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Evaluating the impact of California's marine protected areas on kelp forest recovery from marine heatwaves

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Master of Arts in Geography

by

Emelly Marbella Ortiz-Villa Grajeda

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ABSTRACT OF THE THESIS

Evaluating the impact of California's marine protected areas on kelp forest recovery from marine heatwaves

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Professor Kyle C. Cavanaugh, Chair

Forests of giant kelp (*Macrosystis pyrifera*) and bull kelp (*Nereocystis luetkeana*) in California are experiencing a number of stressors, including increased frequency of marine heatwaves and trophic disruptions that have led to high levels of grazing by sea urchins. These factors have contributed to population declines along the California coast. The State of California has a relatively large network of marine protected areas, and this protection may promote recovery of kelp forests by maintaining populations of predators that can keep urchin populations under control. However, the impact of MPAs on kelp forest recovery has not been examined across large scales. This information is particularly important as heatwave events are expected to occur more frequently. Using a long-term dataset of satellite-derived kelp canopy area estimates, we used BACI (Before-After-Control-Impact) analysis to examine whether the implementation of MPAs lead to increases in kelp abundance compared to reference sites. We identified a reference site for each MPA based on the dynamics of kelp abundance before the MPAs were implemented. We also analyzed kelp canopy recovery and resistance to the 2014-2016 marine heatwave inside and outside of marine protected areas. We characterized resistance and recovery of kelp by comparing kelp abundance during and after the heatwave to a baseline of kelp abundance during a 10-year pre-heatwave period (2003-2012). Our method for identifying reference sites enabled us to efficiently choose sites located outside MPAs that closely resembled the patterns and behavior of kelp observed within the MPA before its establishment. Our BACI analysis did not show a significant effect of MPA protection on kelp abundance over long time scales. However, using the sites from the reference finder, we did find evidence of greater recovery from the 2014-2016 heatwave inside of MPAs as opposed to the unprotected reference sites. These findings contribute to understanding the dynamics within a protected ecosystem and provide insight about kelp-forest recovery and resistance to temperature anomalies.

The thesis of Emelly Marbella Ortiz-Villa Grajeda is approved.

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1. Introduction

Coastal marine habitats are facing a variety of environmental pressures, including marine heatwaves, trophic disruptions, and disease outbreaks (Eger, Marzinelli, Christie, Fagerli, Fujita, Gonzalez, Hong, Kim, Lee, McHugh, et al. 2022; Ling et al. 2015; Rogers-Bennett & Catton, 2019a). As a result, there is a need for conservation and restoration tools to mitigate habitat loss, maintain biodiversity, and promote the resilience of these systems (Bertocci et al. 2015; Caselle et al. 2015; CDFW & COPC, 2018; Foster & Schiel, 2010; Hopf et al. 2022). Marine protected areas (MPAs) are an effective tool for marine management and have been shown to enhance populations of marine species and promote ecological integrity. In addition to direct effects on fished species, MPAs can have indirect effects on non-targeted species through the maintenance of trophic cascades. Trophic cascades refer to changes in the abundance or behavior of organisms at one trophic level, which then propagate through the food chain and impact other trophic levels (Silliman & He, 2018). For example, increases in the abundance of predatory fish can help control populations of prey species such as herbivorous fish, which may reduce grazing pressure on primary producers such as kelp and seagrass (Caselle et al. 2015). Trophic cascades can promote ecosystem health and resilience by controlling herbivorous prey populations and allowing primary producers to proliferate (Malakhoff & Miller, 2021a; Selden et al. 2017; Silliman & He, 2018).

To improve the design and implementation of MPAs, there is a need for effective methods to measure their impacts. A common way to analyze MPA impacts is through a Before-After-Control-Impact (BACI) statistical design. This approach involves comparing affected (Impact) and unaffected (Control) sites before and after some event, e.g., MPA implementation. The data collected from the control sites help control for natural variability in the variable of interest. One major challenge with BACI design is in selecting control sites. Ideally the control and intervention sites will experience similar environmental conditions and therefore show similar trends and variability prior to the impact (Stevens 2002; Wauchope et al,. 2021). This paired-BACI design aids in separating the effects of the disturbance event from other natural variations or confounding factors (Stewart-Oaten, Murdoch, and Parker 1986). However, in many cases there is a lack of pre-intervention data to inform the selection of reference sites, forcing researchers to rely on proxies such as physical site characteristics (Caselle et al. 2015; Hopf et al. 2022; Wauchope et al. 2021) .

