Ocean and Coastal Management 239 (2023) 106595

Contents lists available at ScienceDirect

Ocean and Coastal Management

journal homepage: www.elsevier.com/locate/ocecoaman

Evidence of rebound effect in New Zealand MPAs: Unintended consequences of spatial management measures

Tai Lohrer[a,*], Judi E. Hewitt[a], Andrew M. Lohrer[b], Darren M. Parsons[b,c], Joanne I. Ellis[d], Fabrice Stephenson[d]

[a] Department of Statistics, University of Auckland, New Zealand
[b] National Institute for Water and Atmospheric Research (NIWA), New Zealand
[c] Institute of Marine Science, University of Auckland, New Zealand
[d] School of Science, University of Waikato, New Zealand

ARTICLE INFO

Keywords:
Rebound effect
Marine protected areas
Regression discontinuity
Spatial management
Bottom trawling
Fishing effort

ABSTRACT

Fish size and fish biomass have been shown to increase inside the borders of Marine Protected Areas (MPAs) after the cessation of fishing. However, the effects of marine protection on fishing fleet behaviour and fish catches outside of MPAs are less well understood. Here we investigated changes in total catch and Catch per Unit Effort (CPUE) of bottom trawlers outside the borders of offshore MPAs in New Zealand. We used Regression Discontinuity in Time (before versus after protection) on both aggregate and individual trawl event data for one Marine Reserve, two Benthic Protected Areas, one Closed Seamount Area and one Marine Mammal Sanctuary. Despite the various forms of protection that reduce the total fishable area, total catch tended to increase after MPA implementation. Yet, there was little evidence that this was due to the net-movement of fish from larger populations within MPA boundaries i.e. spillover. Rather, the increases in catch in the post protection period appeared to be a consequence of changes in the behaviours of commercial fishers. This may be an unintended and previously unreported negative consequence of marine protection that could dampen some of the many benefits of MPAs.

1. Introduction

Marine ecosystems worldwide are under threat from overfishing, with 60% of major marine ecosystems being degraded or used unsustainably (OECD 2017). Previous studies have extensively demonstrated the benefits of Marine Protected Areas (MPAs) as a policy tool that can mitigate fishing impacts by increasing biodiversity, spawning stocks, and fish size (Edgar et al., 2014; Baskett and Barnett., 2015). Globally, Marine Reserves have been shown to have multiple positive benefits for species within the boundaries of a reserve. In a review of 87 strict no-take Marine Reserves over ten years old, Edgar et al. (2014) found that they exhibited fish that were twice as large per transect, five times greater large fish biomass and fourteen times more shark biomass than fished areas.

As of 2023, 8.2% of our oceans are classified as MPAs, far below the United Nations Convention on Biological Diversity's draft target to protect 30% of the world's oceans by 2030 (mpatlas.org; CBD 2021). MPAs vary widely in scope, enforcement and regulation, with only 2.9% of global MPAs being designated as strict no-take zones as opposed to less stringent gear or fishing method restrictions (mpatlas.org).

The creation of marine reserves often faces opposition from the fishing industry, who claim that the reduced fishing area and increased travel distances to fishable areas will negatively impact profitability (Boubekri et al., 2022; Lynham et al., 2020; Stevenson et al. 2013). Yet some studies suggest that overall fishing effort is unaffected by Marine Reserve implementation, resulting in a benefit for fishers despite the loss in fishing area (Gell and Roberts 2003; Alcala et al., 2005; Lynham et al., 2020). However, many of these studies focus on subsistence and sport fishing rather than large scale commercial fishing. Commercial fishing generally has a much greater impact on marine ecosystems due to the type of gear and techniques used, the vastly greater area covered, and the sheer amount of biomass caught. In particular, bottom trawling has one of largest anthropogenic impacts on benthic marine ecosystems because of the contact of fishing gear on the seafloor (Thrush and Dayton 2002; Althaus et al., 2009; Worm et al., 2006; Clark et al., 2016).

Marine reserve proponents argue that MPAs have 'spillover' and

* Corresponding author.
E-mail address: tloh305@aucklanduni.ac.nz (T. Lohrer).

https://doi.org/10.1016/j.ocecoaman.2023.106595
Received 21 November 2022; Received in revised form 28 March 2023; Accepted 28 March 2023
Available online 23 April 2023
0964-5691/

'recruitment' benefits outside of their boundaries that result in greater catch yields for local fisheries (McClanahan and Mangi 2000; Di Lorenzo, Claudet, and Guidetti 2016). Spillover refers to the net export of juveniles, and adults of valued species from protected areas to fished areas while recruitment refers to the net export of eggs and larvae (Abesamis and Russ 2005). The extent and magnitude of the spillover and recruitment effects are debated in the literature. In a review of 85 articles attempting to quantify these phenomena, Di Lorenzo, Claudet, and Guidetti (2016) found that 85% of empirical studies reported some level of spillover from MPAs. The extent of spillover in these studies varies greatly depending upon the method of study. Visual census methods estimate spillover up to several hundred metres from reserve boundaries, while fish tagging has shown fish moving out to 5–150 km depending on individual species' behaviour (Di Lorenzo, Claudet, and Guidetti 2016). Another review estimated the reach of spillover to be 600–1500 m from Marine Reserve boundaries (Halpern et al. 2009), however, the analysis was based on observed fish abundance rather than catch data, which may underestimate spillover. Similarly, Green et al. (2015) found that spillover can occur from 100 m to 100 km depending on the species, while recruitment tends to occur within 15 km of MPA boundaries.

Fishing effort and trawling distribution is driven by a multitude of interlinked processes and how fishers respond to management is sometimes unexpected (Stephenson et al., 2018). Ultimately, the decision to fish is a human behavioural response and, consequently, a limited understanding of the social, cultural and technological drivers of fishing can lead to poor management choices (Giakoumi et al., 2017).

Investigating commercial bottom trawling activity surrounding MPAs allows for the examination of the 'rebound effect' after an MPA is implemented. The rebound effect is a counterbalancing response that can occur after conservation policy is implemented (Sorrell and Dimitropoulos 2008). A common example of the rebound effect is improved fuel mileage in automobiles. A 5% increase in fuel efficiency should, in theory, result in a 5% decrease in transport emissions. However, with the reduced cost of travel, consumers may increase their travel distances and thus increase emissions slightly. The rebound effect thus diminishes the expected benefits from an innovation or policy designed to have net positive effects on society or the environment. When the rebound effect is greater than one, it is referred to as the Jevons Paradox e.g. when transport emissions increase after an increase in fuel efficiency (York and McGee 2016).

