary. The sensitivity and reliability of the resultant indices were also evaluated.

2. Material and methods

2.1 Outline of index development and validation

Multimetric biotic indices are generally developed via a common process, the main stages of which are outlined in Fig. 2 (after Simon, 2000). The following subsections describe the approaches employed during the development of fish-based indices for the Swan Estuary, noting that full details of several key stages are published elsewhere (Hallett et al., In press; Hallett and Hall, submitted).

2.2 Sampling of fish communities and collation of data

We utilised fish species abundance data collected both during the current study (2007-2011) and that collected during various other periods since 1976 by different researchers from the Centre for Fish, Fisheries and Aquatic Ecosystem Research at Murdoch University (Table 1). Detailed descriptions of the sampling regimes and methods used in each of the historical studies can be found in the published accounts of the individual studies (see Table 1 for references), but they are briefly summarised below.

Seine nets of different lengths, depths and mesh sizes were employed in the collection of fish from the nearshore, shallow waters (< 2 m depth) throughout the Swan Estuary during the various studies listed in Table 1. Between 1976 and 1982, nearshore fish communities were mostly sampled using either 102.5 m- or 133 m-long seine nets, both of which fished to a maximum depth of 2 m, consisted of 25.4 and 15.9 mm stretched mesh in the wings and pocket, respectively, and swept semi-circular areas of 1,670 m^2 and 2,815 m^2, respectively. However, due to narrowness of the river channel, only half of the latter net was deployed at selected sites throughout the Swan Estuary during this time, thus reducing the area swept by the net to 704 m^2 (Loneragan et al., 1989; Loneragan and Potter, 1990). For each of the studies undertaken between 1995 and 2011, including the current study, nearshore fish were sampled using one or both of two smaller seine nets. The first of these was 41.5 m long, fished to a maximum depth of 1.5 m and swept a semi-circular area of *ca* 274 m^2. The mesh in the wings of this net was 25 mm wide when stretched, and that in the 1.5 m-long bunt was 9 mm (Kanandjembo et al., 2001). The second seine net was 21.5 m long, 1.5 m deep, swept an area of 116 m^2 and comprised two 10 m-long wings (6 m of 9 mm mesh and 4 m of 3 mm mesh) and a 1.5 m-long bunt of 3 mm mesh.

Fish in the offshore, deeper waters (> 2 m depth) of the Swan Estuary were sampled between 1976 and 2011 using sunken, multi-mesh gill nets that consisted of six to eight 20 m-long panels with stretched mesh sizes ranging from 35 to 127 mm in increments of between 12 and 16 mm (Table 1). These nets were deployed at sunset and retrieved after two to three hours.

Sampling for the current study was conducted throughout the estuary during the middle month of each season from winter 2007 to autumn 2009 (for index development) and, for the purposes of index validation, in the middle and last months of both summer and autumn in 2011. Either or both of the 21.5 and 41.5 m-long seine nets were employed in the nearshore waters and multi-mesh gill nets were used in the offshore waters. The nearshore sampling regime was supplemented by additional sampling of fish assemblages with the 21.5 m seine, at selected sites throughout the Canning Estuary and Lower Canning River (CELCR) zone (Fig. 1) during May 2011, in response to an algal bloom event that occurred at that time (see subsection 2.6).

Fish collected during the current study were immediately placed in an ice slurry and taken to the laboratory for processing. All fish were identified to species and the total number of individuals belonging to each species in each sample was recorded. The total length of each fish was measured to the nearest 1 mm, except when a large number of individuals of any one species was encountered in a sample, in which case the lengths of a representative subsample of 50 individuals were measured.

2.3 Metric selection

A full account of the process of metric selection is provided by Hallett et al. (In press). The approach is summarised briefly below, and novel aspects of this approach are highlighted. All fish species encountered in the Swan Estuary during studies of this system between 1976 and 2009 were first allocated to functional ecological guilds (namely 'Habitat', 'Estuarine Use' and 'Feeding Mode') to enable the calculation of various candidate metrics (see Appendix A for a full list of these guilds). Guild allocation was undertaken on the basis of information contained within the Codes for Australian Aquatic Biota (Rees et al., 1999), published literature and FishBase (Froese and Pauly, 2007).

In the absence of reliable, independent measures of estuarine condition against which to test the sensitivity of candidate metrics (either spatially or temporally), multivariate statistical analyses and multi-model inference techniques were employed to select those metric subsets likely to be the most sensitive to inter-annual changes in the health of this ecosystem (Hallett et al., In press). Novel pre-treatment techniques were also applied prior to analysis to (i) down-weight the influence of highly erratic metrics and (ii) minimise the effects of seasonal and spatial differences in sampling upon metric variability. A weight of evidence approach was then adopted to select those nearshore and offshore fish metrics which exhibited the most consistent inter-annual changes between 1976 and 2009 (Hallett et al., In press).

2.4 Reference conditions

We sought to establish 'best available' reference conditions (Harris and Silveira, 1999; Harrison and Whitfield, 2004; Coates et al., 2007) for each selected fish metric using the composite sets of fish community data described above. However, given the divergent nearshore sampling methods employed historically in the Swan Estuary, the resulting nearshore data sets were each affected by differing biases, preventing them from being directly comparable. Therefore, before nearshore reference conditions could be determined, the sampling biases associated with each seine net type were first investigated, and equivalence factors derived to enable the standardization of all species abundance data to those expected in a common net type, i.e. the 21.5 m seine. Hallett and Hall (submitted) provide a detailed description of this standardisation process for representatives of each functional habitat guild of fishes (small benthic, small pelagic, demersal, pelagic) recorded between 1976 and 2009.

