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How did a network of marine protected areas impact adjacent fisheries? Evidence from Australia

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Marine-protected areas (MPAs) are an effective means of improving habitat quality and biodiversity in the world's oceans. While the advantages of MPAs as a mechanism for conservation and biodiversity are well established, the potential improvements to fishery performance resulting from a network of MPAs are still being established. Countries around the world have committed to establishing networks of MPAs within their waters by 2020, in response to the United Nations Convention on Biological Diversity. This, coupled with the increasing global demand for seafood and heavy reliance on fishery resources as a source of economic development for many coastal communities, means that an understanding of how these networks can be expected to impact fishery performance is extremely important. We use a difference-in-difference modelling approach to isolate the change in the fishery performance associated with the south-east marine reserve network in Australia. We find no evidence that the economic performance of adjacent fisheries was negatively impacted by the network. This lack of impact is likely due to a network design explicitly intended to avoid effort displacement in key fisheries, along with fishery management changes intended to remove excess fishing capacity.

Key words: difference-in-differences modelling, fisheries management, marine-protected area network, south-east marine reserve network, Southern and Eastern Scalefish and Shark Fishery.

1. Introduction

Marine-protected areas (MPAs) are increasingly established globally in an effort to improve habitat quality and biodiversity in the world's oceans (Lester *et al*. 2009; O'Leary *et al*. 2016). While the foremost reason for establishing an MPA is the achievement of biological and conservation outcomes (Agardy, di Sciara, and Christie 2011; Wells *et al*. 2016), MPAs are now also expected to generate benefits to various marine resource users, including fisheries (Watson *et al*. 2014). Initially, promotion of MPAs as a means to achieve improvements for both conservation and fishery outcomes predominantly focused on isolated no-take MPAs in which no extractive activity was permitted (Pauly *et al*. 2002; Halpern, Lester, and McLeod

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2010; Caveen *et al.* 2015). The benefits to fisheries stemming from these isolated no-take MPAs is unclear (Hilborn *et al.* 2004; Sale *et al.* 2005; Caveen *et al.* 2015). While some studies demonstrate increased or unchanged catch rates after a no-take MPA is established (Kerwath *et al.* 2013), others suggest that, while the creation of no-take MPAs may lead to long-term conservation and fishery improvements, there is potential for negative short-term socio-economic consequences to fisheries (Oyafuso, Leung, and Franklin 2019).[1]

The desire to minimise the trade-offs between the potentially conflicting goals of conservation improvements and fishery benefits has led to debate over how to design and implement protected areas which achieve conservation outcomes, while at the same time generate benefits for fishery resource users (Sala *et al.* 2002; Charles and Wilson 2009; Gaines *et al.* 2010; Agardy, di Sciara, and Christie 2011; Krueck *et al.* 2017). The concept of MPA networks arose in part from this desire to avoid the trade-offs that may occur when implementing isolated, no-take MPAs.[2] MPA networks are comprised of collections of individual MPAs capable of operating synergistically, at various spatial scales and with a range of protection levels designed to achieve objectives that a single MPA cannot (IUCN-WCPA 2008). These networks have the potential to function collectively to facilitate both ecosystem and fishery improvements above those which might be expected from an isolated MPA (Roberts *et al.* 2001; Gaines, Gaylord, and Largier 2003; Gaines *et al.* 2010; Ballantine 2014; Horigue *et al.* 2015; Roberts, Valkan, and Cook 2018). Recognition that MPA networks may more effectively and efficiently achieve conservation and fishery objectives than isolated no-take MPAs has been reflected in global agreements such as the Convention on Biological Diversity (CBD). In 2011, the CBD set the Aichi Biodiversity Targets, which aim for 10% of coastal and marine areas to be placed within networks of protected areas by 2020 (CBD 2011). The target also expressly includes the need to reconcile conservation of the environment with maintaining the benefits from ecosystem services (Spalding *et al.*2013), of which fisheries comprise a major part (Caveen *et al.* 2015). In response, there has been a global increase in the number of MPA networks, with countries including Australia and the United States implementing large-scale MPA networks within their jurisdictions.

[1] These consequences may include fishing effort displacement (Horta e Costa *et al.* 2013), increased fishing costs (Hannesson 1998) and short-term reduction in catch (Hilborn, Micheli, and De Leo 2006). See also the extensive bioeconomic literature (Holland and Brazee 1996; Smith and Wilen 2003; Grafton, Kompas, and Lindenmayer 2005) exploring the impacts of no-take marine-protected areas on fisheries.

[2] MPA networks are also valuable purely as a conservation tool, as they promote biological connectivity and improved resilience to natural disasters and climate change (IUCN-WCPA 2008).

There is a commensurate increase in the amount of research discussing the design of MPA networks to achieve improved conservation and fishery outcomes (Almany *et al.* 2009; Rassweiler, Costello, and Siegel 2012; Rassweiler *et al.* 2014; Roberts, Valkan, and Cook 2018; Smith and Anna, 2018; Rassweiler, Ojea, and Costello 2020). However, the empirical literature evaluating the *ex post* impacts of an MPA network comprised of MPAs of varying levels of protection on fisheries is relatively scarce. Previous studies examining the effects of MPA networks for fisheries confine their examination to: networks of no-take MPAs (Gell and Roberts 2003; Hopf *et al.* 2016); specific species (Williams *et al.* 2009; Harrison *et al.* 2012); fisher responses and perceptions regarding MPA networks (Arias *et al.* 2015; Cabral *et al.* 2017; Ordoñez-Gauger *et al.* 2018; Ayer *et al.* 2018); or ex ante evaluation of the potential long- and short-term consequences of a network for fisheries (White *et al.* 2013). One exception is Reimer and Haynie (2018) who provided an *ex post* evaluation of the economic impacts on adjacent fisheries of MPAs, the design of which consists of various protective measures for the conservation of Steller sea lions in the North Pacific Ocean. Another exception is Lynham *et al.*(2020) who undertake an *ex post* evaluation of catch rate impacts in the Hawaiian longline fishing fleet resulting from expansions to U.S. National Monuments in the Pacific Ocean.