While typically applied to changes in efficiency resulting from technological innovation, the rebound effect can also be applied to the 'innovation' of government policy. In this case, the implementation of an MPA theoretically results in a decrease in fishing effort to zero within its boundaries (Lynham, 2022). Holding other factors constant, this should result in a decrease in fishing effort and total catch in the area surrounding the MPA. However, as mentioned previously, the spillover and recruitment benefits an MPA provides, in terms of larger, more abundant fish outside the MPA, may offset these losses. Despite the importance of the rebound effect and its potential impacts on conservation policy, few studies have investigated these unintended consequences (see e.g. Lynham 2022; Lenihan et al., 2021).

To address these knowledge gaps and uncertainties, this study examined bottom trawling—the most common commercial fishing method in New Zealand waters—near five offshore MPAs in New Zealand. We investigated whether a measurable rebound effect exists by studying changes in catch and Catch per Unit Effort (CPUE) before and after MPA designation using Regression Discontinuity in Time (RDiT) models from monthly aggregated data for peak fishing seasons. More complex discontinuity models for individual trawl events with controls for depth, climate and current variation, and vessel effects were used to account for variation in other factors that affect catch.

2. Materials & methods

New Zealand was a pioneer in MPA policy, implementing its first no-take Marine Reserve in 1975. Currently, no-take MPAs cover 9.8% of New Zealand's Territorial Seas (Davies et al., 2018). Despite being a small nation by land area, New Zealand has the fourth largest Exclusive Economic Zone (EEZ) in the world, covering over four million km^2. Unsurprisingly, New Zealand's fishing industry is the 7th largest export commodity by value, generating $2.8 billion annually from 410 million kilograms of commercial catch on average between 2015 and 2020 (Dixon and McIndoe, 2022). Despite New Zealand's EEZ's economic and ecological significance, little research has been done to analyse the impact of commercial fishing on the effectiveness of protected areas within it.

2.1. *New Zealand's MPA framework*

New Zealand has three broad classifications of marine protections: Marine Reserve MPAs (type 1), 'other MPAs' (type 2) and 'other marine protections tools' (Department of Conservation and Ministry of Fisheries 2008).

Implemented under the Marine Reserves Act of 1971, Marine Reserves yield the strongest protections and have the specific focus of preserving marine life in order to conduct scientific study on pristine ecosystems. Marine Reserves are designated no-take zones where commercial or recreational fishing activity, seabed mining, and the collecting of materials is strictly prohibited. New Zealand's 44 reserves range in size from 0.17 to 7480 km^2 and border various coastal marine habitats which protect inshore reefs and offshore islands.

'Other MPAs' confer less stringent protections than Marine Reserves and can take a variety of forms such as Benthic Protected Areas and Seamount Closures. However, they all achieve a similar objective: to allow the recovery of biodiversity in an area. These include prohibitions under the 1983 Fisheries Act, which impose restrictions on the type of equipment and methods fishers use, along with other types of protection that indirectly attain these objectives, such as underwater cable protection zones.

The New Zealand government, in collaboration with the fishing industry, implemented a network of Benthic Protected Areas (BPAs) in November 2007. Covering 1.1 million km^2, these type 2 MPAs protect roughly 30% of New Zealand's EEZ (Helson et al., 2010). BPAs are closed to bottom trawling and dredging but still allow for other methods of fishing, hence they do not impose as strict regulation as Marine Reserves. The focus of BPAs is to preserve a wide range of pristine marine ecosystems, and therefore most of them are in areas that have not been extensively fished.

Under the Fisheries Act, 18 areas were closed to trawling and dredging in 2001 protecting 81,000 km^2 within New Zealand's EEZ (Helson et al., 2010). These type 2 MPAs protect 25 underwater topographic features (UTFs), known collectively as the Seamount Closures. Seamounts are highly productive areas, and have been subject to extensive bottom trawling, particularly on the Chatham Rise where they contributed around 65% of total catch in the area as early as 1995 (Clark 1999).

'Other marine protection tools' refers to policies that have similar objectives to the aforementioned groups, but do not sufficiently protect biodiversity to meet those higher standards and were thus excluded from our analysis.

2.1.1. *Study site 1: Auckland Islands Marine Reserve & Marine Mammal Sanctuary*

The Auckland Islands Marine Reserve is one of the largest reserves in New Zealand, with an area of 1046 km^2 (Fig. 1). The main commercial fisheries in the surrounding area are for three deep water species: scampi (*Metanephrops challengeri*), arrow squid (*Nototoarus sloanii*), and hoki (*Macruronus novaezelandiae*).

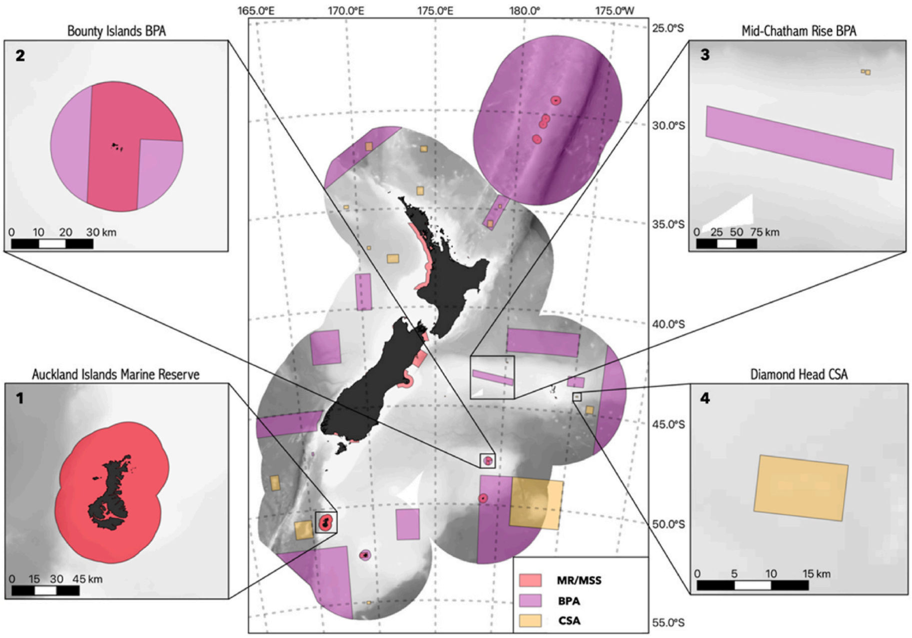

Fig. 1. Location of the study sites within New Zealand's EEZ (coloured). Areas highlighted in red signify marine reserves and marine mammal sanctuaries (MR/MSS), purple signifies benthic protected areas (BPA), and yellow signifies closed seamount areas (CSA). Numbers in the top left of the insets denote the four study sites: 1) The Auckland Islands MR and MMS (implemented in 1993 and 2003 respectively), 2) The Bounty Islands BPA, 3) The Mid-Chatham Rise BPA, and 4) The Diamond Head CSA. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article).

