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

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13 **Abstract**

14 Nature reserves are created to conserve biodiversity and restore populations of
15 harvested species, but it is not clear whether this strategy is successful in all
16 ecosystems. Reserves are gazetted in estuaries to offset significant impacts from
17 burgeoning human populations, however, coastal conservation cannot be optimized
18 because their effectiveness is rarely evaluated. We surveyed 220 sites in 22 estuaries in
19 the Moreton Bay Marine Park, Queensland, Australia, which included all six current
20 estuarine marine reserves within the park. Fish were surveyed using one hour
21 deployments of baited remote underwater video stations twice at each site over
22 consecutive days. We show that although the estuarine reserves in Moreton Bay contain
23 a significantly different fish community, they fail to enhance the abundance of harvested
24 fish species. We posit that performance is limited because reserves protect unique
25 spatial features, or conserve narrow estuaries with weak connections to mangrove
26 habitats and the open sea. Seascapes within reserves are, therefore, not *representative*
27 of estuaries in the region, and reserves primarily support *residual* estuarine habitats
28 (i.e. expansive intertidal flats or shallow creeks) which are not particularly significant to
29 either fish or fishers. We argue that reserve effectiveness can be improved by
30 conserving deeper estuaries, with diverse habitats for fish and strong connections to
31 both freshwater and the open sea. Without incorporating these critical spatial
32 considerations into estuarine reserve design, estuarine reserves are doomed to fail.

33

34 **Key words-** bathymetry, coastal management, fisheries, habitats, landscape ecology.

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 detrimental effects associated with 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). Reserves have been created widely to conserve that

critical ecosystem services that estuaries provide (Barbier et al. 2011; Perkol-Finkel et al. 2012), but because estuarine reserve effectiveness is rarely evaluated we lack the empirical data that is required to optimize conservation (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 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 enhance the abundance of harvested fishes. We then determine whether reserves contain a representative 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, and 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). Currently, a minimum of 10% of each of the 16 recognized habitat types within the bay (e.g. sandy channels, mangrove, intertidal flats) is conserved inside reserves. 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).

Estuaries were surveyed in random order between June and August 2015. Fish were surveyed at ten sites in each of the 22 estuaries twice over two consecutive days. Because salinity is a primary determinant of the distributions of fish 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 2015). 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 3kg weight, a 20 mm PVC pipe to attach baits at a fixed distance of 50 cm from the camera and a GoPro camera recording in high definition. Baits consisted of ~500 g pilchards *Sardinops sagax* placed into a 20 x 30 cm mesh bag with 0.5 cm² openings. A 20 x 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 that was positioned 1m from the camera (i.e. *MaxM*), thereby further minimising the potentially confounding effect of variable visibility. Given the distance between sites (>250m) we considered it unlikely that the same individual was sampled more than once on the same day.

1.3. 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).

2.4. Statistical Analyses- Fish Communities

The effect of reserves on the structure of fish communities was determined using permutational multivariate analysis of variance (PERMANOVA; 'reserve status'=fixed factor, two levels; n=22 estuaries x 10 sites x 2 days=440) on Modified Gower Log2 dissimilarity measures. There was no effect of sampling day on fish assemblages (PERMANOVA; 'day'=random factor, two levels; *d.f.*=1, Pseudo-F=2.07, p=0.09), so the factor of day was excluded from all subsequent analyses.

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 occur in higher abundance, and more often, in the treatments for which they are assigned higher indicator scores in these tests.

2.5. Statistical Analyses- Environmental Attributes

We used distance-based linear modelling (DistLM) to test for effects both of environmental attributes factors *and* reserves on the composition of fish assemblages and PERMANOVA to test for differences in environmental variables between reserves and fished estuaries (Euclidean distance similarity; n=22 estuaries; 'reserve status'=fixed factor, two levels, n=16 fished and 6 reserve estuaries). 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) to determine which environmental attributes most contributed towards differences in the environmental attributes of reserve and fished estuaries.

3. Results

3.1. Effectiveness of reserves for fish

We found no strong evidence that estuarine reserves enhance the abundance of harvested fish species in the region. Four species (yellow-fin bream, sea mullet, common toadfish and weeping toadfish) were good indicators of fished estuaries, and were more abundant in these than in reserves (Fig. 2B). By contrast, two species (estuary perchlet and blue catfish) 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=22.69, $p<0.001$; 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 small species that are not targeted by fisheries (e.g. blue catfish) were more numerous inside reserves (Fig. 2).

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).