The aim of this paper was to extend the above literature by assessing the effect of an MPA network on adjacent fisheries, taking Australia's south-east marine reserve network (SEMRN) as a case study. This network was established in 2007 in the Commonwealth-managed waters around Tasmania, South Australia, Victoria and New South Wales, in waters shared by several Commonwealth-managed fisheries, including the highly valuable Southern and Eastern Scalefish and Shark Fishery (SESSF). The network was primarily established for habitat conservation, while also forming part of a package of management changes in Commonwealth-managed fisheries. The SEMRN was designed explicitly to avoid negative impacts on nearby fisheries (Buxton, Haddon, and Bradshaw 2006), and so beyond the general interest in MPA–fishery interactions, there is specific interest in discovering whether the design of this network was successful in its intention.

We use a panel of data comprised of three sectors and 28 species in the SESSF with a time series of 12 years (2003–2015). We apply a difference-in-differences (DiD) approach to isolate the effect this network had on the performance of adjacent fisheries. The geographical location of this network means there exists a treatment and control group of species, making the SEMRN an appropriate and useful case study for exploring potential fishery impacts of an MPA network. We take as our performance indicators the catch and gross value of production (GVP) of these species. These metrics, often driven by key commercial species within fisheries, are a common measure of fishery performance in

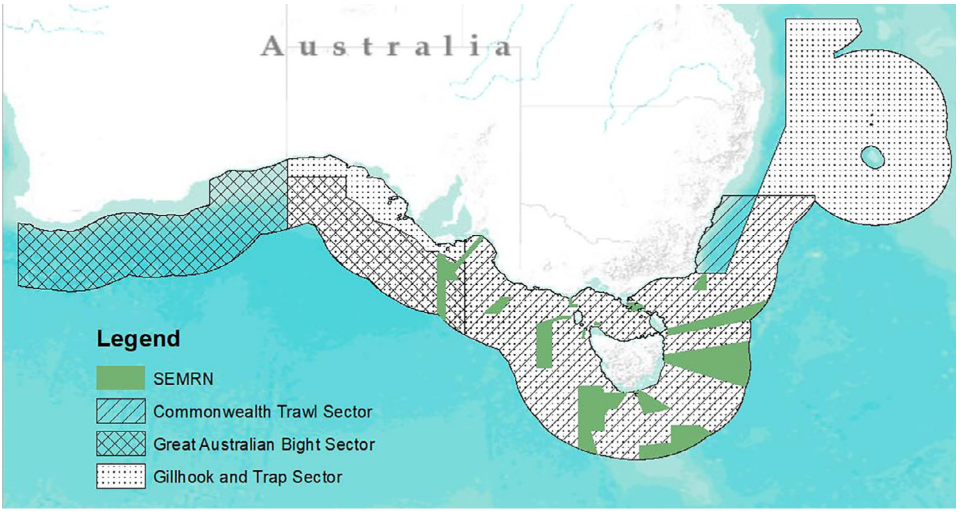

Figure 1 The Southern and Eastern Scalefish and Shark Fishery (SESSF) sectors and the boundaries of the south-east marine reserve network (SEMRN).[1313]Drawn using ArcMap 10.7 with GIS shapefiles sourced from the Commonwealth Fisheries 2006 dataset (https://data.gov.au/data/dataset/commonwealth-fisheries-2006) and the Collaborative Australian Protected Areas Database (CAPAD) 2016 (http://www.environment.gov.au/fed/catalog/search/resource/details.page?uuid=%7B57645456-C5D3-483C-89F4-51A0EC6070EC%7D). [Colour figure can be viewed at wileyonlinelibrary.com]

Australian fisheries (Flood *et al.* 2016) and so provide a convenient measure of fishery performance in this context.

2. Background

2.1 The Southern and Eastern Scalefish and Shark Fishery (SESSF) and the south-east marine reserve network (SEMRN)

The SESSF is a multispecies, multisector fishery located off the southern coast of Australia (Figure 1). It is the largest Commonwealth-managed fishery both in area and in volume of catch, with almost 17,000 tonnes of fish harvested from an area almost half the size of the Australian Fishing Zone (AFZ) in 2016/17 (Mobsby 2018).[3] The three sectors of the SESSF relevant here – the Commonwealth Trawl sector, the Gillnet, Hook and Trap sector and the Great Australian Bight sector – are managed primarily by setting total allowable catches (TACs), with supplementary management including seasonal and spatial closures, individual transferable quotas, gear restrictions, limited entry and monitoring requirements.

[3] This does not include catch from the East Coast Deepwater Trawl sector, which is unreported due to confidentiality requirements.

In late 2005, due to concerns over unsustainability and unprofitability in Commonwealth-managed fisheries, three major initiatives were announced: (1) a statutory direction from the Minister of Fisheries to the Australian Fisheries Management Authority (AFMA) to recover overfished stocks and prevent future overfishing (the introduction of a harvest strategy policy being a core aspect of this direction), (2) a structural adjustment package to remove excess effort from specific fisheries and (3) the introduction of an MPA network in south-east Australia (Rayns 2007). While the harvest strategy policy and structural adjustment package were intended as fisheries management reform, which is the responsibility of the AFMA, the MPA network was established and administered under the Environment Protection and Biodiversity Conservation (EPBC) Act 1999 and is the responsibility of the Director of National Parks.

The SEMRN, as originally proposed by the Australian government in December 2005, was to cover approximately 170,000 square kilometres of ocean in the south-east region off the coasts of Tasmania, South Australia, Victoria and New South Wales (Figure 1). This network was the first step in creating the National Representative System of Marine Protected Areas (NRSMPA) in the Commonwealth marine jurisdiction, which the Australian government had committed to establishing by 2012 as part of their commitment to the CBD. The goal of the NRSMPA has been to contribute to the conservation and maintenance of marine ecosystems and biodiversity in the long term, while minimising any adverse impacts on marine users, both commercial and recreational (ANZECC-TFMPA 1998). After the SEMRN was announced, a study was commissioned to investigate the anticipated impact of the SEMRN to adjacent fisheries and the socio-economic consequences to fishing communities (Buxton, Haddon, and Bradshaw 2006). This study found that the combined effect of the MPAs had potentially high socio-economic consequences for rural fishing communities, due to effort displacement in the highly valuable blue-eye trevalla, blue grenadier, scallop and tiger flathead fisheries, among others. The study also found that the SEMRN could be redesigned to avoid these impacts, without compromising the conservation goals of the NRSMPA. After boundary changes and re-zoning of certain MPAs, the SEMRN was proclaimed in June 2007. This redesigned network consists of fourteen MPAs with varying levels of protection, covering approximately 388,464 square kilometres, or 23.7% of the south-east marine region.