In contrast to the historical fish assemblage studies carried out in the nearshore waters of the Swan Estuary, those undertaken in the offshore waters have employed relatively consistent methods and effort and so are largely free from sampling bias. All fish abundance data obtained from the offshore waters throughout the estuary between 1976 and 2009 were thus converted to equivalent catch rates (fish hr^{-1}) and collated to determine reference conditions for each of the selected offshore fish metrics.

Values for each of the selected nearshore and offshore fish metrics were next calculated from the standardised abundance data for each historical and current fish sample. Season- and zone-specific best available reference conditions for each selected metric were then established from the observed distributions of metric values, minimising the potential for seasonal and zonal differences in fish community structure to impact on the reliability of reference conditions. Identification of these 'best' values for each metric (i.e. whether they were among the lowest or highest of all values ever recorded in a given zone and season) depended on *a priori* hypotheses of metric responses to anthropogenic degradation (Table 2; Hallett, 2010). The upper threshold (95^{th} percentile) of metric values determined the best available reference condition for negative metrics (whose values are predicted to decrease in response to ecological degradation), whilst the lower threshold (5^{th} percentile) defined the best available reference condition for positive metrics (whose values increase with degradation). Upper and lower thresholds were set using percentiles, rather than minima and maxima, to avoid the influence of extreme outliers (Gibson et al., 2000).

2.5 Scoring and index calculation

The appropriate zone*season reference conditions for each metric were used to establish metric scores (0-10) for each sample via continuous scaling, with scores between the upper and lower reference thresholds being calculated by linear interpolation (Hering et al., 2006). For negative metrics, the metric value was divided by the observed range of reference values and then multiplied by 10:

$$Metric\ score = \frac{(Observed\ metric\ value - Lower\ threshold)}{(Upper\ threshold - Lower\ threshold)} \times 10$$

For positive metrics, the quotient was subtracted from 1 before multiplying by 10:

$$Metric\ score = \left(1 - \frac{(Observed\ metric\ value - Lower\ threshold)}{(Upper\ threshold - Lower\ threshold)}\right) \times 10$$

In cases where metric values exceeded the best threshold (i.e. outliers), a metric score of 10 was allocated. When no fish were caught in a sample, all metrics received a score of zero.

Scores for the nearshore and offshore health indices were calculated for each sample by summing the scores for their component metrics, then standardising the resultant value (i.e. dividing the score by the number of metrics in the index and then multiplying by ten) to produce a final index score that ranged from 0-100 (Ganasan and Hughes, 1998).

Finally, thresholds for establishing qualitative estuarine health status were determined by subdividing the range of possible index scores into four equal classes (good ≥75; 75> fair ≥50; 50> poor ≥25; 25> very poor). It was considered that a greater number of classes than this would make decisions regarding management actions more problematic (Ganasan and Hughes, 1998; Qadir and Malik, 2009), whilst fewer classes might allow the health of an estuary to decline markedly before a health status threshold is crossed and management actions are invoked.

2.6 Validation of index sensitivity

An algal bloom that occurred in the CELCR zone during May 2011 provided an opportunity to assess the sensitivity of the nearshore health index to a relatively short-term, spatially discrete environmental perturbation. On the 10th of May, the potentially ichthyotoxic dinoflagellate *Karlodinium veneficum* was recorded at densities above the local management guideline level of 250 cells/mL at locations between Riverton Bridge and Kent St Weir on the Canning River (Fig. 1), with a peak in excess of 30,000 cells/mL at Castledare (Swan River Trust, unpublished data). By May 17th, the densities of *K. veneficum* at Castledare and Riverton Bridge had decreased, whilst those at sites at or upstream of Kent St Weir had increased. By May 24th, the bloom had collapsed and cell densities had decreased markedly at all of the above sites.

Nearshore fish assemblages in the CELCR zone had been sampled immediately prior to the bloom at sites downstream of Riverton Bridge, during the course of the routine sampling described in subsection 2.2. These sites were resampled on May 16th, in the middle of the bloom period, and on May 27th, following the end of the bloom. Nearshore health index scores were calculated from each of these samples as described above, and nearshore index sensitivity was then assessed by comparing the patterns in index scores among samples collected during the bloom (‘mid-bloom’) to those recorded ‘pre-bloom’ (i.e. during April and/or early May) and after the bloom had collapsed (‘post bloom’).

2.7 Validation of index reliability

Month-to-month changes in the nearshore and offshore index scores for each individual site were quantified in each sampling season during 2011, and the resultant changes in qualitative health status examined. Intra-seasonal changes in mean nearshore and offshore scores across each zone, and across the estuary as a whole, were also similarly assessed, to determine the consistency of quantitative index scores and qualitative health classifications. Boxplots were used to examine month-to-month changes in the statistical distribution of all nearshore and offshore index scores in each season. As the index scores from any individual month were not normally distributed, non-parametric Mann-Whitney-Wilcoxon rank sum tests (with Bonferroni corrections for repeated tests) were used to ascertain whether the distributions of index scores among months differed significantly.

