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# Umbrellas can work under water: Using threatened species as indicator and management surrogates can improve coastal conservation

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Species surrogates, the use of particular species to index habitat condition or to represent ecological assemblages are commonly identified in many ecosystems, but are less tested, and therefore less employed in estuaries. Estuaries provide important ecosystem goods (e.g. harvestable species) and services (e.g. carbon processing, coastal armouring), but require protection from multiple human activities, meaning that finding surrogates for estuarine condition or faunal assemblages is a significant knowledge gap. In this study, we test the efficacy of the threatened estuary ray *Hemitrygon fluviorum*, as a suitable indicator of ecosystem condition and management umbrella surrogate species for conservation prioritisation and monitoring purposes within estuaries. We surveyed fish assemblages and ray presence at ten sites within each of 22 estuaries in southeast Queensland, Australia, using one hour deployments of baited video arrays. We then tested for correlations between ray presence, a series of environmental variables considered important to ecosystem management within estuaries (i.e. testing rays as indicator species), and the co-occurring fish species (i.e. testing rays as umbrella species). Estuary rays function as both umbrella species and ecological indicators of habitat status in subtropical Australian estuaries. As umbrellas, ray occurrence concords with elevated species richness. As ecological indicators, ray distribution concords with habitats of good water quality (especially low turbidity) and more natural vegetation remaining in the catchment. These results highlight the potential for other threatened aquatic vertebrates that are both readily detectable and that are reliable proxies for ecosystems status to be become useful management tools in estuaries. The protection of such large, threatened species in coastal seascapes allows managers to address multiple targets for conservation, especially; (1) protecting species of conservation concern; (2) maintaining diversity; and (3) protecting optimal habitats by better placing reserves.



## 1. Introduction

Ecological surrogates are used to represent other ecosystem components or ecosystem functioning (Caro and O'Doherty, 1999; Lindenmayer et al., 2015; Tulloch et al., 2016). There are two broad types of ecological surrogacy; (1) indicator surrogates, which provide information about ecological systems, and (2) management surrogates, which are used to facilitate management goals like no-take reserves and other spatial management techniques (Hunter et al., 2016). Judicious surrogate choice can help to optimise the performance of conservation actions (e.g. selecting sites for reserves) and improve the cost-effectiveness of monitoring programs (Caro and O'Doherty, 1999; Siddig et al., 2016). It is, therefore, imperative that we identify surrogates in ecosystems that are subjected to intense human disturbance, or that are of particular conservation significance.

Within the broader groupings of indicator and management

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surrogates, there are multiple surrogate concepts (Hunter et al., 2016; Lindenmayer et al., 2015). Management surrogates are used to facilitate management goals, especially the maintenance or enhancement of biodiversity (Hunter et al., 2016). One type of management surrogate are umbrella species; species whose preservation extends conservation benefits to the majority of co-occurring species (Hunter et al., 2016). Candidate umbrella species are highly detectable species whose abundance and/or distribution correlates with higher species diversity or other attributes of assemblages or ecosystems that are of conservation interest (see Fleishman et al., 2001), especially when umbrella species range over large, and heterogeneous land- or seascapes (Roberge and Angelstam, 2004). Conversely, ecological indicator species are species whose presence or abundance provide information about ecosystems (Hunter et al., 2016; Lindenmayer et al., 2015; Siddig et al., 2016). Although indicator species are an established tool for monitoring in both terrestrial and marine ecosystems (Hilty and Merenlender, 2000; Pearson and Rosenberg, 1977), and are readily applied in both (Siddig et al., 2016), their efficacy and generality for some marine environments remains uncertain, especially for highly-impacted ecosystems like estuaries (Shokri et al., 2007). Further, there are uncertainties regarding the broad applicability of species-based indicators of ecosystem condition, as it can sometimes be difficult to disentangle natural variability in patterns from human- or management-related changes (Carignan and Villard, 2002; Saetersdal et al., 2005).