Due to by-catch of endangered sea lion (*Phocarctos hookeri*) by the squid fishery, the Auckland Islands was designated as a Marine Mammal Sanctuary in 1993 under the Marine Mammal Protection act of 1978. Most Marine Mammal Sanctuaries are classified as 'other marine protection tools' as they do not directly regulate fishing activities (Department of Conservation and Ministry of Fisheries 2011). However, the Auckland Islands Marine Mammal Sanctuary is a special case, where commercial fishing of all forms was banned within 12 nautical miles of the islands. Thus, in everything but name, it was a Marine Reserve. The Marine Mammal Sanctuary was then granted full Marine Reserve status in December 2003, which implemented some further restrictions such as banning recreational fishing. However, due to the Auckland Islands' geographic remoteness, little to no recreational fishing occurred there prior and, therefore, in effect, the level of marine ecosystem protection was unchanged before and after 2003.

2.1.2. Study site 2: Bounty Islands benthic protected area

The Bounty Islands BPA (Fig. 1) restricted bottom trawling and dredging within 12 nautical miles of the islands. The main commercial fisheries in the area are for three species of deep water, seamount-associated fish: oreo (*Pseudocyttus maculatus*), orange roughy (*Hoplostethus atlanticus*), and southern blue whiting (*Micromesistius australis*). As with other BPAs, the Bounty Islands BPA was implemented in 2007.

2.1.3. Study site 3: Mid-Chatham Rise benthic protected area

The Mid-Chatham Rise BPA is located in one of New Zealand's most prolifically fished waters (Fig. 1). While the area the BPA encloses has never been extensively fished, it is an important breeding ground for commercially caught species on the Chatham Rise (O'Driscoll et al., 2011), thus making it a useful MPA case study. The main commercial fisheries in the area are for hoki, scampi, and hake (*Merluccius australis*).

2.1.4. Study site 4: Diamond Head closed seamount area

The Diamond Head Closed Seamount Area (CSA) is the only seamount closed to bottom trawling among the hundreds of smaller underwater topographic features in the area (Fig. 1). The main commercial fisheries are for orange roughy, alfonsino (*Beryx splendens*), and long-finned beryx (*B. decadactylus*). Diamond Head CSA is the smallest of the four study sites.

2.2. Data

Commercial fishing data were provided by New Zealand's Ministry for Primary Industries (MPI) and used to create timeseries datasets for our analysis. Since 1989, commercial vessels fishing within New Zealand's EEZ have been required by law to supply MPI with catch information for every fishing event (when the nets or longlines are set) along with date of event, start and end coordinates, method of fishing, target species, depth of fishing, number of hooks used and duration of the event. Following other New Zealand fisheries analyses, trawls exceeding 70 km were deemed erroneous as they are beyond what is expected in normal fishing practice (nearly three times longer than the median trawl distance) and were thus excluded from the analysis (Black and L Tilney, 2017; Baird and Wood 2018).

2.2.1. Data overview

To determine the effect of MPA designation on fishing effort and distribution, five data sets were compiled for the four study sites (separate data sets were created for the Auckland Islands Marine Mammal Sanctuary and Marine Reserve). Each data set contained catch data and effort data for events that occurred within 50 km of the MPA boundary. Co-variables such as depth, month of year, and climate/oceanographic proxies (see section 2.2.3) that can affect fish abundance and year-class strength were also included in the data set. All data points were geo-referenced and linked with a unique vessel-ID (i.e., a specific fishing vessel). Thus, distance to the MPA boundary for all individual fishing events and other critical information was available for use in the analysis.

2.2.2. Catch per unit effort calculation

Our analysis was based on the total catch weight reported for each individual trawl event. This metric is an estimate of the total biomass caught during a trawl event measured in kilograms, i.e. it includes both target commercial species as well as all bycatch (other marine species that are not the target species). The rate of bycatch from bottom trawling in New Zealand varies by species targeted, with bycatch making a smaller proportion for fisheries targeting species such as hoki, arrow squid and southern blue whiting but comprising over 80% of total catch in scampi fisheries (Anderson, 2012).

Catch per unit effort (CPUE) was calculated for each fishing event by standardising total catch (kg) by area trawled (km^2, a product of the wingspread of the trawl net by the distance travelled during an event). Trawl events with an effort width of zero (i.e., erroneous) were removed from the dataset.

2.2.3. Ancillary data

Many factors influence fish populations and consequently total catch and CPUE. Studies within New Zealand's EEZ have shown that bottom temperature and depth are good predictors of fish assemblages (Francis et al., 2002; Stephenson et al., 2018). Therefore, spatially explicit estimates of depth (m) and bottom water temperature (^{o}C) (1 km grid resolution, Stephenson et al., 2022) were used to calculate mean depth and bottom temperature for each trawl event. However, due to the high correlation between bottom temperature and depth (R values ranged from 0.84 to 0.98), bottom temperature was excluded from the analysis.

To better account for seasonal changes and fluctuations in global weather patterns, the Southern Oscillation Index (SOI) was included in the analyses. The SOI is a measure of pressure variability between Tahiti and Darwin, Australia, and is an indicator of year-to-year climate variability for the southern hemisphere. Studies on squid fisheries have shown correlations with SOI and other climate and current indices (Waluda et al. 2004; Rodhouse et al., 2014; Thiaw et al., 2017). Given that changes in the SOI are associated with changing weather and ocean current patterns, monthly SOI values will account for between-month variability in environmental factors such as water temperature. SOI data are available online from New Zealand's Ministry for the Environment and were aggregated to a monthly level for the analysis. Values were assigned to individual fishing events corresponding to the month and year they occurred.

2.2.4. Data cleaning & preparation

The trawl data sets were clipped to only include events that came within 50 km of the MPA boundaries (see section 2.4). Any event that overlapped with this buffer was included in the data set. This includes events that occurred within the MPA itself, which was observed both before and after MPA implementation in nearly all data sets.

