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Enhancing the performance of marine reserves in estuaries: Just add water

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ABSTRACT

Nature reserves are created to conserve biodiversity and restore populations of harvested species, but it is not clear whether this strategy is successful in all ecosystems. Reserves are gazetted in estuaries to offset impacts from burgeoning human populations, however, coastal conservation cannot be optimized because their effectiveness is rarely evaluated. We surveyed 220 sites in 22 estuaries in the Moreton Bay Marine Park, Queensland, Australia, including all six current estuarine marine reserves within the park. Fishes were surveyed using one hour deployments of baited remote underwater video stations twice at each site over consecutive days. We show that although the estuarine reserves in Moreton Bay contain a significantly different fish community, they fail to enhance the abundance of harvested fish species. We posit that performance is limited because reserves protect unique spatial features, or conserve narrow estuaries with weak connections to mangrove habitats and the open sea. Consequently, reserves as currently positioned protect only a subset of potential environmental conditions present for fish within the region, and potentially support residual estuarine habitats (i.e. expansive intertidal flats or shallow creeks) which are not particularly significant to either fish or fishers. We argue that reserve effectiveness can be improved by conserving deeper estuaries, with diverse habitats for fish and strong connections to the open sea. Without incorporating these critical spatial considerations into estuarine reserve design, estuarine reserves are doomed to fail.

1. Introduction

"It is not when truth is dirty, but when it is shallow, that the lover of knowledge is reluctant to step into its waters."

Friedrich Nietzsche.

Nature reserves have been created globally to conserve biodiversity, supplement populations of harvested species, and maintain ecosystem functioning (Wood et al., 2008; Boonzaier and Pauly, 2016). Today, the capacity for reserves to increase the abundance of harvested species within their boundaries is well established (Mosqueira et al., 2000; Brashares et al., 2001; Allan et al., 2005). Strategically placed and well-enforced reserves in some marine (e.g. Edgar et al., 2014), freshwater (e.g. Humphries and Winemiller, 2009) and terrestrial (e.g. Joppa et al., 2008) ecosystems can increase the abundance and biomass of harvested species within their boundaries, and drive trophic cascades that alter the ecological condition and functioning of entire ecosystems (e.g. Ripple and Beschta, 2007).

In coastal settings, reserves are often considered the primary tool for conserving biodiversity and species, but their effectiveness has rarely been evaluated in some seascapes (Ban et al., 2014; Schlacher et al., 2015; Olds et al., 2016). This is particularly the case for estuaries, which are surprisingly underrepresented in the spatial conservation literature relative to coral and rocky reefs (see Winberg and Davis, 2014). Estuaries are significantly impacted by the effects of growing coastal cities and populations (e.g. harvesting, habitat loss and degradation) (Barbier et al., 2011). Consequently, estuarine conservation is now considered a management priority (Winberg and Davis, 2014). However, because estuarine reserve effectiveness is rarely reported on, we lack the empirical data that is required to optimize conservation outcomes (Sala et al., 2002; Huijbers et al., 2015).

Reserves usually carry costs for fisheries, mining and other economic activities (Halpern et al., 2013; Klein et al., 2013; Stigner et al., 2016). Attempts at reducing such costs can lead to reserves being placed in *residual* locations, meaning that impacts on industries are lessened, but that conservation outcomes are also poor (Pressey and Bottrill, 2008). Residual reserves might be common in estuaries when massive pressures from fishing and land development relegate reserves to locations that are isolated, shallow, with low habitat diversity and of