Coastal ecosystems are under intense pressure globally from escalating human activity (Halpern et al., 2008). Many estuaries, as significant sites of human settlements, commerce and transport, are subjected to declining water quality, large volumes of terrestrial runoff, habitat loss and modification, and intense fishing (Barbier et al., 2011; Kennish, 2002). The lower reaches of estuaries, in particular, are focal points for human influences on coastal systems as the effects of catchment alterations, habitat loss and pollutant release upstream can concentrate in downstream areas, thereby reducing ecosystem condition (Basnyat et al., 1999; Rodriguez-Iruretagoiena et al., 2016). Identifying effective surrogates for coastal and estuarine environments may, therefore, help to optimise the performance, and cost-effectiveness, of coastal conservation actions (e.g. reserves) and monitoring programs (Shokri et al., 2007; Zacharias and Roff, 2001), especially those that incorporate the critical concepts of habitat quantity and habitat connectivity (Magris et al., 2016; Olds et al., 2014). Further, it has been shown that using solely habitat-based surrogates might not fully represent the subtleties of estuarine fish ecology and habitat requirements, resulting in reserves that are ineffective for fish (Gilby et al., 2017) and that deriving responses of faunal communities from physico-chemical monitoring metrics is difficult given non-linear and species-specific responses (Bunn et al., 2010; Jonzen et al., 2005; Logan and Taffs, 2014). Using species who are under threat as surrogate species for the implementation of spatial conservation techniques such as marine reserves has the added benefit of potentially protecting or managing for the threatened species itself, the surrounding fish assemblage, and areas of higher quality habitats. However, few studies have explicitly set out to test the efficacy of surrogate concepts in these impacted estuarine environments (see, however, Shokri et al., 2009, Shokri et al., 2007). As a consequence, spatial management cannot be optimised, resulting in poor outcomes for reserves in some systems (Gilby et al., 2017). Recent commentaries on the optimisation of coastal management theories have developed conceptual frameworks that seek to identify the human activities responsible for environmental change (i.e. *drivers*), the resulting environmental *pressures* and changes in ecosystem *state*, and then seek to identify the *impacts* on society and potential *responses* (the DPSIR framework) (Elliott et al., 2017). By identifying such indicator and umbrella species in estuaries, we can more effectively identify the *pressures* under which the system's fauna is under, how these pressures change one of the key components of the ecosystems (i.e. the *state* of the fish community), and then prioritise the areas which should be the focus of either catchment revegetation, or should be considered for marine reserves (i.e. *responses*).

In this study, we test whether a large (adults >45 cm disc width) stingray, estuary ray *Hemitrygon fluviorum*, is a suitable surrogate species. Occurring in coastal and estuarine systems across central eastern Australia (IUCN, 2015), this species is thought to be particularly sensitive to habitat loss and declining water quality (IUCN, 2015; Pierce and Bennett, 2010; Pogonoski et al., 2002). Whilst fishing pressure might have historically been a principal threat to this species, their listing on federal and international endangered species lists prevents their removal, targeting, or consumption in Australia (Pogonoski et al., 2002). There have been no reports of this species being removed for consumption in this region (Webley et al., 2015); it is, however, often caught as bycatch by recreational fishers (BG, personal observations). Given these biological attributes, we hypothesise that the estuary ray might be both a useful ecological indicator species (a type of *indicator surrogate*), and umbrella species (a type of *management surrogate*) for nearshore, coastal seascapes. The identification of threatened rays as such surrogates would allow managers to prioritise the protection of optimal habitats with high biodiversity, as well as provide direct protection for species on conservation concern. Therefore, whilst other species or diversity metrics could also be considered appropriate selections as surrogates, it is this additional protection of larger, threatened and wide ranging species that means that such a species is a more effective choice as a surrogate for these sorts of systems. Consequently, we tested two complementary approaches for surrogacy, where we considered that estuary rays would be; (1) candidate ecological indicator species because features of local habitats and conditions of the water column are consistently correlated with the occurrence of rays, and (2) candidate umbrella species because rays occurrence is correlated with a more diverse fish assemblage containing a higher abundance of species who would benefit from conservation techniques (namely, harvested fish species).

## 2. Materials and methods

### 2.1. Study areas

We surveyed the fish assemblages of 22 estuaries along 200 km of coastline in southeast Queensland (SEQ), Australia (Fig. 1), encompassing all estuarine systems wider than 100 m in the region. All estuaries studied exchange with the ocean throughout the year (i.e. none of the estuaries studies are intermittently closed and open). The surveyed estuaries are subject to the full suite of human-related impacts of the coastal zone, and were selected specifically because they range in intensity from the highest possible impacts (e.g. Nerang River; Fig. 1), to towards the lowest recorded in the region (e.g. Noosa River; Fig. 1) for each impact (Tables S1 and S2) (Gilby et al., 2017). Sediment and nutrient runoff into waterways is a significant impact in SEQ estuaries (from both farmlands and urban landscapes, especially construction sites) (Healthy Land and Water Limited, 2016), with studies indicating that catchment revegetation should be a focus of future management interventions the regions coastal zones (Gilby et al., 2016; Olley et al., 2015). Fishing pressure in SEQ include both commercial fishers (including trawl and net finfish fisheries) (Tibbetts et al., 1998; van de Geer et al., 2013) and the largest recreational fishing effort in the state (Pascoe et al., 2014; Webley et al., 2015). Some of the surveyed

