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A data-driven approach to multiple-stressor impact assessment for a marine protected area

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Abstract

The coastal environment is not managed in a way that considers the impact of cumulative threats, despite being subject to threats from all realms (marine, land, and atmosphere). Relationships between threats and species are often nonlinear; thus, current (linear) approaches to estimating the impact of threats may be misleading. We developed a data-driven approach to assessing cumulative impacts on ecosystems and applied it to explore nonlinear relationships between threats and a temperate reef fish community. We used data on water quality, commercial fishing, climate change, and indicators of recreational fishing and urbanization to build a cumulative threat map of the northern region in New South Wales, Australia. We used statistical models of fish abundance to quantify associations among threats and biophysical covariates and predicted where cumulative impacts are likely to have the greatest impact on fish. We also assessed the performance of no-take zones (NTZs), to protect fish from cumulative threats across 2 marine protected area networks (marine parks). Fishing had a greater impact on fish than water quality threats (i.e., percent increase above the mean for invertivores was 337% when fishing was removed and was 11% above the mean when water quality was removed inside NTZs), and fishing outside NTZs affected fish abundances inside NTZs. Quantifying the spatial influence of multiple threats enables managers to understand the multitude of management actions required to address threats.

KEYWORDS
coastal management, fishing pressure, impact maps, multiple threats, New South Wales, no-take zones, temperate reef fish

Una estrategia basada en datos para la evaluación de impacto de múltiples estresores en un área marina protegida

Resumen: Los ambientes costeros no se manejan de manera que se considere el impacto de las amenazas acumulativas, a pesar de que se enfrentan a amenazas de todos los entornos (marinas, terrestres y atmosféricas). Las relaciones entre las amenazas y las especies casi siempre son no lineales; por lo tanto, las estrategias actuales (lineales) para estimar el impacto de las amenazas pueden ser engañosas. Desarrollamos una estrategia basada en datos para evaluar el impacto acumulativo sobre los ecosistemas y la aplicamos para explorar las relaciones no lineales entre las amenazas y la comunidad de peces de arrecifes templados. Usamos datos de la calidad del agua, pesca comercial, cambio climático e indicadores de pesca recreativa y urbanización para construir un mapa acumulativo de amenazas de la región norte de Nueva Gales del Sur, Australia. Usamos modelos estadísticos de la abundancia de peces para cuantificar las asociaciones entre las amenazas y las covarianzas biofísicas y pronosticamos en dónde es probable que los impactos acumulativos sean mayores sobre los peces. También evaluamos el desempeño de las zonas de veda para así proteger a los peces de las amenazas acumulativas en dos redes de áreas marinas protegidas (parques marinos). La pesca tuvo un mayor impacto que la calidad del agua sobre los peces (es decir, el incremento del porcentaje por encima de la media de depredadores de invertebrados fue de 337% cuando se eliminó la pesca y fue de 11% por encima de la media cuando se eliminó la calidad del agua dentro de las zonas de veda) y la pesca fuera de las zonas de veda afectó la abundancia de los peces dentro de ellas. La cuantificación de la influencia espacial de las múltiples amenazas permite que los gestores entiendan la multitud de acciones de manejo que se requieren para abordar las amenazas.

PALABRAS CLAVE

amenazas múltiples, manejo costero, mapas de impacto, Nueva Gales del Sur, peces de arrecifes templados, presión de pesca, zonas de veda

INTRODUCTION

The coastal-marine environment supports rich biodiversity and provides numerous ecosystem services that support coastal communities; however, it is under increasing pressure from a range of anthropogenic land-based, marine-based, and climate change threats (Halpern et al., 2019). Land-based activities introduce nutrients and pollutants into coastal waters that influence estuarine and marine coastal ecosystems (Brown et al., 2017; Lefébure et al., 2013). Marine-based activities, including fishing, have direct impacts on fish biomass and indirect effects on ecosystems through changes in food webs (Fu et al., 2020; Jennings & Polunin, 1997) and have seen considerable growth in the last 2 decades (Halpern et al., 2019). Climate change threats, although regionally unmanageable, have direct and indirect impacts on species and ecosystems (Trisos et al., 2020). Effective management is required to minimize and regulate the impact of manageable activities; however, they are commonly managed independently by different sectors and actors and often involve tools that target single activities, such as marine protected areas that limit fishing, but do not address the impacts of poor water quality (Elliott, 2014). The lack of integrated land–sea plans is a key reason coastal management has failed to address the cumulative impacts of multiple threats (Griffiths et al., 2020; Tulloch et al., 2021).

Lack of integrated planning is partially due to the difficulty of predicting how multiple threats translate into impacts on species and ecosystems, which requires knowledge of their vulnerabilities to the threats (cumulative impact assessment [CIA]). Typically, expert knowledge or empirical knowledge is used to predict impacts from threats (Grech et al., 2012; Halpern et al., 2007; Turschwell et al., 2020). Expert elicitation is commonly used to weight the vulnerabilities of species and ecosystems to threats and thereby interpret the impact based on cumulative threat maps (Loiseau et al., 2021; Stockbridge et al., 2021). However, this process introduces "knowledge-based uncertainty," which can affect the reliability of the results and its application to management (Jones et al., 2018). Impact maps typically also weight threats linearly or log linearly by their vulnerabilities, so these impact maps assume a positive and monotonic relationship between increasing threats and increasing impacts.