Reserves represent a distinct sub-set of habitat features (Fig. 2; PERMANOVA $d.f.=1$, Pseudo-F=3.01, $p=0.027$) that are either not representative of the range of estuarine

conditions in the region, or support habitats that are mostly of low quality for fish. Compared with fished estuaries, reserves extend over narrower and shorter channels, encompass larger areas of mangroves and intertidal flats, but are significantly less urbanised (Fig. 3A, Table 2). Reserve estuaries do, however, contain two distinct types of seascapes: 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, with steep embankments that change little in size with tide variation (e.g. estuary iv, Fig. 3B).

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 fish 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 fish, and which that are not *representative* of estuarine habitats in the region; or (2) they are placed in small creeks (short and narrow) where access to mangroves is limited and there is poor connectivity to freshwaters and 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 fish and fishers; thus these reserves conserve *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. The ecological value of these unique reserve seascapes could be enhanced by modifying reserve boundaries so that they better encompass connectivity with deeper estuarine channels and conserve subtidal refuge habitat for fishes (Nagelkerken et al. 2015; Olds et al. 2016).

Our results indicate that harvested fish species are more abundant in large estuaries, which have wide opening to the sea and large, deep channels. These types of estuarine seascapes have high flow rates and are, therefore, better connected to other fish habitats in fresh and oceanic waters than the more isolated seascapes in which some reserves are located. Placing reserves in small estuaries, with limited flow and poor connectivity to freshwater and the ocean also fails to protect spawning sites and areas of high juvenile settlement at the mouths of larger estuaries (see Blaber 2008). Thus, to adequately represent estuarine seascapes, and protect critical fish habitats, reserves should also conserve a reasonable number of larger estuaries, which are well connected to both upstream freshwater habitats and downstream oceanic ecosystems (Sheaves and Johnston 2008).

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

5. Conclusion

Existing estuarine reserves in the MBMP may represent locations of low habitat value to fish 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.*

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7. References

- Allan, J.D., Abell, R., Hogan, Z., Revenga, C., Taylor, B.W., Welcomme, R.L., Winemiller, K., 2005. Overfishing of inland waters. *Bioscience* 55, 1041-1051.
- Baker, R., Sheaves, M., Johnston, R., 2015. Geographic variation in mangrove flooding and accessibility for fishes and nektonic crustaceans. *Hydrobiologia* 762, 1-14.
- Ban, N.C., Bax, N.J., Gjerde, K.M., Devillers, R., Dunn, D.C., Dunstan, P.K., Hobday, A.J., Maxwell, S.M., Kaplan, D.M., Pressey, R.L., Ardron, J.A., Game, E.T., Halpin, P.N., 2014. Systematic conservation planning: A better recipe for managing the high seas for biodiversity conservation and sustainable use. *Conservation Letters* 7, 41-54.
- Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs* 81, 169-193.
- Blaber, S.J., 2008. Tropical estuarine fishes: ecology, exploration and conservation. John Wiley & Sons.
- Boonzaier, L., Pauly, D., 2016. Marine protection targets: an updated assessment of global progress. *Oryx* 50, 27-35.
- Brashares, J.S., Arcese, P., Sam, M.K., 2001. Human demography and reserve size predict wildlife extinction in West Africa. *Proceedings of the Royal Society B-Biological Sciences* 268, 2473-2478.
- Creighton, C., Boon, P.I., Brookes, J.D., Sheaves, M., 2015. Repairing Australia's estuaries for improved fisheries production - what benefits, at what cost? *Marine and Freshwater Research* 66, 493-507.

291 Devillers, R., Pressey, R.L., Grech, A., Kittinger, J.N., Edgar, G.J., Ward, T., Watson, R., 2015.
 292 Reinventing residual reserves in the sea: are we favouring ease of establishment
 293 over need for protection? *Aquatic Conservation: Marine and Freshwater*
 294 *Ecosystems* 25, 480-504.

295 Dufrene, M., Legendre, P., 1997. Species assemblages and indicator species: the need for
 296 a flexible asymmetrical approach. *Ecological Monographs* 67, 345-366.

297 Edgar, G.J., Stuart-Smith, R.D., Willis, T.J., Kininmonth, S., Baker, S.C., Banks, S., Barrett,
 298 N.S., Becerro, M.A., Bernard, A.T., Berkhout, J., Buxton, C.D., Campbell, S.J., Cooper,
 299 A.T., Davey, M., Edgar, S.C., Forsterra, G., Galvan, D.E., Irigoyen, A.J., Kushner, D.J.,
 300 Moura, R., Parnell, P.E., Shears, N.T., Soler, G., Strain, E.M., Thomson, R.J., 2014.
 301 Global conservation outcomes depend on marine protected areas with five key
 302 features. *Nature* 506, 216-220.