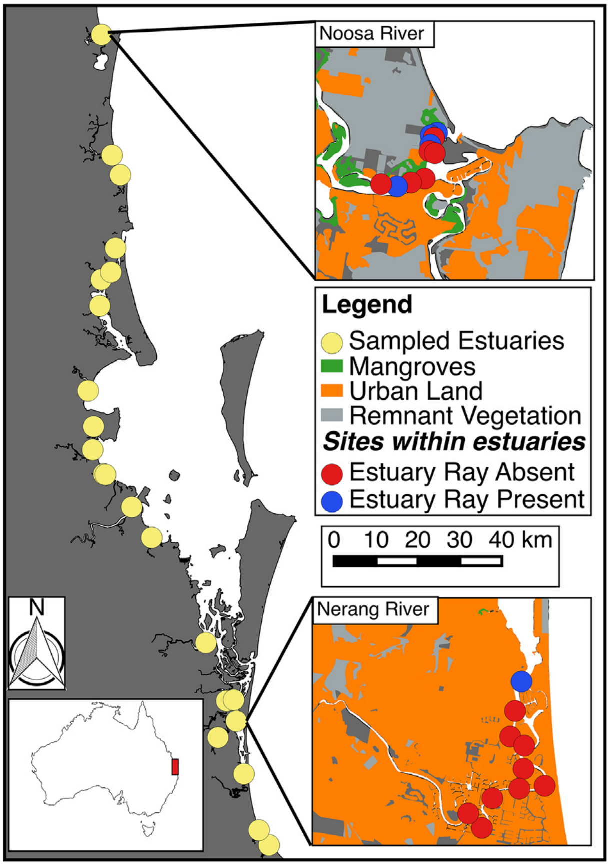


**Fig. 1.** Estuaries sampled in southeast Queensland, Australia. Insets are examples of estuaries with high and low densities of estuary rays. Noosa River: good conditions for estuary rays (abundant mangroves and remnant vegetation in catchment). Nerang River: poor conditions for estuary rays (few mangroves and little remnant vegetation in the catchment).

estuaries are currently no-take areas, however, there is no evidence that they are currently functioning to change fish assemblages (Gilby et al., 2017). Without intervention, these impacts will expand to expand into the future, with the population of SEQ growing exponentially (total population approximately 3 million people increasing by approximately 2% annually) (Australian Bureau of Statistics, 2012).

Ten sites within each estuary were sampled twice over three-day periods between June and August 2015. Salinity levels are a key determinant of species distributions in estuaries, so, sites were spaced evenly from the estuary mouth to the point where salinity declines to 30 ppt during winter (HLWMP, 2017) (Fig. 1). We chose 30 ppt as the upper limits of our sites to standardize the salinity gradients within each estuary, to best match distances sampled between our largest estuaries (which might have tidal limits 25 km upstream) and smallest estuaries (whose lengths might be fully tidal), and to allow for a distance of at least 250 m between sites to minimize detection of the same fish at more than one site. All sites were on non-vegetated muddy substrate, within 30 m of mangroves when they were present to control for seascape scale effects (see Martin et al., 2015), and in water depths of 1.5–2 m.

### 2.2. Fish assemblages

Fish assemblages were surveyed with one hour deployments of baited remote underwater video stations (BRUVS) at each site. BRUVS are now a well-established technique for surveying fish assemblages in coastal areas (Borland et al., 2017; Unsworth et al., 2014). BRUVS were preferred in this study as visual census techniques were not possible in our estuaries, and because they are non-destructive (especially given the threatened status of estuary rays). Studies have shown that whilst BRUVS can increase the numbers of bait-attracted species recorded (including scavengers and predators), the use of bait does not significantly reduce the measurement of herbivores and other species which might not feed directly on carrion (Harvey et al., 2007; Watson et al., 2010). BRUVS were constructed from a 20 × 20 × 5 cm concrete paver, a 1 m length of 2 cm gauge PVC pipe to attach baits at a fixed distance of 50 cm from the camera, and a GoPro camera recording in high definition. Fish were attracted to the camera with baits consisting of 500 g of pilchards (*Sardinops sagax*) placed into a 20 × 30 cm mesh bag with 0.5 cm openings. BRUVS were deployed from a boat, and were buoyed at the surface to prevent the entanglement of the retrieval rope.

A 20 × 20 cm visibility calibration disk, placed 1 m from the camera was used to quantify and standardize visibility among estuaries. The disk was painted with three equal width vertical stripes of white, grey and black paint, with visibility determined by which colours were visible by the observer during the video (black, grey and white seen = high visibility, white and grey = moderate, white only = low).

Each video was analysed by counting the maximum number of individuals of each fish species that was visible within the field of view, and between the camera and the visibility disk (i.e. within 1 m of the camera; *MaxN*), thereby minimising the potentially confounding effect of variable visibility. Given the distance between sites (>250 m) we considered it unlikely that the same individual was sampled more than once. Both estuary rays and surrounding fish assemblages were surveyed during the same deployments at each site.