An alternative approach to assessing impacts is to use statistical models to fit empirically measured ecological responses to threats (Stockbridge et al., 2021; Turschwell et al., 2021). The vulnerability of an ecological component to a threat then becomes a function (potentially nonlinear) rather than a single weight. Empirical assessments are less subject to expert biases and can capture nonmonotonic or decreasing relationships between threats and their impacts on the ecosystem. However, empirical data may cost more and be harder to acquire than expert-elicited vulnerabilities and so have not been as widely explored as an option for assessing cumulative impacts (Johannes, 1998; Teck et al., 2010). Estimating vulnerabilities from data is also challenging because cumulative threats are confounded by their nature (hence being cumulative), so it may be difficult to statistically separate the effect of one threat from another. Therefore, there is a need to further develop cumulative impact mapping approaches that draw directly on empirical measurements of ecological responses.

We explored a new approach to cumulative threat mapping. We predicted how different threats affect reef fish abundance and then made predictions about the impact of cumulative threats on fish abundance inside and outside no-take zones (NTZs). We did this in the northern region of New South Wales (NSW), an area rich with spatial and temporal data on threats and biodiversity that provides an ideal setting in which to explore the empirical approach to CIA (Appendix S1). Specifically, we aimed to build a cumulative threat map based on indicators of water quality, fishing, and climate change; build a predictive model of fish abundance from the threat data and other environmental covariates; apply predictive modeling to predict cumulative human impacts on fish abundance; and assess the performance of NTZs to protect near-shore harvested fish species from cumulative threats. Our exploratory analyses of CIA address the issues of nonlinearity in the vulnerability of ecosystems to threats and confounding of cumulative threats. It also identifies further research needs that will improve the understanding of cumulative impacts in northern NSW and can inform future management priorities.

METHODS

Study region

The northern region of NSW encompasses ∼700 km of coastline and supports a diversity of subtropical and temperate marine habitats (mangroves, seagrasses, saltmarshes, coral reefs, intertidal and subtidal rocky reefs, and estuaries; Figure 1, Appendix S1). The study region is characterized by moderate impacts on the marine environment over a range of threats (moderate urban growth and significant recreational and commercial fisheries) and has been a focal area for long-term monitoring on fish abundance (Knott et al., 2021; Malcolm et al., 2018). The region also includes multiple-use marine state parks, of which Solitary Islands Marine Park (SIMP) and Port Stephens-Great Lakes Marine Park (PSGLMP) were included in our analyses. The SIMP (710 km^2) was established in 1998 and multiple-use zones were applied in 2002. The PSGLMP (980 km^2) was established and zoned for multiple use in 2005 and 2007, respectively. The marine parks extend 3 nautical miles (nm) from shore, which is the seaward limit of the NSW Marine Estate (NSW Government, 2018). Beyond this limit lies Commonwealth waters out to 200 nm. A range of fisheries restrictions are applied in the marine parks, including sanctuary zones, where fishing is prohibited (NTZs), habitat protection zones, where some commercial fishing, aquaculture, and collecting activities are limited, and general use zones, where some netting and longlining activities are limited. Each state park includes multiple areas of variable-sized NTZs (19 in total in the coastal-marine area).

Threats

We identified 9 anthropogenic threats (Table 1) of relevance to the region's nearshore marine environment, particularly to temperate reef fish (Fletcher & Fisk, 2017), and for which comprehensive spatial data were available. The threats represent the common dimensions of human-derived threats for coastal marine species: threats from land pollution and coastal population and development (Allan et al., 2023); threats from marine recreation and commercial fishing; and threats from climate change. As water quality indicators we used nutrient (chlorophyll-*a*) and turbidity data, which are effective measures of water quality and coastal development (Scanes et al., 2007). Recreational fishing was measured based on regional boat ramp usage. We used sea surface temperature (SST) anomalies to represent ocean warming, which is a major effect of climate change. Other climate change variables were also considered (e.g., ocean acidification data), but there was too little variation in the data across the study area for differences to be detected. All threat layers were reprojected to the same base raster with a consistent resolution (500 × 500 m) and projection (UTM zone 56S WGS84). The commercial fishing data were available at 0.01°, and to accurately represent that coarse resolution on maps, we did not smooth the layers relating to commercial fishing. Analyses were performed with R (R Development Core Team, 2018) and its packages raster (Hijmans et al., 2021) and sf (Pebesma, 2018).

Fish monitoring data

We used stereo-baited remote underwater video (stereo-BRUV) at shallow reefs (<50 m) over randomly selected, spatially independent sites from 2010 to 2018 to collect fish abundance data, following the methods of Malcolm et al. (2018). Abundance for each sample was quantified with the maxN statistic—which indicates relative abundance and is the maximum number (maxN) of any individual fish (of a particular species) in the frame at any 1 time during each camera set (Harasti et al., 2018). Shallow, rocky reefs were previously identified (Jordan et al., 2010; Malcolm et al., 2018). Data were selected for fishery-targeted invertivores and carnivores (Appendix S2), which were the most abundant groups counted by BRUV. Data were interrogated for trends across categories (effect of NTZs and others). Invertivores included the invertebrate carnivore feeding guild (10 species, 3616 counts), and carnivores included the piscivore and generalist carnivore feeding guilds (30 species, 2821 counts). There were 952 observations overall.