Although the management plan for this network did not come into effect until 2013, management of these MPAs in the interim was in accordance with the EBPC Act.[4] The management plan made no changes to the design or categorisation of the MPAs but formalised the monitoring and enforcement of the network.

[4] See *Environment Protection and Biodiversity Conservation Regulations 2000*, Schedule 8.

3. Methods

3.1 Data

Data on the catch (in tonnes) and GVP (in 000's nominal AUD) for Commonwealth fisheries are drawn from the Australian fisheries statistics reports for 2012 (Skirtun, Sahlqvist, and Vieira 2013) and 2015 (Savage 2015). These reports provide data for eight Commonwealth fisheries and thirty-nine species, for the financial years 2003/04 to 2014/15.[5] In this paper, we focus on 28 species from three sectors of the SESSF; the Commonwealth Trawl sector, the Gillnet, Hook and Trap sector, and the Great Australian Bight sector (see Appendix S1). Evaluating the impact of the SEMRN requires that species be assigned into treatment and control groups, where the treatment group is expected to have been affected by the introduction of the SEMRN, while the control group is not. We use information regarding the boundaries of the SEMRN (CAPAD 2008) and fishing effort location data sourced from the Australia Bureau of Agricultural and Resource Economics and Sciences (ABARES)[6] to place species within treatment and control groups.

Figure 2 (a) shows fishing effort location[7] for the Commonwealth Trawl and Gillnet, Hook and Trap sectors in 2006, the year prior to the establishment of the SEMRN (in grey). Due to confidentiality requirements, we are unable to access fishing effort location data for the Great Australian Bight sector. However, the area marked with an oval shows the approximate area where fishing effort in the Great Australian Bight sector was concentrated in 2006 (Morison 2007). Prior to the SEMRN, fishing effort in the Commonwealth trawl and Gillnet, Hook and Trap sectors was clustered around the coasts of Victoria and Tasmania, while fishing effort in the Great Australian Bight sector was isolated between the coasts of South Australia and Western Australia. Moreover, the Great Australian Bight sector is mostly a deepwater trawl fishery and does not overlap with other sectors either technically or biologically (Pascoe *et al.* 2020). Based on the geographical proximity of the SEMRN and the Commonwealth Trawl and Gillnet, Hook and Trap sectors, we take these two sectors and the species caught within as our treatment group of species, and the species caught within the Great Australian Bight sector as our control group (see Appendix S1).

Figure 2 (b) shows similar fishing effort location for the three sectors in 2015, post-SEMRN (Patterson *et al.* 2016). This figure demonstrates that the extent of any fishing effort migration is not such that treatment group became control group, or vice versa. Given the spatial scale of the fishery and SEMRN,

[5] Data available on request from the authors.

[6] Data on relative fishing intensity and the total area fished were supplied through private correspondence by ABARES and were derived from fishery logbook data supplied by AFMA.

[7] Unfortunately, the data do not allow us to distinguish between gear types or species, meaning that it is not possible to identify which gear types are catching which species in which area. The data are therefore used only to place sectors (and by extension, species) within the treatment or control group.