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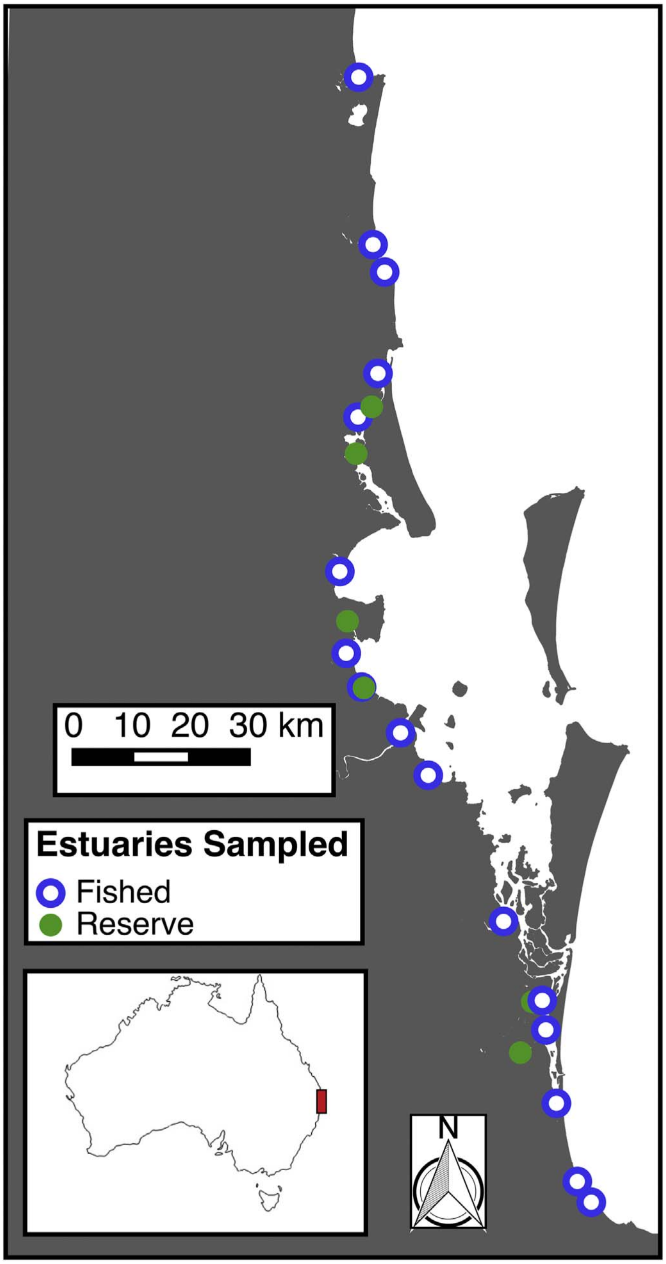

Fig. 1. Map showing the position of estuaries sampled in southeast Queensland, Australia, and their status as fished or reserve estuaries.

lower value to humans (see Devillers et al., 2015).

In this paper we evaluate the effectiveness of a network of six estuarine reserves in the Moreton Bay Marine Park (MBMP) in eastern Australia by testing whether reserves differ from fished reserves in terms of fish community structure and the abundance of harvested fishes. We then determine whether reserves contain a subset of the broader estuarine habitats in the region, and identify spatial attributes of estuaries that influence reserve effectiveness. Protection from fishing is expected to increase the abundance of harvested species inside reserves, and we hypothesise that reserve effectiveness will differ with variation in the spatial properties of estuarine seascapes.

2. Material and methods

2.1. Study design

The marine park was established in 1993 with one reserve in estuaries. In 2008, five additional reserves were added in estuaries as part of an expansion of reserves: we surveyed all six of these estuarine reserves (Fig. 1). All reserves that we sampled are fully no-take marine reserves (i.e. no extractive industries allowed) and are policed by three government agencies. Current reserves are selected primarily on the basis of conserving a minimum of 10% of each of the 16 recognized habitat types within the bay (e.g. sandy channels, mangrove, intertidal flats) within reserves, along with a suite of eight additional biophysical and four socio-economic guiding principles (Queensland Government, 2007). All estuaries are permanently open to the ocean. Pilot surveys of the estuarine reserves indicated that some were characterized by a distinct sub-set of habitat features, such as very large mangrove stands, wide intertidal mudflats, and shallow tidal channels. As a consequence, there are no fished and reserve locations available that could be paired as strict controls for habitat features. Thus, we sampled the full spectrum of estuarine habitats across the region, encompassing 16 fished estuaries in addition to the six reserves (Fig. 1). This design resulted in us sampling all estuarine systems wider than 100 m in high tide width in the region. Therefore, we sampled all estuaries that are likely large enough to potentially support an estuarine reserve.