Hotspots of cumulative threats

We combined the layers of threats to identify hotspots of cumulative threat. Hotspots were defined as grid cells that were in the top 10% of all grid cells for more than 1 threat. The 10% threshold was arbitrary but selected because it excluded most grid cells. Other thresholds (e.g., 1% or 5%) showed similar spatial patterns. The cumulative threat score was the count of the number of threats (9 maximum) for which a cell (pixel) was in the top 10% for each grid. Thus, if a threat did not meet the 10% threshold, it was not included. The hotspot index (H_i), therefore, could take only integer values from 0 to 9 threats and was defined as

$$H_i = \sum_{j=1}^{9} \left(T_{i,j} \geq T_j^{90\%} \right), \tag{1}$$

where $T_{i,j}$ is the value for threat j in grid cell i and $T_j^{90\%}$ is the upper 10% quantile for the threat j. We see this approach as a precursor to a full CIA that does not require vulnerability weights and thus is agnostic about the ecosystem's vulnerability to the threat. We additionally calculated Spearman's pairwise correlations among threats.

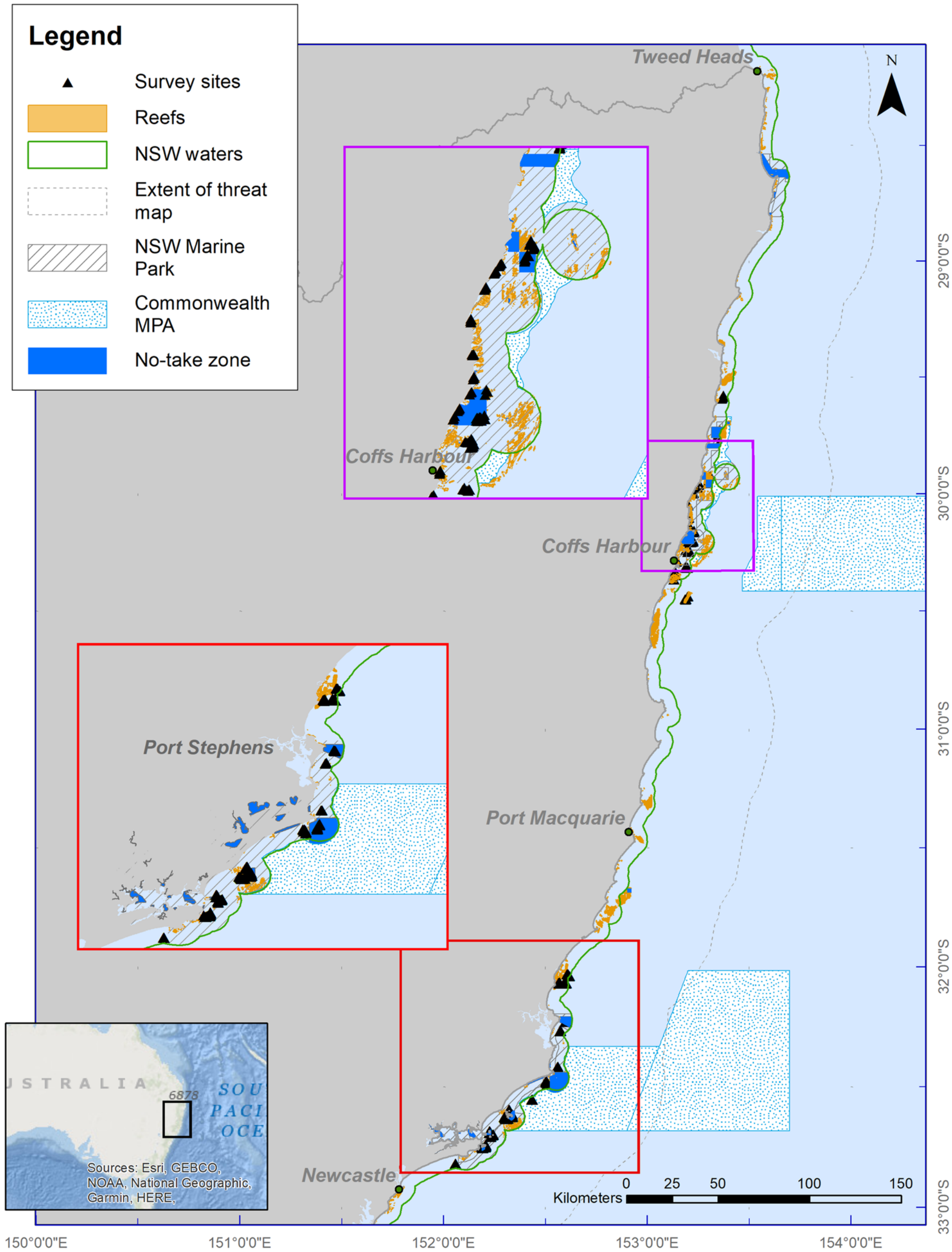

FIGURE 1 Northern region of New South Wales, Australia, showing the study area. Map produced using ArcGIS, ArcMap 10.3 software. Reef polygons and baited remote underwater video survey locations provided by Johnson et al. (2018). Protected area zones sourced from Department of Agriculture Water and the Environment (2020). Sources for the ocean basemap: ESRI, General Bathymetric Chart of the Oceans, National Oceanic and Atmospheric Administration, National Geographic, DeLorme, HERE, Geonames.org, and other contributors.

Model of fish abundance

We used biophysical covariates and threats as predictors to model the abundance of temperate rocky reef fish for both functional groups. Fish abundance was compared in 2 areas subject to different management: NTZs and sites outside NTZs. The biophysical covariates tested were based on 2 a priori hypotheses for biophysical drivers of fish abundance. First, fish abundance is greater when reefs are ecologically connected to adjacent coastal wetlands (Olds et al., 2012). Thus, we used the shortest distance from reef survey sites to each mangrove, seagrass, or saltmarsh location (NSW Department of Primary Industries, 2013). Second, we included depth as a covariate because the BRUV data were collected across shallow (<20 m) and intermediate reefs (>25 to <50 m) and depth has a strong effect on fish abundance (Malcolm et al., 2018).