### 2.3. Environmental factors

We tested 10 factors that have previously been demonstrated as important in the management of ecosystem condition in estuaries (for factor details and references supporting their inclusion, see supplementary material Table S1, for factor values see supplementary material Table S2). These factors could be broadly grouped into three categories: habitat properties at two different scales (site scale and catchment scale) and in-stream water quality. Site-scale factors quantified the cover (in $m^2$) of mangrove forests, sand flats, urbanized terrestrial land and remnant terrestrial vegetation within buffers spanning a radius of 500 m around each site (based on the distance between our sites and our current understanding of the scales of fish movement in estuaries; e.g. Olds et al., 2014). Catchment-scale factors were the proportion of the total catchment classified as urbanized land and remnant terrestrial vegetation, and the total cover (in $m^2$) of mangroves and sand flats for the whole estuary. Water quality measures included bottom turbidity (nephelometric turbidity units) and total nitrogen (mg/L) for each site. Given the large number of estuaries sampled, our surveys encompassed the full range of environmental conditions present within estuaries of the region (Table S2) (Gilby et al., 2017).

Recreational fishing is an important pressure in southeast Queensland, however, information at the scale of estuaries is not available (Webley et al., 2015). A net and trawl fishery currently targets fish throughout southeast Queensland estuaries. There are, however, no current and accurate data quantifying total fishing pressure at an estuary scale throughout this region (a problem not unique to southeast Queensland estuaries; e.g. McPhee et al., 2002, Veiga et al., 2010), so we cannot include this metric in our study. The estuary ray, given its threatened status, is not harvested by either commercial or recreational sectors.

### 2.4. Statistical analyses

To identify which environmental attributes are associated with the distribution of rays, we used generalised additive models (GAM)– this represents our test for 'ecological indicator' status. To assess whether the presence of rays at a site is likely to reflect a more diverse and abundant fish assemblage, we looked for assemblage level (i.e. the type and number of fish in multivariate format) differences using permutational multivariate analysis of variance (PERMANOVA) and compared (using t tests) the mean abundance of fish and species richness between sites with and without rays – this represents the test for 'umbrella species' status. We also used PERMANOVA to assess for the effects of levels of water clarity (as measured by the above-described visibility disc placed 1 m from the camera) on fish assemblages.

*Ecological indicator species* are species whose presence or abundance correlates with some indication of overall ecosystem condition. To test for the estuary ray's candidacy as an ecological indicator species, we used binomial generalised additive models (GAM) in the mgcv package of R (R Core Team, 2017) to determine correlations between our ten environmental factors and whether an estuary ray was detected at our sites (i.e. presence/absence) on either day's surveys (i.e. n = 22 estuaries x 10 sites = 220). GAM overfitting was reduced by modelling all possible combinations of four or fewer factors, and by restricting model knots to 3 or fewer (Burnham and Anderson, 2002). Relative factor importance was determined by taking the sum of weighted Akaike's information criterion corrected (AICc) values for each model containing that factor, with values closer to one indicating greater correlation between that factor and the dependent variable. Best fit models were those with the lowest AICc value, and those within two ΔAICc units (Burnham et al., 2011). Factors were considered as 'important' if they had both an importance value of >0.4, and were included in best-fit models (Burnham and Anderson, 2002). This method was preferred over other model-selection methods as it incorporates an understanding of factors importance across all potential models, and helps in preventing model selection uncertainties (Burnham and Anderson, 2002).

Due to concerns regarding pseudoreplication at the estuary scale, initial GAM analyses also included the fixed factor of 'Estuary'. Estuary was chosen as a fixed factor because we sampled all estuaries larger than 100 m in width within the region, meaning the scale of our 'Estuary' factor is the likely scale at which spatial conservation measures would be applied within these types of systems. Therefore, we were interested in differences between estuaries for this particular analysis. 'Estuary', however, was of low importance in initial GAM models (importance = 0.19), explained a low proportion of variation in estuary ray presence by itself ($R^2$ = 0.23), did not change best fit models (see Supplementary material Table S3), and was therefore lower in explanatory power than all other variables. Therefore, 'Estuary' was not included in subsequent models.

Identifying *umbrella species* (species whose protection may also protect many other species) relies on identifying species whose occurrence is correlated with a more diverse fish assemblage. We used Student's *t* tests to determine differences in average species richness and total fish abundance between estuary ray present and absent sites. We tested for differences in fish assemblage composition (i.e. a multivariate matrix of number and type of species occurring at each site) between estuary ray present and absent sites using the PRIMER 7 multivariate statistics package (Clarke and Gorley, 2015) with the PERMANOVA add on module (Anderson et al., 2008). All PERMANOVA analyses were conducted on square root transformed Bray Curtis measures. We then used the Dufrene-Legendre indicator species analysis (Dufrene and Legendre, 1997), a metric of species occurrence and abundance within site categories, to determine the species driving differences in fish assemblages between sites where estuary rays were present and absent, and whether these differences were statistically significant. Species were considered as an indicator of ray presence or absence if they had both an indicator value >0.2, and a significant p value ($p < 0.05$) from the associated permutation test (Dufrene and Legendre, 1997). Due to very low abundances of some species, and their potential for biasing the results of this test, only species identified on five or more separate videos were included in this analysis.