Estuaries were surveyed in random order between June and August 2015. Fishes were surveyed at ten sites in each of the 22 estuaries twice over two consecutive days. Because salinity is a primary determinant of fish distributions in estuaries, we standardised for salinity by evenly spreading the ten sites from the estuary mouth upstream to where salinity had decreased to 30 psu (based on 10 years salinity data for each estuary; HWMP, 2016). The key harvested species in estuaries within this region (especially bream *Acanthopagrus australis*, moses perch *Lutjanus russelli*, mullet *Mugil cephalus*, and species of whiting *Sillago* spp. and flathead *Platycephalus* spp.; Webley et al., 2015) occur primarily within the lower estuary as they either spawn in these areas, or require linkages to the ocean for spawning migrations (e.g. Pollock, 1982; Davis et al., 2015). Consequently, the distribution of our sites along primarily marine salinities encompasses the majority of these species' ranges within estuaries in the region. Reserves always extend from the estuary mouth to a reserve-specific distance upstream that was always further than our 30 psu upper sampling limits. All sites were located over unvegetated muddy or sandy bottoms, in water depths between 1.5 and 2 m and within 30 m of adjacent mangroves to control for seascape scale effects (for example, see Martin et al., 2015).

2.2. Fish surveys

We used one hour deployments of baited remote underwater video stations (BRUVS) to survey fish communities at each site. BRUVS were constructed of a 3 kg weight and a 20 mm PVC pipe to attach baits at a fixed distance of 50 cm from a GoPro camera recording in high definition. Baits consisted of ~500 g pilchards *Sardinops sagax* placed into a 20 × 30 cm mesh bag with 0.5 cm^2 openings. A 20 × 20 cm visibility calibration disk was placed 1 m from the camera and used to quantify visibility. The disk had three vertical stripes (6.6 cm wide) of white, grey and black paint. When analysing footage, observers noted which stripes were seen and this was used to index visibility (i.e. white only = low visibility, white and grey = moderate, white, grey and black = high): we found that the composition of fish assemblages did not differ significantly between classes of visibility (permutational multivariate analysis of variance; $p > 0.15$) and hence visibility was not included in further analyses. Each video was analysed by counting the maximum number of individuals of each fish species that was visible between the camera and the above-described visibility disk (i.e. *MaxN*). Given the distance between sites (> 250 m) we considered it unlikely that the same individual was sampled at more than one site on the same day.

Table 1
Environmental attributes included in statistical models, and their definitions. All data sourced from Queensland Government regional ecosystem and land use mapping layers (Queensland Government, 2014).

Factor	Definition
Urban shoreline	Proportion of the sampled length of the estuary whose shoreline can be classified as non-natural (e.g. jetty, rock groyne, housing, enriched beach)
Length supporting mangroves	Distance from the mouth of the estuary to the most upstream mangroves in the estuary
Mangroves	Area (in ha) of mangroves in the sampled stretch of the estuary, corrected for the length of the estuary supporting mangroves.
Intertidal flats	Proportion of the sampled stretch of the estuary that is classified as intertidal sandbanks, intertidal flats, and flood and ebb tide delta.
High tide mouth width	Width of the estuary mouth at highest astronomical tide, from bank to bank, usually from the edge of the mangrove fringe or to the level of highest astronomical tide on artificial banks.
Low tide mouth width	Width of the estuary mouth at lowest astronomical tide from bank to bank, usually from the bank of sandy intertidal flats.
High tide average estuary width	Average width of the estuary at the ten sampling points at highest astronomical tide, usually from the edge of the mangrove fringe or to the level of highest astronomical tide on artificial banks.
Low tide average estuary width	Average width of the estuary at the ten sampling points at lowest astronomical tide, usually from the bank of sandy intertidal flats.