For each fish functional group, we fitted a model of threats and biophysical variables with covariates as additive spline effects. Specifically thin-plate regression splines were fit with generalized additive models to allow for nonlinear effects

TABLE 1 Summary of the anthropogenic threats in the northern region of New South Wales (NWS) used to create cumulative impact maps.

Measure	Description	No. of stressors	Original resolution	Date	Source
Nutrients (water quality)	Mean chlorophyll *a* concentration from daily surface multispectral measurements of sunlight from the MODIS sensor (OC3 model) on the Aqua satellite	1	0.02°	2006–2020	Integrated Marine Observing System, 2014a
Turbidity (water quality)	Mean NTU from the sites located closest to the river or estuary mouth measured during routine water quality monitoring of NSW estuaries	1	Modeled 0.5 km^2	2007–2020	Department of Planning Industry and the Environment, 2020
Demersal commercial fishing	Trap, bottom trawl (prawns), and bottom line catch per unit effort from commercial fishing logbook data	3	0.01°	2009–2020	Department of Primary Industries, 2017
Midwater commercial fishing	Midwater line catch per unit effort from commercial fishing logbook data	1	0.01°	2009–2020	Department of Primary Industries, 2017
Recreational fishing	Estimated by linear distance to boat ramps, weighted by an indicator of boat-ramp usage (single vs. multilane ramps)	1	Modeled 0.5 km^2	2020	Transport Roads & Maritime Services, 2021
Sea surface temperature (SST) anomaly (climate change)	Frequency of recent SST (2006–2020) above the historical mean (1992–2005) measured by daily single-sensor NOAA polar-orbiting multisatellite data	1	0.01 degrees	1992–2020	Integrated Marine Observing System, 2014b
Population pressure (urbanization)	Population and housing data from the 2016 Australian Census and followed method of Ostwald et al. (2021)	1	Modeled 0.5 km^2	2020	Australian Bureau of Statistics, 2021

(GAM, R software [R Development Core Team, 2018], mgcv package [Wood, 2023]). All threats used in the threat map were initially included as covariates. However, we used visual inspection of the plots to decide whether a relationship between variables was meaningful, as well as the Akaike information criterion (AIC) to simplify the model. Variables were removed if they caused an increase in the AIC. The model with the lowest AIC was considered the model with the best fit. The best fit model included 3 biophysical variables (depth, and proximity to seagrass and saltmarsh), 6 threats (SST anomalies, urbanization, nutrients [chlorophyll-*a*], turbidity, recreational fishing pressure, and commercial fishing [trap and line fishing]), protection status, year, and site (year and site were random effects).

Data were modeled with a negative binomial family (log link function) because exploratory analyses indicated overdispersion in simpler competing models (Wood et al., 2016). For the estimation, we used the restricted maximum likelihood algorithm (REML) so we could interpret effect sizes and predictions of fish abundance. Concurvity was investigated to ensure that the remaining predictors were not strongly related (Wood, 2017). Most covariates had concurvity values within acceptable limits (<0.5 [Ramsay et al., 2003]), apart from the effect of site variation (Appendices S3 & S4). The assumptions of homoscedasticity and approximate normal distribution of the deviance of residuals were visually investigated and revealed no major violations (Wood, 2017). Plots with confidence intervals were also used to visualize strength of effect on fish abundance for each covariate (visreg package [Breheny & Burchett, 2020]). Highly confounded variables (correlation values >0.5) were not included in the final model (e.g., commercial trawl fishing and distance to mangroves).

Cumulative impact maps for reef fish

We used the fitted models for invertivore and carnivore abundance to create cumulative impact maps for fish abundance (Table 1). We created the impact maps by first predicting fish abundance across the entire region based on predictions made that reflect potential management scenarios: status quo conditions; status quo conditions with no water quality threats (water quality threats set to zero); status quo conditions with no fishing threats (all fishing-related threats set to zero); and status quo conditions with no threats (all threats set to zero). We did not consider status quo with no SST threats because SST is not manageable at the scale of NSW. Water quality threats included nutrients, turbidity, and population pressure. Fishing threats included recreational and commercial fishing.

We defined cumulative impacts at a grid cell to reef fish (C_i) as

$$C_i = \hat{A}_{i,0} - \hat{A}_{i,T}, \tag{2}$$

where $\hat{A}_{i,0}$ is predicted fish relative abundance (maxN) at a grid cell i with no threats and $\hat{A}_{i,T}$ is predicted fish abundance based on all threats, no water quality threats, or no fishing threats (calculated separately). Thus, the cumulative impact index represented the loss of fish abundance that could be attributed to the threat from water quality, fishing, or both. By predicting cumulative impacts in these 3 categories, we avoided problems associated with making predictions when there is confounding among similar types of threats because there is clear separation between threats.

We used spatially comprehensive data for both fish functional groups to create cumulative impact maps. Predictions were made with the terra package in R (Hijmans et al., 2021). Predicted values were capped at the observed maximum value for that variable to overcome problems with extrapolation of nonlinear models.

