Project C2 - Continental-scale tracking of threats to shallow Australian reef ecosystems - Indicator report

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Important Disclaimer

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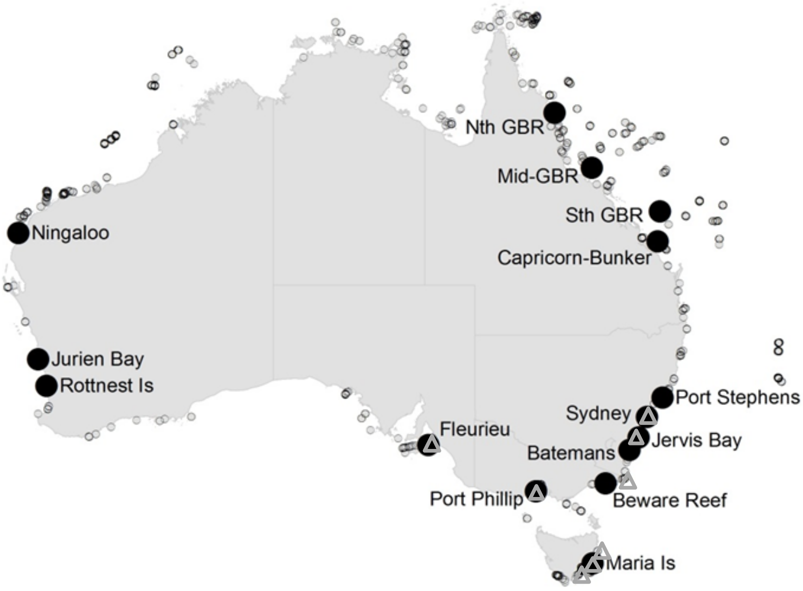
EXECUTIVE SUMMARY

Understanding human pressures and their impacts on marine ecosystems, including how these threats may change in space and time, is important for developing and prioritising management of marine natural resources, overarching biodiversity and associated ecological services. To achieve this goal, the capacity to monitor and detect change, in combination with understanding underlying mechanisms of impact, are both fundamental. However, achieving broad spatial and temporal data coverage, and knowing which key indicators of biodiversity reflect directional change related to specific pressures at local and regional scales represent significant challenges. For this report, we combined the three largest long-term monitoring datasets for Australia's shallow reef communities that provide high taxonomic resolution for tropical and temperate fish, invertebrate and algal species: (1) the Australian Institute of Marine Science Long Term Monitoring Program (AIMS LTM; 276 sites, 23 yrs); (2) Reef Life Survey (RLS; 1,294 sites, up to 9 yrs); and (3) the University of Tasmania Long Term Marine Protected Area Monitoring Programs (LTMPA; 182 sites, up to 25 yrs). All sites and monitoring locations are shown in the map below. These datasets were integrated and analysed with information on major human pressures (fishing, rising sea temperature, introduced/ habitat-modifying species, and pollution) in order to identify impacts and biological indicators most sensitive to these pressures. This comprised the first continental-scale analysis of biogeographic patterns, ecosystem function and the tracking of ecological impacts generated by human-related pressures, and was designed to contribute to State-of-the-Environment (SoE) reporting.



Map of survey sites and monitoring locations. RLS surveyed from 2010-2015 and used in spatial analyses (small symbols, N = 1,294) and long-term monitoring locations from RLS, LTMPA and AIMS used for temporal trend assessment (large filled circles). Triangles show locations in south-eastern Australia where pollutants were sampled and RLS/ LTMPA data used to examine biological indicators of pollution.

Testing of widely-applied fishing indicators confirmed biomass of fishes ≥20 cm and ≥30 cm as the most suitable indicators for assessing fishing impacts on shallow rocky and coral reef communities. These indicators were then mapped nationally, along with indicators for ocean warming, plus threatened and invasive species. Fishing appears most pervasive of human impacts assessed, but decadal trends also showed warming to cause substantial community change. The threatened species index was highest for the Great Australian Bight and Tasman Sea, while invasive species were concentrated near harbours in south-eastern Australia. Examination of candidate biological indicators of pollution impacts for this region revealed reef community metrics associated with reduction in length of species (declining maximum lengths for fishes, invertebrates and biogenic habitats) and a reduction in benthic species richness to be most informative. That is, increased pollution appears to lead to 'short and simple' reef communities whereby constituent species are fewer and only achieve smaller body sizes relative to species found on non-polluted reefs. Heavy metals and high nitrogen levels (eutrophication) correlated with greatest apparent biological impact relative to other pollutants (including other organic, petrochemical, and micro-plastic pollution) for industrialised/ urbanised embayments along the south-eastern Australian coast.

Species-specific indicators of reef impact were also examined by mapping key habitat-modifying species known to collapse shallow reef systems, such as Crown-of-Thorns Seastars (CoTS) on coral reefs and kelp-overgrazing sea urchins on temperate reefs. Furthermore, we showed how these monitoring programs can be used to track change in biomass of herbivorous urchins and kelp bed cover relative to experimentally determined tipping-points of kelp collapse and recovery. This highlights how mechanistic understanding, coupled with long-term monitoring, can enable tipping-points in environmental conditions and resultant change in ecological state to be linked to human-pressures; and ultimately highlights the opportunity for management actions to be enacted before practically-irreversible ecosystem change has already occurred.

Another focus of the project was to understand how well the three major datasets could be integrated for the purposes of indicator calculation and ongoing SoE reporting and other reef biodiversity monitoring needs. Given the development of RLS methods was based on those from the LTMPA, all indicators other than those based on the benthic cover were calculated in a directly comparable means across these two datasets. The AIMS LTM reef fish data were collected only for a subset of fish species (a standardised list is used for consistency along the GBR and through time). While it was possible to calculate the same indicators from the AIMS LTM data and assess trends in these through time on the GBR, possibly important information was lost, and indicator values from AIMS LTM reef fish monitoring were not closely aligned with equivalent values based on the RLS data. As such, combining these datasets for calculation of the SoE indicators recommended here does come with caveats for interpretation across regions.

In summary, we demonstrate continental-scale utility of a suite of biological indicators (summarised in the Table below), and our analyses provide benchmarks of ecological condition, plus highlight the ongoing need for highly-resolved, broad-scale spatial/ temporal biological data so that robust reporting against biodiversity targets and SoE reporting is achievable. Maintaining ongoing monitoring programs is challenging and we highlight RLS citizen-science as a fit-for-purpose low-cost model with highly resolved spatially-extensive data enabling the calculation of numerous biodiversity indicators that can routinely feed into SoE reporting (including the Essential Environmental Measures initiative).

Indicator suite for Australia's shallow reefs.

Pressure	Indicator
Fishing	Biomass of fishes ≥20 cm Size spectrum – Gamma shape parameter
Ocean warming	Community Temperature Index
Invasive species	Density of invasive individuals 50 m^{-2}
Pollution - Heavy metals	Community Shortness Index Invertebrate Lmax
- Nitrogen loading (eutrophication)	Richness of benthic habitats Invertebrate diversity (Shannon)
Crown of thorns sea stars	Density of individuals per Ha
Sea urchins	Overgrazing Index (urchin biomass per m^2 required for incipient barrens formation)

1. INTRODUCTION

Shallow reef habitats harbour the greatest concentrations of biodiversity in the sea (Roberts et al. 2002), and are often the components of the marine environment with which the public interact with and value most. These habitats also overlap the locations where major human pressures are greatest. Fishing, warming, invasive species and pollution have consistently been recognised as the most serious and pervasive threats to marine biodiversity, and can all be present and interacting on coastal reefs (Edgar et al. 2005, Crain et al. 2009).

As a result of biodiversity and social values, and intense human pressures, shallow reef systems have been monitored by Australian research organisations and government management agencies better than any other marine habitat type. The long-term temperate marine protected area (MPA) monitoring program (LTMPA; Barrett et al. 2009) and the Australian Institute of Marine Sciences Great Barrier Reef Long-Term Monitoring program (AIMS LTM; Sweatman et al. 2011) are the two most geographically-extensive reef monitoring programs in Australia that have been operating for more than 20 years. These have focussed on MPAs across temperate Australia and off the QLD coast, respectively, and cover multiple elements of reef biodiversity (*see* methods).

The Reef Life Survey program (RLS; www.reeflifesurvey.com) was established in 2008 using methods designed to be compatible with the LTMPA monitoring. In contrast to the LTMPA and AIMS monitoring by scientific teams, RLS uses a citizen science model that selectively engages, trains and supports the most capable and committed recreational SCUBA divers. It has grown rapidly in scope. In its first eight years, more than 200 divers have contributed 10,000 surveys at >3,000 sites in 47 countries, 7 continents, and 87 of the world's shallow marine ecoregions (as defined by Spalding et al. 2007). In Australia, RLS teams have monitored gaps in the MPA monitoring by the LTMPA program (in terms of MPAs not monitored by that program, or years not covered in the monitoring of that program), as well as filling in gaps in the spatial coverage of data around the Australian continent.

Together, these three monitoring programs provide high resolution spatial coverage of shallow rocky and coral reef biodiversity data across the entire continent, including the offshore reefs and Commonwealth Marine Reserves. They also contain highly detailed species-level data on reef communities, including size structure and composition, although with slight differences in the level of detail between AIMS and the other two (see methods). Importantly, although designed and most often used for particular purposes (e.g. guiding

MPA management), they provide enormous flexibility in terms of potential for answering many scientific and management related questions. This report focusses on using these datasets for the calculation of biodiversity indicators, which summarise detailed multivariate biodiversity data into metrics that describe biodiversity trends in relation to key pressures or particular management or policy goals.

1.1 Selecting biodiversity indicators

A wealth of literature exists on biodiversity indicators, describing different approaches to the calculation, selection and interpretation of hundreds of metrics that differ widely in utility across scales and systems. This report focusses on indicators applicable at the national scale, relevant to needs associated with State of the Environment reporting, assessing progress towards international agreements such as the Convention on Biological Diversity (CBD) and for guiding national level policy. There are some widely-acknowledged requirements/desirable attributes for effective biodiversity indicators at large scales, including responsiveness to particular pressures, specificity to the pressure of interest, ability to guide management or policy responses. These are outlined in detail by Jones et al. (2011).

The two main approaches typically used for selecting indicators have been focussed on modelling of known or expected relationships between ecosystem components (e.g. food web or qualitative conceptual modelling), or through empirical analyses of biodiversity patterns in space or time related to concurrent patterns in pressures (i.e. statistical modelling). The former is arguably most useful for particular locations and when either limited empirical data exist or relationships between ecosystem components are well known (or well-covered in ecosystem models), and also when there is a specific local management question and response clearly identified (Fulton et al. 2005, Hayes et al. 2015). This approach has also been applied for designing monitoring programs and targeting data collection when resources or methods prohibit collection of data covering the system more broadly. Narrowing monitoring to specific components of the system comes with an associated risk, however, that important biodiversity responses are missed if the models omit important links, or when ecological surprises occur.

The alternative approach of statistical modelling (which is common in fisheries) is 'letting data speak for themselves' by looking at empirical relationships between characteristics of the community and the pressures or activities of interest to managers at smaller scales and policy makers at broader scales (Graham et al. 2005, Devictor et al. 2008). This is only possible when sufficient empirical data exist for models to have sufficient power, and may sometimes require complex modelling approaches to account for trends related to environmental gradients and other pressures. It must therefore also be guided by conceptual models or sufficient knowledge of the system.

1.2 Aims

The specific aims of this report were to use the three major reef biodiversity datasets described above to identify:

(1) An initial suite of indicators of biodiversity responses to the major pressures on Australia's shallow coral and rocky reef biodiversity, which can be monitored using data from the largest ongoing monitoring programs,
(2) Distribution and changes in biodiversity responses to these pressures across the continent over the last decade, and
(3) Potential and considerations for integration of datasets from the monitoring programs for ongoing indicator reporting.

Aim (1) is about selecting or developing a set of indicators that characterise biodiversity responses to major stressors on reef communities (e.g. fishing, rising sea temperature, introduced species, crown of thorns sea stars, sea urchin grazing, pollutants), in as specific and responsive ways as possible. Few indicators are completely specific, but suites of responsive indicators with known relationships with major pressures offer the best chance of tracking biodiversity responses through time in a way that can best guide larger scale management and policy, and inform the Australian public.

Aim (2) is about describing the current status of reef biodiversity around Australia, mapping the suite of indicators from (1) and assessing temporal trends over the last decade.

Aim (3) is about arriving at a suite of indicators that can be calculated from each of the three key monitoring programs and interpreted together. It involves understanding quantitative differences between datasets that can be accounted for in indicator calculation, and particular considerations for using each dataset in national indicator reporting.

2. METHODS

2.1 Reef biodiversity data

The globally-standardised RLS methodology is based on underwater visual census on 50 m transect lines set on hard reef substrate (rocky or coral) along a depth contour. Divers undertake three survey methods along each line to cover the majority of large biota that can be surveyed visually (i.e. >2.5 cm in size): fishes (method 1), mobile invertebrates and cryptic fishes (method 2), and photoquadrats of the substrate (method 3). Multiple transects are usually surveyed at each site, often laid parallel at different depths (typically in 4 – 15 m depth). Fishes are surveyed in duplicate 5 m wide belts on either side of the transect line, with abundance and binned size recorded for all species observed during a single swim along each side of the line. Size bins used are 25, 50, 75, 100, 125, 150, 200, 250, 300, 350, 400 and 500 mm total length, with larger individuals estimated to the nearest 125 mm. Fish counts are later converted to biomass estimates using species-specific length-weight relationships provided in Fishbase (fishbase.org), and as described in (Edgar and Stuart-Smith 2009). All species sighted within the blocks are recorded, including unidentified individuals, which are usually photographed for later identification with the assistance of taxonomic experts. Marine mammals, reptiles and birds are also counted if they occur within the 5 m wide blocks. Large mobile invertebrates (echinoderms, molluscs and crustaceans >2.5 cm) are counted in duplicate 1 m wide belts on either side of the line, with divers brushing aside any vegetation and looking closely in crevices, under ledges or amongst corals. Photoquadrats are taken every 2.5 m along the line (20 images per transect) and later processed in the lab by overlaying a grid of 5 points and identifying the category of sessile life or substrate underneath (100 points scored per transect). Categories scored for macroalgal, coral and other sessile invertebrate cover are directly mappable to the standard CATAMI classification system (Althaus et al. 2015). Full details of all RLS survey methods are provided in an online methods manual (at http://reeflifesurvey.com).

LTMPA data are collected using the same methods as for RLS, but with three key distinctions (1) within site replication is more consistent, with 4 x 50 m transects laid end on end at each site centred on either the 5 m or 10 m depth contour, (2) mobile invertebrates (method 2) are surveyed in only a single side of the line (total 200 m^2), and (3) sessile invertebrates and macroalgae are surveyed *in situ*, using a point intercept method by a diver experienced with identification of these difficult taxa, who identifies taxa under each of 50 points in ten 0.25 m^2 quadrats per 50-m long transect.