3. Results

3.1 Metric selection

The respective sets of 11 and seven metrics selected for the nearshore and offshore indices represented a broad range of fish community characteristics, including species composition and diversity, trophic structure, life history and habitat functions and, in the case of the nearshore index, a potential sentinel species, the tolerant, omnivorous Blue-spot Goby, *Pseudogobius olorum* (Table 2; Hallett et al., In press).

3.2 Reference conditions

The zone*season-specific reference conditions for each nearshore and offshore metric are presented in Appendices B and C, respectively. For several of these metrics, there were clear differences in reference condition values both between different zones in a given season, and between seasons within a zone. For example, the reference condition for the nearshore metric *No species* varied from as few as five species in the Upper Swan Estuary (USE) in winter, to as many as 14 species in the CELCR in summer or in the Middle Swan Estuary (MSE) in summer or autumn.

3.3 Validation of index sensitivity

Nearshore index scores for samples collected in the CELCR during late April 2011 (i.e. prior to the *Karlodinium veneficum* bloom) indicated that the health of this zone was fair to good (mean score = 71.5), with most sites exhibiting scores of between 66 and 72 (fair) and two sites scores of 76.8 (good; Fig. 3a). Index scores from sites sampled on May 11th were consistent with those observed on the previous sampling occasion (i.e. a drop of only 0.5 points in the mean score), with individual site scores ranging between 62 and 73 (fair) and one site being characterised as good (Fig. 3b).

At the mid-point of the bloom, however, the scores for each nearshore site had decreased by between two and 29 points. As of May 16th, the ecological health of sites located between Salter Point and Kent St Weir had been considerably impacted and, although the overall health of the CELCR was still assessed as fair at this time, the mean score for the zone had decreased by more than 10 points to 60.8 (Fig. 3c). Most notably, a mid-bloom sample collected from a site immediately downstream of Kent St Weir returned only two fish, with a corresponding score of 42.7 (poor health status).

Following the collapse of the bloom, the health of the CELCR zone recovered towards its pre-bloom condition, with the mean score for the zone reaching 68.1 by the time of the post-bloom sampling (Fig. 3d). Nearshore scores for each individual site had rebounded by two to 16 points between May 16th and 27th, by which time all sites were classified as being in fair health.

3.4 Validation of index reliability

Within both summer and autumn 2011, considerable changes in nearshore index scores were observed from month to month at some sites. During summer, this variation ranged from 0.7 to 26.6 (mean = 8.4) for any individual site and led to a change in the health status classification of ten of the 32 nearshore sites surveyed. In autumn, index scores for any individual nearshore site similarly varied by 0.5 to 25.4 points between months (mean = 6.5), resulting in a change in health status for seven of the 32 sites. Similarly, the intra-seasonal change in index score for any individual offshore site ranged from 1.9 to 28.9 in summer (mean = 10.4), and in autumn changed by as much as 32.8 points between months (mean = 11.4). This variability led to a change in the health status classification of ten of the 23 offshore sites in both seasons.

The extents of intra-seasonal changes in index scores were far less pronounced, however, at the broader scale of estuarine zones, i.e. the minimum spatial scale at which the indices are intended to be used and interpreted. The month-to-month change in the mean nearshore index score for any zone ranged from 0.8 to 7.1 (mean = 3.7) points in summer, and from 3.0 to 6.9 (mean = 4.2) in autumn (Table 3a). Moreover, this level of variability did not result in a change in the nearshore health status of any zone in either season. Similarly, the month-to-month change in mean offshore index score for any zone ranged from 2.1 to 7.9 (mean = 5.4) points in summer, and from 2.5 to 9.7 (mean = 6.0) points in autumn (Table 3b), and did not lead to a change in the offshore health status of any zone in either season. Mann-Whitney-Wilcoxon tests, conducted at the level of estuarine zones, revealed no significant differences in the distributions of either nearshore or offshore scores between months, in either season, for any zone.

The distribution of nearshore index scores across the whole estuary (including those from supplementary sampling around the May 2011 bloom) was broadly similar from month to month in both seasons (Fig. 4). Median nearshore index scores from the first and second sampling occasions during summer were 63.1 and 63.6, respectively. The distributions of scores in the two summer months did not differ significantly (Mann-Whitney-Wilcoxon W = 508, $n_1 = n_2 = 32$, $p = 0.963$). Similarly, the distributions of nearshore index scores from the first (median = 65.6) and second (median = 65.1) sampling occasions during autumn were not significantly different ($W = 719$, $n_1 = 32$, $n_2 = 40$, $p = 0.376$). Moreover, the distribution of nearshore index scores did not differ significantly between seasons ($W = 2314$, $n_1 = 64$, $n_2 = 72$, $p = 0.967$).

For the offshore waters, median index scores observed across all sites from the first and second sampling occasions during summer were 64.1 and 61.7, respectively. The distributions of scores in the two summer months did not differ significantly ($W = 283$, $n_1 = n_2 = 23$, $p = 0.695$), nor did the distributions of offshore index scores from the first (median = 56.9) and second (median = 55.8) sampling occasions during autumn ($W = 311$, $n_1 = n_2 = 23$, $p = 0.315$). However, the distribution of offshore index scores across all samples collected during summer (median = 62.3) differed significantly from that across all autumn samples (median = 56.1; $W = 669$, $n_1 = n_2 = 46$, $p = 0.002$), in that lower median scores were observed during autumn (Fig. 5).