(a) 2006, pre -SEMRN

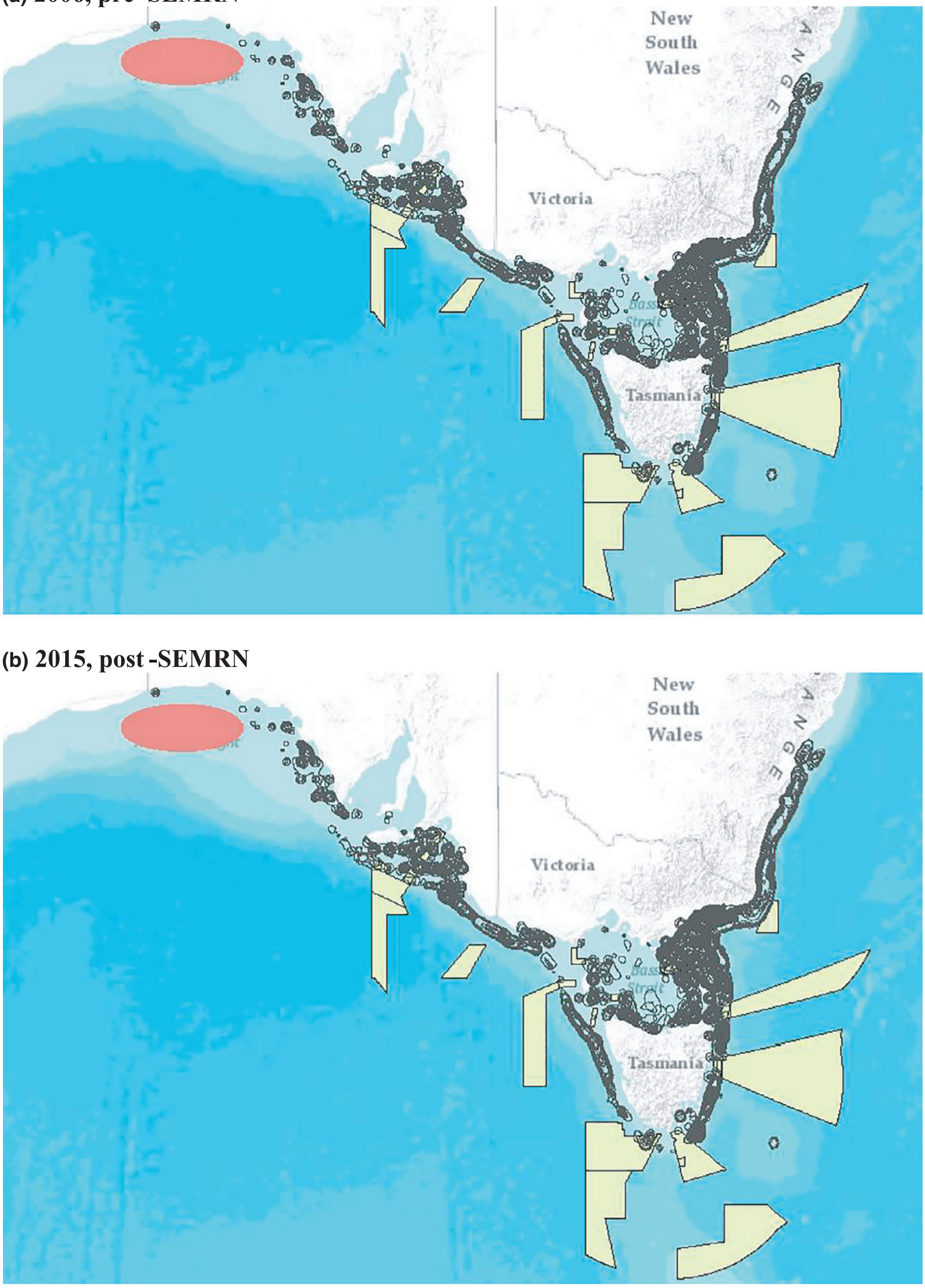

Figure 2 Fishing effort location in 2006 (a) and 2015 (b). Fishing effort location for the treatment group (the Commonwealth Trawl and Gillnet, Hook and Trap sectors) is shown in grey circles. Fishing effort location for the control group (the Great Australian Bight sector) is concentrated in the pink shaded area. [Colour figure can be viewed at wileyonlinelibrary.com]

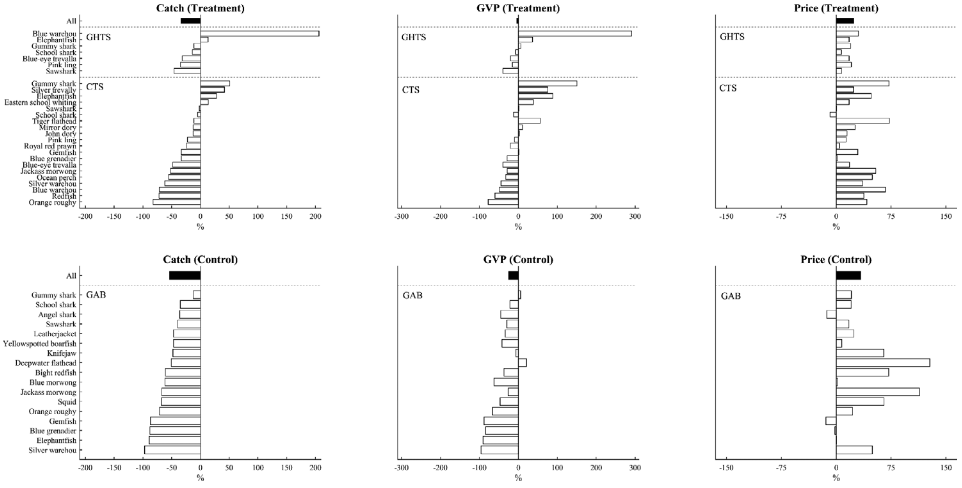

Figure 3 Pre- and post-treatment change in annual catch, GVP and price for each species in the treatment and control group. CTS = Commonwealth Trawl Sector; GHTS = Gillnet, Hook and Trap Sector; GAB = Great Australian Bight Trawl Sector.

the consistency in sector boundaries and the lack of large-scale fishing effort migration over time, this method of determining treatment and control groups was deemed appropriate for the purposes of this analysis. The time series is split into pre- and post-SEMRN periods at the 2008/09 financial year, reflecting the fact that, while the SEMRN was proclaimed in June 2007, it did not commence until August 2007.[8] As such, the 2008/09 financial year is the first in the time series entirely subsequent to the SEMRN proclamation. Sensitivity of results to this choice of post-SEMRN period is given in Section 4.2.

3.2 Data description

Using the data on the catch (in tonnes) and GVP (in 000's nominal AUD) for Commonwealth fisheries, we create a species-level price variable (in nominal AUD per kg) by dividing GVP by catch to enable examination of any price effects on GVP during the time series. Figure 3 shows the percentage change in average catch, GVP and price between the pre- and post-SEMRN periods for each species in the treatment and control group. Of the twenty-seven species in the treatment group, only six species experienced an increase in average catch in the post-SEMRN period, with gummy shark and silver trevally in CTS increasing by over 40%. Certain species also experienced more severe declines in catch than others. Species including blue-eye trevalla, sawshark, jackass morwong and silver warehou all suffered a greater than 40% decline in catch in the post-SEMRN period, with orange roughy experiencing an over 80% decline in catch.

[8] Proclamation of Apollo Marine Park, https://www.legislation.gov.au/Details/F2017C00985, retrieved 11 February 2020.

A similar pattern is found for GVP, although not as extreme. While many species also experienced a decline in GVP in the post-SEMRN period, twelve species experienced an increase. For example, the GVP of gummy shark in CTS increased by 150%, silver trevally increased by 75%, and tiger flathead increased by 56%. While the average decline in catch for the treatment group overall was approximately 33%, the decrease in GVP was only 3%. This was because price increases for certain species offset the declines in catch, such that the decline in GVP was not so pronounced. In the treatment group of species, only one species experienced a price decrease in the post-SEMRN period, while species including gummy shark (CTS), tiger flathead and blue warehou (CTS) increased in price by over 65%. The average price increase for the treatment group was approximately 23%.

All seventeen species in the control group declined in catch, with gemfish, blue grenadier, elephantfish and silver warehou experiencing greater than 80% declines in catch. On average, catch declined by 53% in the control group. Again, there is a similar but slightly less extreme pattern found for GVP, where no species experienced increases in catch in the post-SEMRN period, two species experienced increases in GVP in the post-SEMRN period, including deepwater flathead (21%). Like the treatment group of species, only three species experienced price declines after the SEMRN was established, with deepwater flathead increasing in price by 128%. Figure 3 reveals similar trends in catch, GVP and price between the treatment and control group pre- and post-SEMRN, and the differences in magnitude between groups do not appear to be major. Comparing the changes between groups in this way is not enough to determine causal impacts from the SEMRN due to confounding factors, and so we more formally estimate an average treatment effect using the DiD methodology below.