Given that it can take years to detect spillover and recruitment effects from an MPA, long data sets are required to assess their effects. In addition, examining longer temporal windows allows for more accurate accounting of seasonal effects and other explanatory variables. Thus, to standardise the length of the data sets before and after implementation across the four MPAs, a window of seven years on either side of the month of MPA implementation was chosen in all cases where data were available (see section 2.4). For the Auckland Islands Marine Mammal Sanctuary, a window of three years on either side was used due to its implementation in 1993, as accurate fisheries reporting data is only available from 1989 onwards.

CPUE across all four MPAs exhibited extreme right skewed-ness, and upon closer examination of the observations at the tail, these appeared to be recording errors, where, for example, trawls of 2 km recorded a total catch weight of 60,000 kg. For this reason, events above the 99th percentile CPUE (i.e. likely recording errors) were excluded.

Although the exact date of each trawl event was recorded, the data were aggregated into months so that monthly catch statistics could be analysed both before and after MPA designation, with time (month) treated as a continuous variable that is centred at the month the MPA was implemented (e.g. −3 refers to three months prior to implementation).

2.2.5. Data summary

Five data sets were created for bottom trawling events occurring within 50 km of the MPA boundaries and seven years (three years for the Auckland Islands Marine Mammal Sanctuary) before and after designation. Each observation has the following information: total catch (kg), CPUE (kg/km^2), month (centred on the MPA implementation date), average depth (km), distance to MPA boundary (km), SOI, and the unique vessel ID.

The number of trawl events for each MPA ranged from 993 (Bounty Islands BPA) to 27,688 (Mid-Chatham Rise BPA), indicating the varying extents to which these areas were fished. Full descriptive statistics can be found in the appendix (Table S1-Table S5).

The reductions in fishable area within 50 km of the MPA boundaries ranged from 1%, 10%, 20% and 43% for the Diamond Head CSA, Bounty Islands BPA, Mid-Chatham Rise BPA, and the Auckland Islands Marine Reserve/Marine Mammal Sanctuary respectively.

To address questions regarding the rebound effect and whether the Jevons Paradox can be observed near MPA boundaries, the individual trawl event data was aggregated at the monthly level to detect trends in total catch. In these aggregated data sets, each month has one observation that includes data on the total catch weight (the sum of all catch from trawl events for that month), CPUE (calculated as the total catch weight divided by the trawl area for each month), the total number of trawl events that occurred, the number of vessels, the trawl area (the sum of all individual trawls in km^2 per month) and the SOI value associated with that month.

Across all MPAs, catch was highly seasonal with clear and consistent peaks. As this study pertains to the impact of MPA policy, for these aggregate data, the three peak fishing months in an average year were selected for each MPA to conduct the analysis to get a clearer picture of long-term trends in catch and the drivers of it.

A measure of trawling intensity was also constructed by calculating the average number of trawl events conducted per vessel (calculated by dividing the total number of trawl events by the number of unique vessels) before and after MPA implementation.

In order to visually identify changes in the spatial distribution of bottom trawling surrounding the MPAs, catch from individual trawl events were aggregated into 5 × 5 km grid cells and mapped before and after designation using the *sp* package in R.

2.3. Analysis

To evaluate the impact MPA policy has on fish populations and commercial fisheries, we take advantage of the exogeneity of MPA policy implementation and employ Regression Discontinuity in Time (RDiT) models to analyse changes before and after MPA designation for both the individual trawl event and aggregate datasets. The aggregate models provide insight into overall temporal trends in the outcome variables of interest, while the individual trawl event models introduce additional controls to account for more detailed spatiotemporal patterns.

2.3.1. Aggregate models

Treatment status (the status of protection) was allocated based on the date the MPAs were designated. Since the running variable 'month' is centred at these dates (i.e. the month is a value ranging from −84 to 84, corresponding to seven years before and after MPA designation), the treatment status is defined as:

$$T = \begin{cases} 1 & \text{if } month \geq 0 \text{ (i.e. after MPA implementation)} \\ 0 & \text{if } month < 0 \text{ (i.e. before MPA implementation)} \end{cases}$$

Based on the count data nature of the trawls, negative binomial Generalised Linear Models (GLMs) were used for this analysis, as these can account for non-constant variance and non-normal errors with a more flexible mean-variance relationship than a Poisson model.

The basic specification to estimate the effect of treatment is as follows, where the outcome variable Y_t is the aggregate total catch, CPUE, trawl event count, vessel count, or trawl footprint during month t:

$$\mu_t = exp(\beta_0 + \beta_1 T_t + \beta_2 month_t + \beta_3 month_t * T_t) \tag{1}$$

Where the outcome variable is drawn from a gamma distribution with mean μ_t and variance $\mu_t + {\mu_t}^2/\theta$ where θ is our dispersion parameter.

β_1, β_2 and β_3 are all coefficients of interest, estimating the immediate impact of MPA designation and the temporal trends in the outcome variables before and after. While aggregating the data and selecting peak fishing months results in fewer observations, it allows for simpler models as vessel fixed effects and seasonality can be ignored. Moreover, the mean temperature, depth and SOI values are comparable before and after the MPA is implemented, making this a reasonable comparison. However, SOI was eventually excluded from the regressions as it was not statistically significant for any of the five MPAs.

The designation of an area to be an MPA should have no immediate bearing on biological and physical factors affecting fish catch. Thus, one would expect these factors to continue smoothly at the treatment cut-off point. Given that the implementation of an MPA results in a reduction of fishable area (ranging from 1% to 43% within 50 km of the MPAs of study), one would expect a reduction in catch or no change as fishers are displaced to areas outside of the boundary. In a review of ecological studies within Marine Reserve boundaries, Halpern and Warner (2002) estimated that increases in density, biomass and diversity occur within one to three years of a reserve being established. In addition, Gell and Roberts (2003) note that while there are rapid population build-ups in the first three years of reserve implementation, long-term studies have shown evidence of sustained linear increases in population densities over 10 years after implementation. What these studies suggest is that if spillover occurs from the MPAs, one would expect total catch and CPUE to increase gradually over time, with no major jump at the start date. Therefore, any instantaneous changes in catch may be reasonably attributed to individual fisher behaviours.