In 1999, the State of California passed the Marine Life Protection Act (MLPA), which launched the re-design of MPAs in California into a cohesive, ecologically connected network. Following this act, a network of MPAs was implemented between 2007 and 2012. The goal for this MPA network was to foster and preserve marine productivity, abundance, and diversity (CDFW & COPC, 2018). These MPAs serve as spatial refuges for populations facing environmental threats which include global warming, overfishing, pollution, and sediment and nutrient loading (Halpern et al. 2006; Harley et al. 2006; Lester & Halpern, 2008). Previous research has shown that abundance, diversity, and recovery of targeted species are greater within MPAs, despite having little to no control against the impacts of climate change (Caselle et al. 2015; Lester & Halpern, 2008; Teck et al. 2017) . For example, various rockfish species such as Vermillion and Copper rockfish, along with the Kelp Greenling, and invertebrates like red and black abalone, have exhibited higher biomass inside MPAs than outside (Caselle et al. 2015). However, there have been conflicting reports as to the indirect effects of MPAs that may be driven by trophic cascades. Previous research has shown that abundant and large urchin predators in MPAs control purple urchin populations, leading to increased kelp and understory algal growth (Eisaguirre et al. 2020). Their results highlight that protected trophic redundancy in MPAs maintain stability and prevents undesirable shifts in kelp forest ecosystems during disturbances. However, another study found that urchin biomass increased in some MPAs and observed no positive impact on giant kelp (Malakhoff & Miller, 2021a). This study aims to investigate the impact of MPAs on kelp forests within a larger geographic area. It seeks to provide valuable insights that can help resolve the existing contradictions in the literature concerning this subject. Kelp forests in California are vulnerable to changing climate and anthropogenic conditions. For instance, unfavorable climate conditions, such as the well documented Marine Heat Wave (MHW) that started in 2014, led to declines in kelp abundance across Northern California, with more variable impacts in Central California and Southern California Bight (Tolimieri et al. 2023). Amidst the interconnected challenges the MHW caused, a concerted effort has been undertaken by scientists and conservationists to unravel the complexities of sea star wasting disease and its cascading effects on the delicate balance of California's kelp forests. The spread of sea star wasting disease (SSWD) in California caused a decline in sea star populations. As a result, sea urchin populations increased and overgrazed kelp, leading to a reduction in kelp abundance. The absence of sea stars disrupted the balance between sea urchins and kelp, resulting in negative impacts on kelp forests (Eisaguirre et al. 2020; Harvell et al. 2019; Hewson et al. 2018). Although climate has been confirmed as a major driver of kelp abundance, regional changes in kelp abundance are rapid and extremely variable (Cavanaugh et al. 2011; Eger et al., 2022; Rogers-Bennett & Catton, 2019b). MPAs may enhance the recovery of kelp forests to disturbances such as marine heatwaves by maintaining trophic cascades and reducing sea urchin grazing pressure (Eisaguirre et al. 2020). However, the dynamic nature of kelp abundance may make it difficult to detect MPA effects (Rassweiler et al. 2021).

In this study, we examined the indirect effects of MPAs on kelp abundance in California using a 30-year time series of satellite observations of kelp abundance. To account for drivers of kelp abundance other than protection status, we developed a method to identify paired reference sites for each MPA that exhibited similar trends and variability in kelp abundance prior to MPA implementation. We then performed a series of analyses to examine the impacts of MPA protection on kelp abundance. We conducted a BACI analysis to determine if kelp abundance increased in MPAs relative to reference sites after MPA implementation. We also examined the impacts of sea star wasting disease and the 2014-2016 marine heatwave on kelp in MPAs and the paired reference sites.