Finally, we sought to test whether the factors that most influenced estuary ray presence were also important for fish assemblages more generally. We tested for correlations between the ten environmental variables and fish assemblages at each site separately (to find whether factors describe a significant proportion of variation in the fish community separately), and then all together to find the best combination of variables using the BIOENV procedure in PrimerE. Analyses were conducted on square root transformed Bray Curtis measures for the fish assemblage, and normalised environmental metrics. Factors were considered important if they both explained a significant proportion of variation individually, and were included in best fit models.

## 3. Results

### 3.1. Surveyed fish assemblages

Water column visibility (as measured by visibility disc placed 1 m from the camera) did not affect the composition of the fish assemblage ($p > 0.15$), and so was excluded from all analyses.

### 3.2. Ecological indicators: habitat associations

Estuary rays occurred at 42 out of 220 sites and four habitat attributes were consistently associated with ray occurrence. Bottom turbidity was found to be the most important variable overall, with an importance value of 0.76 (Fig. 2A). Bottom turbidity had little effect on estuary ray presence up to approximately 10 NTU. Values higher than 10NTU resulted in a decline in the likelihood of ray presence. Each of the remaining three important factors, the cover of remnant vegetation in the catchment (importance = 0.65), total nitrogen concentration in the water column (importance = 0.61) and total mangrove cover in the catchment (importance = 0.5) had positive relationships with ray presence (Fig. 2). These importance values were supported by two best-fit models, which both contained combinations of these four factors and sand flat cover at the site scale (Supplementary material Table S3).

### 3.3. Umbrella species: fish assemblage associations

Sites where estuary rays occur contained 55 out of the total 59 identified fish species (93.2% of total diversity). Eleven fish species were found only at sites where estuary rays were present, and 'ray sites' had a more diverse fish assemblage (Fig. 3E, Student's *t*-test, t = 2.15, p = 0.03) but did not consistently contain a higher abundance of fish overall (Fig. 3E, Student's *t*-test, t = 1.63, p = 0.1). Three of the four environmental variables that were important in predicting the occurrence of estuary rays (i.e. mangrove area-catchment, bottom turbidity, total nitrogen) (Fig. 2), were also important in explaining spatial variation in the composition of the broader fish assemblages (i.e. the multivariate matrix of number and type of species occurring at each site) (Table 1). Fish

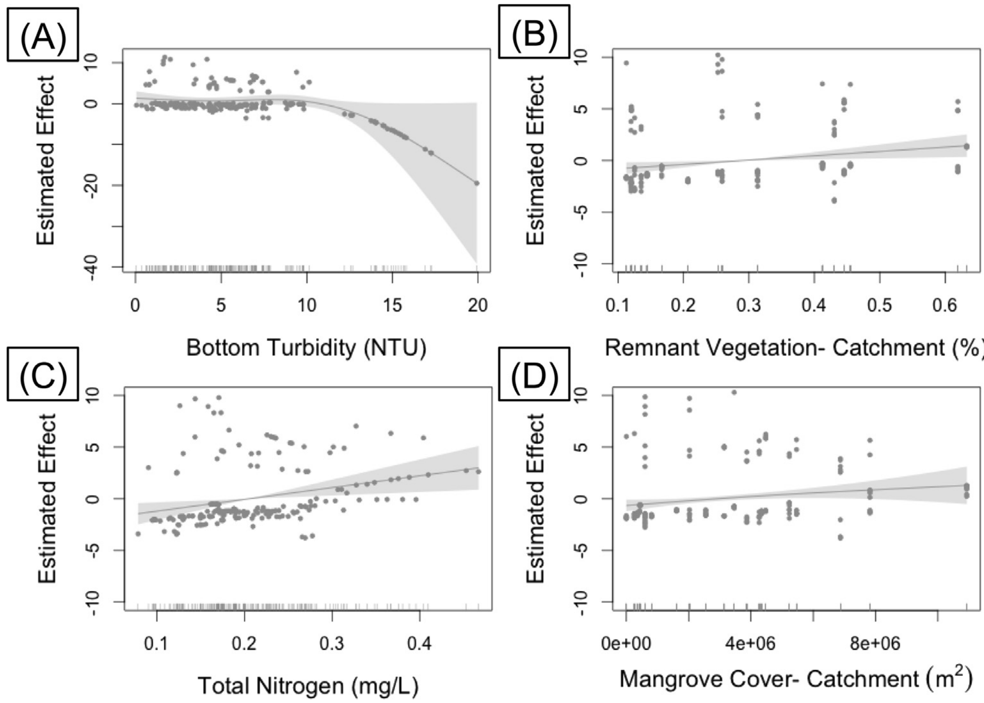


**Fig. 2.** Generalised additive model correlation plots from the best-fit model for presence/absence of estuary rays in estuaries in southeast Queensland. Shaded grey areas indicate 95% Bayesian intervals. Images are ordered from factors with highest importance values to those with the lowest; A) bottom turbidity = 0.76, B) remnant vegetation-catchment = 0.65, C) total nitrogen concentration in water column = 0.61, and D) mangrove cover in estuary = 0.5. Please note the different scale on the y-axis for bottom turbidity. Y-axis values denote the estimated degrees of freedom of the term being plotted.