303 Fernandes, L., Day, J., Lewis, A., Slegers, S., Kerrigan, B., Breen, D., Cameron, D., Jago, B.,
 304 Hall, J., Lowe, D., Innes, J., Tanzer, J., Chadwick, V., Thompson, L., Gorman, K.,
 305 Simmons, M., Barnett, B., Sampson, K., De'ath, G., Mapstone, B., Marsh, H.,
 306 Possingham, H., Ball, I., Ward, T., Dobbs, K., Aumend, J., Slater, D., Stapleton, K.,
 307 2005. Establishing representative no-take areas in the Great Barrier Reef: Large-
 308 scale implementation of theory on marine protected areas. *Conservation Biology*
 309 19, 1733-1744.

310 Halpern, B.S., Klein, C.J., Brown, C.J., Beger, M., Grantham, H.S., Mangubhai, S.,
 311 Ruckelshaus, M., Tulloch, V.J., Watts, M., White, C., Possingham, H.P., 2013.
 312 Achieving the triple bottom line in the face of inherent trade-offs among social
 313 equity, economic return and conservation. *Proceedings of the National Academy*
 314 *of Sciences* 110, 6229- 6234.

315 Harris, P.T., Heap, A.D., Whiteway, T., Post, A., 2008. Application of biophysical
 316 information to support Australia's representative marine protected area
 317 program. *Ocean & Coastal Management* 51, 701-711.

318 Huijbers, C.M., Connolly, R.M., Pitt, K.A., Schoeman, D.S., Schlacher, T.A., Burfeind, D.D.,
 319 Steele, C., Olds, A.D., Maxwell, P.S., Babcock, R.C., Rissik, D., 2015. Conservation
 320 benefits of marine reserves are undiminished near coastal rivers and cities.
 321 *Conservation Letters* 8, 312-319.

322 Humphries, P., Winemiller, K.O., 2009. Historical Impacts on River Fauna, Shifting
 323 Baselines, and Challenges for Restoration. *Bioscience* 59, 673-684.

324 HWMP, 2015. Healthy Waterways Monitoring Program,
 325 <http://www.healthywaterways.org/>. Accessed July 2015.

326 Joppa, L.N., Loarie, S.R., Pimm, S.L., 2008. On the protection of "protected areas".
 327 *Proceedings of the National Academy of Sciences of the United States of America*
 328 105, 6673-6678.

329 Klein, C.J., Tulloch, V.J., Halpern, B.S., Selkoe, K.A., Watts, M.E., Steinback, C., Scholz, A.,
 330 Possingham, H.P., 2013. Tradeoffs in marine reserve design: habitat condition,
 331 representation, and socioeconomic costs. *Conservation Letters* 6, 324-332.

332 Ley, J.A., Halliday, I.A., 2004. A key role for marine protected areas in sustaining a
 333 regional fishery for Barramundi Lates calcarifer in Mangrove-dominated
 334 estuaries? Evidence from northern Australia, In *Aquatic Protected Areas as*
 335 *Fisheries Management Tools*. ed. J.B. Shipley, pp. 225-236.

336 Lindenmayer, D., Pierson, J., Barton, P., Beger, M., Branquinho, C., Calhoun, A., Caro, T.,
 337 Greig, H., Gross, J., Heino, J., Hunter, M., Lane, P., Longo, C., Martini, K., McDowell,
 338 W.H., Mellin, C., Salo, H., Tulloch, A., Westgate, M., 2015. A new framework for

339 selecting environmental surrogates. *Science of the Total Environment* 538, 1029-
340 1038.

341 Louzao, M., Pinaud, D., Peron, C., Delord, K., Wiegand, T., Weimerskirch, H., 2011.
342 Conserving pelagic habitats: seascape modelling of an oceanic top predator.
343 *Journal of Applied Ecology* 48, 121-132.

344 Martin, T.S.H., Olds, A.D., Pitt, K.A., Johnston, A.B., Butler, I.R., Maxwell, P.S., Connolly,
345 R.M., 2015. Effective protection of fish on inshore coral reefs depends on the
346 scale of mangrove-reef connectivity. *Marine Ecology Progress Series* 527, 157-
347 165.