Performance of NTZs to protect reef fish

We assessed the performance of NTZs, in terms of how much fish abundance could increase if threats were reduced, by applying Equation (2) to grid cells inside NTZs (SIMP and PSGLMP). The expected impacts of fishing and water quality on fish abundance in NTZs were then calculated as mean (C_i)/SD (C_i), where mean and standard deviation were taken across all grid cells. Thus, the performance metric was the increase in fish abundance expected from reducing the threat (fishing or water quality) as a fraction of the spatial variation in the predicted increase. Values of the performance statistic >100% indicated increases in fish abundance that were highly consistent across all marine park locations.

RESULTS

Hotspots of cumulative threats

We found cumulative threats operating throughout the NSW and Commonwealth waters, with a greater concentration of threats in state waters. A maximum of 7 threats (Figure 2) were apparent on any grid cell throughout the region (individual maps of threats in Appendix S5). We found the highest number of threats in the state waters south of Coffs Harbour and in Commonwealth waters south of Port Macquarie. Waters farther than 20 km offshore contained the least number of cumulative threats, including areas with no threats. The map of threats ranked by the top 5% of grid cells showed similar spatial patterns, except that hotspots were clustered closer to the coast than in the top 10% map (Appendix S6).

A comparison of correlations between different threat layers (Appendix S7) indicated that some threats were highly correlated. Indicators associated with urban areas were positively correlated (population pressure, recreational fishing pressure, and water quality).

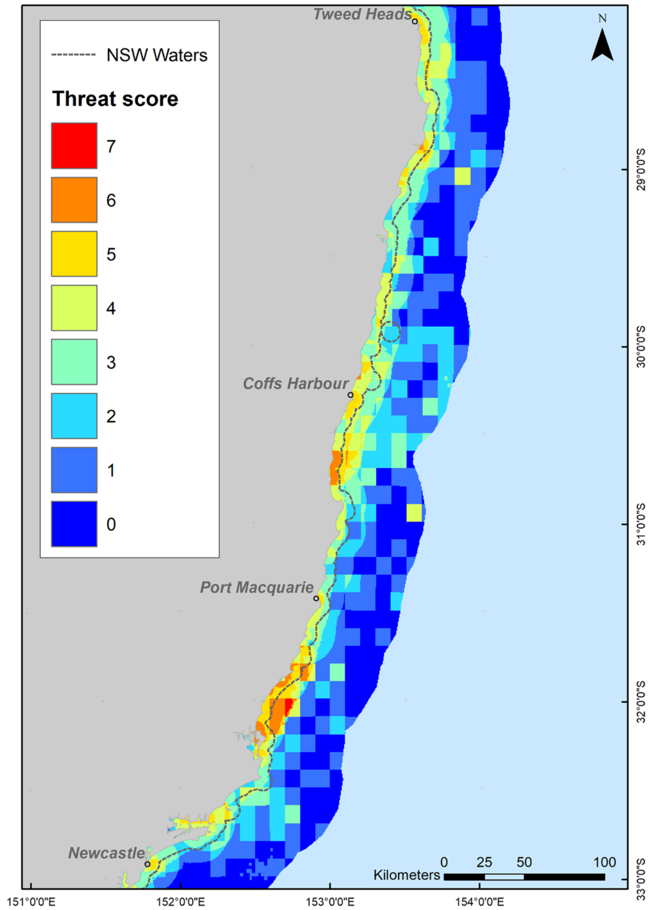

FIGURE 2 Hotspot index of cumulative threats to the northern region of New South Wales (H_i, hotspot index defined as the count of threats in the top 10% of all values for each grid; red, highest threat level; dark blue, lowest threat level). A maximum of 7 threats were apparent on any grid cell throughout the region. Table 1 contains the list of threat layers included. Map produced using ArcGIS ArcMap 10.3 software.

Model of fish abundance

The model explained 48% and 50% of the deviance in fish abundance for invertivore and carnivore groups, respectively. The effect of NTZs had a strong positive influence on carnivores (mean [SE] increase of 1.37 [1.06] inside NTZs) and weaker, but significant, influence on invertivores (mean [SE] increase of 1.11 [1.06] inside NTZs). Carnivorous fish abundance declined as SST anomalies increased and when closer to centers of urbanization but increased when closer to vegetated areas. Carnivorous fish abundance increased as recreational and commercial midline fishing increased. However, these variables were confounded with other variables, which made it difficult to interpret their effects independent from one another (Appendix S7). Similar confounding occurred in the model of invertivores because most of the significant threats had a positive influence