4. Discussion

The fish-based multimetric indices we have constructed, which are the first such indicators to be developed for assessing the health of estuaries in Australia, provide robust and informative tools for management and communication. The framework of index development may be applied to construct similar health indices for any estuary in south-western Australia, and is based on widely accepted and objective approaches, assumptions and techniques. Where novel methodologies were employed, these were developed and applied with a focus on statistical rigour and subjected to scientific peer-review (e.g. Hallett et al., In press; Hallett and Hall, submitted). Below, we evaluate both the process by which these indices were developed, and their resulting reliability and sensitivity.

4.1 Metric selection

Hallett et al. (In press) have evaluated the selection of metrics for the current indices and noted that, while the approach provides an avenue for circumventing any *a priori* demonstration of the relationships between the selected metrics and independent measures of anthropogenic degradation (i.e. where the latter data is not available), *a posteriori* tests of index sensitivity are essential to demonstrate the ecological relevance of the resulting index. The sensitivity of the indices we have developed is thus addressed in subsection 4.3.

4.2 Reference conditions

Ideally, the health of an ecosystem should be assessed in comparison to a pristine system that has not been modified by anthropogenic influences (Harris and Silveira, 1999; Gibson et al., 2000). However, given that few estuaries are free from human impacts, many studies have selected least disturbed or best available sites as a reference (Oberdorff and Hughes, 1992; Deegan et al., 1997). Moreover, in systems which have been heavily modified, such as the Swan Estuary, it is often difficult to distinguish the least impacted sites. We therefore adopted an approach in which biological reference conditions are defined from some ‘best’ fraction of the observed metric values across a large number of samples collected throughout the system over time (Gibson et al., 2000; Blocksom, 2003; Harrison and Whitfield, 2004; 2006; Coates et al., 2007).

In the present case, the resultant reference conditions do not, and cannot, characterize a pristine state, given that the Swan Estuary (like most other estuaries across south-western Australia) has been heavily modified by a range of anthropogenic pressures since the mid-1800s. Instead, they represent a measure of the best biological status observed over the past 30 years, and thus provide a sound reference point against which to assess the ecological health of the system. Under this approach, the specific, ‘best-available’ reference value established for each metric will depend on the statistical criterion applied to the distribution of metric values although, given that environmental management aims to improve or maintain the ecosystem, reference conditions should be set as high as the data will reliably allow (Hughes, 1995). Whereas several authors have suggested using the maximum (or, where relevant, minimum) value of a metric as a reference in order to eliminate subjectivity (Hering et al., 2006; Roset et al., 2007), such an approach may be unduly influenced by extreme outliers and was thus avoided in the current approach (Gibson et al., 2000).

Several authors have highlighted problems associated with the use of historical data for establishing reference conditions, including a lack of quantity or quality of data and a lack of standardised methods for data collection (Hughes, 1995; Harrison and Whitfield, 2004). In the case of the indices presented here, the combined data set used to establish reference conditions comprised several thousand samples collected throughout the Swan Estuary over three decades. However, the divergent gears used to sample fish in the nearshore waters necessitated the use of complex data standardisation procedures. Hallett and Hall (submitted) considered the efficacy of these standardisation procedures, and judged that the benefits of having such a large data set outweighed the potential issues of wide confidence intervals associated with the equivalence factors.

In addition to standardising catch data to overcome gear-related biases, we have also accounted for the natural spatio-temporal variability of fish assemblages by defining appropriate reference conditions for each zone in each season (Karr, 1999; Kennard et al., 2006a; Coates et al., 2007). Although several authors have reported that fish-based multimetric indices for assessing the biotic integrity of riverine systems were unaffected by intra-annual variability in fish community composition (Karr et al., 1986; Pyron et al., 2008; Qadir and Malik, 2009), the effects on estuarine biota of highly seasonal freshwater flows and strong physico-chemical gradients potentially impact the reliability of indicators developed for these ecosystems (Lobry et al., 2006; Chainho et al., 2007; Pérez-Ruzafa et al., 2007; Bilkovic and Roggero, 2008; Mazor et al., 2009; Rashleigh et al., 2009) and must be taken into account when setting reference conditions.

4.3 Index validation

A crucial, final step in the development of an effective biotic index is validating its sensitivity (the degree to which it responds to degradation) and reliability (the consistency and repeatability of index assessments). With respect to index reliability, month-to-month changes in mean nearshore or offshore index scores did not result in a change in the health status of any zone in either season. This indicates that the nearshore and offshore indices are robust to the effects of natural, intra-seasonal variability in environmental conditions, and thus provide reliable tools for quantifying and classifying the ecological health of the Swan Estuary and its constituent management zones. Moreover, they demonstrate that repeated sampling across multiple months within a season is not necessary to reliably capture the health status of the estuary, or that of a particular zone. However, given that summer and autumn have previously been identified as the optimum period in which to implement the index (Hallett, 2010), and that the health of the estuary may change between seasons due to short-term perturbations such as algal blooms, it is recommended that any future monitoring regime for the Swan Estuary should include both summer and autumn sampling.