2. Methods

2.1 Data Collection

All MPA boundaries were obtained from the Department of Fish and Wildlife (Fig. 1; https://wildlife.ca.gov/Conservation/Marine/MPAs). MPAs were implemented in the Central Coast in 2007, North Central Coast in 2010, South Coast in early 2012, and North Coast at the end of 2012. 16 of the 117 sites had some form of protection implemented prior to the initiation of the MLPA, and subsequently had their regulations updated by the MLPA. Each MPA designation comes with varied amounts of activities and protections, which include State Marine Reserves (SMRs), State Marine Conservation Areas (SMCA), State Marine Conservation Area (No-Take), and Special Closure.

Landsat satellite imagery with 30-meter pixel resolution was used to examine trends and variability in kelp abundance inside and outside of MPAs between 1984 and 2021. Canopy area of giant kelp and bull kelp was estimated following methods described in Bell et al. (2020) and Bell et al. (2023). Briefly, a decision tree classifier was combined with multiple endmember spectral unmixing (MESMA) to estimate the fraction of kelp cover in each 30 m pixel each quarter from 1984 to 2021. This method has been used to map both giant kelp and bull kelp (Bell et al. 2020; Hamilton et al. 2020), although it is not able to differentiate between the two species. We narrowed our analyses to MPAs with at least 10 pixels of kelp habitat (9,000 m^2), which resulted in a total of 61 MPAs. This removed MPAs without kelp habitat such as protected lagoons and intertidal wetlands. For each MPA, we summed kelp area within the MPA boundary and calculated an annual time series of maximum kelp area by selecting the quarter with the maximum kelp area for each year.