AIMS LTM fish data are also collected along 50 m transects, with five transects laid end on end at each site in 6 – 9 m depth. AIMS LTM surveys only include a subset of reef fish species from 10 families (see Emslie et al. 2014; http://www.aims.gov.au/c/document_library/get_file?uuid=29d6a8ae-2ae9-4311-a2de-742d6fdc9a6e&groupId=30301). Size information was not collected for fish species in AIMS LTM monitoring prior to 2005, but since then has been collected consistently for a subset of species, including some members of the family Labridae, and all members of Lethrinidae, Lutjanidae and Serranidae.

2.2 Indicators

Indicator selection was undertaken by assessing the most informative available metric for each of the key pressures separately. Results of previous research using these datasets and existing indicators from the literature were used first, before undertaking additional empirical analyses using the national RLS dataset as a second stage, as required. The RLS data provided the most comprehensive geographic coverage in a standardised fashion and in most detail, allowing assessment of sensitivity and specificity of indicators across the national scale, which includes a very large temperature gradient and covers rocky and coral habitats. Data from 1,294 sites surveyed by RLS divers in Australian waters between 2010 and 2015 were used in spatial analyses for this study (Figure 1). A subset of 372 RLS sites at nine locations were used for the time-series analyses, complemented by fish survey data from the LTMPA and the AIMS LTM programs.

Analysis of pollution, involving new empirical measurements of pollutants at 43 RLS/ LTMPA reef sites, was restricted to south-eastern Australia (Figure 1) where some of the most polluted reef environments occur, particularly within industrialised and/ or urbanised embayments. While reefs in such environments typically experience relatively low water exchange compared to exposed coastal reefs, making these communities intuitively more prone to plumes of pollution, all sites investigated were influenced by coastal waters in terms of the subset of species present and by virtue of the sites meeting the minimum requirement of visibility of ≥ 5 m such that effective visual census of reef biota was possible according to standard RLS/ LTMPA methods. Reefs occurring higher-up estuarine systems, including coastal harbours such as those along the Queensland and Northern Territory coasts, typically become turbid with insufficient visibility for visual surveys of reef fishes, and thus our assessment of reef community indicators of pollution was focussed on sub-maximally exposed coastal reefs in south-eastern Australia.

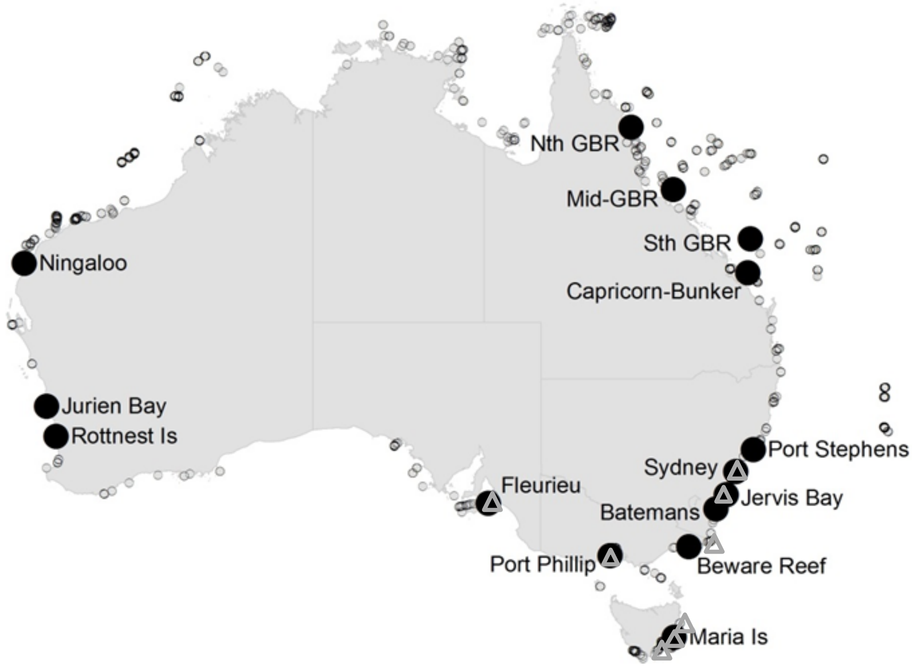


Figure 1. Map of survey sites and locations.
RLS survey sites surveyed from 2010-2015 and used in spatial analyses (small symbols, N = 1,294) and long-term monitoring locations from RLS, LTMPA and AIMS used for temporal trend assessment (large filled circles). Triangles show locations in south-eastern Australia where pollutants were sampled and RLS/ LTMPA data used to examine biological indicators of pollution.

2.2.1 Fishing

Numerous indicators have been developed which have a conceptual basis for assessing community-level responses to fishing pressure, and typically applied for fisheries-independent assessment of impacts of large commercial fisheries. A literature review was used to guide potential indicators and determine a short-list, but previous studies have typically been undertaken in the northern hemisphere or developing countries with subsistence coral reef fisheries, and therefore provide limited insight into which indicators are most sensitive and specific to the types of fishing pressure that occur on Australian shallow reefs (which include substantial recreational fishing pressure).

The assessment covers tropical and temperate locations with completely different species composition, thus indicator species (or similar taxonomic-based metrics) would not be comparable across the full range of sites. This reduces the list of potential indicators to community-level or trait-based metrics. The short-list of candidate fishing indicators was compiled based on an extensive literature search, and an initial screening for applicability to underwater visual census data. The vast majority of fishing indicators can be calculated from visual census data, many with less bias than when calculated using the typically-used trawl data. Few of the key studies comparing fishing indicators have used data as rich in detail as from visual censuses (Rochet and Trenkel 2003, Fulton et al. 2005). The shortlist (Table 1) includes those based on trophic level or group, biomass, exploited status, and size based indicators (Blanchard et al. 2005). For the latter, the slope of the linear size (biomass) spectrum was specifically included due to its widespread use, relative specificity to fishing impacts and broad applicability (Blanchard et al. 2005, Graham et al. 2005, Shin et al. 2005). A new metric also trialled that was based on fitting a gamma distribution to the size spectrum of fishes to account for a consistent non-linearity evident in visual census data (Thomson et al, unpublished). This was developed within the NERP Marine Biodiversity Hub research program, but has not yet been published.

Effective marine protected areas (MPAs), human population density and two metrics of geographic isolation were used as proxies for fishing pressure, and broad-scale environmental trends (sea surface temperature (SST), SST range, turbidity, nitrates and depth) were accounted for using data from Bio-Oracle (Tyberghein et al. 2012). Proxies for fishing pressure were necessary as no catch data were available to quantify the intense recreational fishing pressure on many of the shallow reef systems around Australia, nor were they available at an appropriate resolution or scale from most commercial fisheries operating in this environment. Isolation (by distance) from recreational fisher access has been shown to be a useful predictor of fishing impacts in shallow rocky reef communities in Tasmania (Stuart-Smith et al. 2008), while effective MPAs (Edgar et al. 2009, Edgar et al. 2014) comprise an experimental removal of fishing pressure. Isolation was measured using two metrics for each site: the shortest distance by water to the nearest boat ramp (measured on Google Earth), and a shore fishing pressure index. The latter was based on three categories for distance a site was offshore (0 = >500 m, 1 = 100-500 m, 2 = <100 m), and along shore from the nearest road access (0 = >5 km, 1 = 500 m - 5 km, 2 = <500 m). Multiplying the two categories gave an index value that was zero if a site was inaccessible to shore fishing and four if a site was both close to shore and close to a road access point. A human population density index was calculated to provide a relative value of the density of people living within close proximity to each reef site. This involved fitting a smoothly tapered surface to each settlement point on a 2010 world population density grid (CIESIN et al. 2005) using the quadratic kernel function (Silverman 1986), after screening for a density greater than 500 people per 30 arc-second grid cell and with the model boundary set at 100 km.

A linear mixed effects model was used to test for the effects of the four fishing pressure proxies described above on the spatial distribution of indicator values at RLS sites after accounting for environmental covariates. The most appropriate fishing indicators were then selected using the following procedure: (1) candidate indicators were ranked based on their ability to describe spatial variation in fishing pressure, as inferred from the goodness of fit of the model with, versus without, the four fishing pressure proxies, (2) those indicators for which none of the fishing pressure proxies showed significant relationships in the direction consistent with fishing pressure (when proxies were considered individually) were excluded, (3) those indicators that were strongly and significantly related to mean annual SST were also excluded, as variation with SST will reduce interpretability of spatial and temporal trends in indicator values (Blanchard et al. 2005), and may confound warming and fishing impacts in the longer-term.

2.2.2 Ocean warming

Few effective indicators of ecological state in relation to ocean warming have been developed or proposed (Gregory et al. 2009). Recent research on birds and butterflies (Devictor et al. 2008, Tayleur et al. 2016) has found the Community Temperature Index (CTI) to capture biodiversity responses to long-term warming, and studies on marine fishes and invertebrates using the RLS and LTMPA data have tested CTI and found it to be a sensitive and specific indicator of reef biodiversity responses to ocean warming (Bates et al. 2014, Stuart-Smith et al. 2015a). The CTI for fishes was used for this study, as preliminary analyses have shown this to be more responsive to temperature change than CTI calculated for invertebrates. Fish CTI was calculated for each site as described in Stuart-Smith et al. (2015a) as a community-weighted mean (CWM) of the midpoint of the realised thermal range of each species, weighted by the log of their abundance.

2.2.3 Pollution

Marine environments are ultimate end points (sumps) for many pollutants which, depending on pollutant type, typically enter these environments as a plume leading to high local concentrations that can impact the distribution of marine organisms and ecological functions. Our aim was to measure a range of pollutants across a range of industrialised, urbanised and more pristine reef environments in south-eastern Australia to identify the most suitable biological indicator(s) of pollution impacts on sub-tidal rocky reef communities.

Our study was focussed on the major urban centres of Sydney (New South Wales), Melbourne (Victoria), Adelaide (South Australia) and Hobart (Tasmania); supplemented by more remote areas along adjacent coasts, with lower levels of apparent human activity. These cities have major ports and industry, and substantial known heavy metal pollution as a legacy from historical industrial pollution and through contemporary, but ostensibly reduced, inputs of heavy metals, organic enrichment and other pollutants from storm water runoff and effluent discharges from urbanised / agricultural dominated sub-catchments (Birch, 2000; Johnston and Keough, 2002; Townsend and Seen, 2012).

Specifically, we sampled concentrations of heavy metal, organic (total nitrogen, nitrogen 14:15 isotope ratio, total organic content), petro-chemicals, and plastic pollution on and within sediments adjacent to 43 south-eastern Australian rocky reef sites (NSW, n=13; SA, n=6; Vic, n=8; Tas, n=16). Within each south-eastern Australian state, sites were distributed across the contrasting polluted and relatively pristine sub-locations of Sydney Harbour, Jervis Bay and Eden in NSW; from adjacent to the city of Melbourne towards The Heads in Port Phillip Bay, Victoria; from Port Adelaide south along the Adelaide metropolitan coast in South Australia; and from the Derwent Estuary south to the D'Entrecasteaux Channel plus more pristine sites in eastern Tasmania (Figure 1). Because pollutants generally have an ongoing legacy of impact regardless of potential reductions of pollutant inputs in recent years due to improved environmental management, reef community data for individual RLS/ LTMPA sites were aggregated for all available sampling periods (1991-2015 inclusive). Comparisons between aggregated and the most recent community-level data for each site revealed broad qualitative similarity.

Investigated sites were spread as evenly as logistically-practical across pollution gradients. Where multiple time points were sampled at particular reef sites, data were standardised as densities per individual transect. Measurements of pollutants at each site involved sampling duplicate sub-sites spread 50 m apart, enabling averaged conditions to be estimated at the site level for all pollutants except for micro-plastics, for which extraction and enumeration was highly time consuming and thus only a single sample was processed within the time-frame of the study.

Pollutants were sampled across all sites during Sep-Dec 2015, with laboratory determination of pollutant levels occurring from Oct 2015 to Dec 2016. Labile pollutants (e.g. nutrients and petro-chemical compounds) were assessed within 2 weeks of collection, while non-labile material such as micro-plastic concentrations were processed within ~12 months of sample collection.

Pollutants measured included heavy metals, specifically Antimony, Arsenic, Cadmium, Chromium, Copper, Cobalt, Lead, Manganese, Nickel, Selenium, Silver, Vanadium, Zinc and Mercury. Heavy metal and organic pollution samples (i.e. total organic carbon) were analysed by ALS Environmental Pty Ltd Australia (http://www.alsenviro.com; 277-289 Woodpark Rd, Smithfield, NSW, 2164). Analysis of Nitrogen and N15 enrichment, indicating urban sources of N, was performed by Environmental Isotopes Pty Ltd (http://www.isotopic.com.au/). Micro-plastics were extracted from marine sediments using density separation by NaCl plus size-graded sieving (38 µm to > 4 mm) and centrifuging with all plastics collected onto filter paper and enumerated under dissecting microscope. Counts distinguished plastic particles from filaments such as polyesters shed from clothing made from synthetic fabrics (Ling et al. *In Prep.*).

In order to obtain signals of pollutants directly from reefs where fish, invertebrates and macroalgal abundance was scored by RLS divers, divers also sampled fine sediment layers trapped within algal turfs by suctioning with 50 ml syringes. Comparison of heavy metal pollution measurements for turf-trapped sediments on reefs and conventional Van Veen grabs of sediment from adjacent sandy/silty habitats (within 300 m of the reef site) revealed high concordance. Heavy metal levels in the turf-sediment environment on reefs were therefore used in statistical analyses as these showed the most direct relationship to the reef community. By contrast, isotopic signals of organic pollutants, petro-chemical surrogates and micro-plastics required larger volumes of sediment than was readily obtainable from the reef surface, consequently soft-sediment habitats adjacent to reef sites were sampled by Van Veen grabs for these purposes.

Reef community metrics were examined across pollutant gradients using multiple linear regression to identify the most informative biological indicators of pollution. The list of candidate indicators included diversity indices of species richness, Shannon diversity, number of effective species, and evenness for each reef community component of fishes, cryptic fishes, invertebrates, benthic habitats, and whole communities (inclusive of all taxonomic components). Functional traits were also explored singularly and combined within community groupings (*after* Stuart-Smith et al 2013), with each trait explored for abundance and biomass based community weightings using means at the site level where pollutants were measured.