2.3. Environmental attributes of estuaries

The environmental variables measured and analysed represented attributes known to influence the abundance of fishes in estuaries: habitat type and extent (e.g. mangrove area, intertidal flats) and estuary size (e.g. the width of the estuary at the mouth and throughout the estuarine stretch of the waterway) (see Table 1 for further detail, justification and data sources). We did not include a metric of channel depth in our analyses because channel depth is highly temporally variable, making accurate bathymetry maps difficult to obtain. Further, estuarine depth and distance to adjacent deeper waters is a poor metric for connectivity between habitats in this shallow system (see Gilby et al., 2016). Given these points, aerial metrics relating to estuarine channel widths at various states of the tide and the extent of intertidal sans flats are considered a better metric of depth within this system (Meyer and Posey, 2009; Lacerda et al., 2014; Becker et al., 2016).

2.4. Statistical analyses - fish communities

The effect of reserves on the structure of fish communities was determined using two-way permutational multivariate analysis of variance (PERMANOVA; $n = 22$ estuaries × 10 sites × 2 days = 440) calculated on Modified Gower Log2 dissimilarity measures ($\alpha = 0.05$), and visualised using non-metric multidimensional scaling (nMDS) ordinations. Our two factors were 'reserve status' (fixed factor; 2 levels, fished and reserve estuaries) to determine the effects of reserves on estuarine fish, and 'sampling day' (random factor; two levels, two sampling days) to determine if fish communities differed between our two sampling days.

Species accounting for differences in community structure between fished estuaries and reserves were identified using the Dufrene-Legendre indicator species analysis (Dufrene and Legendre, 1997) in the labdsv package of R. Here, species are assigned higher indicator scores for the treatments in which they occur more often, and in higher abundance.

2.5. Statistical analyses - environmental attributes

We used distance-based linear modelling (DistLM; model based on stepwise selection and evaluated using Akaike's Information Criterion (AICc)) on Modified Gower Log2 dissimilarity measures and normalised environmental attributes to test for effects both of environmental attributes and reserves on the composition of fish assemblages. We used PERMANOVA to test for differences in environmental variables between reserves and fished estuaries (Euclidean distance; $n = 22$ estuaries; 'reserve status' = fixed factor, two levels, $n = 16$ fished and 6 reserve estuaries). Differences in environmental variables between estuaries were visualised using nMDS ordinations with overlaid bubble plots. As indicator species analyses account for both abundance and occurrence, they cannot be used for environmental attributes. Consequently, we used the similarity percentage analysis (SIMPER) on normalised environmental attributes data to determine which environmental attributes most contributed towards differences in the environmental attributes of reserve and fished estuaries. We used cluster analysis with similarity profile (SIMPROF) analysis to determine groupings of estuaries according to environmental attributes. Finally, we used linear regression to look for correlations between important environmental factors of interest.

3. Results

3.1. Effectiveness of reserves for fishes

We found no strong evidence that estuarine reserves enhance the abundance of harvested fish species in the region. Four species (yellow-fin bream *Acanthopagrus australis*, sea mullet *Mugil cephalus*, common toadfish *Marilyna pleurosticta* and weeping toadfish *Torquigener pleurogramma*) were good indicators of fished estuaries, and were more abundant in these than in reserves (Fig. 2B). By contrast, two species (estuary perchlet *Ambassis marianus* and blue catfish *Neoarius graeffei*) were good indicators of reserves, and were more abundant in these than fished estuaries (Fig. 2B), however, these species are not harvested in the region. Although the composition of fish assemblages differed between reserves and fished estuaries (PERMANOVA *d.f.* = 1, Pseudo-F = 22.71, $p \leq 0.001$; Fig. 2A), the abundance of yellow-fin bream and sea mullet (the two most heavily harvested species in the region) were significantly lower ($P < 0.05$) in reserves by 2.8 and 3.4 times, respectively. Only species that are not targeted by fisheries (e.g. blue catfish) were more numerous inside reserves (Fig. 2). Importantly, there was no main effect of sampling day on fish assemblages (PERMANOVA; *d.f.* = 1, Pseudo-F = 2.18, $p = 0.07$), and no interaction between sampling day and reserve status (*d.f.* = 1, Pseudo-F = 0.25, $p = 0.89$).