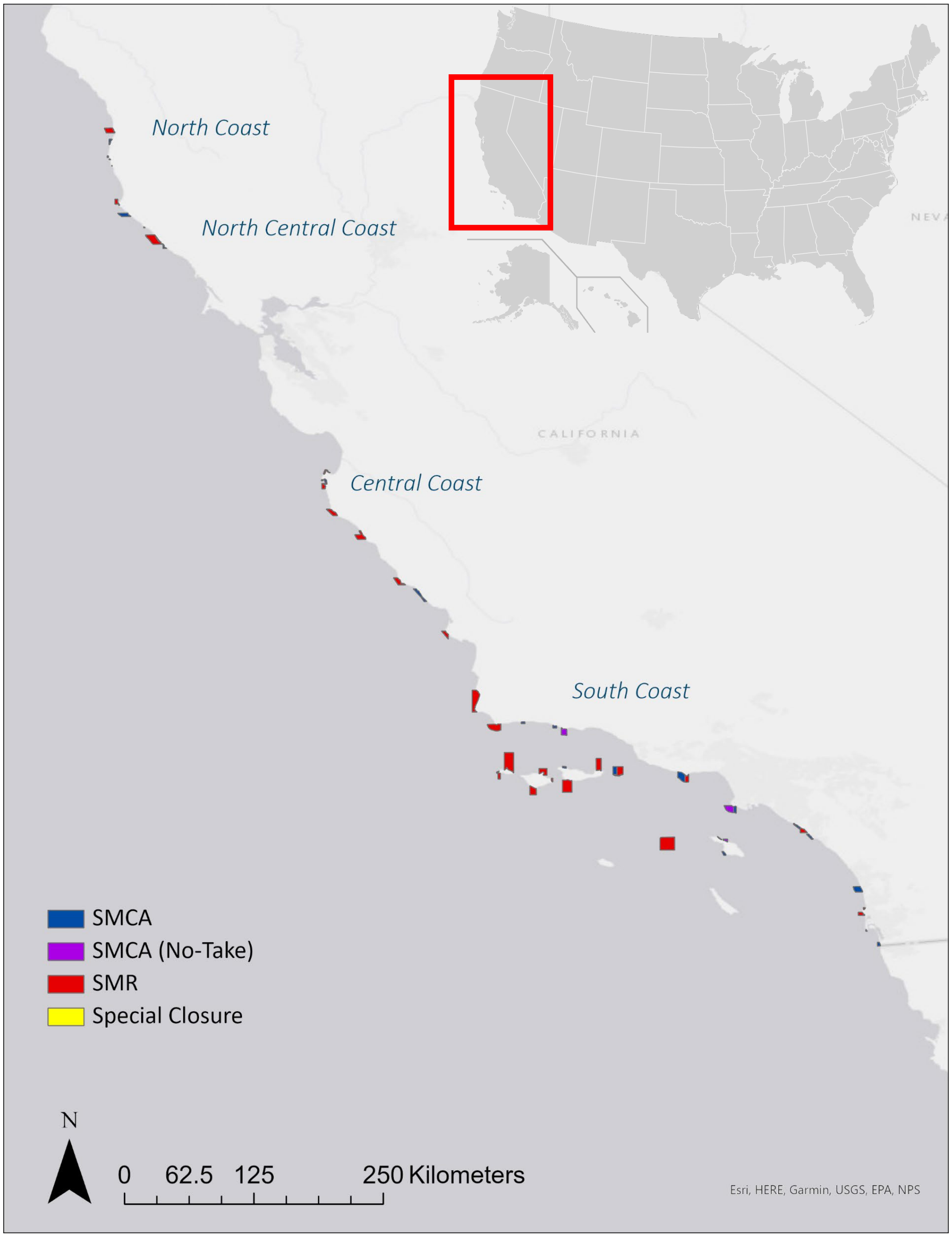

Fig. 1. Sampled MPAs in California, Channel Islands. Designation colors correspond to the type of protected area which include State Marine Conservation Areas (SMCA), No-Take State Marine Conservation Areas (SMCA No-Take), State Marine Reserves (SMR), and Special Closures.

2.2 Reference Finder

To assess the impacts of MPAs on kelp abundance, we identified a paired reference site for each MPA using the Landsat kelp canopy time series. Our general approach was to identify a contiguous region of kelp habitat outside of each MPA that exhibited similar kelp canopy dynamics to the MPA prior to MPA implementation. We first calculated the total amount of kelp habitat within each MPA. Kelp habitat was defined as 30 m Landsat pixels that contained kelp at some point during the time series. We then used a sliding window technique to find a group of the same number of pixels of kelp habitat outside of the MPA that exhibited kelp canopy area (m^2) dynamics that were strongly correlated to the dynamics within the MPA, prior to MPA implementation (Fig. 2). We constrained our search to a 0.5 x 0.5-degree window centered on the MPA. For each potential reference site (combination of pixels with the same number of pixels as was in the MPA) we calculated the annual time series of maximum kelp area as described for the MPA time series. We iterated through each combination of grouped kelp canopy pixels and calculated the Spearman correlation of the annual time series of kelp canopy inside the MPA and in the potential reference site for the years from 1984 to the appointed MPA implementation year. The reference site regions had to be contiguous, but they could be separated by the MPA (i.e., in some cases the reference site had parts of either side of the MPA. The optimal reference site was determined by selecting the reference time series with the highest correlation with the MPA time series. This approach ensured that the chosen reference site closely aligned with the patterns and dynamics observed within the MPA prior to MPA implementation. We removed any MPAs from further analysis where the correlation between the MPA and paired reference site prior to MPA implementation was not similar in kelp canopy dynamics (<0.5).