The response of the nearshore index to a spatially and temporally discrete algal bloom has also confirmed that it is sufficiently sensitive to quantify ecological health responses to local-scale environmental perturbations, and also to track the subsequent recovery of the system following their removal. Nearshore index scores at sites affected by the algal bloom exhibited a clear decrease from pre-bloom conditions. In the absence of any observed fish kill, it is argued that this reflects the movement of fish away from these affected areas to escape the overall decline in habitat quality which would accompany such a bloom. As the bloom senesced and collapsed, and environmental conditions returned to a pre-bloom state, the fish fauna that typify a more healthy CELCR zone recolonised the bloom-affected areas, leading to a recovery in health index scores. Moreover, the consistency of index scores across sampling occasions prior to the bloom (Fig. 3a and b) provides further confirmation that the nearshore health index is consistent and robust (i.e. is not overly sensitive to natural, background variability).

5. Conclusions

The indices we have developed provide a simple, objective method for quantifying and communicating the ecological health of estuaries, monitoring temporal changes in estuarine health and identifying those zones of the system at greatest risk of environmental decline. Application of these indices to the Swan Estuary in WA has addressed a critical need for managers of that system, and could do so for other estuaries across the region. Validation of these indices has shown that classification of the health status of the estuary and its component zones is reliable and robust, despite natural and sampling-related variability. Moreover, the sensitivity of these indices to relatively short, localised environmental perturbations related to human-caused stressors (i.e. algal blooms), has now been demonstrated.

Despite the complexity of the process by which these indices have been developed, their future implementation and use for assessing estuarine health is, in contrast, conceptually simple and technically straightforward. Index outputs can be communicated both quantitatively and qualitatively (e.g. good, fair, poor, very poor), with the latter being very easily understood by managers and the public alike. These indices are thus well-suited to inclusion in future ecosystem report cards planned for the Swan Estuary, akin to those produced for estuaries in Queensland and the US (e.g. EHMP, 2007; Longstaff et al., 2010). More broadly, the approach we have described could easily be modified for application to other estuaries across the south-west bioregion of Australia and beyond. Given the lack of quantitative, biological indicators currently available to estuarine managers, there is considerable potential for the multimetric indices we have developed to advance the field of estuarine health assessment in Australia and to form a crucial component of state and federal national estuarine assessment programs.

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Figure captions

Figure 1. Locations of the Swan-Canning Estuary, Western Australia (inset), and of the nearshore (<2 m depth; closed circles) and offshore (>2 m depth; open circles) sites throughout this system at which fish communities were sampled historically and during the current study. Ecological management zones and locations referred to in the text: Lower Swan-Canning Estuary (LSCE), Canning Estuary/Lower Canning River (CELCR), Middle Swan Estuary (MSE), Upper Swan Estuary (USE); Salter Point (SAL); Riverton Bridge (RIV); Castledare (CAS); Kent St. Weir (KEN).

Figure 2. Typical stages in the development of multimetric biotic indices (after Simon, 2000).

Figure 3. Maps of the Canning Estuary & Lower Canning River (CELCR) zone of the Swan Estuary, illustrating nearshore health index scores (circled) and health classifications (green, good; yellow, fair; orange, poor; red, very poor) for sites sampled (a, b) before, (c) during and (d) after a *Karlodinium veneficum* bloom in May 2011. Numbers outside circles illustrate

changes in index scores from the previous sampling occasion. Boxed text presents mean index score (± SE) for the CELCR zone, coloured to reflect the accompanying health classification. SAL, Salter Point; RIV, Riverton Bridge; CAS, Castledare; KEN, Kent St Weir.

Figure 4. The distributions of nearshore index scores obtained during each month of sampling in summer and autumn 2011. Sample sizes (*n*) for each month are shown above boxplots. Median scores are represented by dark horizontal bars and the first and third quartiles of the data as upper and lower bounds of the boxes, respectively. Dashed whiskers illustrate either the maximum observed values or *ca* two standard deviations (whichever is the smaller value), and any remaining outliers are plotted individually.

Figure 5. The distributions of offshore index scores obtained during each month of sampling in summer and autumn 2011. Sample sizes (*n*) for each month are shown above boxplots. Median scores are represented by dark horizontal bars and the first and third quartiles of the data as upper and lower bounds of the boxes, respectively. Dashed whiskers illustrate either the maximum observed values or *ca* two standard deviations (whichever is the smaller value), and any remaining outliers are plotted individually.

Tables

Table 1. Fish community data sets employed in the selection of metrics sensitive to temporal ecosystem change in the Swan Estuary, illustrating the zones of that system sampled consistently during each study and the methods employed to sample them; Lower Swan-Canning Estuary (LSCE), Canning Estuary/Lower Canning River (CELCR), Middle Swan Estuary (MSE), Upper Swan Estuary (USE).