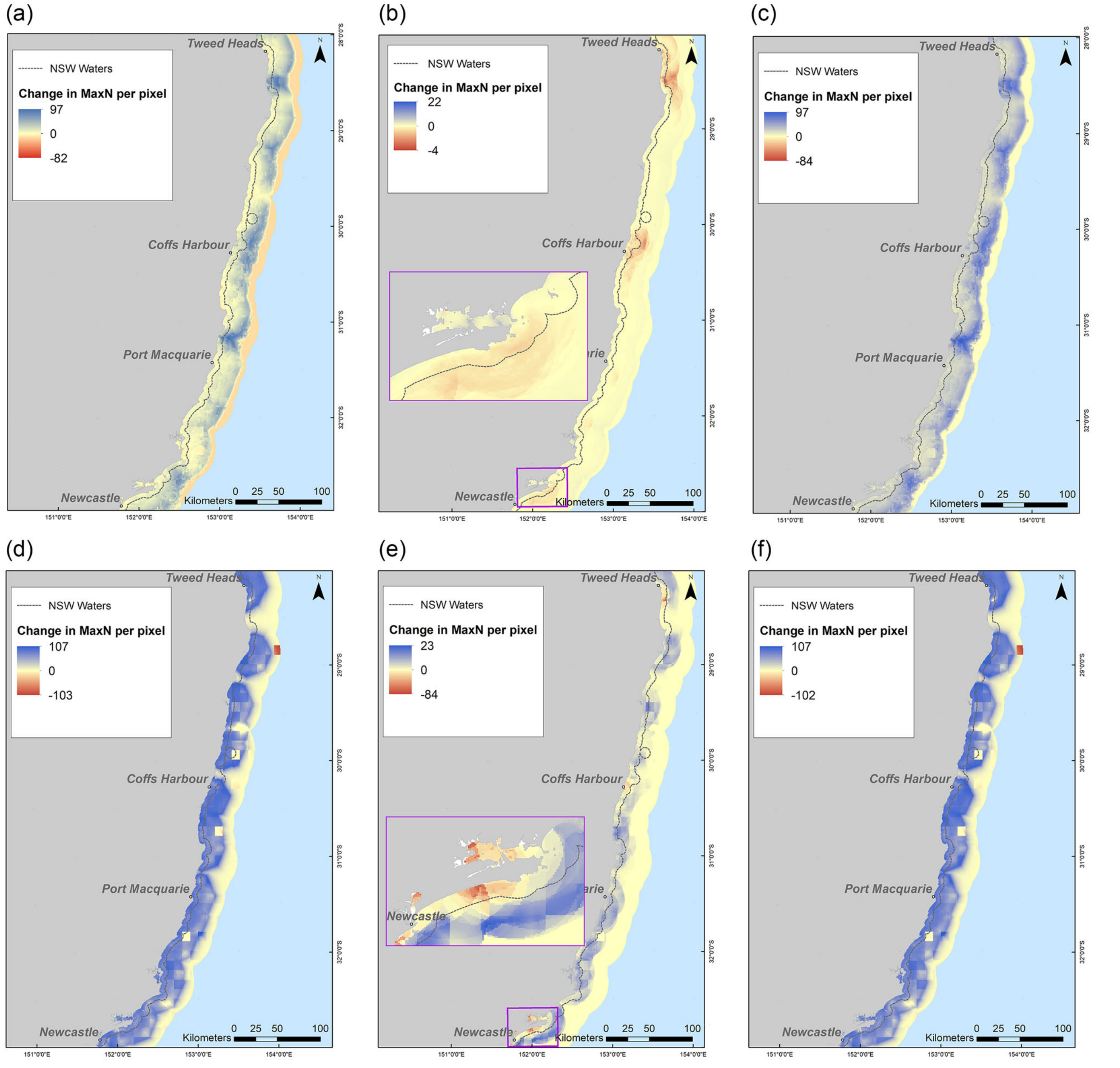

FIGURE 3 The simulated cumulative effect of removing threats on the relative abundance of (a, b, c) carnivorous fish and (d, e, f) invertivorous fish: (a, d) removal of fishing threats, (b, e) water quality threats, and (c, f) both water quality and fishing threats (blue, predicted increase in relative fish abundance with removal of threat; red, relative decrease in predicted fish abundance with removal of threat). Maps produced using ArcGIS ArcMap 10.3 software.

on fish abundance. The effects of water quality covariates, however, were not confounded with fishing pressure covariates or the climate change covariate. Therefore, it was possible to separate the cumulative effects of all fishing-related threats from water quality threats.

Cumulative impact map for reef fish

Fish abundance increased when fishing threats were removed (Figure 3a,d). Both functional groups showed a similar response (increase per grid cell of maxN up to 107 and 97 fish from the baseline for invertivores and carnivores, respectively). The cumulative effect of fishing pressure on carnivores was patchily distributed throughout the northern and central regions of NSW (Figure 3a). The pattern for invertivores was more consistently spread within the 40-km contour and was the maximum extent of recreational fishing pressure used in the model (Figure 3d).

Regions of high cumulative impact from water quality threats were mainly isolated to the coastal areas north of Newcastle, surrounding Coffs Harbour and south of Tweed Heads for carnivores and invertivores (Figure 3d,e). Estuarine areas in Newcastle (Fullerton Cove) and farther north (Port Stephens) also had areas of high cumulative impact for invertivores (Figure 3e). In these regions, fish abundance increased as water quality threats increased, indicating that some species responded in counterintuitive ways to indicators of water quality.

Marine park performance for reef fish

The effect of modeled removal of all fishing threats (recreational and commercial fishing) from inside NTZs and outside NTZs along the boundary had a greater predicted benefit for invertivores inside NTZs than carnivores inside NTZs. For example, there was an increase in the predicted relative abundance for invertivores of 337% compared with a 213% increase for carnivores inside NTZs. The difference between the functional group responses to no fishing was driven by the effect on invertivores in PSGLMP, where an increase in relative abundance of 957% was predicted, compared with an increase of 178% for carnivores (Appendix S8). The predicted increase in relative abundance was similar between carnivores and invertivores in SIMP (349% and 379%, respectively).

When all water quality threats were removed (turbidity, population pressure indicator, and chlorophyll-*a*), the model predicted an increase in the relative abundance of functional groups inside NTZs, but the levels of effect differed. For example, the relative abundance of carnivores was predicted to increase by 51% inside NTZs, but invertivore maxN was predicted to increase by only 11%. When comparing the effect among NTZs, a different pattern was observed. For example, carnivore relative abundance was predicted to decrease inside NTZs in SIMP by 12% and increase inside PSGLMP by 103% when water quality threats were removed. The removal of water quality threats for invertivores inside SIMP and PSGLMP showed limited effect (0.013% predicted increase in SIMP and 1.35% predicted decrease in PSGLMP) (Appendix S8).