Initially, pollutant types were examined by cross-correlation, with exclusion from subsequent analyses of highly-correlated (i.e. R > 0.70 across the 43 sites) pollutant components, using the criterion that only the most informative/cost effective pollutant variable was retained (correlation matrix shown in Appendix I). Due to known relationships of fishes, invertebrates and habitat type with sea temperature at large scales, and with wave exposure at local scales (*see* Stuart-Smith et al 2015a&b), both average sea surface temperature (SST) and wave exposure (derived via wave fetch modelling, *after* Hill et al 2010) were initially included in multiple regression models. These models possessed the following general form:

$$\textbf{bio\_indicator}\ (y_{\mathrm{i}}) = \beta_0 + \beta_1 x_1(\mathrm{SST}) + \beta_2 x_2(\mathrm{Exposure}) + \beta_3 x_3(\mathrm{Metals}) + \beta_4 x_4(\mathrm{d15N})$$

$$+\beta_5 x_5(\mathrm{TOC}) + \beta_6 x_6(\mathrm{Sed\_N}) + \beta_7 x_7(\mathrm{Petrochems}) + \beta_8 x_8(\mathrm{Microplastics}) + \varepsilon_i$$

Where y_i is a reef community metric (e.g. fish abundance, invertebrate richness, habitat diversity etc), β_0 is a vector of the predictor variable, *SST* is Sea Surface Temperature in °C (which can account for large proportions of variability in faunal biodiversity patterns and was thus included first, e.g. Stuart-Smith et al 2015), *Exposure* is wave exposure (*after* Hill et al 2014; again a key environmental variable known to account for reef biodiversity patterns, Stuart-Smith et al 2015); *Metals* is total heavy metals (inclusive of those stated above which were aggregated due to high correlation between some metals and to also provide generality and to reduce the number of independent variables to be estimated in model), *d15N* is delta 15 Nitrogen (an isotopic form of Nitrogen enriched by anthropogenic N synthesis), *Sed_N* is total sediment nitrogen (inclusive of natural and anthropogenic sources), *TOC* is Total Organic Carbon (inclusive of natural and anthropogenic sources); *Petrochems* is the concentration of petro-chemical surrogates of ethylene dichloride, Toluene-d_8, 4-Bromofluorobenzene; *Microplastics* is total micro-plastic concentration in marine sediment immediately adjacent to reef, and ε_i is the error term. For spatial maps of modelled non-pollutant and pollutant variables across the south-eastern Australian sampling sites *see* Appendix II.

All correlation and regression analyses were undertaken using *R* (R Development Core Team 2015) in *RStudio* (RStudio Team 2015); specifically, the *R* package '*Relaimpo*' (https://cran.r-project.org/web/packages/relaimpo/relaimpo.pdf) was used to partition the contribution of each predictor variable to the overall R^2 of each model fit. Furthermore, the stepwise statistical regression routine was then subsequently used to identify predictor variables defining the most parsimonious models, with *Relaimpo* then used to further examine the contribution of predictor variables to the most parsimonious models.

2.2.4 Invasive species

Ecological impacts of invasive species can be difficult to tease apart from those due to numerous other pressures. This pressure can also be considered an ecological response, and so is not clearly placed in standard indicator frameworks (e.g. Driver-Pressure-State-Indicator-Response). The proportion of invasive species was used as an indicator for this study, calculated from the individuals of species that are not native to Australia amongst the mobile invertebrates and bottom-dwelling small fishes surveyed in 50 m x 1 m transect blocks (using RLS method 2). This was also applied to the RLS data by Stuart-Smith et al. (2015b), and is the most direct metric of ecological 'state' relating to this pressure. Other than the cyptic species recorded in RLS method 2, no other non-native fishes were recorded elsewhere in Australia, meaning that the addition of fish data from 50 m x 5 m transect blocks (method 1) provided no additional information relating to this pressure. The photoquadrat images have not been scored to species-level to date, and thus any relevant data from method 3 on introduced seaweeds or sessile invertebrates were unavailable at the time of this report.

2.2.5 Crown of thorns sea stars

Crown of thorns sea star (CoTS) outbreaks represent one of the biggest threats to Australian coral reefs. Similar to the invasive species indicator, this is another case in which the 'pressure' is part of the monitored biodiversity and provides an unusually direct link to ecosystem 'state'. Densities of CoTS directly translate to the magnitude of coral mortality observed on reefs at local scales (De'ath et al. 2012), and so provide a reasonable biodiversity indicator for impacts of this species. Densities measured on reef surveys were aggregated to represent the number of individual per hectare of reef, as this is the scale most often reported in other studies.

2.2.6 Sea urchin overgrazing

Change in sea urchin herbivory represents one of the strongest drivers of reef state both globally and for shallow Australian reef ecosystems (*reviewed by* Ling et al 2015). Increase in sea urchin abundances and resultant overgrazing is also an indirect cascading response to ecological overfishing, mediated via reduction in abundance of large 'urchinivorous' predators (e.g. Ling et al 2009; 2015). Similar to both the invasive species and CoTS indicators, sea urchin abundance is another case in which the pressure is part of the monitored biodiversity and provides an unusually direct link to ecosystem biodiversity 'state' (Ling 2008).

The impact of sea urchin grazing follows non-linear tipping-points, whereby transformation of temperate kelp beds to sea urchin barrens through overgrazing occurs at biomass of ~700 g m^{-2}, while recovery of kelp beds only occurs once sea urchin biomass drops below ~70 g m^{-2} (Ling et al 2015). Given the existence of this non-linear collapse/ recovery dynamic, we here estimate urchin biomass for long-term monitored reefs in south-eastern Australia (based on allometery of average size for each overgrazing species of sea urchin) to broadly assess reef sites relative to established upper and lower tipping-points for overgrazing species. For specific kelp overgrazing species on temperate Australian reefs, namely *Centrostephanus rodgersii* and *Heliocidaris erythrogramma*, densities on reef sites where assessed relative to experimentally determined overgrazing densities of 2.2 and 8.0 urchins m^{-2} (~700 and 450 g m^{-2}), and kelp recovery densities at below 0.27 and 4.0 urchins m^{-2} (70 and 213 g m^{-2}) for each species respectively (Ling et al 2015; Kriegisch et al 2016).

Notably, urchin densities between upper and lower tipping-points can exist as either kelp beds or urchin barrens states (i.e. the range over which bifurcation exists) and thus assessment against this range gives an index of where overgrazing could be problematic, but for which monitoring through time is required for urchin density to be an effective indicator of ecosystem state, given the context state-dependent nature of this indicator (Ling et al 2015). While sea urchin barrens can be remarkably stable through time, for example extensive *Centrostephanus* barrens across NSW (Andrew & Byrne 2007), change in sea urchin abundance and subsequent change in reef ecosystem state has been documented to occur via range-extension and population increase of *Centrostephanus* in eastern Tasmania (Ling et al 2009; Stuart-Smith et al 2010). Changes in reef state have also been observed for mobile feeding fronts of *Heliocidaris* in NSW (Wright et al. 2005), Tasmania (Ling et al 2010) and Port Phillip Bay (Kriegisch et al 2016), and also for *Tripneustes* on Lord Howe Island (Valentine & Edgar 2010). Therefore, monitoring of sea urchin abundance represents a single taxonomic group indicator of ecosystem state which is tractable through time.

2.2.7 Species vulnerability

An index of threatened species was calculated as the proportion of all species recorded on RLS method 1 and 2 (belonging to the bony fishes, elasmobranchs, marine mammals, reptiles, echinoderms and molluscs) that are listed by the IUCN as threatened (i.e. Vulnerable, Endangered, or Critically Endangered). Use of the IUCN listings provided indicator values that are globally comparable. Marine species are poorly covered by the Australian threatened species listing system (the EPBC Act, 1999), making it a less consistent basis for an indicator.

2.3 Mapping indicator values

Data from 1,294 sites surveyed by RLS divers in Australian waters between 2010 and 2015 (inclusive) were used in spatial analyses for this study (Fig 1). The indicators in the final suite resulting from the process above were each calculated using data from individual RLS surveys from 2010-2015, averaged among multiple surveys per site. Values were mapped around Australia, using inverse distance weighting interpolation to represent averaged values in regions with dense data points and to interpolate to regions with sparser data points. The interpolation was completed at a resolution of 300 by 300 pixels (approx. 23 km by 12 km cell size), and values were extended to a maximum of 100 km from survey sites to enable visualisation of a broader strip of colour around the coastline. Values only apply to reef habitats within the coloured areas of maps. The invasive species and CoTS indicators were zero around much of the continent, and so only individual sites with values >0 were plotted (these are often locally concentrated, and so overlap considerably).

For sea urchin overgrazing indicators, i.e. for the sea urchins *C. rodgersii* and *H. erythrogramma*, mapping was performed on the frequency of sites within 1 degree bins where urchin density exceeded that required to overgraze a 5 m^2 area of reef, i.e. to form ‘incipient barrens’ (*after* Johnson et al 2005). For *C. rodgersii* and *H. erythrogramma* respectively this equates to 11 and 20 urchins per 50 m^2 instead of 110 or 200 given that the overgrazing density of 2.2 or 4.0 urchins m^{-2} would be locally exceeded at the 5 m^2 scale if all individuals along a 50 m^2 transect were densely aggregated.

2.4 Integration of data from the three programs to assess temporal trends in indicators

A key aim of this study was to determine whether and how the datasets from the three major reef monitoring programs (and any others compatible with each) could be integrated for the purpose of assessing long-term trends in biodiversity indicators. The LTMPA and RLS datasets are directly compatible, only requiring common level of replication achieved in calculation of indicators. Thus, integration only requires calculating indicators per 500 m^2 reef area when using data on fishes, or per 50 m^2 using invertebrate data (but note sea urchin and CoTS densities are calculated over a different area to allow direct comparability with other studies, as described above). The AIMS LTM data uses different methods, however, so assessment of the ability to calculate the same indicators, and quantitative differences with the other datasets when doing so, required specific comparative analyses. Differences were investigated by two approaches: (1) Direct comparison of indicator values from co-located RLS and AIMS LTM surveys; and (2) Using the RLS data from along the entire GBR to re-calculate and compare indicator values based only on the same subset of species recorded in AIMS surveys.

For the first approach, two RLS divers joined the AIMS fish monitoring team for surveys of the southern GBR in October 2015. They surveyed 24 transects at 12 sites around four reefs, with RLS divers using standard RLS methods along two of the five transects laid along each reef site by the AIMS team. Indicators were calculated from AIMS and RLS surveys at each of these sites and linear regression used to determine the strength and nature of relationships between the two.

For the second approach, restriction of RLS data to AIMS species list was undertaken statistically. Indicator values were calculated from the full RLS data from 202 sites along the Great Barrier Reef, and then again from the same RLS data but using only observations of the subset of species (from 10 families) that are recorded in AIMS LTM surveys. Linear regression of the two values from the same sites was used to provide a correction factor to be applied to AIMS surveys for the study.

Once appropriate corrects had been made to allow integration, time series analyses were based on RLS data from 372 sites at nine locations, LTMPA data from 182 sites at five locations, and AIMS data from 276 sites along the GBR (which were divided into four regions along the Great Barrier Reef to allow independent examination of regional trends). All monitoring locations assessed using data from the three programs are shown as large circles in Figure 1.

Only fishing and warming indicators were assessed for temporal trends, as few CoTS, invasive and threatened species were present at any locations with adequate time-series. Data within each dataset were mostly collected in the same season each year, or sometimes across two seasons (e.g. summer/autumn). Any data from a third season were omitted from temporal trend assessments to prevent confounding of seasonal and inter-annual trends.

Trends in CTI were based on means of raw values across sites in each year, as values from different locations were all on the same scale (^oC). Fishing indicator values differed considerably from site to site and location to location, so were standardised to the maximum value for each site across the years of survey. The mean standardised values among sites in a given year were plotted.

A caveat associated with time-series analyses for fish biomass indicators based on AIMS data is that they are based on a relatively small subset of the species included in biomass indicators from the other two datasets. Although standardisation for each site and consistent application of AIMS methodology for time series in the GBR mean temporal trends are comparable, comparison with trends from other regions requires consideration that biomass changes in species not covered by AIMS methods will not be captured.

3. RESULTS

Results for analyses designed to test responsiveness of fishing and pollution indicators are below, followed by analyses involving integration of datasets.

3.1 Fishing indicator selection analyses

The fishing indicator selection process found model fit was significantly improved with the addition of the fishing proxies for all indicators tested (Table 1). The community-weighted mean (CWM) of the vulnerability index of Cheung et al. (2005), as provided in Fishbase, showed the best fit to modelled fishing pressure when weighted by species abundance, and the third-best fit when weighted by biomass of species. This index is based on a range of life-history parameters including age at maturity, fecundity, longevity and range size (Cheung et al. 2005), and was developed for the purpose of identifying species that are vulnerable to fishing. However, for data-poor species (i.e. most species, including virtually all unexploited populations), the vulnerability index reduces to an index of maximum size (Lmax), which is the only life history parameter available for all species. When assessed separately, the Lmax CWM was second in the rankings based on model fit. Thus, the top three fishing indicators effectively describe the same effect of fishing, in changing the composition of fishes observed on RLS surveys around Australia based on the maximum size they can attain.

Table 1. Ranking and model results for fishing indicators calculated from RLS surveys around Australia. Vulnerability, Lmax, and Trophic Level values are calculated as community-weighted means, with the mean index value of members of the community weighted by the log of their abundance ('B' indicates biomass weighting instead of abundance). The χ^2 goodness of fit (column B) is from the likelihood ratio between models with all four fishing pressure variables versus models including environmental variables but no variables related to fishing pressure. The significant individual proxies of fishing pressure for which the trend was in the direction consistent with fishing are shown in column C (MPA = no take vs fished, Pop = human population index, BR = distance from nearest public boat ramp, SF = shore fishing index). Values in column D represent standardised beta coefficient values for the effect of mean annual sea surface temperature. *NS*: $p > 0.05$, *$p < 0.05$, *** $p < 0.001$. Final rank is shown for the top four indicators, following rationale provided in the text.

A. Indicator	B. χ^2 goodness of fit	C. Significant fishing effects	D. Significant SST effect	Rank
Vulnerability Index[1,2]	69.9***	3 (MPA, Pop, SF)	-0.87***	
Lmax[2]	58.9***	2 (MPA, Pop)	-0.59***	
Vulnerability Index[1,2] (B)	58.3***	3 (Pop, BR, SF)	-0.80***	
B20[3]	45.6***	3 (MPA, Pop, BR)	*NS* (0.07)	1
Total Biomass[3]	42.8***	2 (Pop, BR)	0.23*	
Lmax[2,4] (B)	41.3***	2 (MPA, Pop)	-0.33***	
Gamma Scale[5]	41.2***	2 (Pop, SF)	*NS* (-0.07)	2
Trophic Level[2,6]	38.3***	-	-0.30***	
Mean Length[4]	35.6***	2 (Pop, SF)	0.62***	
Max of Lmax[7]	34.7***	2 (MPA, Pop)	*NS* (0.12)	3
B30[3]	34.6***	2 (MPA, Pop)	*NS* (0.09)	4
Mean biomass	32.1***	2 (Pop, SF)	-0.49***	
B Exploited[8]	32.0***	2 (MPA, Pop)	0.18*	
Proportion pelagic[9,10]	28.9***	1 (MPA)	*NS* (0.05)	
Elasmobrach B[11]	23.7***	2 (MPA, Pop)	*NS* (0.05)	
Proportion piscivorous[10]	23.3***	2 (MPA, Pop)	0.57***	
Trophic Level[6] (B)	21.2***	-	*NS* (-0.13)	
Proportion B Exploited	16.5*	1 (MPA)	*NS* (0.08)	
B spectrum slope[7]	16.0*	2 (Pop, BR)	-0.35***	
Large Fish Index (20 cm)[11]	15.9*	2 (Pop, SF)	-0.52***	
Richness spectra slope	10.2*	1 (MPA)	-0.42***	

[1] Cheung et al. (2005), [2] Fishbase.org, [3]Edgar et al. (2014), [4]Jennings et al. (1999), [5]Thomson et al (in prep), [6]Pauly et al. (1998), [7]Shin et al. (2005), [8]Willis et al. (2003), [9]Rochet and Trenkel (2003), [10]Methratta and Link (2006), [11]Cury and Christensen (2005).