3.2. Environmental attributes of reserves

All environmental attributes explained a significant proportion of the variation in fish assemblage composition among estuaries (see Table A1 in Supporting information).

Reserves represent a distinct sub-set of habitat features (Fig. 3; PERMANOVA *d.f.* = 1, Pseudo-F = 3.01, $p = 0.027$). Compared with fished estuaries, reserves are less urbanised, extend over narrower and shorter channels, and encompass larger areas of mangroves and intertidal flats (Fig. 3A, Table 2). Reserve estuaries represent two distinct types of seascapes (Fig. A1): 1) broad shallow estuaries, with extensive intertidal flats and large variation in width between high and low tide (illustrated by estuary ii, Fig. 3B); and 2) small, narrow creeks (e.g. estuary iv, Fig. 3B).

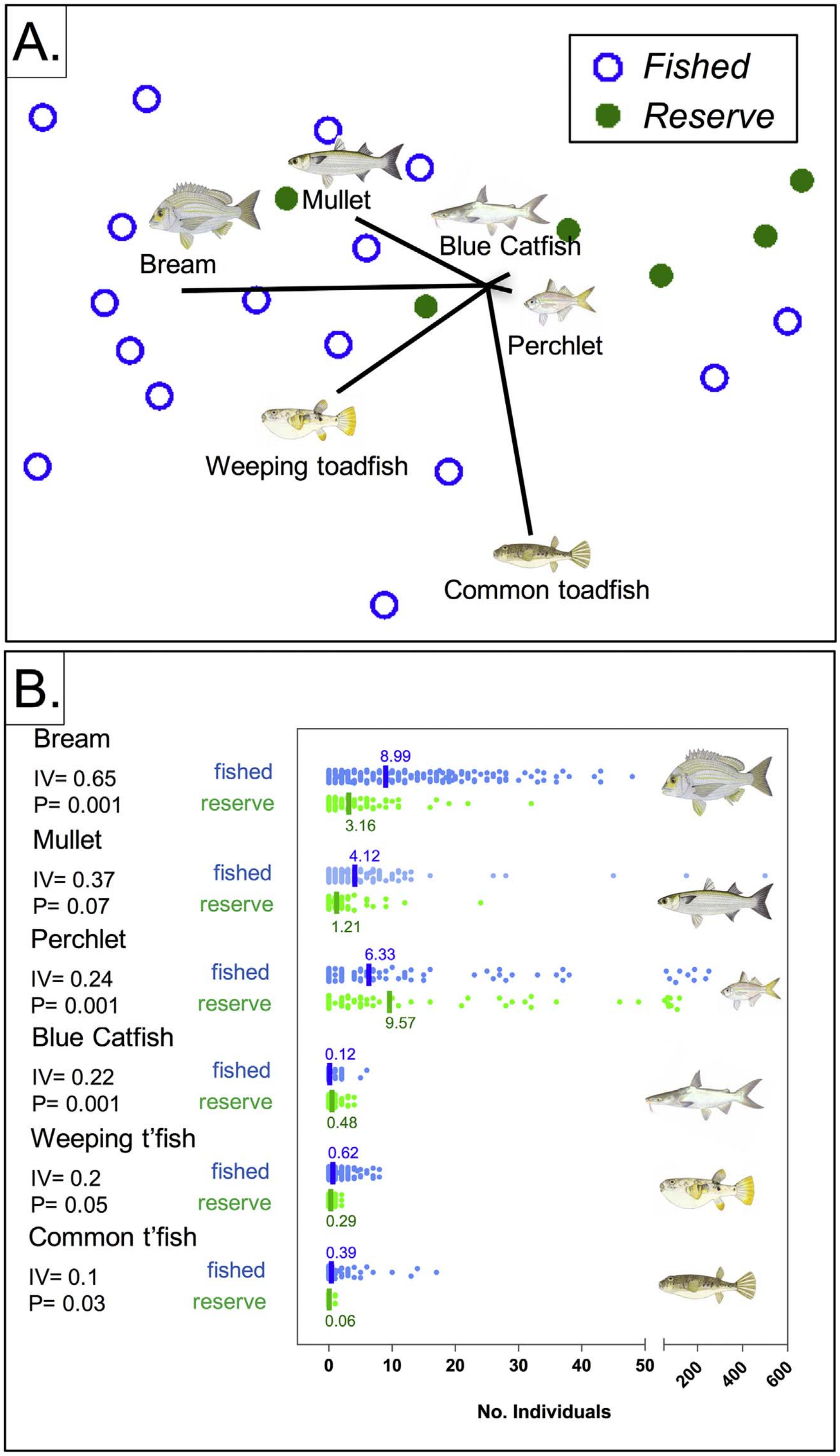

Fig. 2. Fish assemblages differ significantly between reserve and fished estuaries. A) These differences (as illustrated with nMDS ordination of centroid values for each estuary) are driven by most fish species are more abundant in fished estuaries than in reserves (B). Dots represent individual samples with mean values shown as horizontal bars and actual values. IV are indicators values (and associated p values) from Duffrene-Legendre indicator species analysis contrasting fished estuaries and reserves.

4. Discussion

Fish are heavily harvested by fishers in estuaries globally (Creighton et al., 2015), including several species in the marine park studied by us (Webley et al., 2015). Despite, the ecological role of estuaries as critical habitat for fishes and the economic importance of these areas to fisheries, estuaries are typically underrepresented in coastal conservation (Sala et al., 2002; Huijbers et al., 2015). We found that the reserves studied here do not enhance the abundance of harvested fish species within their boundaries. Estuarine reserves may fail to promote fish abundance and diversity if: (1) they support unique seascapes that are of little ecological value to fishes, (2) if reserves protect only a subset of environmental conditions present in estuaries within the region; or (2) they are placed in small creeks (short and narrow) where access to mangroves is limited and there is poor connectivity with the ocean. All of the estuarine reserves that we assessed support one of these two types of seascapes (i.e. shallow or small) and are, therefore, likely to be of limited value to both fishes and fishers; thus, these reserves conserve