2.3.2. Individual trawl event models

To control for factors mentioned above that may undermine identification of a treatment effect, similar discontinuity-based models were applied to the individual trawl event data sets. In comparison to conventional regression discontinuity analysis of cross-sectional data, RDiT models have time as the running variable, which requires controls to prevent biased results (Hausman and Rapson 2017; Bernal et al., 2016). Moreover, while the previous models demonstrate the impact of MPA implementation on aggregate, investigating individual trawl events provides additional insight into the impact MPA implementation has on individual fishers and whether catch increases as one approaches MPA boundaries. A negative binomial GLM is used as in equation (1) but with the addition of control variables:

$$\mu_{i,t} = exp\big(\beta_0 + \beta_1 T_t + \beta_2 \text{month}_t + \beta_3 month_t{}^*T_t + \beta_4 dist_i + \beta_5 dist_i{}^*T_t + \beta_6 depth_i + \beta_7 depth_i{}^*T_t + \beta_8 SOI_t + \beta_9 SOI_t{}^*T_t + \beta_{10} monthFE_t + \beta_{11} vesselFE_i\big) \quad (2)$$

Where the outcome variable is drawn from a gamma distribution with mean $\mu_{i,t}$ and variance $\mu_{i,t} + \mu_{i,t}{}^2/\theta$.

The response variable $Y_{i,t}$ is the total catch or the CPUE for an individual trawl event i at time t. As in equation (1), the coefficients of interest are β_1, β_2 and β_3. The coefficients for $dist_i$ (β_4 and β_5) are also of interest, quantifying the relationship between the distance of event i to the MPA boundary in kilometres and total catch/CPUE before and after protection.

As mentioned previously, abiotic factors such as temperature, depth, climate and current patterns influence catch, which the variables $depth_i$ (the average depth of trawl event i measured in kilometres) and SOI_t (the SOI index value for month t) attempt to capture. In order to control for the seasonal patterns in total catch, the month of year is added as a fixed-effects factor in the variable $monthFE_t$. The last fixed effect control, $vesselFE_i$ (the ID number unique to each fishing vessel) relates to how total catch and CPUE are affected by the specific vessel undertaking the trawl event. This may proxy for a variety of unmeasured factors such as the size of the vessel, the engine size, the width of trawl net used, the efficiency of crew members, and the knowledge of the skipper.

2.4. Sensitivity analysis

There are various caveats to implementing RDiT that differ from cross sectional regression discontinuity methods. For one, choosing observations close to the discontinuity is generally preferred. However, with RDiT, this is difficult to achieve as precision falls with decreasing sample size. Moreover, due to the highly seasonal nature of commercial fishing and the ecological fluctuations from year to year, reducing the window of observation may further confound estimates of the treatment effect. Choosing a window of seven years on either side of the MPA implementation date was chosen to balance these considerations. Thus, in order to investigate the time varying effects, sensitivity analysis was conducted to see whether the treatment effect varies based on the specification of year and area of study. This was done by re-running these models with a window of three years either side and with 1.5 and 15 km buffers (details available in the supplementary materials).

3. Results

Visualisations created from the spatially explicit datasets demonstrate that fishing activity increased after MPA designation (Fig. 2, Fig. S3-Fig. S6). However, the distribution of catch remained largely unchanged after MPA designation. This was further confirmed in the aggregate and individual trawl event models.

3.1. Aggregate models

The results for total catch in the aggregate models (Table S6-Table S10) indicate that there was a discontinuous increase in total catch

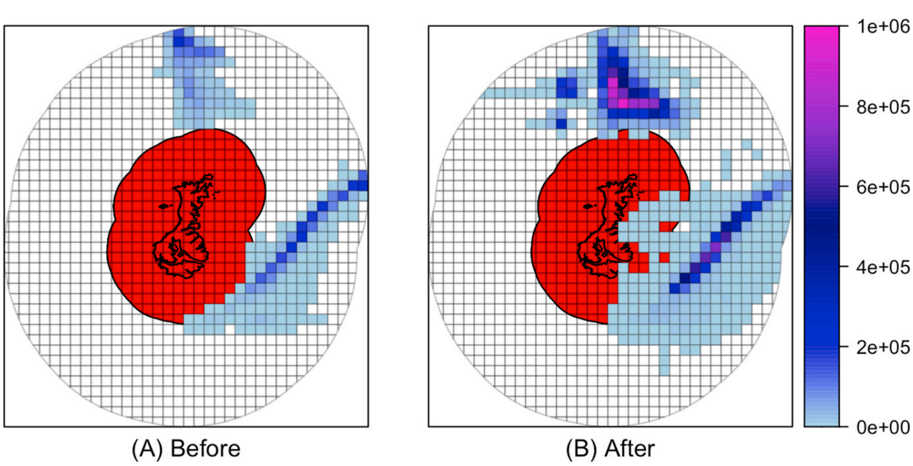

Fig. 2. Catch (in kg) from bottom trawling three years before (a) and after (b) the Auckland Islands Marine Mammal Sanctuary (red polygon) was implemented. To comply with New Zealand regulations around commercially sensitive data release the maps only show fishing events within grid cells within which more than two unique vessels fished. Reported fishing was provided by fishers; it is unclear whether reported fishing activity within protected areas represents accidental misreporting of fishing positions or non-compliance with regulations.