The vulnerability index and Lmax were also significantly related to mean annual SST, with strong natural gradients towards lower values in northern Australia (smaller, short-lived fishes dominate by abundance in warm areas). Although it might be possible to standardise CWM values of the vulnerability index by local SST to provide a metric that was comparable among regions, the tight relationship with temperature indicates that future changes in its values may be influenced by ocean warming. Further to this, a recent study found the CWM of the vulnerability index based on RLS data (as used here) was very closely associated with a pollution gradient, with a trend in the opposite direction to that expected from fishing pressure (i.e. increasing prevalence of vulnerable species with greater fishing pressure; Stuart-Smith et al. 2015b). This and the strong relationship with SST imply that this indicator has poor specificity for fishing impacts (Shin et al. 2010), and that its interpretation as a fishing indicator could be confounded by changes arising from pollution or warming, when these pressures overlap with fishing pressure.

The biomass of all fishes in size classes 20 cm and above, hereafter referred to as B20, was the most sensitive indicator that was not significantly related to mean SST at the continental scale (Table 1), and was followed closely in rankings by the scale parameter from a gamma model of the size spectrum of fishes. The latter also described the trend for a reduced density of larger fishes in the size spectrum (regardless of species identity) with increasing fishing pressure, and is theoretically specific to fishing impacts, although complete specificity is unlikely to be realistic for any fishing indicator.

3.2 Relative importance of pollutants and candidate indicators

For reefs in south-eastern Australia, analysis of a suite of potential indicators of pollutants revealed heavy metals to have the greatest and most consistent effects on reef community metrics relative to other pollutants; accounting for 6 of the top 10 parsimonious and regionally consistent multiple regression models (Table 2a). Overall, potential biological indicators were negatively associated with increasing heavy metal concentrations, with heavy metals implicated as having deleterious biological effects for 21 of 26 parsimonious models that included heavy metals (Appendix III). Across all indicators, the next ranked pollutant associated with negative biological impacts was total Nitrogen (notably derived from both natural and anthropogenic sources), which contributed the remainder of the top 10 models. Across all multiple regression models, Nitrogen was negatively associated with 12 potential indicators (Appendix III). Furthermore, the top-ranked indicator of any pollutant was the community metric of *Benthic Functional Diversity*, which was negatively associated with increasing Nitrogen (Table 2a; *see* also Appendix IV). Summed across all indicators, the non-pollutant variable of wave exposure showed greatest importance to multiple regression models by contributing to 29 of 42 parsimonious models, while sea surface temperature contributed to 23 models (Appendix II). Intuitively countering the effects of pollution, wave exposure had an apparent ameliorating effect on 25 of 29 candidate pollution indicators (Appendix III).

Relative importance decreased for the pollutant variables of d15N, Microplastics, Petrochem surrogates, and TOC; with the top-ranked indicator for each of these pollutants having poor overall fit and small apparent contributions by each pollutant, with as many positive effects as negative effects on candidate indicators across the different locations (Table 2b; see also Appendix III). For heavy metals, the best performing indicator was the *Reef Community Shortness index* (i.e. a community weighted mean of maximum lengths for all fish and invertebrate species and biogenic habitats present in the community), with communities becoming progressively shorter on reefs with increasing heavy metal concentration (Table 2a; Appendix IV).

Table 2. Model results and ranking of pollution indicators for rocky reef communities.
Calculated from RLS and LTMPA surveys spanning 43 sites in temperate south-eastern Australia where pollutant levels were co-measured. (a.) Pollutant indicators listed by contribution to parsimonious model fit and consistency in regional trends across NSW, Vic, SA and Tasmania; note that a total of 42 candidate biological indicators were regressed against measured pollutant levels (see Appendix I for full results). Final rank is shown for the top ten pollution indicators that demonstrated greatest contribution to model fit plus also showed consistent trends across the 4 south-eastern Australian locations NSW, Vic, SA, Tas. (b.) Top-ranked indicators for the pollutant variables of d15N, TOC and Petrochemicals (which ranked well outside the top-10) explained low variability in reef community patterns relative to heavy metals and/ or nitrogen. All diversity/ functional diversity values were calculated as community-weighted means, with the mean index value of members of the community weighted by abundance ('B' indicates biomass weighting instead of abundance). Note the saturated model includes the independent variables SST, Exposure, Metals, d15N, TOC, Sed_N, Petrochems, Microplastics; with parsimonious terms identified using "backward" stepwise regression. Further note that all pollutant types were uncorrelated (max. Pearson correlation coefficients < 0.70 or > -0.70; *see* Appendix II). All primary pollutants had negative effects on the biological indicator of interest, except where highlighted with an asterisks in which case the primary pollutant had a positive effect on the indicator.

	Indicator	Saturated model fit R^2	Saturated model *P*	Parsimonious model terms (*italics* = negative; black=positive; **bold** = significant effect)	Parsimonious model fit R^2	Parsimonious model *P*	Primary Pollutant	Primary pollutant R^2 contribution	Regional consistency in primary pollutant trends	Rank
a	Habitat Functional Richness	0.37	**0.035**	SST + ***Sed_N***	0.29	**0.001**	Sed_N	0.26	Yes (4/4)	1
	Turf Cover	0.53	**0.001**	**SST** + ***Exposure*** + d15N + **Sed_N** + **Microplastics**	**0.49**	**0.000**	Sed_N*	0.26	No (3/4)	
	Canopy Seaweed Cover	0.37	**0.037**	*SST* + Exposure + ***Sed_N***	**0.34**	**0.001**	Sed_N	0.25	No (3/4)	
	Habitat Richness	0.50	**0.002**	***SST*** + ***Metals*** + **d15N**	**0.44**	**0.000**	Metals	0.24	No (2/4)	
	Community Shortness Index	0.39	**0.022**	Exposure + **Metals**	**0.33**	**0.000**	Metals*	0.23	Yes (4/4)	2
	Fish Functional Richness	0.29	0.155	***Metals***	0.22	**0.002**	Metals	0.22	No (3/4)	
	Fish Length	0.61	**0.000**	SST + **Exposure** + **TOC** + ***Sed_N***	**0.58**	**0.000**	Sed_N	0.21	No (3/4)	
	Invertebrate Lmax	0.25	0.240	***Metals***	0.20	**0.003**	Metals	0.20	Yes (4/4)	3
	Invertebrate diversity H	0.40	**0.020**	SST + ***Exposure*** + d15N + ***Sed_N***	0.33	**0.004**	Sed_N	0.17	Yes (4/4)	4
	Whole Community Richness	0.36	**0.040**	**Exposure** + ***Metals***	0.31	**0.001**	Metals	0.16	Yes (4/4)	5
	Invertebrate & fish Lmax	0.37	**0.035**	*SST* + **Exposure** + ***Metals***	**0.36**	**0.001**	Metals	0.15	Yes (4/4)	6
	Fish Biomass	0.47	**0.004**	**Exposure** + ***d15N*** + ***Sed_N*** + *Microplastics*	**0.47**	**0.000**	Sed_N	0.14	No (3/4)	
	Invert. & cryp. fish diversity H	0.35	**0.055**	**Exposure** + Metals + *Sed_N*	0.24	**0.014**	Sed_N	0.14	Yes (4/4)	7
	Invert. & cryp. fish diversity H'	0.36	**0.041**	SST + Exposure + Metals + d15N + ***Sed_N***	0.32	**0.014**	Sed_N	0.13	Yes (4/4)	8
	Fucoid seaweed Cover	0.52	**0.001**	***SST*** + Exposure + ***d15N*** + ***Sed_N***	**0.52**	**0.000**	Sed_N	0.13	No (2/4)	
	Invertebrate Richness	0.42	**0.011**	***SST*** + **Exposure** + ***Metals***	**0.39**	**0.000**	Metals	0.13	Yes (4/4)	9
	Laminarian Kelp Cover	0.25	0.231	***Metals*** + ***TOC***	0.24	**0.005**	Metals	0.13	Yes (4/4)	10
b	Invertebrate & cryptic fish abundance	0.43	**0.010**	***SST*** **+** ***Metals*** + **d15N** + **Sed_N** + Petrochem + **Microplastics**	0.43	**0.002**	d15N*	0.09	No (3/4)	
	Fish Trophic Level (B)	0.42	**0.011**	***SST*** + Exposure + TOC	0.38	**0.000**	TOC*	0.09	No (2/4)	
	Fish Trophic level	0.47	**0.003**	***SST*** + **Exposure** + Metals + **Petrochem**	**0.41**	**0.000**	Petrochem*	0.05	Yes (4/4	
	Large Fish Index (30 cm)	0.28	0.165	Exposure + *Microplastics*	0.20	**0.012**	Micro-plastics	0.05	No (3/4)	

3.3 Integration of datasets

Tests for relationships between the AIMS LTM and RLS data for the GBR were undertaken for the two indicators described above that relate to the fish community – B20 and CTI. Direct comparison of these two indicators between RLS and AIMS LTM surveys undertaken together in the southern GBR during Oct 2015 suggested only a very poor relationship in B20 (R^2 = 0.03), and only a moderate relationship in CTI values (R^2 = 0.5) derived using the two methods at the same time and place. Averaging values derived from individual transects across multiple transects at each site improved the relationship noticeably for B20 (R2 = 0.65; Fig 2a).

Using the RLS data from 202 sites along the entire GBR to re-calculate and compare indicator values based only on the same subset of species recorded in AIMS surveys suggested a comparatively closer relationship in B20, but poorer relationship in CTI between the two survey methods (Fig 3). RLS surveys covered a greater range in CTI (from 24.4 to 27.2°C *cf.* 24.9 to 26.9°C for AIMS subset) and averaged 0.23°C 'warmer', but increased less with each degree of increase in CTI from the AIMS subset.

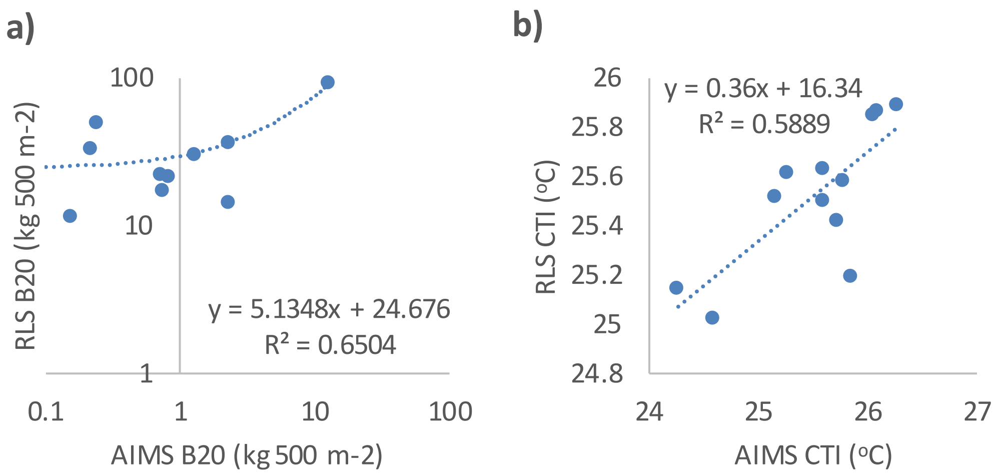


Figure 2. B20 (a) and CTI (b) from co-located AIMS LTM and RLS reef fish surveys, averaged across surveys at each site. Both axes in the B20 plot are on a logarithmic scale.

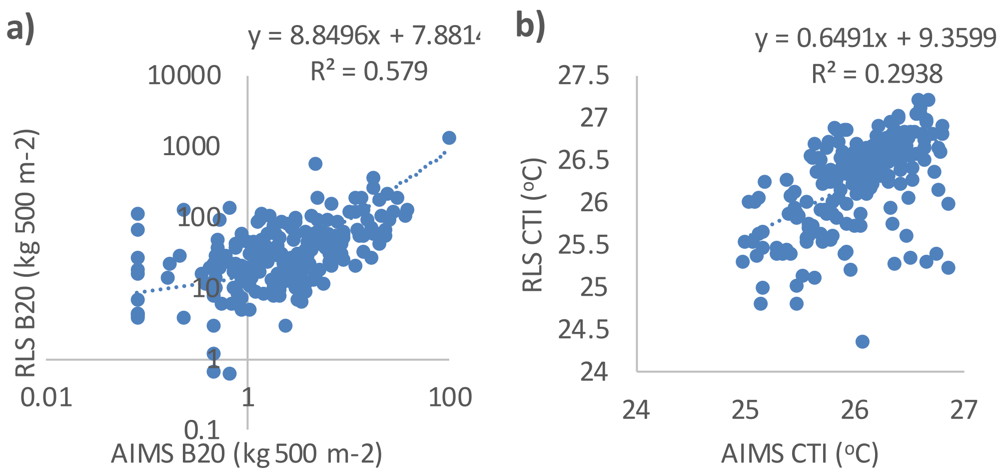


Figure 3. B20 (a) and CTI (b) from RLS reef fish surveys using data from all species (y-axis), or the subset of AIMS species (x-axis). Both axes in the B20 plot are on a logarithmic scale.

3.4 Mapping indicator values

Maps of indicator values for ecological state relating to fishing, warming, invasive species, and for threatened species reveal clear patterns in the state of Australian rocky and coral reefs, where the footprint of human population centres is visible (Fig 4). The map of B20 suggests some relationship with mean SST, even though this was not significant in the linear mixed model and B20 had one of the lowest standardised coefficients for SST of all indicators tested (Table 1). Reef fish communities in southern Australia are typified by lower biomass of fishes 20 cm and over. Clear local deviations can be seen from natural gradients, however, with localised areas of depressed B20 relative to surrounding areas, observable at population centres along the east coast and in the south-west, and at Ashmore and Hibernia Reefs in the north-west, and with some depression at Ningaloo Reef. High values of B20 were found, on average, on reefs in the Gulf of Carpentaria, despite lower mean SST than the GBR.