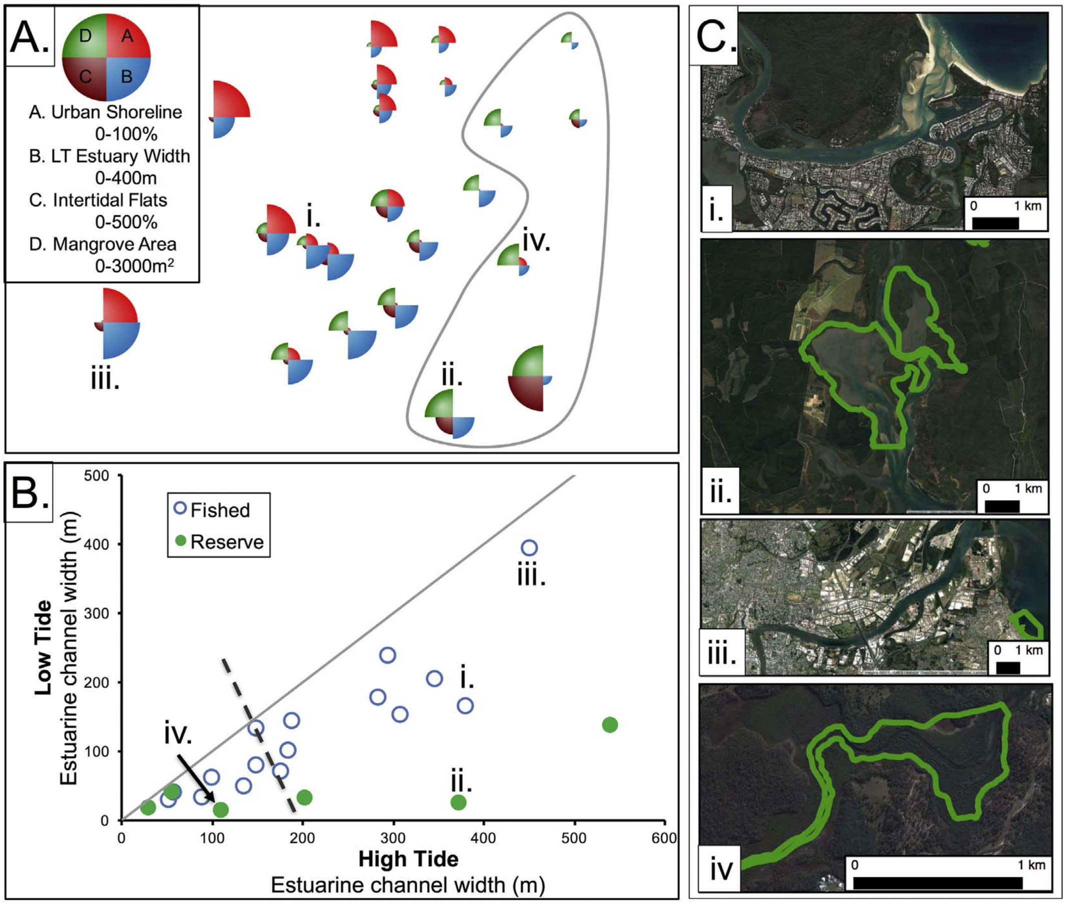

Fig. 3. Environmental attributes of reserve and fished estuaries. A) Bubble plots illustrate the four environmental attributes of estuaries that differed most between reserves and fished locations (Table 2). Grey ellipses indicate significant ($p < 0.05$) groupings. B) Linear regression of low tide estuarine channel width against high tide estuarine channel width. Reserve estuaries to the right of the dashed line grouped separately ($\alpha = 0.05$) to reserve estuaries to the right of the line in cluster analysis. C) Examples of reserves (shown with green boundaries) and fished estuaries. Labels i.–iv. show the position of example estuaries in panels A and B. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

residual estuarine locations (Rife et al., 2013; Pressey et al., 2015).

It appears that reserve selection in MBMP may have targeted locations that possess a ‘unique’ set of geomorphic features, especially broad and shallow areas with extensive sand flats (e.g. estuary ii. and other outliers in Fig. 3). This might reflect social or economic pressures, or a desire to conserve iconic marine habitats (Fernandes et al., 2005; Harris et al., 2008). Nevertheless, because these unique estuaries are of low value to fishers and contain seascapes that are of limited ecological value as fish habitat, this approach has resulted in the inclusion of residual estuaries in the reserve network.

Our results indicate that harvested species are more abundant in large estuaries, which have wide openings to the sea at low tide. Reserves were best characterized by lower urbanization, lower low tide estuary widths and a higher proportion of intertidal sand flat coverage. Such estuaries must therefore be considered as being less connected with the ocean, as fish movement with tides is more restricted than wider estuaries (Meyer and Posey, 2009; Lacerda et al., 2014; Becker et al., 2016). Seascapes within fished estuaries therefore have high flow rates and are better connected to other fish habitats in oceanic waters than the more isolated reserve seascapes (see Becker et al., 2016). Placing reserves in small estuaries with limited flow and poor connectivity to the ocean also fails to protect spawning sites for marine species and areas of high juvenile settlement at the mouths of larger estuaries (see Blaber, 2008). Thus, to adequately conserve estuarine seascapes, and protect critical fish habitats, reserves should also conserve a reasonable number of larger estuaries, which are well

Table 2

Similarity percentage analysis (SIMPER) output for differences in environmental attributes between reserve and fished estuaries. Mean values and 95% confidence intervals are provided for fished and reserve estuaries for each attribute to provide information on the range of values within Moreton Bay.