Study (Years)	Sampling method			
	21.5 m seine net	41.5 m seine net	102-133 m seine net	Gill net
Loneragan [a] (1976-1982)			LSCE, CELCR, MSE, USE	
Sarre [b] (1993-1994)				MSE, USE
Kanandjembo [c] (1995-1997)		MSE, CELCR		MSE
Hoeksema [d] (1999-2001)	MSE, USE			MSE, USE
Hoeksema [e] (2003-2004)	MSE, USE	LSCE, CELCR, MSE		MSE, USE
Valesini [f] (2005-2007)	LSCE, MSE, USE			
Current study [g] (2007-2009)	LSCE, USE	LSCE, CELCR, MSE		LSCE, CELCR, MSE, USE
Current study (2010-2011)	LSCE, CELCR, MSE, USE			LSCE, CELCR, MSE, USE

[a] Loneragan et al., 1989; Loneragan and Potter 1990; [b] Sarre, unpublished data; [c] Kanandjembo et al., 2001; [d] Hoeksema and Potter, 2006; [e] Hoeksema, unpublished data; [f] Valesini et al., 2009; [g] Hallett, 2010.

Table 2. Fish metrics selected (√) for the nearshore and offshore fish assemblage-based estuarine health indices developed for the Swan-Canning Estuary (from Hallett et al., In press). Hypothesised metric responses to ecological degradation, i.e. positive (+) or negative (-), are shown in parentheses.

Metric	**Abbreviation**	**Nearshore Index**	**Offshore Index**
Number of species (-)	*No species*	√	√
Dominance (+)	*Dominance*		
Shannon-Weiner diversity (-)	*Sh-div*		√
Proportion of trophic specialists (-)	*Prop trop spec*	√	
Number of trophic specialist species (-)	*No trop spec*	√	√
Number of trophic generalist species (+)	*No trop gen*	√	√
Proportion of detritivores (+)	*Prop detr*	√	√
Feeding guild composition (-)	*Feed guild comp*		
Proportion of benthic-associated individuals (-)	*Prop benthic*	√	√
Number of benthic species (-)	*No benthic*	√	
Proportion of estuarine spawning individuals (-)	*Prop est spawn*	√	√
Number of estuarine spawning species (-)	*No est spawn*	√	
Proportion of *Pseudogobius olorum* (+)	*Prop P. olorum*	√	
Total number of *Pseudogobius olorum* (+)	*Tot no P. olorum*	√	

Table 3. Mean (± SE) index scores across (a) nearshore and (b) offshore sites sampled during the middle months (month 1) and final months (month 2) of summer and autumn 2011 in each zone of the Swan Estuary (Lower Swan-Canning Estuary [LSCE], Canning Estuary/Lower Canning River [CELCR], Middle Swan Estuary [MSE], Upper Swan Estuary [USE]). Numbers in parentheses represent the numbers of sites sampled.

	Summer		**Autumn**	
Zone	Month 1	Month 2	Month 1	Month 2
(a) Nearshore				
LSCE (*n* = 8)	70.0 ± 6.6	63.0 ± 3.8	64.8 ± 2.1	61.7 ± 2.3
CELCR (*n* = 8)	59.8 ± 3.2	61.7 ± 3.9	71.5 ± 1.4	68.5 ± 2.9
MSE (*n* = 8)	60.2 ± 3.3	59.4 ± 2.6	62.7 ± 2.6	66.3 ± 1.3
USE (*n* = 8)	67.5 ± 3.9	72.5 ± 2.3	62.5 ± 3.4	55.6 ± 3.6
(b) Offshore				
LSCE (*n* = 5)	60.7 ± 5.9	68.7 ± 3.7	60.9 ± 3.5	55.8 ± 5.5
CELCR (*n* = 5)	57.3 ± 3.5	50.3 ± 4.4	56.2 ± 2.2	53.7 ± 5.4
MSE (*n* = 6)	67.9 ± 3.5	63.5 ± 3.0	51.2 ± 3.3	57.9 ± 5.7
USE (*n* = 7)	65.1 ± 5.0	64.0 ± 2.6	61.0 ± 2.1	51.3 ± 5.4

Appendix A. List of fish species identified from the Swan Estuary during previous (1976-2007) and current (2007-2011) studies, and the functional guilds to which they were allocated. Abbreviations: P – large pelagic; D – demersal (species closely associated with substrate, rocks or weed); BP – bentho-pelagic; SP – small pelagic; SB – small benthic; MS – marine straggler; MM – marine migrant (includes marine estuarine opportunists); SA – semi-anadromous; ES – estuarine species; FM – freshwater migrant or straggler; PV – piscivore; ZB – zoobenthivore; ZP – zooplanktivore; DV – detritivore; OV – omnivore; HV – herbivore; OP – opportunist.