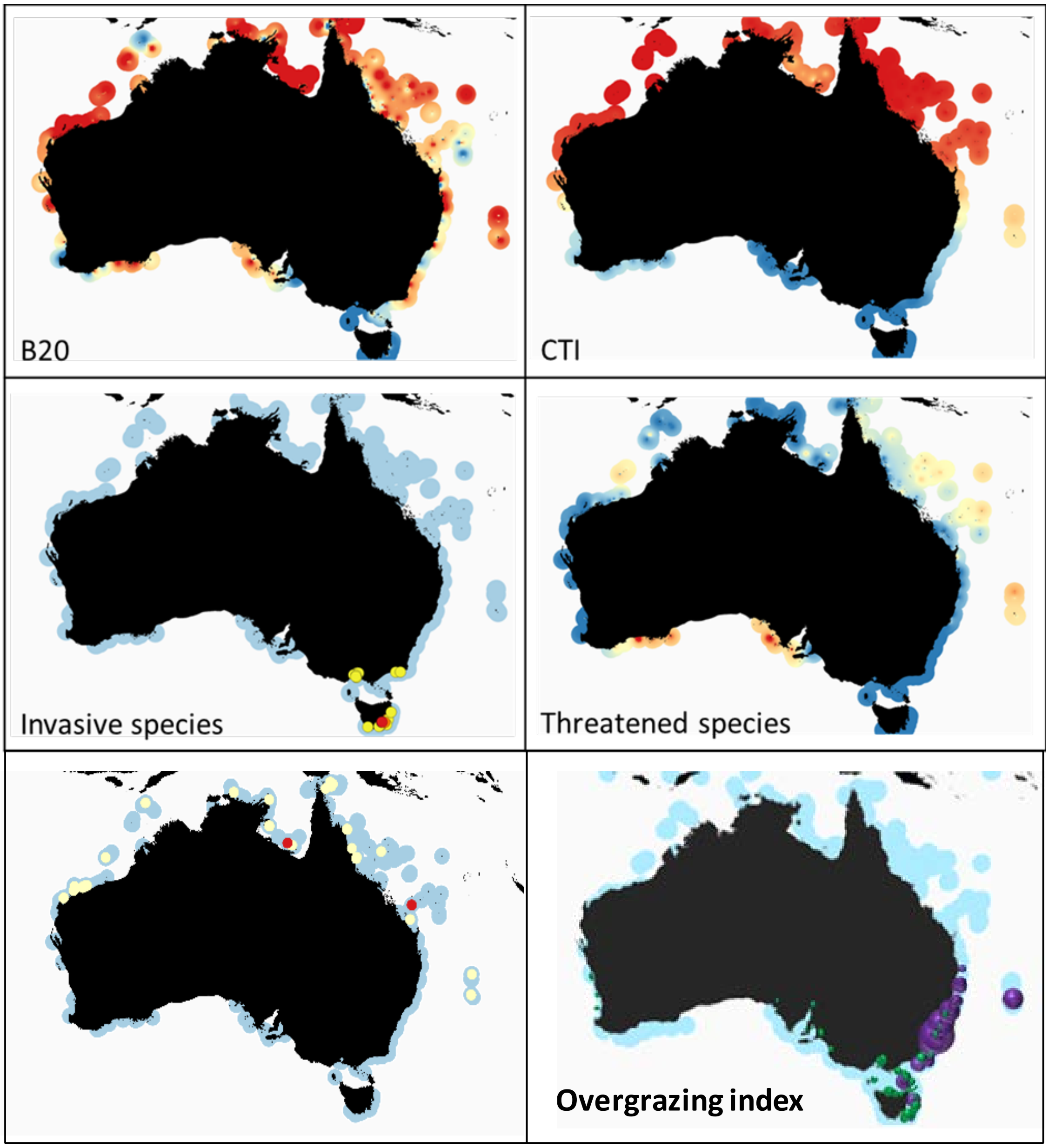


Figure 4. Distribution of indicators of reef biodiversity.
In relation to fishing pressure (top left), ocean warming (top right), invasive species (middle left), threatened species (middle right), crown of thorns sea stars (CoTS; bottom left) and sea urchin overgrazing index (bottom right) based on quantitative surveys of coral and rocky reefs (N=1,294 sites). B20 is total biomass of fishes 20 cm or larger, and CTI is the community temperature index. The CTI represents the current mean thermal affinity of reef fish communities rather than implying any warming-related change (shown in Fig 5). Invasive species and CoTS were only plotted for sites at which they were recorded, with yellow indicating a range from invasive species presence up to 30% of individuals belonging to invasive species or CoTS densities 1-200 per Ha, and red indicating values from 30 to 95% of individuals of invasive species and >200 CoTS per Ha. Otherwise colour scales are interpreted as red being the highest values in the dataset and blue as the lowest (zero for invasive and threatened species). For the Urchin Overgrazing Index, bubbles indicate the frequency of sites within local areas (0.1° bins) whereby the biomass of herbivorous sea urchins exceeds the known tipping-points of kelp bed overgrazing for the species *Centrostephanus rodgersii* (purple bubbles; max. bubble size is 30 sites) and *Heliocidaris erythrogramma* (green bubbles; max. bubble size is 10 sites). Note the Urchin Overgrazing Index was only established for temperate regions given that tipping-points for tropical urchin species in Australia are yet to be established.

Invasive species were absent from reef survey data across most of the continent, but localised high densities were found in the south-east, ranging up to 100% of individuals surveyed (in RLS method 2; reported in Stuart-Smith et al. 2015b). These were made up of nine non-native species from four phyla (Arthropoda, Chordata, Echinodermata, Mollusca).

Crown-of-thorns seastars exceeded 200 per ha in only two locations, southern Great Barrier Reef and on Mornington Island in the Gulf of Carpentaria (Fig 4). On temperate reefs, sea urchin densities exceeding that required for local overgrazing were frequently observed along the south-eastern region of the continent, with the sea urchin *Centrostephanus rodgersii* regularly triggering the Overgrazing Index along the NSW coastline and into north eastern Tasmania. In contrast, *Heliocidaris erythrogramma* frequently achieved densities sufficient for overgrazing in eastern and northern Tasmania and into Victoria (predominantly Port Phillip Bay) and east into NSW where overgrazing by both species regularly overlaps (Fig 4).

Thirty-four species listed as threatened on the IUCN Red List were recorded on reef surveys around Australia, with high values of the threatened species index also relatively localised. Western blue groper (*Achoerodus gouldii*; VU) and Australian sea lions (*Neophoca cinerea*; EN) were recorded on many surveys in the Great Australian Bight, leading to high index values in this region (Fig 4). This was also amplified by relatively low local species richness, which meant that each threatened species formed a greater proportion of the community compared with tropical locations. The maximum number of threatened species recorded on any survey block was four, at three sites in the Coral Sea and two offshore sites in the north-west, both areas with relatively high species richness, leading to moderate index values for these sites (4 to 7.5%) (Fig 4).

Spatial patterns in CTI provide little indication of the current distribution of warming impacts, but provide a baseline for future assessment. Importantly, the CTI map (Fig 4) reveals a lack of any obvious north-south gradients along the GBR and the north-west coastline, despite regional gradients in SST along these coasts. The fact that community patterns do not simply conform to the SST trend emphasises the importance of tracking a community metric such as CTI instead of inferring ecological change from changes in SST without a baseline such as this (Stuart-Smith et al. 2015a).

3.5 Temporal trends in indicator values

Temporal trends in B20 and CTI suggest that some reef communities have changed over the past decade because of fishing pressure and warm-water events (Figs 5 & 6). Despite some variation, only four of 15 monitored locations show an increasing trend in B20, while decreases are apparent in at least eight. Some of the declines are very steep, with B20 values dropping by more than 60% at the Capricorn Bunker Group (Queensland), Fleurieu Peninsula (South Australia), Beware Reef (Victoria) and Port Stephens (New South Wales) at some point during the monitored time-series, although values appear to be increasing in the last three years for the latter two locations (and at Sydney, Port Phillip and Rottnest Island).

Values in CTI have been remarkably stable through time in most tropical locations, but distinct impacts of a marine heatwave are evident in the temperate Western Australian locations of Rottnest Island and Jurien Bay in 2011 (and subsequent warm year in 2012). The change in CTI at Rottnest Island over the course of the heatwave was equivalent to the difference in fish communities observed between Rottnest Island and locations more than 250 km further north.

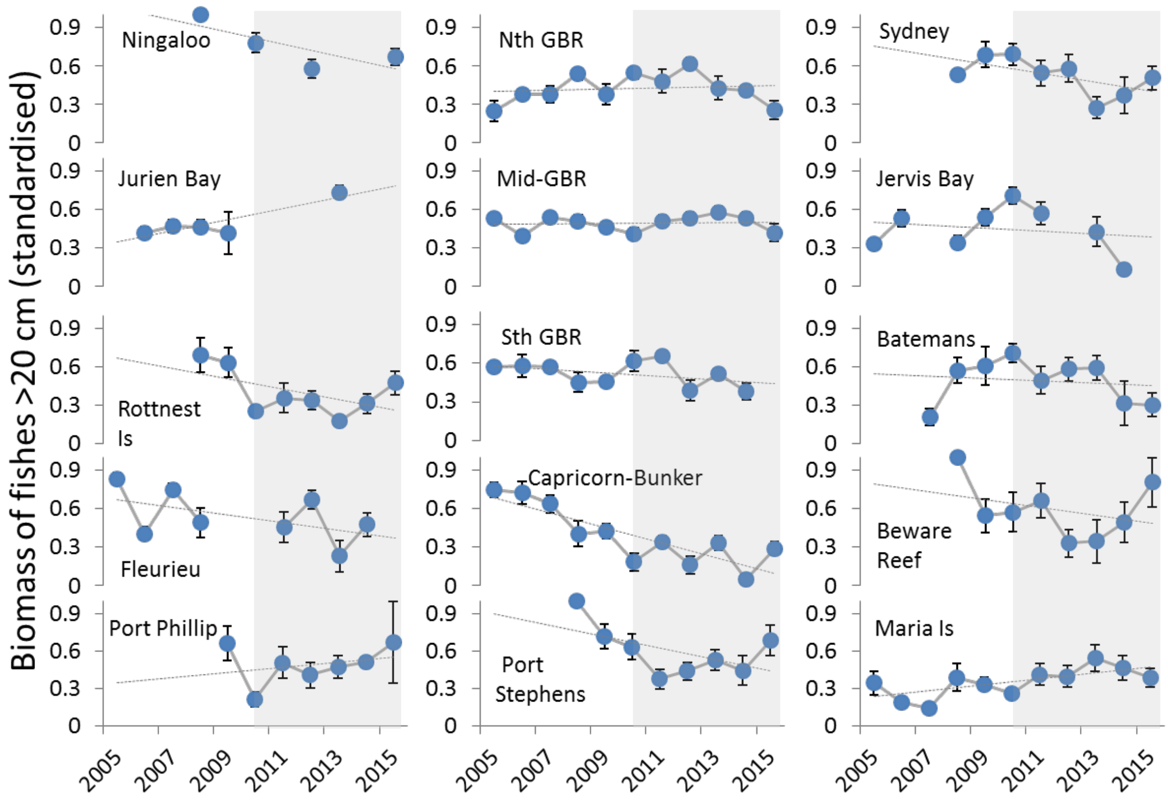


Figure 5. Trends in biomass of large reef fishes (≥20 cm) at monitoring locations from 2005-2015.
Values for each site have been standardised by the maximum for that site over the time series, and means (±SE) of standardised values among a number of sites at each location are shown (mean of 23 sites per location per year). Long-term trends shown by the dotted grey line are linear smoothers. Locations at which MPAs were monitored include sites within and outside MPA boundaries.

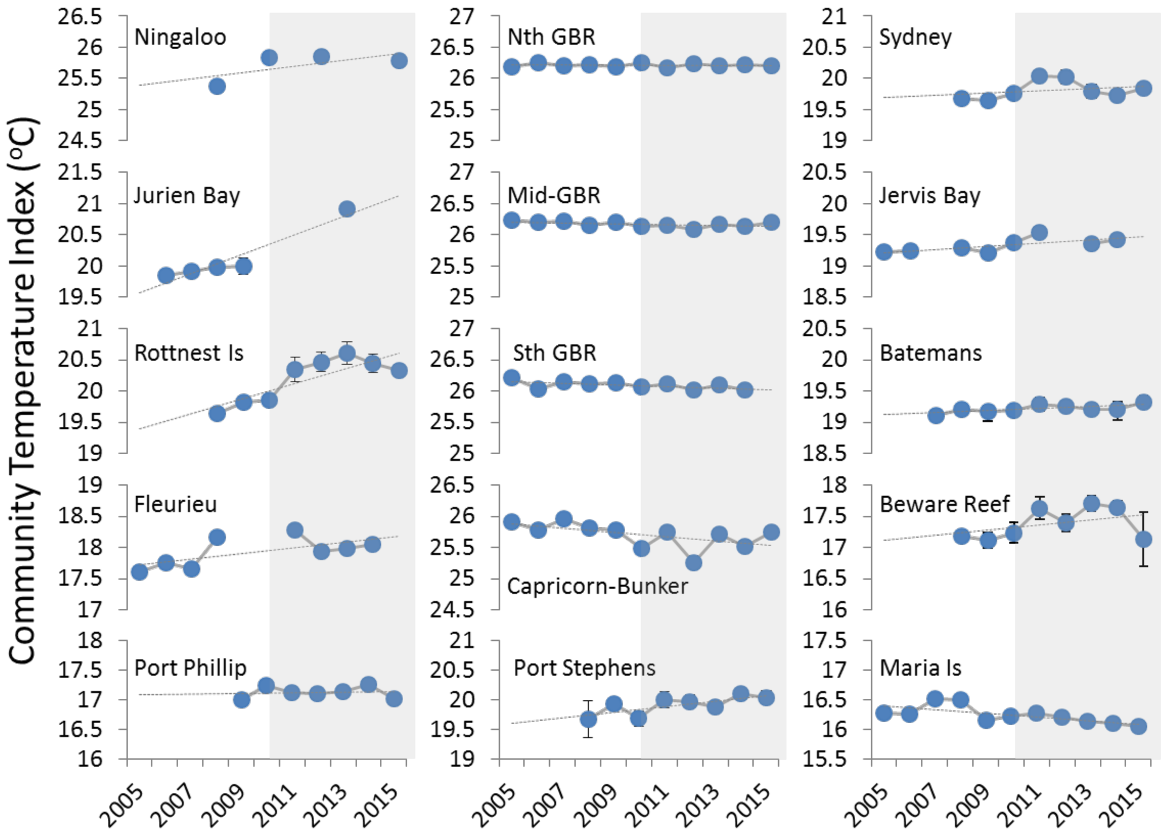


Figure 6. Trends in the community temperature index (CTI) for reef fishes at monitoring locations from 2005-2015.
Each point represents the mean (±SE) of CTI values among sites surveyed at each location in that year (mean of 23 sites per location per year). Long-term trends shown by the dotted grey line are linear smoothers. Locations at which MPAs were monitored include sites within and outside MPA boundaries.

Providing an example of how knowledge of tipping points in ecological condition can allow tracking of critical ecological change, long-term trends in sea urchin *Heliocidaris erythrogramma* biomass at LTMPA sites in Tasmania confirm recovery of kelp beds once urchin biomass drops below the experimentally-determined kelp recovery tipping-point inside the Maria Island (Fig 7a) and Tinderbox (Fig 7c) Marine Reserves; and conversely kelp bed collapse when urchin biomass builds beyond the overgrazing tipping-point at a fished reference site adjacent to Maria Island (Fig 7b). In addition, *H. erythrogramma* at more wave exposed sites can achieve densities required to achieve overgrazing, yet overgrazing is not observed to occur due to supply of drift-algae (Ling et al 2010; Kriegisch et al In Prep.), as shown for reference site 6 “Green Head” (Fig 7d). Thus, while the Overgrazing Indicator may undergo substantial increase at offshore sites, overgrazing may not manifest for this facultative grazing species. For *Centrostephanus rodgersii*, tracking of urchin biomass to either below the kelp recovery or above the point of overgrazing are not available. Instead, long-term monitoring shows high stability in *C. rodgersii* density for LTMPA sites, including urchin biomass maintained above the kelp recovery tipping-point and thus stability of urchin barrens (Fig 7e); but building towards, yet not currently exceeding the overgrazing tipping-point (Fig 7f).