3.3 Estimating the average treatment effect

The DiD approach finds the impact of a policy change by analysing differences in a treatment group, where the treatment group is expected to have been affected by the policy, before and after the policy change, along with differences in a control group at matching times.[9] We take catch and GVP as the outcome variables to assess the potential impact of the SEMRN for the treatment group, which corresponds to an econometric test of whether the policy treatment causes a change in these variables relative to the change in the control group within the same time frame. Given Figure 3, which seems to

[9] The DiD approach has diverse application in economics, with the approach used to analyse policy changes in areas such as education (Card and Krueger 1992) and health (Wing, Simon, and Bello-Gomez 2018). The before–after control–impact (BACI) method is an alternative methodology commonly used in the ecology literature (Sciberras *et al.* 2013; Kerr *et al.* 2019). Both BACI and DiD are a quasi-experimental research design that is considered one of the most rigorous ways to isolate the effect of policy changes or other human and environmental perturbations when treatment and control groups cannot be established through randomization (Kerr *et al.* 2019; Angrist and Pischke 2009). .

suggest price movements during the time series had some compensatory effect on GVP, we also include price as an outcome variable in these analyses, even though price is not generally considered a performance indicator for fisheries.

To find the average treatment effect for the treatment group, we first estimate a pure DiD model shown by equation (1) below:

$$\begin{aligned} Y_{i,s,t} = {} & \alpha + \beta_1 Post2007_t + \beta_2 Treatment_{i,s} + \beta_3 Post2007_t \times Treatment_{i,s} \\ & + \theta_t + \sigma_i + d_s + \varepsilon_{i,s,t} \end{aligned} \tag{1}$$

where $Y_{i,s,t}$ is the catch, GVP or price for sector i, species s and year t. $Post2007_t$ and $Treatment_{i,s}$ are dummy variables equal to one indicating an observation occurs after the SEMRN was implemented and that it is a treatment species, respectively. $Post2007_t \times Treatment_{i,s}$ is an interaction variable, the coefficient of which (β_3) gives the average treatment effect per species per year in the post-SEMRN period (i.e. 'DiD estimator'). More precisely, the DiD estimator measures the average treatment effect as:

$$\begin{aligned} \beta_3 = {} & \underbrace{\{E[Y_{i,s,t}|i = treated, t = post - SEMRN] - E[Y_{i,s,t}|i = treated, t = pre - SEMRN]\}}_{Average\ change\ in\ Y_{i,s,t}\ in\ the\ treatment\ group} \\ & - \underbrace{\{E[Y_{i,s,t}|i = control, t = post - SEMRN] - E[Y_{i,s,t}|i = control, t = pre - SEMRN]\}}_{Average\ change\ in\ Y_{i,s,t}\ in\ the\ control\ group} \end{aligned}$$

This way, the potential bias in the treatment effect estimate due to the time effect that is unrelated to the introduction of the SEMRN is removed (Angrist and Pischke 2009). In equation (1), year fixed effects (θ_t), sector fixed effects (σ_i) and species fixed effects (δ_s) further control for all time-invariant factors that affect both treatment and control groups. Factors controlled for by these fixed effects include time trends related to unobserved factors (year fixed effects), differences in efficiency of fishing gear used or in scale between sectors (sector fixed effects) and physiological differences between species which remain constant over time (species fixed effects).

Given the pronounced declines in catch and GVP for commercially important species such as blue grenadier and blue-eye trevalla, which are larger than the overall declines for the treatment group, we anticipate the potential for heterogeneity in treatment effects between commercially important and less important species in the treatment group (i.e. 'heterogeneous treatment effects'). To explore this potential, we first identify commercially important species based on the proportion of the value contributed by each species to the total GVP of each sector[10] (see

[10] The SESSF currently uses a definition of 'commercially valued' species which encompasses primary and secondary species (Knuckey *et al.* 2017), with percentage contribution ($\geq$1.7%) to GVP being the primary means of distinguishing between these primary and secondary species. In this paper, we use a stricter definition of 'commercially important' species, although the means of determining these species is consistent with SESSF management.

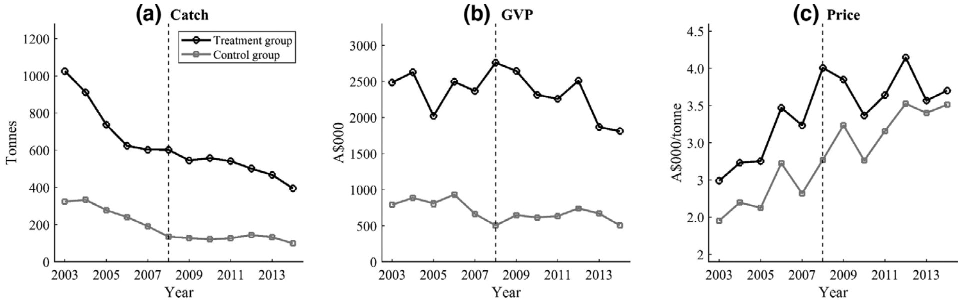

Figure 4 Average catch, GVP and price for the treatment and control group, 2003–2014.

Appendix S2), and then, we use a modified DiD model as shown in equation (2) below:

$$Y_{i,s,t} = \alpha + \beta_1 Post2007_t + \beta_2 Treatment_{i,s} + \beta_3 Post2007_t \times Treatment_{i,s} \\ + \beta_4 CI_{i,s} + \beta_5 Post2007_t \times Treatment_{i,s} \times CI_{i,s} + \theta_t + \sigma_i + \delta_s + \varepsilon_{i,s,t} \tag{2}$$

where $CI_{i,s}$ is a dummy variable equal to one if a treatment species is considered commercially important. The DiD estimator, β_3, is as in equation (1) and gives the average treatment effect for the less important species post-SEMRN. The average treatment effect for those commercially important species in the treatment group is measured by $\beta_3 + \beta_5$, with the coefficient of the commercially important interaction variable, β_5, giving the additional treatment effect for the species deemed commercially important. Using the estimated parameters in equations (1) and (2), we test two hypotheses. First, we test a null hypothesis that the DiD estimator for this model, given by β_3, is zero. The null hypothesis, that there has been no effect due to the SEMRN, is unchanged regardless of whether we account for possible heterogeneity in treatment effects within the treatment group (equation 2) or not (equation 1). The second hypothesis is that the coefficient on the CI interaction variable, β_5, is equal to zero, such that there is no heterogeneity in treatment effects.