assemblages differed significantly between areas that supported estuary rays and those that did not (PERMANOVA, df = 1, Pseudo-F = 2.6, p = 0.01). Three fish species (*Acanthopagrus australis*, *Marilyna pleurosticta* and *Tetractenos hamiltoni*) were significantly more abundant and more likely to occur at sites where estuary rays were present (Dufrene-Legendre indicator species analysis; Supplementary material Table S4). Only one species (*Mugil* spp.) was found to have higher abundance and prevalence at sites where estuary rays were absent (Dufrene-Legendre indicator species analysis; Supplementary material Table S4).

## 4. Discussion

Estuaries are under significant threat from human activities (Halpern et al., 2008), which means that we need more effective mechanisms to prioritise estuarine conservation areas (Gilby et al., 2017), and to monitor the effects that we have on estuarine ecosystems more generally (Caro and O'Doherty, 1999; Siddig et al., 2016). One potential method that we can use to overcome such challenges is that of surrogate species. The concept of surrogate species is well established in many ecosystems, but it has been less tested, and therefore less frequently adopted, in estuaries and coastal waters (Tisseuil et al., 2013). In this study, we show that estuary rays are an appropriate indicator species for estuarine in southeast Queensland; their presence corresponds to areas where habitat quality is relatively high (Fig. 2, Table S3), and correlates with similar factors to those that are most influential for the fish assemblage (Table 1). Estuary rays are a candidate umbrella species because placing reserves in areas where estuary rays occur would protect areas that contain the majority of estuarine fish diversity in the region (93.2% of total species richness for the region), and sites that have higher average species richness. Finally, factors that best explained estuary ray distribution were also important in structuring the broader fish assemblage. In these estuaries, no other species of conservation concern was detected in high enough abundance to justify their selection as surrogates. Given these results, estuary rays qualify under established definitions for both indicator and umbrella management species (Caro and Girling, 2010; Hunter et al., 2016). In other systems where rays are not present, similar species, which are easy to detect, are large, or threatened, and which provide integrated measures of ecosystem condition, especially the realised outcomes of environmental conditions or change, should be considered as candidate umbrella and/or indicator species. Importantly, selecting such species allows managers to prioritise protection for areas of optimal habitat and high biodiversity, as well as allowing for the direct protection species of conservation concern.

Determining how surrogate species respond to habitat features and associated management interventions is a vital consideration in surrogate species selection (Caro and Girling, 2010). In this study, estuary ray presence correlated with a less developed catchment, higher mangrove cover and factors important for water quality, meaning that it is a surrogate for generally good estuarine ecosystem condition. Water column nutrient concentrations are often slightly elevated in areas of high mangrove forests, and the benefits of slightly elevated nitrogen on food web productivity are well understood (Oczkowski and Nixon, 2008; Rabalais, 2002). In any case, the sites that we sampled did not surpass the threshold of nitrogen concentrations that would cause detrimental effects in these systems (max total nitrogen concentration of ~0.5 mg/L) (Rabalais, 2002). Similarly, threshold effects (Fig. 2A) of turbidity on fish assemblages are also well established (e.g. Lunt and Smee, 2015), and responses found here fit well with what is known of estuary ray biology (Pogonoski et al., 2002). Estuary rays responded positively to the coverage of mangroves and remnant vegetation within the catchment, likely due to higher food availability