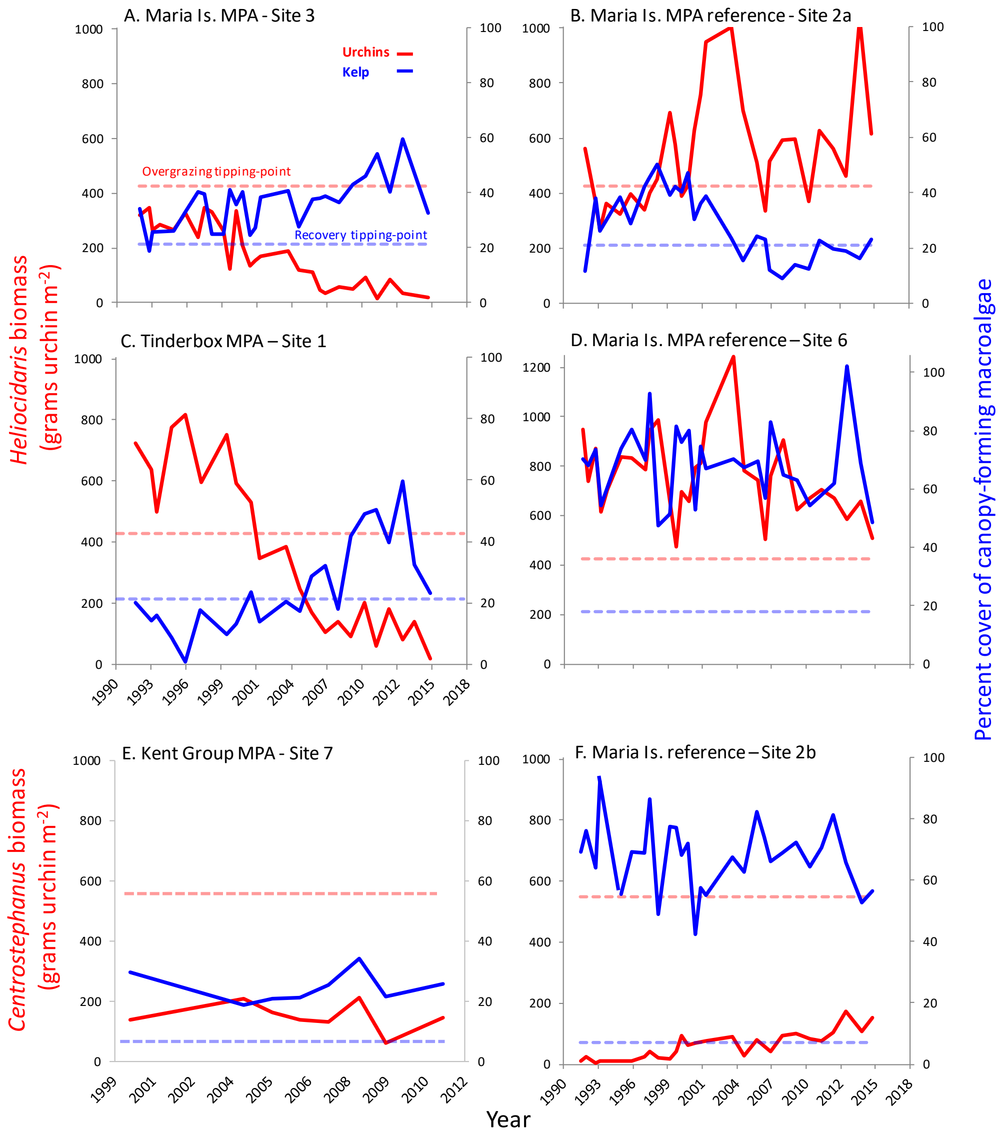


Figure 7. Trends in overgrazing indicators for sea urchins *Heliocidaris erythrogramma* (A-D) and *Centrostephanus rodgersii* (E-F) and subsequent ecological response of kelp beds for example Tasmanian sites surveyed as part of the Long-Term Marine Protected Area monitoring program. Dashed horizontal lines show the respective urchin biomass required for overgrazing (red dash) and kelp bed recovery (blue dash).

4. DISCUSSION

4.1 National marine biodiversity assessment

Our assessment of anthropogenic influences affecting the current state of Australian rocky and coral reef biodiversity suggests that impacts of fishing are the most detectable and widespread among the pressures examined here. Of the fishing indicators tested, the biomass of fishes over 20 cm (and to a lesser extent over 30 cm) and the gamma scale parameter for the size spectrum appear to be most useful for the purpose of assessing trends in shallow reef communities relating to fishing pressure around Australia, and can easily be calculated from a range of available datasets. The vulnerability index also offers an informative means to measure fishing impacts at the national scale, but the interaction with pollution, and potentially with warming, would require careful interpretation of trends.

By contrast, the mean trophic level, or Marine Trophic Index (MTI; Pauly et al. 1998, Pauly and Watson 2005), based on fisheries catch data, has so far used as a headline indicator for measuring progress towards the relevant target for the Convention on Biological Diversity (Aichi Target 6). MTI based on catch data poorly covers impacts on non-commercial species and inshore marine systems where recreational and unreported subsistence fisheries typically operate, and where fishing is generally much less regulated than for large offshore commercial fisheries. This study provided little empirical support for using mean trophic level to track changes in shallow reef fish communities due to fishing pressure (in line with the findings of Branch et al. 2010).

The variable nature of trends in the fishing indicator (B20) at monitored locations suggest that either fishing pressure is highly dynamic or, more likely, that longer time-series are needed to separate true fishing impacts from other sources of natural variation that affect fish production. The spatial analyses integrate observations over multiple years in many locations (2010-2015) and thus smooth out shorter-term variation. The time-series needed to disentangle fishing impacts from natural variation in fish communities has previously been noted to be an important consideration for most fishing indicators (Piet and Jennings 2005). Caution is required in interpreting year-to-year changes in B20 shown in Fig 5, and more emphasis should be placed on longer-term trends of over five years. The purpose of assessing B20 here is not to guide an immediate fisheries management response, but to identify locations where fishing pressure is having greatest impact and provide insight into its magnitude in relation to other pressures. The longer-term trends in B20 suggest that few improvements have occurred in ecological condition around Australia over the last decade, other than in some areas where MPAs form part of the seascape, and where B20 was initially low by national standards (including Jurien Bay and Maria Island).

Changes in B20 are evident across more locations in temperate and tropical regions than are warming driven changes, as measured by CTI (Fig 6). Figure 2 suggests fishing impacts appear to be greatest in localised patches close to large population centres on the east coast, in the south-west, and also at the remote Ashmore and Hibernia Reefs in the north-west (the “MOU Box”), where traditional fishing by fishers from nearby Indonesian islands is permitted.

Long-term ocean warming is beyond the 10 year scope of this assessment, but the impacts of short-term warming events were very clear. In particular, the CTI changes at Rottnest Island and Jurien Bay in Western Australia following a marine heatwave were substantial; the mean thermal affinities of fish communities at these locations has changed more than at other monitored locations presented in Fig 6. A number of impacts of this marine heatwave have been documented (Smale and Wernberg 2013, Wernberg et al. 2013), but more investigation is needed to determine the extent and longevity of changes to ecological functions and local species loss. CTI has been quite stable at most tropical locations, which is consistent with SST trends at these locations over the monitoring periods (which were stable or slightly cooling). The warming trend in the fish community on the east coast of Tasmania that was previously observed (1992 - 2012; Bates et al. 2014) appears to have stalled, with a slight cooling trend evident when only the last 10 years are considered and CTI is abundance-weighted (The CTI values reported in the previous study were richness-based). Longer time-series will likely show this to represent a temporary downward portion of the decadal cycle that overlays a longer-term warming trend (Stuart-Smith et al. 2010, Bates et al. 2014).

The national baseline of CTI provided in Fig 4 will be invaluable for visualising detailed spatial trends in future, providing insights for targeted conservation (Tayleur et al. 2016). In the meantime, reporting on changes in CTI, as well as other indicators assessed here, is important for improving public and policy-makers’ awareness of biodiversity change, guiding where long-term changes in behaviour and management practices are required, and assessing success of current policies. Assessing progress towards Aichi target 10 for the

CBD also requires knowledge of impacts on biodiversity of coral reefs and other vulnerable ecosystems that are related to climate change, and this study represents the first nation-wide assessment of marine biodiversity related to this pressure. It supports recent calls for CTI to be included in the CBD indicator suite (Devictor et al. 2012), and demonstrates a cost-effective mechanism for ongoing reporting for marine communities.

Identification of trends in invasive species provide a basis to evaluate Aichi Target 9, but are not monitored by any national system in Australia - despite having substantial impacts (Bax et al. 2003). Although temporal trends in invasive reef species were unable to be assessed in this study, this was largely due to the rarity of invasive species in reef surveys at long-term monitoring locations. The paucity of invasive species in the reef data suggests that this pressure is not currently having as widespread an impact on biodiversity as are fishing or warming. Nevertheless, invasive species assessed here only cover mobile species recorded on hard substrates, and the map in Figure 4 overlooks aggregations of invasive species amongst anthropogenic structures and soft sediments in Sydney and Melbourne (Hewitt et al. 1999, Glasby et al. 2006), but includes highly abundant invasive species populations such as the northern Pacific seastar *Asterias amurensis* beneath Hobart Wharves aggregated over mixed cobble and mussel shell hard structures (Ling et al 2012). The threat posed by invasive species should not be regarded as negligible outside locations identified in Figure 4 (e.g. Dunstan & Bax 2007; Ling et al 2012); however, highlighted locations appear to be most at risk. A recent global study also identified the south-east as the Australian hotspot for high ecological impact from invasive species (Molnar et al. 2008).

South-east of Australia is also a hotspot for sea urchin overgrazing and ecosystem transformation to urchin barren grounds though population increases of both *Centrostephanus rodgersii* and *Heliocidaris erythrogramma*. Urchin impacts include reduced species diversity and declines in commercial fisheries for rock lobster and abalone that depend on kelp beds (Ling 2008; Johnson et al 2013). Experimental manipulations and long-term monitoring inside/outside Tasmanian MPAs reveals that removal of predators, in particular large rock lobsters, has enabled urchin populations to build to abundances causing overgrazing and conversely allowed recovery of kelp beds where large urchinivorous predator abundances have rebuilt following the cessation of fishing (Pederson & Johnson 2006; Ling et al 2009; Ling & Johnson 2012; Babcock et al 2010).

Crown of thorns sea star populations are known to be characterised by periods of ‘boom and bust’, which are patchy in space as well as time. Only two locations surveyed by RLS divers around Australia from 2010-2015 (in the Gulf of Carpentaria and Swains Reefs on the GBR) had densities of CoTS that would be considered outbreaks, and which may be leading to mass local coral mortality. In contrast to the boom and bust dynamics of CoTS, sea urchin population increases can be sustained in longer-term with urchin grazing collapsing kelp beds but this collapse is long-lived as urchins do not “eat themselves out of house and home”, but instead switch diet from large nutritious kelps to filamentous and encrusting coralline algae. The result being persistent urchin barrens ground, which represent an alternative stable state of temperate reef ecosystems (*reviewed by* Ling et al 2015).

Furthermore, the urchin overgrazing phenomenon highlights a distinct management problem because while sustained high abundances are required to overgraze kelp beds, regrowth of kelp on overgrazed reefs requires an almost complete removal of urchins. This phase-shift dynamic, involving ecological hysteresis, means that preventing overgrazing in the first instance is a far more effective management strategy than attempts to restore reefs (Ling et al 2009). This dynamic therefore requires early detection of increasing urchin abundances and management response prior to the overgrazing tipping-point being reached. The Overgrazing index was therefore conservatively set such that early warning of increasing urchin abundances to the point where incipient barrens (5 m^2 in area) can be detected (Fig 4).

For specific indicators of ecologically important “habitat-modifier” species, detailed investigation of local reefs should be initiated as indicators of abundance/impact approach management trigger-levels, including investigation of appropriate management responses (such as tactical culling programs, e.g. Tracey et al 2015; Sanderson et al 2016). Indicator information is ideally required more frequently than through 5-yearly State of the Environment reporting, otherwise potential management responses may not be as timely as required. Thus, regular reporting of invasive species, CoTS and sea urchin overgrazing indicators is arguably more critical than for fishing and warming indicators given the importance of avoiding critical transitions through early warning and response.

The distribution of threatened species in Fig 2 indicates Australian locations with relative high global conservation value. Locations where threatened species constitute a greater proportion of species present on reefs are clearly important for conservation, but appropriate management still relies on considering the most important pressures on the particular threatened species in that area. In the case of the Great Australian Bight and Tasman Sea, some of the key threatened species are threatened primarily by exploitation (e.g. western blue groper and doubleheader wrasse), and thus MPAs and carefully targeted fisheries regulations or closures are likely effective conservation strategies in these areas. The Tasman Sea reefs (Lord Howe Island and Elizabeth and Middleton Reefs) already have considerable area within no-take MPAs, and thus appear well-placed in this regard.

Values of the threatened species indicator were low, however, with the highest value at any single site being 9% of the species recorded (from the six animal classes included in calculation of the index). Generally low values could be seen as promising, in terms of suggesting that only a small proportion of the mobile reef species making up ecological communities around Australia are globally-threatened. But low values may also relate to the fact that historically-limited population trend data has prohibited effective threat assessment for the majority of unexploited or less charismatic marine species. Low indicator values and the natural rarity of threatened species also made it difficult to assess trends in this aspect of marine biodiversity at a national scale, even with the RLS dataset which includes site and species level abundance data and covers >2,500 Australian species. An additional important consideration for tracking changes in this indicator through the future is that improvements in data availability and knowledge may result in more species being listed as threatened, which could result in increases in the indicator value, even if some species are lost to extinction. Thus, this indicator will not provide a substitute for tracking population trends in individual threatened species, as is also required for reporting against Aichi Target 12 (prevention of extinction of known threatened species).

An important limitation of this pressure-specific assessment of the state of Australian reefs is that we have not directly considered habitats over the national scale. Habitat integrity is an important component of ecological condition, and degradation may also lead to changes in values of any of the indicators reported here. In particular, it is likely that the observed decline in B20 in the Capricorn-Bunker Group may be at least in part a result of coral loss associated with recent cyclones, rather than purely due to increasing fishing pressure. This reinforces a process whereby indicator trends trigger detailed local investigation before applying specific local management actions. Complimentary investigation of trends in small fishes could help separate influences of habitat loss versus fishing impacts in the particular case of the Capricorn-Bunker group B20 trend.