To interpret these parameters as the causal effect of MPA networks, the DiD approach requires two assumptions: (1) the dependent variable, $Y_{i,s,t}$, follows a common time trend (conditional on the covariates that may lead to different time trends) across the treatment and control groups and (2) the composition of the treatment and control groups does not change after the implementation of MPA networks (Cameron and Trivedi 2005). The first assumption is required so that a deviation of the dependent variable from the trend is attributed to the effects of MPA networks. The second assumption rules out that the composition of the treatment and control groups is affected by the effects of MPA networks (see Section 3.1). Figure 4 shows the average catch, GVP and price for the treatment and control group before and after the introduction of the SEMRN.

Figure 4 broadly supports the assumption of a common time trend across the treatment and control groups, although there appears to be some divergence in catch and GVP across the time series. Other factors that may affect our estimate include management changes, the presence of treatment species that were subject to stock rebuilding plans and the choice of post-SEMRN year. We examine the sensitivity of our results to these assumptions in Section 4.2.

4. Results

4.1 Average effect of the SEMRN on the treatment group

We find no statistical evidence that the SEMRN impacted performance of fisheries adjacent to the network (Table 1). The coefficient on the DiD estimator indicates that in the sample, catch of the treatment group of species was 118 tonnes lower per species per year on average, compared to catch of the control group of species, but that this difference between groups is statistically insignificant. The same is true of the DiD estimator for GVP and price. The DiD estimator for GVP suggests that in the sample, the SEMRN had a positive impact of $146,000 on the gross value of production on average, but that this treatment effect is statistically insignificant. The coefficient on the price equation shows that in the sample, while the average price in the post-SEMRN period was higher than that in the pre-SEMRN period, the treatment effect of the SEMRN is negligible (approximately 3

Table 1 Effects of MPA networks on catch, gross value of production and price

	Dependent variable		
	Catch (tonnes)	GVP ($A000)	Price ($A/kg)
Post2007_t	−401**	−651	1.28****
	(177)	(476)	(0.22)
$\text{Treatment}_{i,s}$	895****	2197****	0.03
	(146)	(401)	(0.12)
$\text{Post2007}_t \times \text{Treatment}_{i,s}$	−119	147	−0.03
	(111)	(364)	(0.15)
Constant	354**	935	3.05****
	(180)	(614)	(0.24)
Fishery fixed effects	Yes	Yes	Yes
Species fixed effects	Yes	Yes	Yes
Year fixed effects	Yes	Yes	Yes
R-squared	0.59	0.61	0.86
Number of pooled observations	522	522	519

Note: This table reports the DiD estimation results of equation (1). Heteroskedasticity-robust standard errors in parentheses. Statistical significance at the **** 0.1% level, *** 1% level, ** 5% level and * 10% level. Subscripts t, i and s denote year, fishery and species, respectively. All models use the sample period 2003–2014.

cents per kg) and statistically insignificant. This suggests that price increases in the time series may be attributable to something other than the SEMRN.

The coefficient of the commercially important interaction variable, β_5, in equation (2) is statistically insignificant for catch and GVP (Table 2). This result suggests that there exists no heterogeneity in treatment effects; that is, the catch and GVP of commercially important species were not impacted by the SEMRN more or less than the effect observed for the less important group of species. However, this coefficient for price is positive and statistically significant, suggesting that the price of commercially important species increased by 37 cents per kg on average compared to less important species due to the SEMRN. We confirm the result for equation (1), where the SEMRN had no significant effect on catch or GVP. The DiD estimator (β_3) for catch decreases slightly compared to equation (1), while the DiD estimator for GVP decreases substantially. The coefficient on the dummy variable indicating a commercially important species (*CI*) is positive for both catch and GVP, suggesting that catch and GVP for commercially important species were 187 tonnes higher and $1,618,000 higher on average than less important species, although neither of these coefficients are statistically significant. The highly significant coefficient for price suggests the price of commercially important species were $1.93 per kg on average higher than less important species.

Table 2 Heterogeneous treatment effects on commercially important species

	Dependent variable		
	Catch (tonnes)	GVP ($A000)	Price ($A/kg)
Post2007_t	−385**	−773	1.23****
	(164)	(476)	(0.22)
$\text{Treatment}_{i,s}$	887****	2257****	0.05
	(143)	(387)	(0.12)
$\text{CI}_{i,s}$	187	1619	1.93****
	(256)	(1137)	(0.23)
$\text{Post2007}_t \times \text{Treatment}_{i,s}$	−106	47.5	−0.06
	(110)	(342)	(0.15)
$\text{Post2007}_t \times \text{Treatment}_{i,s} \times \text{CI}_{i,s}$	−126	990	0.37**
	(265)	(895)	(0.16)
Constant	345*	1004*	3.07****
	(177)	(605)	(0.24)
Fishery fixed effects	Yes	Yes	Yes
Species fixed effects	Yes	Yes	Yes
Year fixed effects	Yes	Yes	Yes
R-squared	0.59	0.61	0.86
Number of pooled observations	522	522	519

Note: This table reports the estimation results of equation (2). Heteroskedasticity-robust standard errors in parentheses. Statistical significance at the **** 0.1% level, *** 1% level, ** 5% level, and * 10% level. Subscripts *t*, *i* and *s* denote year, fishery and species, respectively. All models use the sample period 2003–2014.