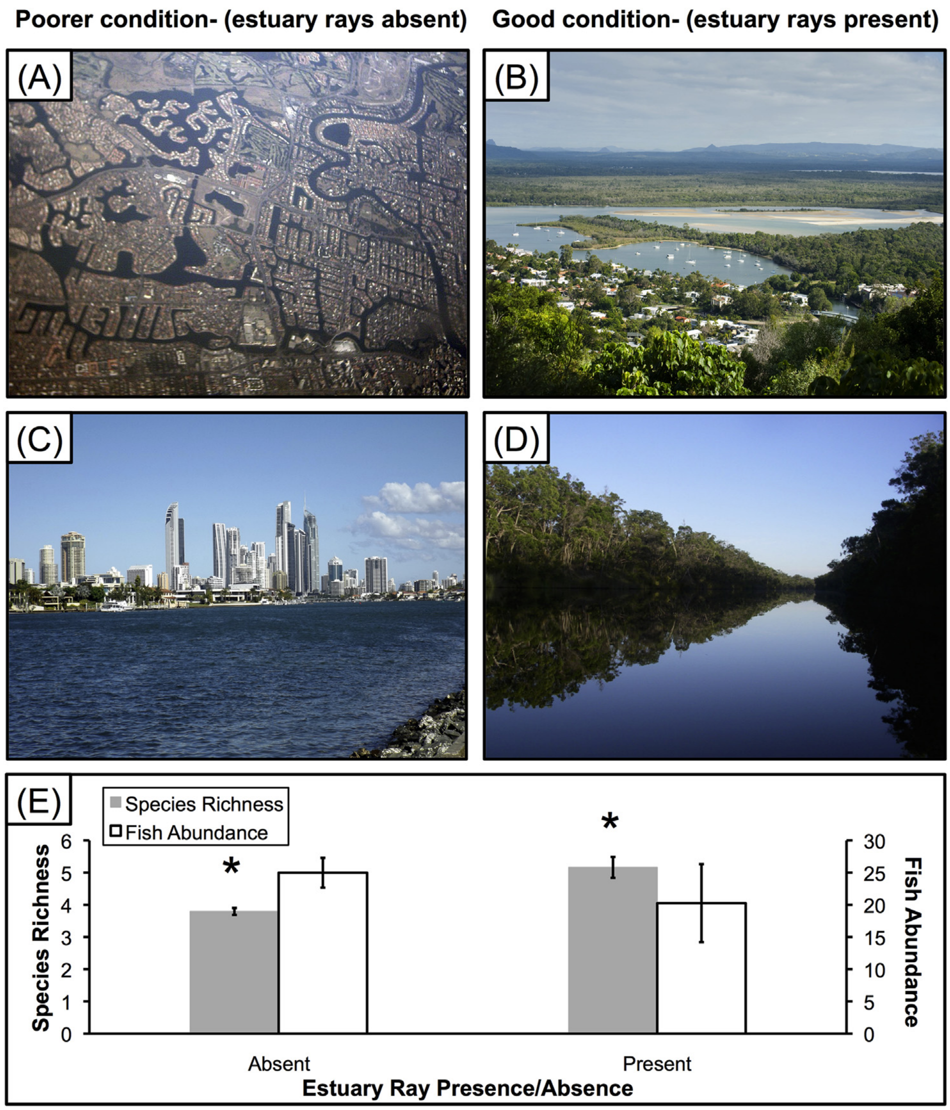


**Fig. 3.** Estuary rays may be useful as an indicator species for monitoring estuarine ecosystem condition because their distribution is positively correlated with the cover of remnant vegetation in an estuaries' catchment (compare A and B) the abundance of mangroves in an estuary (compare C and D), and negatively correlated with turbidity. Estuary rays may also be important surrogate species for estuarine conservation because their distribution correlates with fish species richness (E). Fish abundance (i.e. the sum of MaxN for all species at a site) was not significantly different between estuary ray present and absent sites. * indicates bars significantly different to each other at $\alpha = 0.05$. Photos- I. Toksave (CC BY 3.0), Shiftchange (CC BY 1.0), AussieStock (CC BY 3.0) G. Burns (CC BY 4.0).

(especially crabs, polychaetes and bivalves; Pardo et al., 2015) and access to high-tide refugia (Pierce et al., 2011) in areas of higher mangrove cover. Importantly, each of these factors operate at the estuary scale (i.e. 1–10 km), corresponding to scales at which spatial conservation actions, especially marine reserves, will likely be applied, thereby adding further support for the choice of this species as an umbrella species (Roberge and Angelstam, 2004).

Maximising cost-effectiveness, without compromising performance, is the common goal of all conservation and monitoring programs. For this reason, integrated measures of ecosystem condition are attractive (Cooper et al., 2009; Woodward et al., 1999). Historically, monitoring programs in coastal zones have focused on habitat extent and water quality parameters, including pulse impacts from floods and other major events (e.g. HLWMP, 2017). However, studies have shown that simply correlating the condition of assemblages with traditional monitoring techniques (e.g. fish assemblages with water quality) can often be poor in explanatory power (Bunn et al., 2010; Jonzen et al., 2005; Logan and Taffs, 2014). Therefore, the addition of indicator species to coastal monitoring programs provides direct information on ecological responses rather than having to infer these from nutrient and habitat proxies. This is especially pertinent for a species such as the estuary ray for which the key pressures behind their threatened status are well understood (principally habitat loss and declining water quality; IUCN, 2015, Pierce and Bennett, 2010, Pogonoski et al., 2002), and for which the effects of fishing pressure are low (Webley et al., 2015). Although identifying surrogate species initially requires a series of environmental data and should be verified for each study system, such concepts provide an inexpensive and reliable option for improving the precision and scope of existing programs. Whilst BRUVS might only be appropriate in estuaries with high water clarity, the advantage of using such large species like estuaries rays is their detectability using this or other simple sampling techniques (netting, visual count/inspections or angling) and metrics (species occurrence). A complementary approach is to index the health of ecosystems by measuring the health of organisms and this