DISCUSSION

Effective management requires knowledge about the spatial distribution of threats as well as the impact of threats to species and ecosystems (Halpern et al., 2008). Our results highlighted threat hotspots of potential concern that require further investigation. We also demonstrated that this data-driven approach to CIA revealed some interesting nonlinear relationships between threat and impact, particularly the finding that fish abundance was higher in some areas with putatively higher water quality threat. Understanding how threats affect reef fish abundance helps managers prioritize their actions through regional planning instruments (NSW Government, 2018) and stimulates cross-jurisdictional conversations between state and local councils (Crowder et al., 2006; Griffiths et al., 2020). For example, threats that interact synergistically (i.e., the interaction increases the impact [Brown et al., 2014]) may require a coordinated management approach that can transverse different sectors.

The greatest cumulative threats occurred along the coastline in NSW state waters (3 nm). These included water quality threats in combination with other threats, such as fishing. For example, the threat hotspot in the area south of Port Macquarie, which is the center of a significant agricultural district, was the result of low water quality (turbidity and total nitrogen) and demersal line fishing. Large coastal impacts are widely recognized within Australia and globally and driven by catchment-related processes (Halpern et al., 2019; Ostwald et al., 2021). Fifty percent of the 133 sources of input into the coastal waters of the region (rivers, creeks, etc.) are considered modified or extensively modified, whereas only 10% are in pristine condition (OzEstuaries, 2003). This value is likely to be modest given the increase in coastal human footprint over the last 2 decades (Sanderson et al., 2022). The long history of agricultural use in the region likely explains this (Townsend, 2020), despite a low-to-medium population density of most coastal communities (mean population density of 400 people per 1-km grid cell [Australian Bureau of Statistics, 2021]). Areas of localized hotspots for nutrients (up to 100 μg/L) and turbidity (up to 15 NTU) occur in and adjacent to the Clarence River, which is the largest estuary in the region and one of the largest rivers in mainland Australia (NSW Department of Primary Industries, 2006).

We predicted the removal of cumulative impacts from fishing would benefit invertivores inside NTZs more than carnivores, despite the expectation that carnivores would benefit the most. There are 3 potential explanations for the difference in response between functional groups. First, there were approximately 30% more BRUV captures of invertivores than carnivores (Appendix S2), suggesting that there was greater statistical power to detect impacts on invertivores compared with carnivores. Second, complex predator–prey and competitive relationships are a stronger driver of community structure than fishing intensity (Boaden & Kingsford, 2015). Therefore, the absence of a large increase in carnivores when fishing is removed could be related to compensatory response mechanisms operating between predators and prey or between competitors (Hunsicker et al., 2011; Mitchell & Harborne, 2020). Third, there is likely to be a level of preferential selection of the 30 targeted carnivorous species by fishers, despite all species being targeted by fishers (Malcolm et al., 2018). This requires analysis at the species level because it is anticipated that highly selected carnivores, such as pink snapper (*Chrysophrys auratus*) and yellowtail kingfish (*Seriola lalandi*), would benefit more from the cessation of fishing than invertivores This analysis is recommended for future studies.

Fish abundance was predicted to increase as water quality decreased for carnivores and in some localized areas for invertivores. This finding could be an artifact of other more important variables acting on fish abundance that we did not include. Alternatively, it could suggest that fish benefit from low levels of increased nutrients, an effect that has been observed for pink snapper (Rees et al., 2021). The relationship between water quality threats and fish responses is complex and does not always act linearly (Meador & Frey, 2018). Furthermore, temperate reef fish are mobile and thus have a wide exposure gradient to water quality threats because they can range from within 100 m to up to >5 km from the mouth of estuaries and outlet sources. Therefore, coastal reef fish are unlikely to be exposed to acute concentrations observed in lakes and semienclosed watersheds because of the dispersion into the coastal environment (Meador & Frey, 2018; Parker et al., 2016; Uriarte & Borja, 2009).

We assessed the expected benefit inside NTZs and along the boundary by removing the threat of fishing from areas in and adjacent to NTZs to represent poaching and the effect of fishing around the edge of NTZs, which can be a significant hindrance to fish protection inside NTZs (Harasti et al., 2019; Ohayon et al., 2021). Our predictions indicated that NTZs supported more carnivore and invertivore groups than areas outside NTZs (up to 1.5 and 1.3 times, respectively), which is consistent with previous findings (Harasti et al., 2018; Knott et al., 2021; Malcolm et al., 2018). The predicted difference in fish abundance indicates that the cumulative impacts of fishing outside of NTZs are affecting fish abundance inside NTZs because the relative abundance of fish was predicted to increase with the alleviation of fishing at the boundary. The predicted increase in fish abundance with simulated removal of fishing also suggests that illegal fishing could be occurring in NTZs, which has been identified previously for NTZs in PSGLMP (Davis & Harasti, 2020; Harasti et al., 2019). Strong differences between functional group responses were also predicted between SIMP and PSGLMP. Site-specific differences in abundances may be driven by unmeasured influences on productivity, such as habitat complexity and stochastic variation in early-life-stage recruitment (Knott et al., 2021; Malcolm et al., 2018).