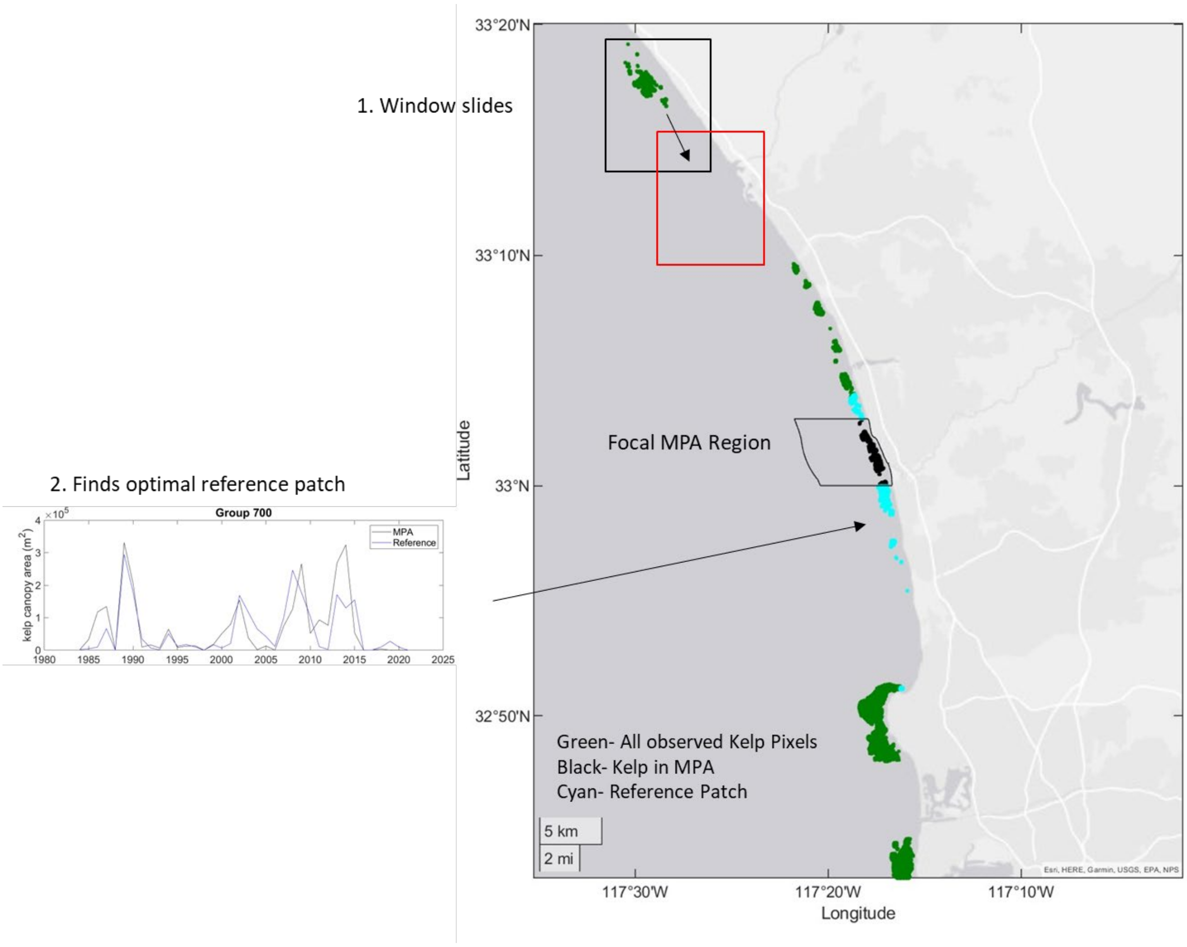

Fig. 2. Reference site selection method using a sliding window technique to find a patch of kelp canopy pixels outside of the MPA that is similarly sized to the patch inside the MPA. Annual time series of maximum kelp canopy are compared between MPA and potential reference sites to determine the patch of pixels where maximum annual kelp dynamics in the reference site has the highest correlation to the dynamics in the MPA.

2.3 Environmental Validation

To further validate our selected reference sites, we compared four environmental conditions between each MPA and its paired reference site. We examined mean annual max sea surface temperature (°C), depth (m), sea surface nitrate (µmol L-1), and wave height (m). Sea surface temperature was calculated using the mean daily surface temperature from the NOAA Coral Reef Watch SST data (5 km resolution), available from 1985- 2022. Sea surface nitrate was derived from the daily temperature data following Snyder et al. 2020, including a new temperature to nitrate relationship derived from CalCOFI data in Northern CA (north of San Francisco), available from 1985-2022. Depth of each 30m pixel was derived from the Seafloor Mapping project, except for areas around Port San Luis from NOAA NCEI, and to the far south near Imperial Beach from GEBCO. Maximum significant wave height was modeled by the CDIP MOPv1.1 wave model on an hourly time scale at 1 km coastline segments. All wave data prior to 2004 was hindcasted by developing a non-linear statistical model (GAM) between CDIP data and data from one of 18 offshore US Army Corp Wave Information Study (WIS) model sites (1984 – 2019). The site that produced the best model estimating CDIP wave height from WIS wave height, period, and direction was used to model the daily maximum significant wave height back to 1984. For each variable we calculated the average of all grid cell values within the boundaries of each MPA and reference site. We then calculated the correlation between the MPA and reference site for each environmental variable.