348 Mosqueira, I., Cote, I.M., Jennings, S., Reynolds, J.D., 2000. Conservation benefits of
349 marine reserves for fish populations. *Animal Conservation* 3, 321-332.

350 Nagelkerken, I., Sheaves, M., Baker, R., Connolly, R.M., 2015. The seascape nursery: a
351 novel spatial approach to identify and manage nurseries for coastal marine
352 fauna. *Fish and Fisheries* 16, 362-371.

353 Olds, A.D., Connolly, R.M., Pitt, K.A., Maxwell, P.S., 2012. Habitat connectivity improves
354 reserve performance. *Conservation Letters* 5, 56-63.

355 Olds, A.D., Connolly, R.M., Pitt, K.A., Pittman, S.J., Maxwell, P.S., Huijbers, C.M., Moore,
356 B.R., Albert, S., Rissik, D., Babcock, R.C., Schlacher, T.A., 2016. Quantifying the
357 conservation value of seascape connectivity: a global synthesis. *Global Ecology*
358 *and Biogeography* 25, 3-15.

359 Perkol-Finkel, S., Ferrario, F., Nicotera, V., Airoidi, L., 2012. Conservation challenges in
360 urban seascapes: promoting the growth of threatened species on coastal
361 infrastructures. *Journal of Applied Ecology* 49, 1457-1466.

362 Pressey, R.L., Bottrill, M.C., 2008. Opportunism, threats, and the evolution of systematic
363 conservation planning. *Conservation Biology* 22, 1340-1345.

364 Pressey, R.L., Visconti, P., Ferraro, P.J., 2015. Making parks make a difference: poor
 365 alignment of policy, planning and management with protected-area impact, and
 366 ways forward. *Philosophical Transactions of the Royal Society B-Biological*
 367 *Sciences* 370, 7420-7425.

368 Queensland Government, 2014. Queensland landuse mapping program- southeast
 369 Queensland natural resource management region. Queensland Government,
 370 Brisbane, Australia.

371 Rife, A.N., Erisman, B., Sanchez, A., Aburto-Oropeza, O., 2013. When good intentions are
 372 not enough ... Insights on networks of "paper park" marine protected areas.
 373 *Conservation Letters* 6, 200-212.

374 Ripple, W.J., Beschta, R.L., 2007. Restoring Yellowstone's aspen with wolves. *Biological*
 375 *Conservation* 138, 514-519.

376 Sala, E., Aburto-Oropeza, O., Paredes, G., Parra, I., Barrera, J.C., Dayton, P.K., 2002. A
 377 general model for designing networks of marine reserves. *Science* 298, 1991-
 378 1993.

379 Schlacher, T.A., Weston, M.A., Lynn, D., Schoeman, D.S., Huijbers, C.M., Olds, A.D.,
 380 Masters, S., Connolly, R.M., 2015. Conservation gone to the dogs: when canids
 381 rule the beach in small coastal reserves. *Biodiversity and Conservation* 24, 493-
 382 509.

383 Sheaves, M., Johnston, R., 2008. Influence of marine and freshwater connectivity on the
 384 dynamics of subtropical estuarine wetland fish metapopulations. *Marine Ecology*
 385 *Progress Series* 357, 225-243.

386 Sheaves, M., Johnston, R., Baker, R., 2016. Use of mangroves by fish: new insights from
 387 in-forest videos. *Marine Ecology Progress Series* 549, 167-182.

388 Shokri, M.R., Gladstone, W., 2013. Limitations of habitats as biodiversity surrogates for
 389 conservation planning in estuaries. *Environmental Monitoring and Assessment*
 390 185, 3477-3492.

391 Stigner, M.G., Beyer, H.L., Klein, C.J., Fuller, R.A., 2016. Reconciling recreational use and
 392 conservation values in a coastal protected area. *Journal of Applied Ecology* 53,
 393 1206-1214.

394 Sundblad, G., Bergstrom, U., Sandstrom, A., 2011. Ecological coherence of marine
 395 protected area networks: a spatial assessment using species distribution models.
 396 *Journal of Applied Ecology* 48, 112-120.

397 Webley, J., McInnes, K., Teixeira, D., Lawson, A., Quinn, R., 2015. Statewide Recreational
 398 Fishing Survey 2013-14. Queensland Government, Brisbane, Australia.

399 Winberg, P.C., Davis, A.R., 2014. Ecological response to MPA zoning following cessation
 400 of bait harvesting in an estuarine tidal flat. *Marine Ecology Progress Series* 517,
 401 171-180.