Variable	Fished		Reserve		SIMPER	
	Mean	(95% CI)	Mean	(95% CI)	Sq. Dist/SD	Contrib%
Urban shoreline (proportion)	26.6	(9.9–43.2)	0.8	(− 1.1–2.8)	1.11	15.67
LT average estuary width (m)	131	(80–182)	44	(− 5–93)	0.92	13.55
Intertidal flats (proportion)	0.2	(0.1–0.3)	1	(− 0.9–2.9)	0.92	12.76
Mangroves (ha)	286	(138–435)	997	(183–1810)	0.73	12.75
HT mouth width (m)	249	(164–333)	298	(34–563)	0.94	12.42
HT average estuary width (m)	208	(144–272)	219	(8–430)	0.9	11.9
Length supporting mangroves (m)	17,617	(8978–26,257)	6390	(3030–9751)	0.64	11.45
LT mouth width (m)	172	(83–261)	81	(23–139)	0.65	9.51

connected to downstream oceanic ecosystems (Sheaves and Johnston, 2008).

Because comprehensive spatial data are seldom available on the ecological attributes of species and ecosystems, conservation planning often uses surrogates, particularly geomorphic or other habitat surrogates (Lindenmayer et al., 2015). For example, protecting extensive mangrove forests is widely considered to be beneficial for estuarine species (Ley and Halliday, 2004). Unexpectedly, in our study, the large area of mangroves in some reserves did not result in higher numbers of harvested fish species. This may result from a lack of connectivity between mangroves and channels in shallow reserves, because some fish species do not use mangroves extensively (Baker et al., 2015; Sheaves et al., 2016), or because some fish, especially juveniles, inhabiting mangroves were not readily detected by our methods. Alternatively, other features of seascapes, such as low connectivity downstream might reduce the otherwise positive effects of mangroves. Selecting reserves in estuaries primarily on the basis of surrogates (such as mangroves or sandy estuarine bottoms in MBMP) may, therefore, not be particularly effective for fishes if key ecological processes (e.g. spawning, migration, ontogenetic habitat changes, predator refuges, nursery areas) are not also represented (Sheaves and Johnston, 2008; Nagelkerken et al., 2015). The use of habitat surrogates in coastal ecosystems also requires specific information on how estuarine fishes use these seascapes (Olds et al., 2012); for example, the degree to which structural complexity in mangrove forests structures fish assemblages (Sheaves et al., 2016). Thus, improving the performance of coastal reserves for fishes requires that we identify effective ecological surrogates for the assemblages which we seek to protect (Zacharias and Roff, 2001; Louzao et al., 2011; Sundblad et al., 2011; Shokri and Gladstone, 2013) which incorporate an understanding of how fish species utilize complex habitat mosaics (Olds et al., 2012).

5. Conclusion

Existing estuarine reserves in the MBMP may represent locations of low habitat value to fishes and hence do not significantly increase the abundance of harvested fish species. To improve reserve performance, we argue that future reserve configurations use ecological data to complement abiotic surrogates in the design process; this can be usefully extended to incorporate habitat suitability data for threatened species and those heavily fished. We also argue that expansion of the estuarine reserve network should protect reaches of high connectivity with the ocean, spawning sites, and habitats of elevated structural habitat complexity in the subtidal. Last but not least, expanding reserve coverage from shallow intertidal embayments to deeper waters mirrors a fundamental biological truth: *reserves that seek to enhance fish must contain water*.

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.biocon.2017.03.027.

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