2.4 BACI trend/ Immediate Effect

We conducted a before-after-control-impact (BACI) analysis following Wauchope et al. (2021) to examine the impact of MPA implementation on kelp abundance. To assess whether the implementation of the Marine Protected Area (MPA) led to a significant increase in kelp, we compared the before-after measurements of the intervention group (MPA) to those of the control group (reference site). This was done by calculating the difference between the before-after measurements of the intervention group and subtracting the difference between the before-after measurements of the control group. The equation for this generalized linear model is represented as:

$$\text{Value} \sim \text{BA} + \text{CI} + (\text{BA} \times \text{CI})$$

The interaction term between BA, CI captures average chance in abundance following MPA implementation (Wauchope et al. 2021). A control-intervention variable (CI) is included, which is 0 for the control time series (reference sites), and 1 for the intervention time series (MPAs). B and A represent time periods before and after the MPAs were established, considering the different dates of implementation for each region.

2.5 Recovery and Resistance

We also examined how kelp abundance responded to and recovered from the severe marine heatwaves that occurred between 2014 and 2016, which coincided with the loss of seastars across the State due to seastar wasting disease. To characterize the initial response of kelp abundance to these disturbances, we calculated a resistance metric as the mean annual maximum kelp canopy area from 2014-2016 (during the heatwave) divided by the mean annual maximum kelp canopy area from 2004 to 2013, the 10-year period before the heatwave. To characterize the recovery of kelp following the heatwave, we divided the mean annual maximum kelp canopy area from 2016 to 2022 (post heatwave) by the mean annual maximum kelp canopy area from 2004 to 2013. We calculated the resistance and recovery metrics for each MPA and paired reference site. Rank sum tests were performed to determine whether resistance/recovery were significantly higher in MPAs as compared to reference sites. We also compared the response inside and outside of MPAs by dividing the resistance and recovery metrics for each MPA by its paired reference site. Values greater than 1 indicate higher resistance/recovery inside of the MPA.

3. Results

3.1 Reference Finder

We identified paired reference sites for each of the 61 MPAs in our study area that had more than 10 pixels of kelp habitat (9,000 m^2) (Table.1). We removed 4 MPA-reference pairs where correlations between kelp dynamics before MPA implementation were <0.5, leaving us with 57 MPA-reference pairs . The mean correlation between kelp dynamics for MPA-reference pairs before MPA implementation was 0.79. Mean correlation between MPA-reference pairs after MPA implementation was 0.68. MPAs and their paired reference sites experienced similar temperature, nutrient, and wave conditions (r = 0.99, 0.99, and 0.73 respectively; Fig. 3). Depth was less correlated between MPA and reference sites (r = 0.32), with reference sites having greater depth on average (Fig. 3B).