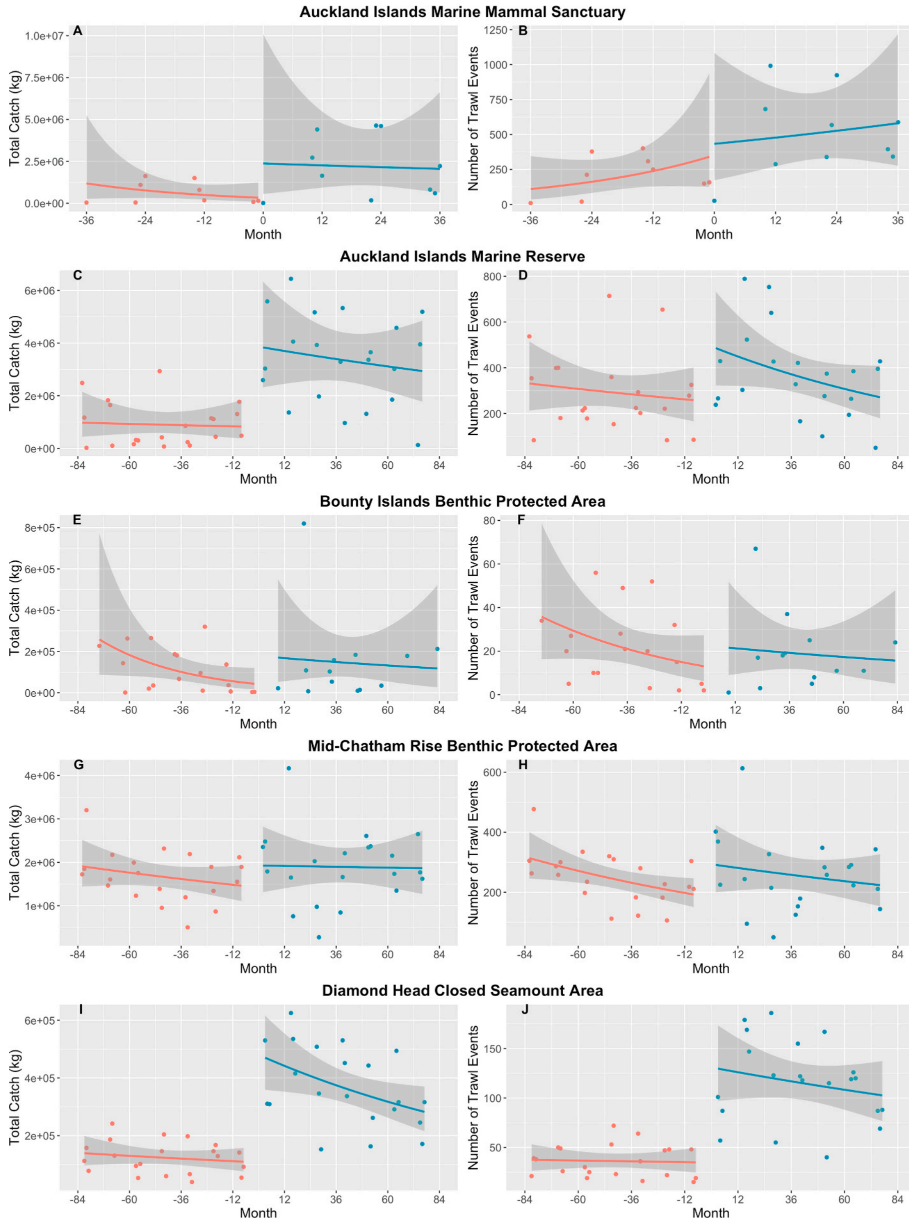

Fig. 3. Total Catch (kg) (left) and number of trawl events (right) aggregated monthly for peak fishing seasons for the Auckland Island Marine Mammal Sanctuary (a–b), the Auckland Islands Marine Reserve (c–d), the Bounty Islands BPA (e–f), the Mid-Chatham Rise BPA (g–h) and Diamond Head CSA (i–j). Trend lines and confidence intervals were generated from the model specified in equation (1).

at the time of MPA implementation for all study locations ranging from 35% to 640% (though it was not statistically significant for the Bounty Islands and Mid-Chatham Rise BPAs) (Fig. 3).

These increases in total catch were accompanied by a jump in monthly trawl events ranging from 23% to 277% (Table S6-Table S10). All five MPAs exhibited a discontinuous increase in the number of trawl events after implementation (Fig. 3), with the Auckland Islands Marine Reserve and Diamond Head CSA demonstrating significant results.

The discontinuities in total catch and number of events were largely mirrored in the number of unique vessels (Table S6-Table S10), though the changes in the number of vessels appeared to be less pronounced (Fig. S1). Only the Auckland Islands Marine Reserve exhibited a significant increase in vessels after MPA designation.

Similarly, changes in trawled area (Table S6-Table S10) follow the patterns of total catch and trawl events. Only Diamond Head CSA exhibited a statistically significant discontinuity. These changes were greater in magnitude than those of the number of unique vessels (Fig. S2).

Despite the observed initial jump in the number of vessels at the time of treatment, apart from the Auckland Islands Marine Mammal Sanctuary, the total number of different vessels fishing decreased in the seven-year period after MPA implementation. The Auckland Islands Marine Reserve exhibited the largest decrease, with 15 fewer vessels in operation in the seven years after the reserve was implemented, while the Marine Mammal Sanctuary saw a three vessel increase after three years.

Due to the low number of vessels that fished near the Bounty Islands BPA and Diamond Head CSA, (14 and 24 unique vessels across the windows of study respectively) it is difficult to discern an impact from the change in the number of vessels alone. Apart from the Bounty Islands BPA, the MPAs saw substantial increases in the number of trawl events per vessel (Table 1).

Diamond Head CSA saw nearly a four-fold increase in trawl intensity after MPA implementation, where on average vessels conducted 186.92

Table 1
Trawl intensity (measured as the mean number of trawl events per unique vessel) before and after MPA designation.

MPA	Trawl Intensity Before	Trawl Intensity After	Percent Change
Marine Mammal Sanctuary	47.18	119.60	152.22%
Auckland Islands Marine Reserve	118.42	209.43	76.85%
Bounty Islands BPA	35.55	35.14	−1.15%
Mid-Chatham Rise BPA	124.60	173.61	39.33%
Diamond Head CSA	38.00	186.92	391.89%

trawl events compared to 38 before MPA implementation. Likewise, the Auckland Islands Marine Mammal Sanctuary saw greater than a 1.5-fold increase in trawl events per vessel in the three years after the MPA was implemented. Both the Bounty Islands BPA and Mid-Chatham Rise BPA exhibited the smallest changes after implementation.

To understand what might be driving these positive treatment effects and changes in trawl intensity, we turn to the results for CPUE (Table S6-Table S10). As expected, most of the MPAs did not exhibit a statistically significant increase in CPUE at the implementation month, except for the Auckland Islands Marine Reserve and Diamond Head CSA, which exhibited an increase with treatment (Fig. 4).

There were no statistically significant temporal trends after MPA designation across all five MPAs (Table S6-Table S10). However, there appeared to be negative temporal trends in total catch after MPA designation for the two sites that exhibited statistically significant discontinuities in CPUE: the Auckland Islands Marine Reserve and Diamond Head CSA (Fig. 3). These trends are mirrored in the number of trawl events, unique vessels, and trawl area (Fig. 3, Fig. S1, Fig. S2).

As with total catch, there were no statistically significant temporal trends in CPUE after MPA implementation. Yet, visually there appear to be upward trends in CPUE after the implementation of the Auckland Islands Marine Reserve, Bounty Islands BPA and the Mid-Chatham Rise BPA (Fig. 4). These temporal trends appear to be inversely related to the temporal trends observed in total catch, the number of events and number of vessels.