Existing monitoring programs in Australia cover different elements of habitat integrity, but as yet, no nationally-coordinated means to collate and report on habitat trends exists in Australia. Quantifying habitat degradation through coral bleaching and storms in the tropics, as well as ecological interaction and climate-driven loss of kelp canopies in temperate locations, are both needed. The recent mass bleaching event observed on the Great Barrier Reef has initiated a thorough assessment of coral loss in 2016, but further indicator development and inclusion in this suite is needed for habitat loss.

Impacts of pollution from metropolitan point sources may be locally-severe, but often dissipate relatively quickly with distance from the source (Shahidul Islam and Tanaka 2004, Stuart-Smith et al. 2015b). Sedimentation and pollution from run-off from intensively managed landscapes may have more widespread impacts, however, but these are still arguably less likely to be as widespread as those from fishing and warming. Pollution covers a broad suite of pressures that are typically referred to in aggregate as a single pressure, however of the 6 pollutant metrics we investigated (i.e. heavy metals, Nitrogen, d15N, TOC, petrochemicals, & microplastics), none were correlated across the 43 study reefs (Appendix I). Thus, rather than a generalised “pollution” impact for temperate reef communities, our results suggest “oils ain’t oils”, that investigations focussed on pollution impacts must be specific to particular pollutants.

The most appropriate broad-scale indicators for the impacts of different types of pollution on reef biodiversity appeared to be those associated with a shortening of the length of species (i.e. changing species composition to those which do not grow as large) within reef communities and a reduction in biogenic habitat richness, which best predicted increasing heavy metal and nitrogen levels respectively (Table 3). Indeed, 6 of the top 10 indicators were related to species richness and 3 of the top 10 related to species lengths (Table 3). Thus we predict that a ‘general’ increase in pollution, for which heavy metals and eutrophication appear to have greatest effects, will lead to ‘short and simple’ reef communities whereby constituent species are fewer and only achieve smaller sizes. In comparison, pristine unpolluted reefs are expected to be ‘long and complicated’, containing more species including those growing to larger/ longer sizes.

4.2 Integration of data from monitoring programs

An important component of this study was to assess how well the different monitoring programs could be used in combination for ongoing reporting of biodiversity indicators. Each clearly has its own strengths and weaknesses, and AIMS and LTMPA were designed specifically for MPA monitoring/performance assessment, rather than tracking indicators for State of the Environment reporting, for example. RLS and LTMPA can be fully integrated and provide equal opportunities for reporting against indicators presented here. The AIMS LTM data, however, appear to provide only a vaguely comparable picture of fish biomass (B20) and mean thermal affinity of fish communities (CTI). AIMS LTM data were still used for tracking temporal trends in these indicators here, but direct comparison between locations monitored by AIMS and the other two programs may not be reliable for these indicators (or a few of the useful alternatives tested – see below).

The direct comparison of AIMS and RLS surveys done at the same time and place could potentially be confounded by instantaneous variation in fish community - i.e. local, reef-scale patchiness of fishes in time and space, naturally, or as a result of two dive teams operating along the same transects. But the comparison of subset RLS data from along the entire GBR provide a method of comparison that cannot be confounded in the same way. Thus more emphasis should be placed on results from the second method (Figs 2 & 3).

Equivalent trials of other fishing indicators identified in Table 1 as potentially useful did not indicate any better matches between AIMS and RLS data, and it is likely that some important information relating to community size structure, large fish stocks, and even the thermal composition of the fish community, is lost when reducing the species monitored to a subset of families. For example, the B20 plot in Figs 2 & 3 show almost an order of magnitude greater range in biomass of fishes 20 cm+ in the RLS data than in the AIMS LTM data. Whether this additional biomass comes from targeted species has not been assessed and is likely to vary greatly. Given the reason for its establishment, it is likely that the AIMS LTM program selected some of the most important targeted families on the GBR for inclusion, so these biomass differences may not be as important in measuring change in the direct effects of fishing, even if substantial ecological information is lost. Thus, temporal trends in B20 from the AIMS data should provide insight into fishing impacts along the GBR (and other phenomena that reduce large fish biomass, as discussed above). But direct comparison between datasets appears to require careful interpretation.

CTI varied more than expected between the two methods, and unusually more in the comparison using the subset of RLS data (Fig 3) than in the direct comparison using fewer surveys (Fig 2). The comparison suggested that the subset of species surveyed in the AIMS LTM are biased towards species with cooler thermal affinities than is typically present in RLS surveys. This may reflect an original choice of monitored species based on species lists from the southern sections of the GBR, before expanding monitoring further northwards, or potentially unexplored correlations between phylogeny and thermal affinity, such that ‘warmer’ families have been among those omitted. Regardless of the causes, as for B20, comparison of CTI values between datasets appears unreliable. Whether temporal trends in CTI based only on the AIMS LTM data provided an equivalent picture to that based on all fish species surveyed will be complicated and is not explored here. Such an assessment does not seem important at this stage, as there has been no long-term warming trend along the GBR during the period assessed here, and therefore no substantial changes in CTI expected. If future warming trends result in the loss of ‘warm’ species, and these are not covered in the AIMS survey methods, then the CTI will not reflect such changes. However, if ‘cooler’ species covered well in the AIMS surveys are extirpated, there will be a disproportionately large decrease in CTI.

4.3 Recommendations

Based on present knowledge, the indicators suggested here to provide representation of biodiversity trends in relation to key threats on Australian coral and rocky reefs are shown in Table 3. The details of their calculation are provided in the methods section, and interpretation broadly discussed above. Importantly, further research is needed to provide more rigorous tests of temporal responsiveness of these indicators (particularly those for which temporal trends could not be examined here), and provide a basis for determining ecologically-relevant reference levels and targets. Research to determine ecological thresholds and tipping points in relation to indicator values was beyond the scope of this project, and represents an important next step, along with modelling of alternative management scenarios (Collen and Nicholson 2014).

Given the trends in B20 and need to distinguish trends associated with habitat loss, it is suggested the gamma shape parameter from the size spectrum is also monitored as an additional indicator for fishing impacts, given it can be used to confirm or distinguish equivalent responses in smaller fishes. Thus, B20 and the gamma parameter can provide a more complete picture of changes in the fish community, at least until an indicator of habitat is added to this suite.

No single indicator showed a general response to pollution, rather, indicators correlating most strongly with pollution were specific to particular pollutant types (Table 3). Essentially, a suite of indicators should be tracked through time, and only when corresponding change in pressures are known, could trends in a suite of indicators be used to attribute specific impacts. Furthermore, attribution would only be possible if baselines for particular reefs are known prior to impact, and external reefs, not incurring the same pressure, remain relatively unchanged and act as effective control sites (achieving a Before-After-Control-Impact design). Further still, due to potential interactive effects between multiple pressures, plus the potential that pressures may manifest in similar ways, detection of change in indicators should be used to inform possible impacts rather than providing definitive cause and effect. For example, shortening of the reef community caused by size reductions of component fishes, invertebrates and biogenic habitats could be caused by harvesting some or all of these reef components (B20+ and the gamma shape parameter for the size spectrum can detect changes to the fish component due to fishing, Table 3). Alternatively, as revealed by the analysis of pollutants, such a shortening of the community could also occur due to elevated pollution whereby larger longer-lived species locally decline in abundance. Therefore, while a specific indicator may suggest a particular pressure has reached a critical level, further context of all potential pressures acting / or not acting on particular reefs would be required to clearly identify the likely driver of such change. In the same way, analysing trends in a suite of indicators is likely to be much more informative than examining one indicator in isolation, which may manifest in a similar or subtly different manner to other properties of the community.

Table 3. Indicator suite for Australia's shallow reefs.

Pressure	Indicator
Fishing	Biomass of fishes ≥20 cm Size spectrum – Gamma shape parameter
Ocean warming	Community Temperature Index
Invasive species	Density of invasive individuals 50 m^{-2}
Pollution **- Heavy metals**	Community Shortness Index Invertebrate Lmax
- Nitrogen loading (eutrophication)	Richness of benthic habitats Invertebrate diversity (Shannon)
Crown of thorns sea stars	Density of individuals per Ha
Sea urchins	Overgrazing Index (urchin biomass per m^2 required for incipient barrens formation)

The monitoring programs discussed in this report contain the detail necessary to calculate numerous other indicators retrospectively, should future research determine that alternative indicators are more informative. Such further research is needed, and it is anticipated that the indicator suite will be refined through time.

In terms of long-term integration of data from multiple monitoring programs, clearly a valuable synergy exists between RLS and the LTMPA program, plus a number of programs run by state government management agencies (not assessed here). Surveys are undertaken in numerous locations in Victoria, Western Australia, New South Wales and South Australia undertake using compatible methods, although most have tended to contribute to the LTMPA through collaboration on Australian Government ARC Linkage projects over the last 20 years.

The AIMS LTM data provide invaluable coverage and long-term consistency for the GBR, with extremely high power to detect changing patterns associated with fishing pressure, CoTS outbreaks, cyclone damage and other factors through the long term. While these data show low comparability to RLS and LTMPA datasets for tracking the biodiversity indicators tested here (B20 and CTI), other components of the AIMS monitoring program (e.g. crown-of-thorns sea stars and coral cover) may better integrate with RLS and LTMPA data, in addition to providing a better picture of trends in the GBR over a longer time scale than any other ecological monitoring program for Australian reefs. Thus, even though fishing and warming indicators derived from AIMS LTM data have little comparability with other monitoring data for between site comparisons, data from all programs are complimentary and important when attempting to understand trends at the continental-scale.

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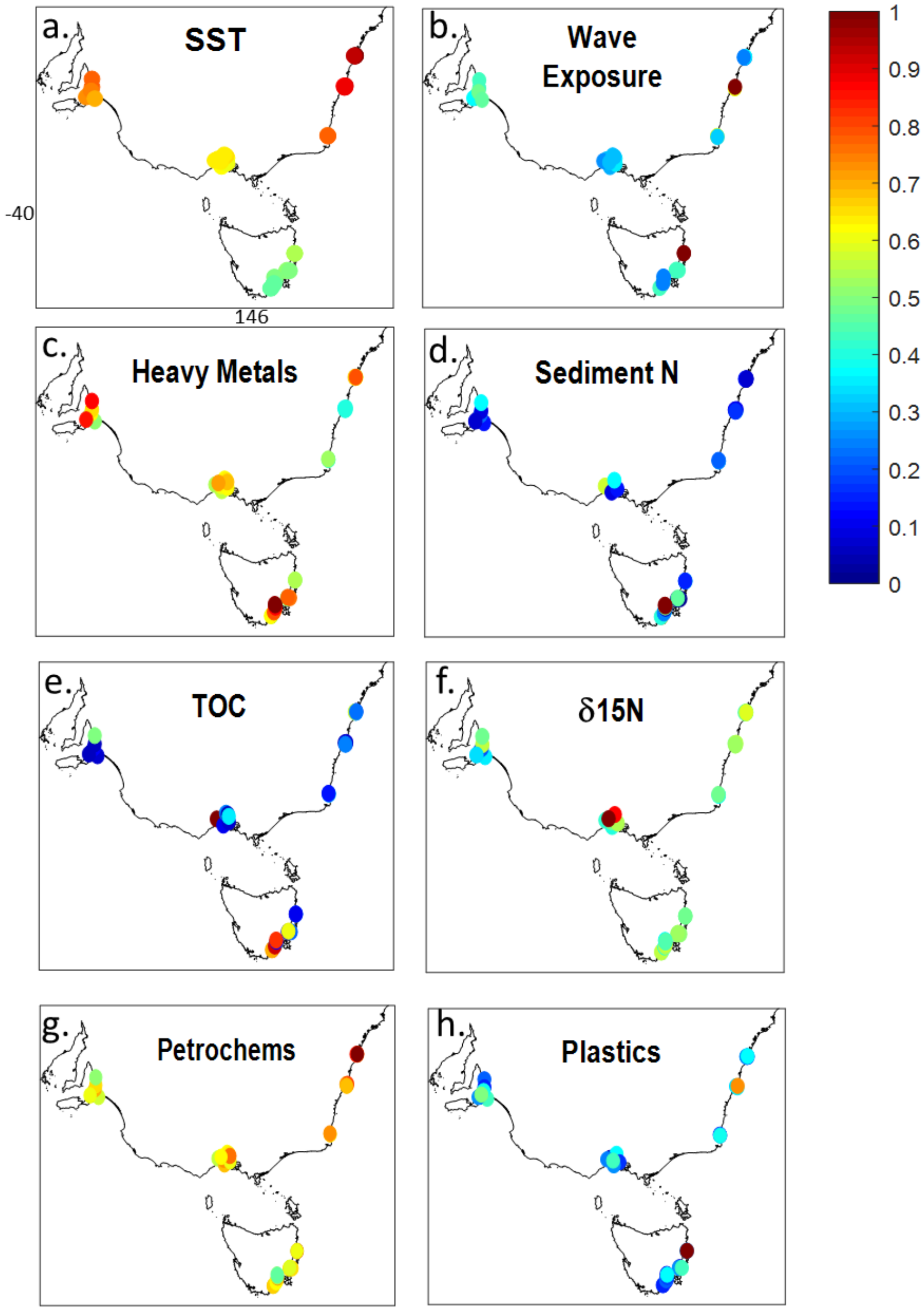
APPENDIX I. Pearson correlation coefficients between levels of pollutant types and sea surface temperature (SST) and wave exposure as measured for the 43 south-eastern Australian reef sites.

	metals	d15N	TOC	Sed_N	Petrochem surrogates	SST	Wave exposure	Micro-plastic conc.
metals	1.00							
d15N	-0.09	1.00						
TOC	0.27	0.00	1.00					
Sed_N	0.64	-0.07	0.58	1.00				
Petrochem surrogates	-0.43	0.15	-0.32	-0.59	1.00			
SST	-0.28	-0.07	-0.32	-0.46	0.53	1.00		
Exposure	-0.29	-0.07	-0.28	-0.08	0.11	-0.02	1.00	
Micro-plastic conc.	-0.18	-0.22	-0.35	-0.13	-0.09	0.11	0.58	1.00