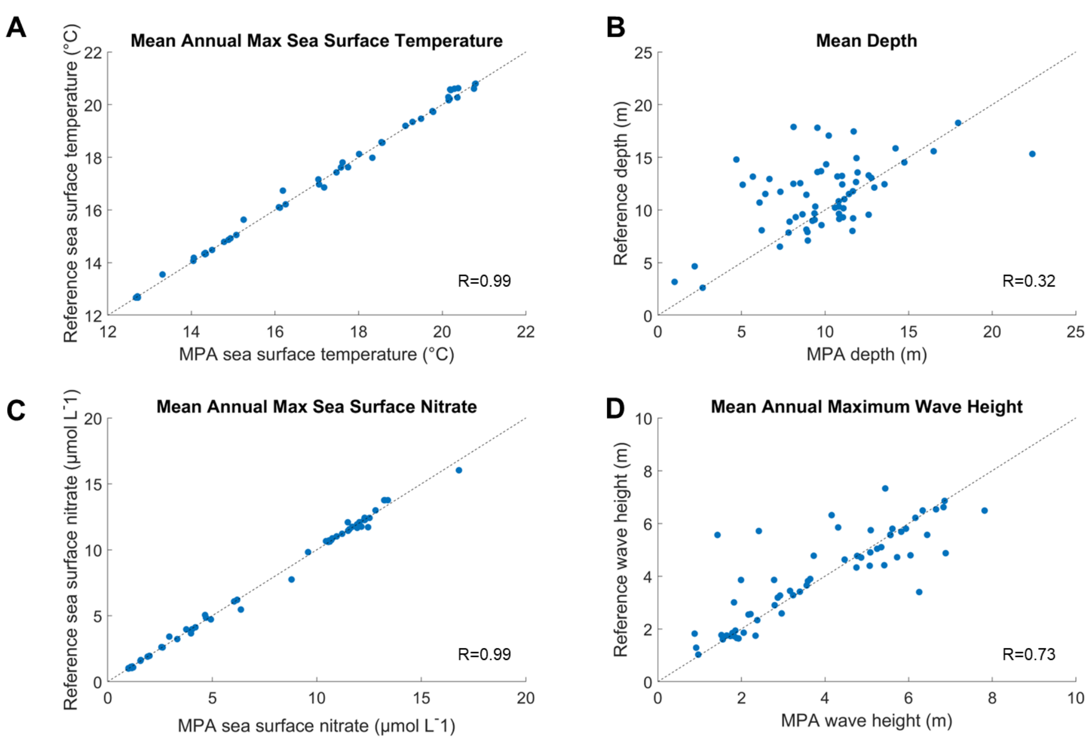

Fig. 3 Mean annual max sea surface temperature, sea surface nitrate, wave height, and depth plotted against MPAs and their respective reference sites to measure correlation of environmental variables between sites.

Table 1. BACI results for immediate impact of implementation on MPAs showing the Standard Error or the Estimate, Standard Error, t-statistic, and p value for each MPA. The p values in bold show the sites with significant immediate effect.

MPA Name	EST	SE	tStat	pValue
Ten Mile SMR	-860.004	1758.94	-0.48893	0.626417
Point Cabrillo SMR	-1499.97	2033.982	-0.73746	0.463309
Russian Gulch SMCA	577.2759	7156.707	0.080662	0.935941
Van Damme SMCA	1240.681	2288.553	0.542125	0.589454
Point Arena SMR	-1369.09	8350.378	-0.16396	0.870239
Sea Lion Cove SMCA	942.2667	2842.108	0.331538	0.741229
Saunders Reef SMCA	**-105247**	**51951.27**	**-2.02587**	**0.046591**
Carmel Pinnacles SMR	-6951.19	11428.33	-0.60824	0.545024
Vandenberg SMR	**-30831.4**	**9472.654**	**-3.25478**	**0.00175**
San Miguel Island Special Closure	2713.222	59869.07	0.045319	0.963982
Harris Point SMR & FMR	52088	60548.96	0.860263	0.392581
Carrington Point SMR	90855.86	52562.83	1.728519	0.088304
Skunk Point SMR	14825.51	4004.879	3.701862	0.000423
South Point SMR & FMR	-5823.48	96653.56	-0.06025	0.952127
Painted Cave SMCA	-188.905	552.81	-0.34172	0.733587
Gull Island SMR & FMR	6778.016	83964.65	0.080725	0.935891
Scorpion SMR & FMR	3208.385	2195.502	1.461345	0.148395
Anacapa Island Special Closure	2863.552	12066.01	0.237324	0.813099
Anacapa Island SMCA & FMCA	1187.893	978.4605	1.214043	0.228811
Anacapa Island SMR & FMR	**3210.48**	**1588.741**	**2.020771**	**0.04713**
Santa Barbara Island SMR & FMR	23756.08	14548.71	1.632866	0.106988
Arrow Point to Lion Head Point SMCA	-362.218	1718.966	-0.21072	0.833719
Blue Cavern Onshore SMCA (No-Take)	-102.968	313.4815	-0.32846	0.743616
Farnsworth Onshore SMCA	-141.943	3792.055	-0.03743	0.970249
Swami''s SMCA	44328.79	46690.01	0.949428	0.34567
Matlahuayl SMR	-1608.91	4273.529	-0.37648	0.707696
South La Jolla SMR	-66605.7	125182.6	-0.53207	0.596364
Cabrillo SMR	-5233.23	5163.554	-1.01349	0.314315
Tijuana River Mouth SMCA	504.7778	2296.169	0.219835	0.82664
MacKerricher SMCA	561.3319	1953.055	0.287412	0.774646
Del Mar Landing SMR	33.87778	974.2175	0.034774	0.972359
Salt Point SMCA	-6181.13	27640.17	-0.22363	0.823698
Lovers Point - Julia Platt SMR	18399.91	19800.03	0.929287	0.35598
Pacific Grove Marine Gardens SMCA	-6238.54	50614.14	-0.12326	0.902262
Asilomar SMR	48404.55	43458.68	1.113806	0.269284
Carmel Bay SMCA	73683.31	117149.1	0.62897	0.531416
Point Lobos SMR	70100.77	100036.5	0.700752	0.485781
Point Sur SMR	171052.7	388162.5	0.440673	0.660808
Big Creek SMR	125925.4	131416.8	0.958214	0.341253
Piedras Blancas SMR	75115.96	127343.5	0.589869	0.557178
Cambria SMCA	14494.04	250595.4	0.057838	0.954042
White Rock SMCA	-39995.4	221930.2	-0.18022	0.857503
Point Buchon SMR	26269.32	45035.31	0.583305	0.561563
Point Conception SMR	60726.71	48836.66	1.243466	0.217845
Naples SMCA	-37210.6	56965.5	-0.65321	0.515758
Campus Point SMCA (No-Take)	53675.77	80732.68	0.664858	0.508326
Judith Rock SMR	-13953	64251.27	-0.21716	0.828713
Point Dume SMCA	-2918.79	21628.11	-0.13495	0.893036
Point Dume SMR	11152.44	17862.22	0.624359	0.534421
Abalone Cove SMCA	6212.718	6322.77	0.982594	0.329191
Point Vicente SMCA (No-Take)	28176.47	16893.9	1.667849	0.099814
Crystal Cove SMCA	-10719.1	13903.56	-0.77096	0.443326
Laguna Beach SMR	8727.802	10786.63	0.809131	0.421181
Dana Point SMCA	-5458.49	31338.49	-0.17418	0.862228
Stewarts Point SMCA	505.5889	24796.94	0.020389	0.983791
Stewarts Point SMR	-3640.08	28745.14	-0.12663	0.899594