Species name	**Common name**	**Habitat**	**Estuarine Use**	**Feeding Mode**
Carcharinas leucas	Bull shark	P	MS	PV
Myliobatis australis	Southern eagle ray	D	MS	ZB
Elops machnata	Giant herring	BP	MS	PV
Hyperlophus vittatus	Sandy sprat	SP	MM	ZP
Spratelloides robustus	Blue sprat	SP	MM	ZP
Sardinops neopilchardus	Australian pilchard	P	MS	ZP
Sardinella lemuru	Scaly mackerel	P	MS	ZP
Nematalosa vlaminghi	Perth herring	BP	SA	DV
Engraulis australis	Southern anchovy	SP	ES	ZP
Galaxias occidentalis	Western minnow	SB	FM	ZB
Carassius auratus	Goldfish	BP	FM	OV
Cnidoglanis macrocephalus	Estuarine cobbler	D	MM	ZB
Tandanus bostocki	Freshwater cobbler	D	FM	ZB
Hyporhamphus melanochir	Southern sea garfish	P	ES	HV
Hyporhamphus regularis	Western river garfish	P	FM	HV
Gambusia holbrooki	Mosquito fish	SP	FM	ZB
Atherinosoma elongata	Elongate hardyhead	SP	ES	ZB
Leptatherina presbyteroides	Presbyter's hardyhead	SP	MM	ZP
Atherinomorus vaigensis	Ogilby's hardyhead	SP	MM	ZB
Craterocephalus mugiloides	Mugil's hardyhead	SP	ES	ZB
Leptatherina wallacei	Wallace's hardyhead	SP	ES	ZP
Cleidopus gloriamaris	Pineapplefish	D	MS	ZB
Stigmatophora nigra	Wide-bodied pipefish	D	MS	ZB
Vanacampus phillipi	Port Phillip pipefish	D	MS	ZB
Phyllopteryx taeniolatus	Common seadragon	D	MS	ZB
Hippocampus angustus	Western Australian seahorse	D	MS	ZP
Stigmatophora argus	Spotted pipefish	D	MS	ZP
Urocampus carinirostris	Hairy pipefish	D	ES	ZP
Filicampus tigris	Tiger pipefish	D	MS	ZP
Pugnaso curtirostris	Pugnose pipefish	D	MS	ZP
Gymnapistes marmoratus	Devilfish	D	MS	ZB
Chelidonichthys kumu	Red gurnard	D	MS	ZB
Platycephalus laevigatus	Rock flathead	D	MS	PV
Platycephalus endrachtensis	Bar-tailed flathead	D	ES	PV
Leviprora inops	Long-head flathead	D	MS	PV
Platycephalus speculator	Southern blue-spotted flathead	D	ES	PV
Pegasus lancifer	Sculptured seamoth	D	MS	ZB
Amniataba caudavittata	Yellow-tail trumpeter	BP	ES	OP
Pelates octolineatus	Eight-line trumpeter	BP	MM	OV
Pelsartia humeralis	Sea trumpeter	BP	MS	OV
Edelia vittata	Western pygmy perch	BP	FM	ZB
Apogon rueppelli	Gobbleguts	BP	ES	ZB

Siphamia cephalotes	Woods siphonfish	BP	MS	ZB
Sillago bassensis	Southern school whiting	D	MS	ZB
Sillago burrus	Trumpeter whiting	D	MM	ZB
Sillaginodes punctata	King George whiting	D	MM	ZB
Sillago schomburgkii	Yellow-finned whiting	D	MM	ZB
Sillago vittata	Western school whiting	D	MM	ZB
Pomatomus saltatrix	Tailor	P	MM	PV
Trachurus novaezelandiae	Yellowtail scad	P	MS	ZB
Pseudocaranx dentex	Silver trevally	BP	MM	ZB
Pseudocaranx wrightii	Sand trevally	BP	MM	ZB
Arripis georgianus	Australian herring	P	MM	PV
Arripis esper	Southern Australian salmon	P	MS	PV
Gerres subfasciatus	Roach	BP	MM	ZB
Pagrus auratus	Snapper	BP	MM	ZB
Acanthopagrus butcheri	Southern black bream	BP	ES	OP
Rhabdosargus sarba	Tarwhine	BP	MM	ZB
Argyrosomus japonicus	Mulloway	BP	MM	PV
Pampeneus spilurus	Black-saddled goatfish	D	MS	ZB
Enoplosus armatus	Old wife	D	MS	ZB
Aldrichetta forsteri	Yellow-eye mullet	P	MM	OV
Mugil cephalus	Sea mullet	P	MM	DV
Sphyraena obtusata	Striped barracuda	P	MS	PV
Haletta semifasciata	Blue weed whiting	D	MS	OV
Siphonognathus radiatus	Long-rayed weed whiting	D	MS	OV
Neoodax baltatus	Little weed whiting	D	MS	OV
Odax acroptilus	Rainbow cale	D	MS	OV
Parapercis haackei	Wavy grubfish	D	MS	ZB
Petroscirtes breviceps	Short-head sabre blenny	SB	MS	OV
Omobranchus germaini	Germain's blenny	SB	MS	ZB
Parablennius intermedius	Horned blenny	D	MS	ZB
Istiblennius meleagris	Peacock rockskipper	D	MS	HV
Cristiceps australis	Southern crested weedfish	D	MS	ZB
Pseudocalliurichthys goodladi	Longspine stinkfish	D	MS	ZB
Eocallionymus papilio	Painted stinkfish	D	MS	ZB
Nesogobius pulchellus	Sailfin goby	SB	MS	ZB
Favonigobius lateralis	Long-finned goby	SB	MM	ZB
Afurcagobius suppositus	Southwestern goby	SB	ES	ZB
Pseudogobius olorum	Blue-spot / Swan River goby	SB	ES	OV
Amoya bifrenatus	Bridled goby	SB	ES	ZB
Callogobius mucosus	Sculptured goby	SB	MS	ZB
Callogobius depressus	Flathead goby	SB	MS	ZB
Papillogobius punctatus	Red-spot goby	SB	ES	ZB
Tridentiger trigonocephalus	Trident goby	SB	MS	ZB
Pseudorhombus jenynsii	Small-toothed flounder	D	MM	ZB
Ammotretis rostratus	Longsnout flounder	D	MM	ZB
Ammotretis elongata	Elongate flounder	D	MM	ZB
Cynoglossus broadhursti	Southern tongue sole	D	MS	ZB
Acanthaluteres brownii	Spiny-tailed leatherjacket	D	MS	OV
Brachaluteres jacksonianus	Southern pygmy leatherjacket	D	MS	OV
Scobinichthys granulatus	Rough leatherjacket	D	MS	OV
Meuschenia freycineti	Sixspine leatherjacket	D	MM	OV
Monacanthus chinensis	Fanbellied leatherjacket	D	MM	OV
Eubalichthys mosaicus	Mosaic leatherjacket	D	MS	OV
Acanthaluteres vittiger	Toothbrush leatherjacket	D	MS	OV
Acanthaluteres spilomelanurus	Bridled leatherjacket	D	MM	OV
Torquigener pleurogramma	Banded toadfish	BP	MM	OP
Contusus brevicaudus	Prickly toadfish	BP	MS	OP
Polyspina piosae	Orange-barred puffer	BP	MS	OP
Diodon nichthemenus	Globefish	D	MS	ZB
Scorpis aequipinnis	Sea sweep	P	MS	ZP
Neatypus obliquus	Footballer sweep	P	MS	ZP