**Table 1**
Results of BIOENV analyses showing Spearman's correlations between the fish community (square root transformed, Bray-Curtis dissimilarity measures) and environmental variables of interest (normalised, Euclidean distance measures; for further details on factors, see Table S1). Values in bold are significant at $\alpha$=0.05.

| Model | ρ | p |
|---|---|---|
| *Individual Factors* | | |
| Sand flats- catchment | 0.14 | **0.01** |
| Bottom turbidity | 0.13 | **0.01** |
| Mangrove area- catchment | 0.1 | **0.01** |
| Sand flats- site | 0.08 | **0.03** |
| Total nitrogen concentration | 0.05 | **0.02** |
| Urbanized land- site | 0.05 | 0.97 |
| Remnant vegetation- site | 0.03 | 0.21 |
| Proportion urbanized land- catchment | 0.03 | 0.94 |
| Proportion remnant vegetation- catchment | 0.02 | 0.12 |
| Mangrove area- site | 0.01 | 0.42 |
| *Best Models* | | |
| Sand flats-site, Sand flats- catchment, Bottom turbidity | 0.2 | **<0.01 for all** |
| Sand flats-site, Sand flats- catchment, Bottom turbidity, Remnant vegetation-site | 0.19 | |
| Sand flats-site, Mangrove area-catchment, Sand flats- catchment, Bottom turbidity | 0.19 | |
| Sand flats- catchment, Bottom turbidity | 0.19 | |
| Sand flats-site, Sand flats-catchment, Bottom turbidity, Remnant vegetation-catchment | 0.19 | |

approach, using fish, can reliably detect pollution signals in estuaries (Schlacher et al., 2005, 2007).

There are three key mechanisms through which using umbrella or indicator species can help in the conservation and management of estuaries. Firstly, as umbrella species, managers can use them to select areas where reserves should be implemented (i.e. areas where they are often present), or, perhaps, where habitat restoration should occur (i.e. areas where they are not often present, and there is a lack of key habitats) (McAlpine et al., 2016; Pouzols and Moilanen, 2014). Secondly, as umbrella and indicator species, managers can use them in association with traditional metrics (e.g. habitat extent or quality, water quality metrics) to help identify, and then prioritise, which estuaries require which management actions. For example, estuaries where the species is in low abundance, or never occurs, can be targeted for catchment revegetation, reduced nutrient releases, and improved in-stream habitats, perhaps through restoration, or additional restrictions on area usage (e.g. anchoring, trampling, other habitat protection methods). By using these surrogate species in association with 'traditional' techniques we can add additional support to existing decision making tools by incorporating a simple measure of animal community response (Butler et al., 2012; Caro and Girling, 2010). Finally, as indicator species, managers can use them to monitor the outcomes of reserve implementation, or of other catchment or estuarine management techniques for the fish population and fish habitat quality more generally (Caro and Girling, 2010; Lindenmayer et al., 2015). Such species might be incorporated into monitoring plans, or citizen science programs at a regional scale (Bevilacqua et al., 2015; Smale et al., 2011).

Our findings demonstrate a situation where umbrella species can be used to improve our choice of target conservation areas, especially through marine reserves, in estuarine and coastal ecosystems (Zacharias and Roff, 2001). Furthermore, indicator species can be used to improve the accuracy and cost effectiveness of ecological assessments by providing a measure of realised impact on animals (rather than simply assuming faunal impacts from other environmental factors), and can therefore increase the accuracy and validity of monitoring programs in coastal systems if monitored over longer periods (Sheaves et al., 2012; Whitfield and Elliott, 2002). Although rays might not be applicable surrogates in all systems (they might be uncommon, not present, or extirpated from some estuaries), other threatened groups with similar biological and ecological attributes (i.e. large, wide ranging, under direct threat from human actions) are likely able to function as surrogates in aquatic ecosystems. For example other threatened elasmobranchs (e.g. shovelnose and requiem sharks, sawfish, whiprays), or bony fish (e.g. large freshwater cod, grouper) and marine and freshwater turtles might also be appropriate surrogate and indicator species in aquatic systems given their likely threats and relative ease of detection (both high detectability, and simple metrics-presence/absence- and sampling techniques), and the direct benefits of also protecting threatened or vulnerable species (IUCN, 2015; Pogonoski et al., 2002). Whilst the current study was restricted in scope to mostly marine stretches of estuaries, the chosen methods (fish surveys, followed by seeking correlations between fish and a suite of environmental metrics) are applicable to any coastal ecosystem worldwide. Therefore, we suggest that candidate surrogates should be present in most estuarine and coastal systems globally, and even if the full benefits of this method are not fulfilled (for example, if the species is not threatened). By protecting species that are threatened and, which also have merit as integrated surrogates for conservation and monitoring, managers might address multiple targets for conservation: (1) protecting populations of a species of conservation concern; (2) maintaining diversity; and (3) better place reserves in heterogeneous seascapes to more effectively protect estuarine habitats.

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## Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.ecss.2017.10.003.

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