3.2. Individual trawl event models

In four of the five MPAs, there was a discontinuous change in total catch at the time of treatment (Table S11-Table S15), with the exception of Diamond Head CSA. These increases ranged from 20% for the Auckland Islands Marine Mammal Sanctuary to over 2000% for the Bounty Islands BPA. The Mid-Chatham Rise BPA exhibited a decrease in total catch of 45%. These findings largely reinforce those found in the aggregate models for total catch, and demonstrate that in most of the MPAs, catch increased immediately after MPA implementation both in aggregate and at the individual event level. Only the Auckland Islands Marine Mammal Sanctuary exhibits a statistically significant change in temporal trends in total catch, increasing before MPA implementation followed by a decrease.

The results for the relationship between total catch and distance from MPA boundaries are mixed (Table S11-Table S15). All five MPAs exhibited statistically significant differences in the relationship between these variables before and after MPA implementation, yet only two of these differences resulted in a change in sign. Diamond Head CSA

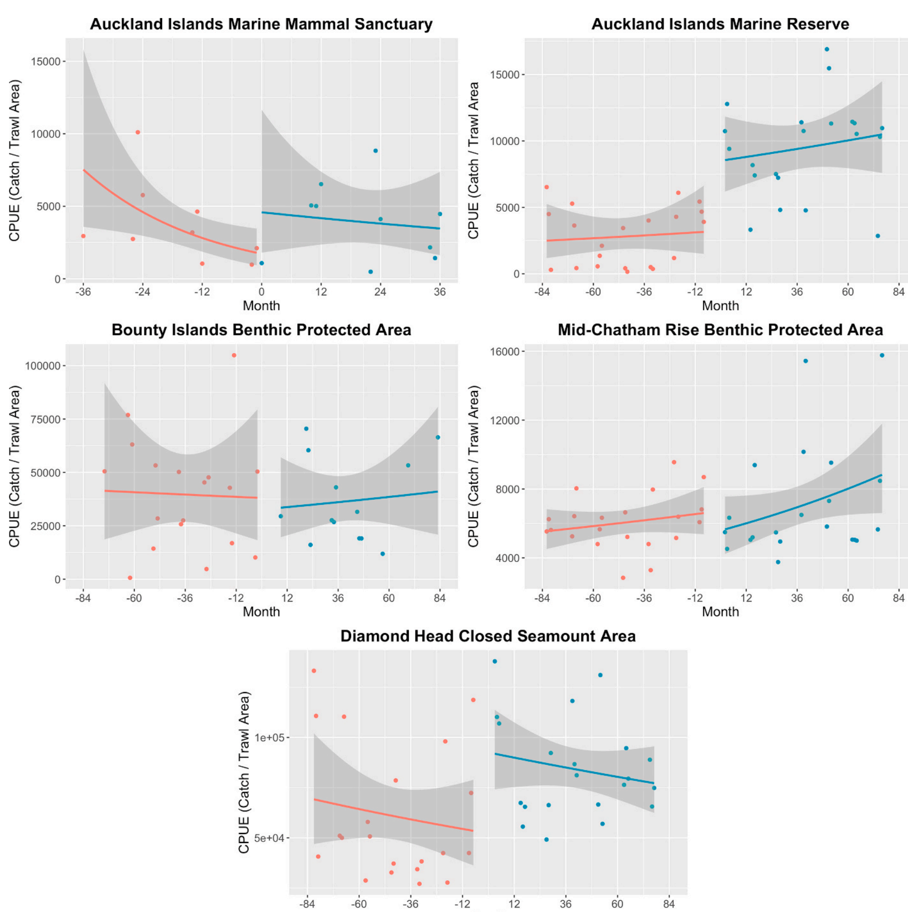

Fig. 4. Catch per unit effort (kg/km^2) aggregated monthly for peak fishing seasons. The trend lines and confidence intervals were generated from the model specified in equation (1).

exhibited a negative relationship before protection (i.e., total catch decreased the further from the MPA the trawl event was) which was followed by a positive relationship while the Auckland Islands Marine Mammal Sanctuary exhibited a positive relationship followed by a negative one. The Auckland Islands Marine Reserve and Bounty Islands BPA both had positive relationships between catch and distance from the MPA prior to protection followed by a decrease, but still positive relationship. The Mid-Chatham Rise also exhibited a positive relationship beforehand, which was followed by a negative relationship after protection that was of the same magnitude (i.e. there was no relationship between total catch and distance after MPA implementation).

Contrary to the expectation that MPA implementation should have no discontinuous effect on catch per km^2 trawled, the Auckland Islands Marine Mammal Sanctuary, Auckland Islands Marine Reserve, Bounty Islands BPA and Diamond Head CSA yielded statistically significant increases in CPUE at the time of MPA implementation (Table S11-Table S15), while the Mid-Chatham Rise exhibited a statistically significant discontinuous decrease in CPUE.

In controlling for abiotic factors, seasonality, and vessel fixed effects that influence CPUE, the temporal trends for the individual trawl events provide more informative results in comparison to the temporal trends in CPUE for the aggregate models. When these factors are included, the Auckland Islands Marine Mammal Sanctuary, Auckland Islands Marine Reserve and Mid-Chatham Rise exhibited statistically significant changes in CPUE over time after MPA implementation (Table S11-Table S15). Of these MPAs, only the Auckland Islands Marine Mammal Sanctuary's relationship between CPUE and time resulted in a change in sign, going from a positive to a negative trend. The Auckland Islands Marine Reserve exhibited a negative relationship which became less negative after MPA implementation while the Mid-Chatham Rise had a positive relationship which decreased after protection. Both the Bounty Islands BPA and Diamond Head CSA did not exhibit statistically significant temporal trends after treatment. In both cases the relationships with CPUE were negative before MPA implementation.

4. Discussion

For the first time, we have investigated the drivers of changes in fishing effort over time near five offshore MPAs in New Zealand. We found that, despite reductions in fishing area of 1–41% within 50 km of the MPA boundaries, total catch increased in aggregate across all areas of study ranging from 35 to 640%. This suggests that the Jevons Paradox occurred, where the rebound effect is greater than one i.e., biomass extraction increased despite the reductions in permitted fishable area. The individual trawl event model findings reinforce those found in the aggregate models for total catch, demonstrating that in most of the MPAs, biomass extraction increased immediately after MPA implementation both in aggregate and at the individual event level. In a similar analysis of the spiny lobster (*Panulirus interruptus*) fishery in Southern California, Lenihan et al. (2021) observed similar patterns in fishing effort and total catch. In contrast, our findings attribute increased catch to fisher behaviours rather than increased productivity, suggesting that fishery specific factors should be accounted for by environmental managers when considering MPA implementation.