4.2 Sensitivity analysis

Our baseline results above show that the establishment of the SEMRN had no meaningful impact on the performance of the adjacent SESSF sectors as measured by changes in catch and GVP. We now examine the sensitivity of the results in Table 1 to various model assumptions. First, the estimated treatment effects in equation (1) may be confounded by the incidence of fishery management changes occurring in the same time period. In November 2005, the Australian government announced the Securing Our Fishing Future initiatives, aimed at reducing the level of fishing effort and the development of a formal harvest strategy policy for Commonwealth fisheries. A key part of this package was a structural adjustment package, to allow fishers to voluntarily exit targeted fisheries and so reduce the overall level of fishing effort through a vessel buyback scheme (Vieira 2010). The other major change to Australian fishery management was the development of the Commonwealth Harvest Strategy Policy in 2007 (Smith *et al.* 2008; Smith *et al.* 2014). These management changes indicate an increase in the level of precaution used when managing Commonwealth fish stocks which may confound the effects of MPA networks (Punt 2006). All species in the sample were subject to these management changes to varying degrees, and so it is not possible to separate the effects of these management changes from those of MPA networks with the current sample.[11] To address the issue, the sample of fisheries is expanded to include a larger control group of fisheries. Specifically, the control group of fisheries for this regression are the Great Australian Bight sector of the SESSF, the Northern Prawn Fishery, and the Eastern and Western Tuna and Billfish Fisheries, while the treatment group of fisheries remains unchanged (see Appendix S3). Second, the DiD estimator relies on the assumption that the outcome variable follows a common time trend across the treatment and control groups. For example, the estimated treatment effect in equation (1) is not reliable if the pretreatment characteristics, which are associated with the treatment outcome, are different between the treatment and control groups or if the composition of the treatment and control group is affected by the treatment (Angrist and Pischke 2009). To account for a possible violation of these assumptions, we perform a sensitivity analysis using Abadie (2005)'s semiparametric DiD approach, which weighs the treatment outcome based on the propensity score (i.e. probability to be in the treatment group), to make the common trend assumption more credible.

Third, the treatment effects of MPA networks may be overestimated if the treatment group includes species which were subject to stock rebuilding plans and thus the catch was lower due to constraints on the total allowable catch independent of the SEMRN. To address the issue, we re-estimate equation (1) where those treatment and control species under rebuilding plans during the

[11] These may be found by including various interaction variables controlling for species and sector-specific effects in the regression equations, with the trade-off being a loss of statistical power in the analyses.

sample period (i.e. orange roughy, redfish, blue warehou, gemfish and school shark) are removed from the sample. The final sensitivity analysis is to test whether the baseline results are affected by the choice of the SEMRN time period. In this regression, we use the 2007/08, instead of 2008/09, financial year to split the sample into pre- and post-SEMRN periods given that the SEMRN was proclaimed in June 2007.

We find overall that the baseline result, that there is no significant treatment effect for catch or GVP resulting from the SEMRN, is robust to these re-estimations (Table 3). When our sample is expanded to control for fishery management changes, we observe a slightly increased negative impact on catch within the sample (a decrease of 158 tonnes on average for the treatment group) and a statistically insignificant and negative impact for GVP. The coefficient for price becomes positive and statistically significant. When rebuilding species are removed from the sample, we see similar treatment effects for catch, GVP and price compared to the baseline model. As one would expect, the DiD estimator for catch decreases slightly once these species under rebuilding plans are removed from the sample, but the coefficient remains statistically insignificant. When the network is considered to have been established in the 2007/08 financial year, not 2008/09, we find near identical results for catch, GVP and price, as in the baseline model. The only change to our baseline result comes when we re-estimate equation (1) using Abadie (2005)'s semiparametric DiD estimator. Here, we find a statistically significant decline in catch of 290 tonnes per species per year on average due to the SEMRN, but no statistically significant impact on GVP. We find a significant increase in price of $0.84 per species per year on average.

Table 3 Sensitivity of treatment effects

	Catch (tonnes)	GVP ($A000)	Price ($A/kg)	Number of observations
(1) Baseline (Table 1)	−119	147	−0.03	522
	(111)	(364)	(0.15)	
(2) Fishery management	−158	−103	1.01****	679
	(99)	(517)	(0.19)	
(3) Semiparametric DiD	−290**	109	0.84****	522
	(126)	(396)	(0.22)	
(4) Stock rebuilding plan	−98.1	375	−0.11	402
	(134)	(439)	(0.16)	
(5) Post-SEMRN year	−119	147	−0.03	522
	(111)	(363)	(0.15)	

Note:: This table reports the average treatment effects and heteroskedasticity-robust standard errors in parentheses. Statistical significance at the **** 0.1% level, *** 1% level, ** 5% level, and * 10% level. All models use the sample period 2003–2014. In (2), the control group is expanded, and dummy variables of the harvest strategy policy and vessel buyback schemes are included as explanatory variables. In (3), Abadie's (2005) semiparametric DiD estimator is used. In (4), the treatment and control species under rebuilding plans are removed from the sample. In (5), the post-SEMRN year is changed to the 2007/2008 financial year.

5. Discussion

MPA networks are being established worldwide in response to global commitments to conservation targets, such as the Aichi 11 target of the Convention on Biological Diversity. In addition to achieving conservation benefits, it is anticipated that these MPA networks will generate benefits for marine resource users. The increase in the number of MPA networks around the world means that a better understanding of how these networks can be expected to affect fishery performance is important. However, the literature on *ex post* estimation of the treatment effect of MPA networks on fishery performance is extremely scarce (Ferraro, Sanchirico, and Smith 2019). In this paper, we explore the impacts of MPA networks on adjacent fisheries, using Australia's SEMRN as a case study. We use a panel of data comprised of three sectors of the SESSF, 28 species and 12 years and apply a difference-in-differences modelling approach to isolate the effect of this network for adjacent fisheries.

We find no evidence that the SEMRN had any significant impact on the economic performance of sectors within the SESSF, as indicated by the catch and GVP within the sectors and the price of species. This conclusion is robust to both the removal of species subject to rebuilding plans and fishery management changes that reduced fishing effort and introduced formal harvest strategies. One exception to this result occurs where we examine the difference between control and treatment groups using a semiparametric estimator. This outcome, that the SEMRN had no significant impact on either catch or GVP in the SESSF, would tend to accord with the results from similar studies. Lynham *et al.* (2020), who undertook a difference-in-difference analysis of catch and catch per unit effort in Hawaiian fisheries after an expansion of existing MPAs in the region, found that these performance indicators had improved postexpansion. Similarly, Reimer and Haynie (2018) in their analysis of the North Pacific stellar sea lion closures, found little evidence of a decline in overall fishery performance resulting from those closures.