Appendix B. Reference conditions for each of the selected nearshore fish metrics, determined from standardised historical and current seine net data collected from each zone of the Swan Estuary (Lower Swan-Canning Estuary [LSCE], Canning Estuary/Lower Canning River [CELCR], Middle Swan Estuary [MSE] and Upper Swan Estuary [USE]) in each season. *n* = number of samples per zone*season combination. See Table 2 for metric abbreviations.

		Metric										
Zone*season	*n*	*No species*	*Prop trop spec*	*No trop spec*	*No trop gen*	*Prop detr*	*Prop benthic*	*No benthic*	*Prop est spawn*	*No est spawn*	*Prop P. olorum*	*Tot no P. olorum*
LSCE*summer	174	11	0.99	8	1	0	1.0	9	0.96	5	0	0
LSCE*autumn	156	13	0.99	8	1	0	1.0	9	0.83	5	0	0
LSCE*winter	173	8	1.0	6	0	0	1.0	6	0.79	4	0	0
LSCE*spring	179	11	0.98	7	1	0	1.0	8	0.76	5	0	0
CELCR*summer	66	14	0.99	9	1	0	1.0	9	1.0	9	0	0
CELCR*autumn	68	13	0.99	8	0	0	1.0	6	1.0	7	0	0
CELCR*winter	79	10	0.99	5	0	0	1.0	5	1.0	6	0	0
CELCR*spring	84	12	0.98	8	1	0	1.0	7	1.0	8	0	0
MSE*summer	119	14	0.96	8	1	0	1.0	9	1.0	9	0	0
MSE*autumn	123	14	1.0	9	0	0	1.0	9	1.0	8	0	0
MSE*winter	115	10	0.98	6	0	0	1.0	7	1.0	6	0	0
MSE*spring	144	13	0.93	8	1	0	1.0	9	1.0	8	0	0
USE*summer	108	10	0.98	6	1	0	0.98	7	1.0	8	0	0
USE*autumn	111	9	1.0	5	0	0	1.0	6	1.0	7	0	0
USE*winter	99	5	0.99	3	0	0	0.95	3	1.0	4	0	0
USE*spring	132	9	0.98	5	1	0	1.0	6	1.0	7	0	0

Appendix C. Reference conditions for each of the selected offshore fish metrics, determined from historical and current gill net data collected from each zone of the Swan Estuary (Lower Swan-Canning Estuary [LSCE], Canning Estuary/Lower Canning River [CELCR], Middle Swan Estuary [MSE] and Upper Swan Estuary [USE]) in each season. *n* = number of samples per zone*season combination. See Table 2 for metric abbreviations.

		Metric						
Zone*season	*n*	*No species*	*Sh-div*	*No trop spec*	*No trop gen*	*Prop detr*	*Prop benthic*	*Prop est spawn*
LSCE*summer	11	6	1.51	4	0	0	1.0	1.0
LSCE*autumn	12	6	1.63	4	0	0	1.0	0.92
LSCE*winter	12	8	1.87	5	0	0	1.0	0.41
LSCE*spring	8	5	1.47	5	0	0	1.0	1.0
CELCR*summer	10	7	1.71	4	0	0.20	1.0	0.83
CELCR*autumn	8	8	1.69	4	0	0.36	1.0	0.72
CELCR*winter	10	4	1.36	3	0	0	1.0	1.0
CELCR*spring	8	9	1.71	4	0	0	0.96	1.0
MSE*summer	37	6	1.67	2	0	0.09	1.0	1.0
MSE*autumn	45	6	1.44	3	0	0.16	1.0	1.0
MSE*winter	42	5	1.44	2	0	0	1.0	1.0
MSE*spring	42	5	1.29	2	0	0.20	1.0	1.0
USE*summer	35	5	1.18	2	1	0	1.0	1.0
USE*autumn	39	5	1.55	3	0	0	1.0	1.0
USE*winter	39	4	1.18	1	0	0	1.0	1.0
USE*spring	37	4	1.27	1	1	0	1.0	1.0