There is the possibility that commercial vessels changed their fishing habits coincidentally with the MPA implementation and that the changes in catch and CPUE were unrelated. For example, fishers may have spatially relocated to inherently more productive areas, within the area, that have similar depth and temperature profiles which could explain a discontinuity in the outcome variables. Another possibility could be that other biological factors that affect fish populations changed discontinuously, resulting in a spike in population around the implementation date. Commercial species such as arrow squid that have short lifespans are known to have wide interannual population size variability, responding quickly to changes in environmental conditions (Waluda et al. 2004; Rodhouse et al., 2014; Thiaw et al., 2017). Thus, a sudden change in environmental conditions could explain a jump at the time of treatment. However, the spatial distribution of catch before and after MPA implementation shows that fishers did not relocate to new areas upon MPA designation, rather, this increase in catch was concentrated in areas that were previously fished. Additionally, the fact that these discontinuous increases in total catch were observed across all five study sites which were implemented at various times spanning from 1993 to 2007 suggest that it is unlikely environmental factors are responsible for these findings.

The CPUE results for the individual trawl event models suggest that spillover and recruitment effects are unlikely to explain these discontinuities in catch across the sites and timeframe of our study, with the potential exceptions of the Auckland Islands Marine Reserve and the Mid-Chatham Rise BPA. The lack of a clear and consistent relationship between CPUE and distance from the MPAs may confirm that spillover and recruitment cannot be observed at this scale. However, even if these phenomena were to occur at these distances, the fact that the protected areas did not have extensive trawl fishing activity within them before implementation means that spillover and recruitment effects were unlikely to occur regardless. This observation also highlights that these discontinuous increases in catch cannot be explained through displacement of fishing from with the MPA boundaries.

The change in the number of monthly trawl events after MPA designation suggest a behavioural component to the discontinuities in catch. Upon hearing the news that an MPA has been established, fishers may have increased trawling intensity in the surrounding area. This could be a consequence of the reputation MPAs have in exhibiting larger, more abundant fish or because the MPA locations were chosen because they were rich in biodiversity (Halpern and Warner 2002; Gell and Roberts 2003). Studies have shown that fishers respond to incentives in terms of both changing conditions, such as the decline of certain fish stocks, and to regulatory changes, in order to maintain profitability (Stephenson et al., 2018). Moreover, McDermott et al. (2019) have documented what they term the "Blue Paradox," where fishers pre-emptively fish in areas that are designated to be an MPA. While Fig. 3 does not appear to show any extensive pre-emptive fishing before these MPAs were established, it is clear that fishing habits have changed afterwards.

Analysis of the fishing vessels data revealed that, immediately after an MPA is announced, more vessels fish near MPAs with the idea that fishing opportunities may be greater than elsewhere. Moreover, the initial arrival of new vessels may also lead the legacy vessels to increase their fishing effort in response to the increased competition (Geer et al., 2013; Smith and Wilen 2003). However, after one to two fishing seasons, the new vessels cease operations in the area, reducing the total vessel count. This might offer an alternative explanation for the increasing temporal trends in CPUE after protection also, as the three MPAs that exhibit increasing trends in CPUE (the Auckland Islands Marine Reserve, Bounty Islands BPA, and Mid Chatham Rise BPA) all see the number of events and vessels decrease to pre-MPA levels after an initial jump. Thus, the increase in CPUE could be a result of biomass recovery after extensive trawling activity immediately after MPA implementation.

The change in trawl intensities (the number of trawl events per unique vessel before and after protection) also appear to be consistent with the aggregate and individual trawl event results for each MPA. The Bounty Islands BPA did not exhibit a significant change in the number of trawls per vessel before and after MPA implementation, yet the aggregate results for total catch depict a discontinuous increase. The substantial discontinuous increase in total catch at the individual event level appears to explain these discrepancies. On the other hand, Diamond Head CSA exhibited nearly a four-fold increase in the number of trawls per vessel along with a significant discontinuous increase in aggregate total catch, while catch at the individual trawl event level did not significantly change.

The findings for the Auckland Islands Marine Reserve in particular support the notion that the changes in total catch are due to fisher behaviour. This MPA exhibited some of the strongest discontinuities of the five MPAs, yet the implementation of the Marine Reserve conferred no additional benefits to the Auckland Islands' ecosystems given that it had already been a Marine Mammal Sanctuary for ten years. Thus, it seems unlikely that the Marine Reserve would have any additional effects on biodiversity both within and outside the reserve boundary. Consequently, the observed changes in total catch, CPUE and number of trawl events may have been influenced by the reputation Marine Reserves have for exhibiting better biological outcomes in comparison to other forms of protection.

5. Conclusions

Our results have some important implications for policy and planning linked to MPAs and for future research to underpin successful conservation. While we find that rebound effects can be severe, the reasons underlying them are complex as they are primarily related to fisher behaviour. This suggests that, in order to get a clearer picture of the commercial impacts of MPA implementation, and thus to design policy that will support the aims of the MPA, understanding the links between changes in fisher behaviour and equipment, monetary values, individual species abundances and biomass, and effects on other ecosystem components is required.

Policies supporting MPAs should require an explicit statement of underlying aims, such as the reduction in total fish catch or an individual species, the provision of refuge or spillover, or to decrease in damage to the biodiversity of the seafloor. Our results suggest that understanding these aims are critical for determining the focus of data collection and analysis to monitor the success of MPA policies.

Little to no trawl fishing activity occurred within the MPA boundaries prior to their implementation across all five sites of study, yet fishing activity and total biomass extraction increased in the wider area surrounding these MPAs after designation. While the long-term impacts of these fishing behavioural changes are unknown, this study provides evidence of an unintended and previously unreported consequence of MPA implementation that requires more scrutiny. Our findings suggest that accounting for the behavioural responses of commercial fishers is an important consideration when designing effective MPAs.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data that has been used is confidential.

Acknowledgements

We extend our gratitude to New Zealand's Ministry for Primary Industries for providing the commercial fishing data. This work was developed as an undergraduate economics honours thesis supervised by Professor Clair Brown at the University of California, Berkeley. Funding for further analysis and writing of this manuscript was provided by New Zealand's Sustainable Seas National Science Challenge project 3.2: "Communicating Risk and Uncertainty to Aid Decision Making"; Ministry of Business, Innovation and Employment (MBIE) grant number: C01X1901.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ocecoaman.2023.106595.

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