402 Wood, L.J., Fish, L., Laughren, J., Pauly, D., 2008. Assessing progress towards global
 403 marine protection targets: shortfalls in information and action. *Oryx* 42, 340-351.

404 Zacharias, M.A., Roff, J.C., 2001. Use of focal species in marine conservation and
 405 management: a review and critique. *Aquatic Conservation-Marine and*
 406 *Freshwater Ecosystems* 11, 59-76.

Tables

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 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 our 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 our ten sampling points at lowest astronomical tide, usually from the bank of sandy intertidal flats.

Table 2. Similarity percentage analysis (SIMPER) output for differences in environmental attributes between reserve and fished estuaries. P values in bold are significant at 0.05.

Variable	Fished		Reserve		PERMANOVA		SIMPER	
	Mean	(95% CI)	Mean	(95% CI)	Pseudo F	P	Sq.Distance/SD	Contrib%
Mangroves (ha)	286	(138 - 435)	997	(183 - 1810)	10.56	0.01	0.73	12.75
LT Average Estuary Width (m)	131	(80- 182)	44	(-5 - 93)	4.45	0.04	0.92	13.55
Urban Shoreline (proportion)	26.6	(9.9 - 43.2)	0.8	(-1.1 - 2.8)	3.95	0.05	1.11	15.67
Intertidal Flats (proportion)	0.2	(0.1 - 0.3)	1	(-0.9 - 2.9)	3.28	0.08	0.92	12.76
Length supporting mangroves (m)	17617	(8978 - 26257)	6390	(3030 - 9751)	2.75	0.09	0.64	11.45
LT Mouth Width (m)	172	(83- 261)	81	(23 - 139)	1.66	0.19	0.65	9.51
HT Mouth Width (m)	249	(164 - 333)	298	(34 - 563)	0.31	0.58	0.94	12.42
HT Average Estuary Width (m)	208	(144 - 272)	219	(8 - 430)	0.02	0.89	0.90	11.9

Figure Captions

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

Figure 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, however, not as expected: most fish species are more abundant in fished estuaries than in reserves (B). In panel 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.

Figure 3. Environmental attributes of reserve and fished estuaries. A) reserve estuaries (enclosed by grey ellipse over nMDS ordination) have significantly different environmental attributes to fished estuaries. Star plots illustrate the four environmental attributes of estuaries that differed most between reserves and fished locations (Table 2). B) there are two types of reserves in estuaries of the MBMP: those with extensive intertidal flats that cause large differences in the width of estuaries between low and high tide (100s of m); and those that encompass small creeks with little difference in width between low and high tide. 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.

A.

B.

Bream

IV= 0.65

P= 0.001

fished

reserve

Mullet

IV= 0.37

P= 0.07

fished

reserve

Perchlet

IV= 0.24

P= 0.001

fished

reserve

Blue Catfish

IV= 0.22

P= 0.001

fished

reserve

Weeping t'fish

IV= 0.2

P= 0.05

fished

reserve

Common t'fish

IV= 0.1

P= 0.03

fished

reserve

Gilby et al. Biological Conservation Highlights

- The effects of nature reserves on communities remains unclear for some ecosystems
- Estuarine reserves cannot be optimised because their effectiveness is rarely tested
- Estuarine reserves in Moreton Bay Marine Park fail to enhance harvested fish
- We show that this is because they're placed in shallow, narrow, residual areas
- Improving reserves in estuaries requires excellent integration of spatial ecology

Table S1. Results of distance based linear model (DistLM) relating the composition of estuarine fish assemblages with the conservation status of estuaries (i.e. reserve or fished) and variation in the environmental attributes of estuaries (Table 2). Model based on stepwise selection and evaluated using Aikake's Information Criterion (AICc).

Factor	SS	Pseudo-F	P	Proportion explained
Mangroves	43.933	24.955	0.001	5.39E-02
Marine reserve status	40.141	22.689	0.001	4.93E-02
High Tide Estuary Width	33.26	18.635	0.001	4.08E-02
High Tide Average Estuary Width	25.026	13.875	0.001	3.07E-02
Low Tide Estuary Width	13.458	7.3537	0.001	1.65E-02
Length supporting mangroves	8.8265	4.7953	0.001	1.08E-02
Sandflats	7.3116	3.9648	0.01	8.97E-03
Low Tide Average Estuary Width	7.0277	3.8096	0.011	8.62E-03
Urban Shoreline	6.7157	3.639	0.015	8.24E-03