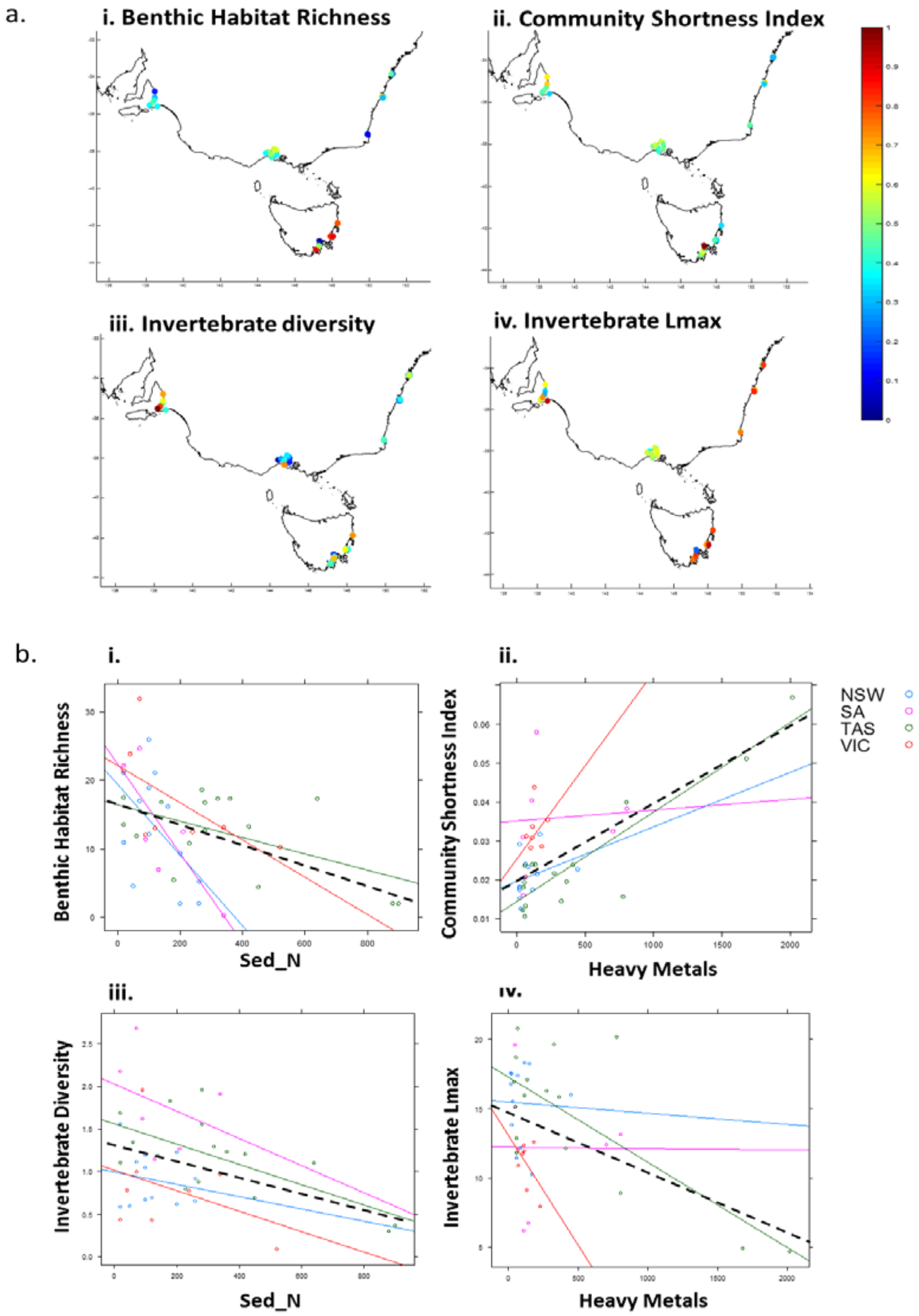
APPENDIX II. Heat maps of non-pollutant, and pollutant variables included in multiple regression modelling of candidate biological indicators of pollution. For optimal display, all variables have been re-scaled from 0 – 1.



APPENDIX III. Saturated and parsimonious multiple regression model results of candidate pollution indicators for rocky reef communities (listed alphabetically), as calculated from RLS and LTMPA surveys of 43 sites in temperate south-eastern Australia where pollutant levels were co-measured. Red values indicate model terms with negative effects, while significant effects are highlighted in **bold**.

	Saturated model											Parsimonious model terms												
	Environment		Pollutants									Environment		Pollutants										
Indicators (listed alphabetically)	SST	Wave Expos-ure	Metals	d15N	TOC	Sed_N	Petro-chem	Micro-plastics	model R^2	*P*- value	Max. pollutant R^2	SST	Wave Exposure	Metals	d15 N	TOC	Sed_ N	Petro-chem	Micro-plastics	model R^2	*P*- value	Max. pollutant R^2	Pollutant (direction of effect)	Consistency in trends between regions
Canopy Seaweed Cover	0.05	0.03	0.04	0.01	0.05	**0.15**	0.02	0.01	0.37	**0.037**	0.15	0.03	0.06				**0.25**			0.34	**0.001**	0.25	Sed_N -	No (3/4 regions)
Community Shortness Index	0.03	0.09	0.17	0.01	0.01	0.04	0.03	0.02	0.39	**0.022**	0.17		0.11	**0.23**						0.33	**0.000**	0.23	Metals +	Yes (4/4 regions)
Cryptic Fish Abundance	0.16	0.08	0.06	0.01	0.05	0.02	0.09	0.02	0.49	**0.003**	0.09	**0.16**	**0.08**	**0.08**		**0.06**		**0.08**	0.02	0.48	**0.001**	0.08	Petrochem +	No (2/4 regions)
Cryptic Fish Richness	**0.12**	0.09	0.03	0.01	0.03	0.01	0.02	0.01	0.33	0.078	0.03	0.09	**0.13**					0.05		0.27	**0.007**	0.05	Sed_N -	No (2/4 regions)
Fish & Invert. Lmax	0.01	**0.03**	**0.13**	0.00	0.01	0.03	0.07	0.01	0.37	**0.035**	0.13	0.04	**0.16**	**0.15**						0.36	**0.001**	0.15	Metals -	Yes (4/4 regions)
Fish & Invert. Richness	0.01	0.11	0.07	0.01	0.01	0.01	0.04	0.04	0.31	0.10	0.07		**0.17**	0.09						0.26	**0.003**	0.09	Metals -	Yes (4/4 regions)
Fish Abundance	0.03	0.04	0.01	0.03	0.01	0.05	0.11	0.03	0.32	0.090	0.11	0.04			0.03		**0.06**	**0.13**	0.05	0.30	**0.019**	0.13	Petrochem +	No (3/4 regions
Fish Biomass	0.01	**0.14**	0.03	**0.14**	0.03	0.07	0.03	0.02	0.47	**0.004**	0.14		**0.17**		**0.13**		**0.14**		0.03	0.47	**0.000**	0.14	Sed_N -	No (3/4 regions)
Fish Diversity (Shannon)	0.00	0.05	0.01	0.01	0.04	0.01	0.00	0.01	0.15	0.658	0.01		**0.08**			0.04	0.03			0.14	0.127	0.04	TOC +	No (3/4 regions)
Fish Functional Richness (B)	0.01	0.02	0.03	0.00	0.02	0.03	0.04	0.02	0.17	0.590	0.04		**0.17**	0.09				0.07		0.07	0.093	0.07	Petrochem +	No (3/4 regions)
Fish Length	**0.06**	**0.23**	0.11	0.01	0.02	**0.12**	0.05	0.03	0.61	**0.000**	0.12	0.04	**0.30**			**0.03**	**0.21**			0.58	**0.000**	0.21	Sed_N -	No (3/4 regions
Fish Lmax	0.08	0.14	**0.10**	0.00	0.00	0.01	0.01	0.02	0.37	**0.039**	0.10	**0.08**	**0.16**	**0.10**						0.34	**0.001**	0.10	Metals -	Yes (4/4 regions)
Fish no. Effective Species	0.01	0.06	0.02	0.01	0.02	0.01	0.00	0.01	0.14	0.72	0.02		**0.07**							0.07	0.10	nil	nil	N/A
Fish Proportion Pelagic	0.01	0.06	0.01	0.01	0.01	0.01	0.03	0.01	0.14	0.69	0.03		0.09							0.09	0.052	nil	nil	N/A
Fish Richness	0.01	0.08	0.04	0.01	0.01	0.01	0.05	0.04	0.25	0.246	0.05		**0.14**					0.07		0.21	**0.009**	0.07	Petrochem +	No (3/4 regions
Fish Trophic Level	0.24	0.09	0.03	0.04	0.00	0.01	**0.05**	0.01	0.47	**0.003**	0.05	**0.24**	**0.09**	0.03				**0.05**		0.41	**0.000**	0.05	Petrochem +	Yes (4/4 regions)
Fish Trophic Level (B)	0.22	0.03	0.01	0.00	0.05	0.07	0.02	0.01	0.42	**0.011**	0.07	**0.26**	0.03			0.09				0.38	**0.000**	0.09	TOC +	No (2/4 regions)
Fish Vulnerability Index	0.07	0.14	0.06	0.00	0.01	0.01	0.01	0.02	0.31	0.10	0.06	0.06	**0.17**	0.08						0.30	**0.003**	0.08	Metals -	No (3/4 regions)
Fish Vulnerability Index (B)	0.02	0.00	0.02	0.00	0.01	0.10	0.02	0.00	0.17	0.587	0.10						**0.11**			0.11	**0.031**	0.11	Sed_N -	No (3/4 regions)
Fish Functional Eveness	**0.17**	0.00	0.00	0.01	0.07	0.01	0.03	0.01	0.30	0.127	0.07	**0.18**					0.07	0.03		0.28	**0.006**	0.07	Sed_N -	No (2/4 regions)
Fish Functional Richness	0.01	0.03	0.13	0.00	0.01	0.03	0.07	0.01	0.29	**0.155**	0.13			**0.22**						0.22	**0.002**	0.22	Metals -	No (3/4 regions)
Fucoid Seaweed Cover	0.17	0.11	0.03	0.07	0.03	0.07	0.01	0.04	0.52	**0.001**	0.07	**0.15**	**0.17**		**0.07**		**0.13**			0.52	**0.000**	0.13	Sed_N -	No (2/4 regions)
Habitat Functional Richness	**0.05**	0.01	0.09	0.01	0.02	0.14	0.05	0.02	0.37	**0.035**	0.14	0.03					**0.26**			0.29	**0.001**	0.26	Sed_N -	Yes (4/4 regions)
Habitat Height max.	0.03	0.06	0.07	0.01	0.01	0.10	0.08	0.01	0.38	**0.029**	0.10		**0.08**	**0.13**				**0.13**	0.02	0.35	**0.002**	**0.13**	Petrochem +	No (3/4 regions)
Habitat Richness	**0.18**	0.01	**0.15**	0.05	0.02	0.06	0.01	0.01	0.50	**0.002**	0.15	**0.16**		**0.24**	**0.05**					0.44	**0.000**	0.24	Metals -	No (2/4 regions)
Habitat Weediness	0.02	0.01	0.09	0.00	0.01	0.09	0.10	0.00	0.32	0.090	0.10			**0.14**				**0.15**		0.29	**0.001**	0.15	Petrochem -	No (3/4 regions)
Invert. & Cryptic Fish Diversity (Shannon)	0.02	0.06	0.02	0.03	0.09	0.09	0.01	0.01	0.35	0.055	0.09		**0.08**	0.03			**0.14**			0.24	**0.014**	0.14	Sed_N -	Yes (4/4 regions)
Invert. & Cryptic Fish no. Effective Species	0.02	**0.06**	0.02	0.08	0.07	**0.09**	0.01	0.02	0.36	**0.041**	0.09	0.02	0.07	0.02	0.07		**0.13**			0.32	**0.014**	0.13	Sed_N -	Yes (4/4 regions)
Invert. Abundance	**0.12**	0.06	0.04	**0.08**	0.01	**0.08**	0.01	0.02	0.43	0.009	0.08	**0.13**		**0.05**	**0.09**		**0.08**		0.04	0.39	**0.003**	0.09	d15N +	No (3/4 regions)
Invert. Diversity (Shannon)	0.03	0.09	0.02	0.04	0.08	**0.11**	0.01	0.02	0.40	**0.020**	0.11	0.03	**0.10**		0.04		**0.17**			0.33	**0.004**	0.17	Sed_N -	Yes (4/4 regions)
Invert. Lmax	0.01	0.01	**0.11**	0.01	0.01	0.08	0.01	0.01	0.25	0.240	0.11			**0.20**						0.20	**0.003**	0.20	Metals -	Yes (4/4 regions)
Invert. no. Effective Species	0.01	0.07	0.01	0.08	0.06	0.09	0.01	0.02	0.31	0.102	0.09		0.09	0.02	**0.08**		**0.13**	0.02	0.02	0.35	**0.013**	0.13	Sed_N -	Yes (4/4 regions)
Invert. Richness	0.08	0.14	**0.11**	0.00	0.04	0.02	0.01	0.03	0.42	**0.011**	0.11	**0.06**	**0.20**	**0.13**						0.39	**0.000**	0.13	Metals -	Yes (4/4 regions)
Invert. & cryptic fish abundance	**0.10**	0.06	0.04	**0.09**	0.01	**0.07**	0.02	0.03	0.43	0.010	0.09	**0.11**		**0.06**	**0.09**		**0.09**	0.03	**0.05**	0.43	**0.002**	0.09	d15N +	No (3/4 regions)
Invert. & cryptic fish richness	0.08	0.11	**0.08**	0.00	0.02	0.02	0.02	0.04	0.37	**0.035**	0.08	**0.07**	**0.17**	0.08						0.32	**0.002**	0.08	Metals -	Yes (4/4 regions)
Laminarian Kelp Cover	0.01	0.01	0.07	0.00	0.07	0.07	0.02	0.01	0.25	**0.231**	0.07			**0.13**		**0.12**				0.24	**0.005**	0.13	Sed_N -	Yes (4/4 regions)
Large Fish Index (20 cm)	0.00	**0.17**	0.09	**0.06**	**0.07**	0.02	0.01	**0.05**	0.48	**0.003**	0.09	**0.17**		**0.10**	**0.07**	**0.07**			**0.04**	0.45	**0.000**	0.10	Microplastics -	No (1/3 regions)
Large Fish Index (20 cm) (B)	0.01	0.08	**0.10**	0.03	0.03	0.02	0.00	0.02	0.30	0.127	0.10	0.15		**0.10**	0.00		**0.02**		**0.02**	0.29	**0.024**	0.10	Metals -	No (3/4 regions)
Large Fish Index (30 cm)	0.02	0.13	0.03	0.02	0.01	0.01	0.00	0.05	0.28	0.17	0.05		0.15						0.05	0.20	**0.012**	0.05	Microplastics -	No (3/4 regions)
Large Fish Index (30 cm) (B)	0.01	**0.11**	0.03	0.01	0.00	0.00	0.01	0.05	0.21	0.37	0.05		0.13						0.05	0.18	**0.019**	0.05	Microplastics -	No (3/4 regions)
Turf Algae Cover	0.08	0.07	0.09	0.03	0.02	**0.18**	0.02	0.05	0.53	**0.001**	0.18	**0.08**	**0.08**		0.03		**0.26**		**0.05**	0.49	**0.000**	0.26	Sed_N +	No (3/4 regions)
Whole Community Richness	0.02	0.10	0.12	0.00	0.01	0.03	0.05	0.04	0.36	**0.040**	0.12		**0.14**	**0.16**						0.31	**0.001**	0.16	Metals -	Yes (4/4 regions)

APPENDIX IV. (a) Spatial maps of best performing pollution indicators (i. Benthic Habitat Richness; ii. Community Shortness Index; iii. Invertebrate Diversity; iv. Invertebrate Lmax). (b) Trends in pollution indicators by location (coloured lines, as per legend) and overall trends for south-eastern Australia (dashed line), indicators (i-iv) as per (a).



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