The caveats in our study limit interpretation of the results and require further investigation to improve the understanding of cumulative impacts on temperate reef fish in northern NSW. The patterns of modeled fish abundance in response to cumulative threats are a test of a method where the effect of threats on the ecosystem was driven by the data, rather than being based on the assumption that all threats are negative, as is common practice (Halpern et al., 2008; Korpinen & Andersen, 2016). Therefore, there are some unexpected and unexplained outcomes in the prediction maps, such as the positive response (or lack of response) of fish abundance to a high level of threat (Figure 3b,d). For example, some areas appeared to maintain a high carnivore abundance despite being subject to high fishing pressure, so it is possible that these are productive areas that attract fishing. The regional stability of stocks despite high fishing pressure has been observed in small pelagic fisheries (Hilborn et al., 2022); however, it is not well documented in coastal reef fish. These uncertainties and any potential confounding between fishing pressure and fish stocks suggest that defining the vulnerability value based on a combination of expert opinion and data could link threats to impacts in more meaningful ways (Ferraro et al., 2018). Alternatively, it could suggest that we missed key anthropogenic threats that were driving the predictions.

We considered only additive effects in the model of threat impact when they could have been interactive (Brown et al., 2014; Cote et al., 2016; Crain et al., 2008; Stock & Micheli, 2016). Including covariate interactions would help explain other nonlinear outcomes between threats (Brown et al., 2014), but in this case we were limited by the region having insufficient contrasts of different stressors, which are required to model interactions. Dispersion parameters for the water quality threats were also applied consistently across the seascape, as is common practice with cumulative threat maps (Halpern et al., 2008, 2019). However, the effect of hydrodynamic variation (currents, wind, and waves) is likely to influence dispersion at the watershed level. Although we considered the effect of catchment size, we did not include other parameters, such as flow volume and rate, currents, exposure, and so forth, which are known to have an effect on dispersion (Deignan-Schmidt et al., 2021; Tseng, 2002; Yu et al., 2016). The omission of these influences in the cumulative threat map may change the footprint and distribution of water quality impact.

Interpretation of the findings would have benefited from more spatially resolved data, use of measured data, rather than proxies, and inclusion of other covariates. For example, commercial fishing data were only available at a 1° grid cell, whereas all other threats were more resolved. Although the use of proxies is common in the development of large-scale threat maps (Halpern et al., 2008; Ostwald et al., 2021), spatially explicit measured data (e.g., for recreational fishing [Keramidas et al., 2018; Lynch et al., 2020; Taylor et al., 2018]) would provide a clearer link between threat and impact.

Other variables to consider in future cumulative impacts for the region include other drivers of fish abundance and distribution, for example, seafloor rugosity (Burgos et al., 2017; Emslie et al., 2014; Rees et al., 2021); exposure and currents (Fulton & Bellwood, 2004; Williams & Bax, 2001); food-web dynamics (Fu et al., 2020; Lefébure et al., 2013); habitat type and condition (Fulton et al., 2016; Quaas et al., 2019); and larval and recruitment variability (Planes et al., 2009). Sociocultural, economic, and political factors, such as community support, enforcement, and degree of remoteness (Edgar et al., 2014), also influence the performance of marine parks (Glew et al., 2012; Noble et al., 2020, 2021). We also did not consider indirect pressures, such as shipping and mining exploration, because they are primarily of concern for pelagic species (e.g., deeper benthic species [van der Grient & Drazen, 2021] and cetaceans). Major shipping routes, for example, are in depths >30 m, which is beyond the depths we assessed. Emerging industries in NSW yet to be implemented, such as renewable energy projects, were also not included. However, there is a need to understand if and how pressures associated with these industries act on coastal fish species (Copping et al., 2021). The inclusion of social variables in the models of marine park performance could help explain the differences observed between functional groups and marine parks, and better inform management actions.

The approach we propose addresses 2 limitations common in CIA. First, our method overcomes issues of nonlinearity in the vulnerability of ecosystems to threats by modeling the spatial gradient in fish abundance in response to threats. This approach has the advantage over the expert elicitation approach because it does not require identifying the vulnerabilities of each threat in a preconceived way but, rather, allows for a data-driven approach to explore relationships between threats and patterns in fish abundance. This is important because it is often assumed that all threats are negative when in some cases, they could have benefits. For example, the relative abundance of invertivores was predicted to decrease in some locations by up to 56% when water quality threats were removed. This suggests that additional nutrients may be beneficial to invertivores; the influence of water quality on fish abundance has been shown elsewhere (Meador & Frey, 2018; Rees et al., 2021). Second, the approach offers a clearer path for management action because it quantifies correlations between threats and partitions out the numbers of different types of threat rather than the values of threats. In this way, the number of actors and stakeholders responsible for managing the threat and degree of management action required to address threats are identified.

We used fisheries, water quality, and climate change data to develop a data-driven approach to predict the impact of cumulative threats to fishery targeted coastal, temperate, reef fish. Cumulative threats have not been mapped in NSW previously. However, our work is exploratory and offers an alternative to assessing cumulative impacts that does not rely on expert elicitation. Further research is needed to assess the impacts on temperate reef fish from other cumulative threats we did not consider, such as shipping (and related anchoring), offshore mining, emerging renewable energy industries, and other climate change threats, as well as other spatially explicit variables. The methodology we used could also be used to inform cumulative impacts on other important coastal areas, such as seagrasses, saltmarshes, and mangroves, and their associated estuarine fish assemblages, which are important environmental, economic, and social features of the NSW coastal zone. The inclusion of these indicators would improve understanding of cumulative impacts for the NSW region and could help inform the evaluation of threat against management objectives. Such work needs to be carried out before conservation resources are invested to mitigate threats. Despite this, our results iterate the importance of NTZs in protecting reef fish from threats and provide support for management action to be directed at enforcement activities in NTZs and along the boundaries of NTZs.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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