There are several reasons why we would not observe any economic impact, either positive or negative, resulting from the SEMRN. One reason is that the SEMRN was intended to be a tool for habitat conservation, not fisheries management and, so far as fisheries were considered in the design of the network, the intention was to avoid negatively impacting catch in adjacent fisheries (Buxton, Haddon, and Bradshaw 2006). This highlights the fact that, if benefits are to accrue to fisheries from an MPA network, the design of that network needs to explicitly aim to achieve both habitat conservation and improved fishery outcomes (Green *et al.* 2014). The network also does not appear to have led to significant fishing effort displacement, as shown by Figure 2 above. This was likely by design and suggests that fishers were not forced to adapt to the network by searching for new fishing grounds or fishing differently than they were before, and so may not have incurred increased search times or loss of fishing effort as a result of the network. Another reason may be that the conservation benefits anticipated from MPAs, including enhanced biomass and density within the boundaries of both multiuse and no-take MPAs (Sciberras *et al.* 2015) have not yet eventuated. This is perhaps reflected in the total allowable catch for species within the treatment group, which has stagnated in the post-2007 era. While the TAC for some species has increased, the overall TAC for species within the treatment group has been relatively stagnant, with quota latency also evident throughout the time period (Woodhams, Vieira, and Stobutzki 2013; Patterson *et al.* 2017). This suggests that increases in biomass, which should be reflected in increases in TAC, have not yet eventuated.

Our results may be influenced by the potential impacts of climate change in the region. The south-east of Australia is subject to an ocean warming 'hotspot' (Hobday *et al.* 2006; Hobday and Lough 2011; Hobday and Pecl 2014; Popova *et al.* 2016), in which ocean temperatures increase at a relatively faster rate in response to climate change than other regions (Pecl *et al.* 2014)[12]. There is evidence of increases in sea surface temperature and an increase in the southward range of the East Australian current (EAC) driven in part by climate change (Ridgway 2007), and biological responses to climate change are likely to have occurred prior to the establishment of the SEMRN (Hobday *et al.* 2006). While the impacts of ocean warming on individual species are highly variable (Harley *et al.* 2006), impacts for species in the south-east region have included changes in distribution or migration of species (Ridgway 2007; Ling *et al.* 2009; Last *et al.* 2011). Additional impacts may include changes in growth rates, reproductive output and an increased susceptibility to disease (Pecl *et al.* 2011), all of which may contribute to declines in catch (Brander 2007).

There are two ways in which climate change impacts may influence our estimate of the treatment effect. The first is that, if every species in the treatment group was negatively impacted by ocean warming after the establishment of the SEMRN, with the control group of species unaffected (or less affected) by ocean warming, this will be incorporated into the treatment effect, and so the treatment effect on catch attributable to the SEMRN will be biased downward. The second is that given the potential effects MPAs have in hedging against the effects of climate change (Roberts *et al.* 2017), it is possible that the decline in catch of species in the region would have been greater if not for the MPA network. That is, if every species in the treatment group was negatively impacted by ocean warming, but the SEMRN has acted as a buffer against these adverse impacts, the treatment effect will be understated. It is also possible that, if some species in the treatment group were negatively influenced by ocean warming, and others

[12] In Hobday and Pecl (2014), a warming threshold rate of 10% was identified, which equates to an increase in temperature at a rate of 1.48°C per 100 years.

positively influenced, then overall the effects of climate change may not have a significant confounding effect on our results.

Our results also show no heterogeneity in treatment effects for either catch or GVP between the treatment group overall and species deemed commercially important. There is no indication the species which were major contributors to GVP prior to the network were affected differently by the establishment of the SEMRN compared to less important species. In addition to this, the commercially important species in the treatment group identified in the year immediately prior to the network commencing remained key drivers of GVP in subsequent years (Appendix S2). This suggests that fishers did not adapt their targeting behaviour in response to the network to the extent of changing the key target species in the treatment fisheries. We do find a statistically significant increase in the price of these commercially important species in the post-SEMRN period which, coupled with the avoidance of any specific negative impact on the catch of these key species, may help to explain the lack of a significant treatment effect on GVP overall as a result of the SEMRN. This further suggests that changes in market conditions as demonstrated by the significant increase in price of those species in the post-SEMRN period may have compensated for declines in catch, such that we observe a negative treatment effect for catch in the sample, but not for GVP.

Several caveats need to be noted when interpreting our results, and more work is needed to further explore empirically the impact of MPA networks on fisheries. First, an issue stemming from our choice of case study is the lack of vessel-level location data throughout the time series for all sectors of the SESSF. The availability of such data would have both enhanced the choice of treatment and control species and enabled explicit analysis of change in vessel movements or change in target species in response to the MPA network. Given the literature on fishing effort relocation in response to MPAs, both *ex ante* and *ex post* (Holland and Brazee 1996; Hannesson 1998; Smith and Wilen 2003; Murawski *et al*. 2005), incorporating this level of data into future analyses is of great interest and may lead to different conclusions regarding the effects of MPA networks. Second, our modelling framework does not include any variable other than fixed effects that explicitly control for ecosystem changes associated with climate change. Since the effects of an MPA network is sensitive to climate change impacts on marine ecosystems (Brander 2007), future work to disentangle the effects of climate change from the estimation of the average treatment effect of the MPA network would be highly relevant. Third, data availability confines our analysis to the use of catch and GVP as proxies for the performance of fisheries. However, there are other possible performance indicators which could be used to indicate the overall performance of a fishery, such as the net economic return of fisheries. Net economic return (NER) is a performance indicator which includes costs as well as value, and so provides an indication of the profitability of a fishery. NER is considered a more appropriate metric to assess fishery performance

(White *et al.* 2008). Future research which includes performance measures that reflect not only changes in revenue, but also changes in costs, may yield different results to the ones found here. Finally, while catch and GVP are common measures of fishery performance, they are not the only considerations from a fishery management perspective, which is concerned with the wider biological and social sustainability of the resource (Anderson *et al.* 2015). Incorporating a wider range of indicators when assessing the effects of MPA networks, such as indicators for biomass and habitat health and fishery-related employment, may lead to different conclusions concerning the benefits derived from MPA networks.

Data availability statement

Data available on request from the authors (see footnote 5).

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. List of species by treatment and control groups.

Appendix S2. The proportion of value contributed by each species to total GVP of each treatment sector in the pre- and post-SEMRN periods.

Appendix S3. List of additional control fisheries and species.