3.2 Before and After MPA Implementation BACI

BACI analysis indicated that most MPAs did not show a significant increase or decrease in kelp abundance as compared to their paired reference site following MPA implementation. Of the 57 sites, two showed a significant positive effect of MPA protection on kelp abundance and 2 showed a significant negative effect. The sites where a significant negative effect was observed were Vandenberg SMR and Saunders Reef SMCA. The sites with a significant positive effect on MPA protection were Skunk Point SMR and Anacapa SMR and FMR.

3.3 Recovery and Resistance

During the 2014-2016 heatwave, kelp abundance was low across California (Figs 4a, Fig 5a and 5b). One exception to this pattern was the coastline in central California between 34.5°N and 35.75°N, where kelp abundance remained near pre-heatwave levels. A rank sum test showed no significant difference in resistance between MPAs and reference sites ($p = 0.42$). The median resistance values in MPAs and reference sites were 0.31 and 0.24 respectively. However, when compared to their individually paired references sites, MPAs had higher resistance, i.e., the mean proportion of MPA to paired reference site resistance was 2.8. The mean proportion of MPA to paired reference site resistance was highest in the North and North Central Coast regions: South Coast = 1.95, Central Coast = 1.68, North Central Coast = 5.89, North Coast = 8.11.

Recovery following the heatwave was more variable across California. The Central Coast and South Coast regions showed the highest recovery, and the North coast region showed the lowest recovery values (Fig. 6). This pattern held for both MPA and reference sites. A rank sum test also showed no significant difference in recovery between MPAs and reference sites ($p = 0.17$). The medians for MPA and reference recovery were 0.37 and 0.20, respectively. As with resistance, the MPAs showed higher recovery when compared to their paired reference site. The mean proportion of MPA to paired reference site recovery was 3.56 across the entire study area (4.25 for the South Coast, 2.76 for the Central Coast, 1.15 for the North Central Coast, and 5.20 for the North Coast).

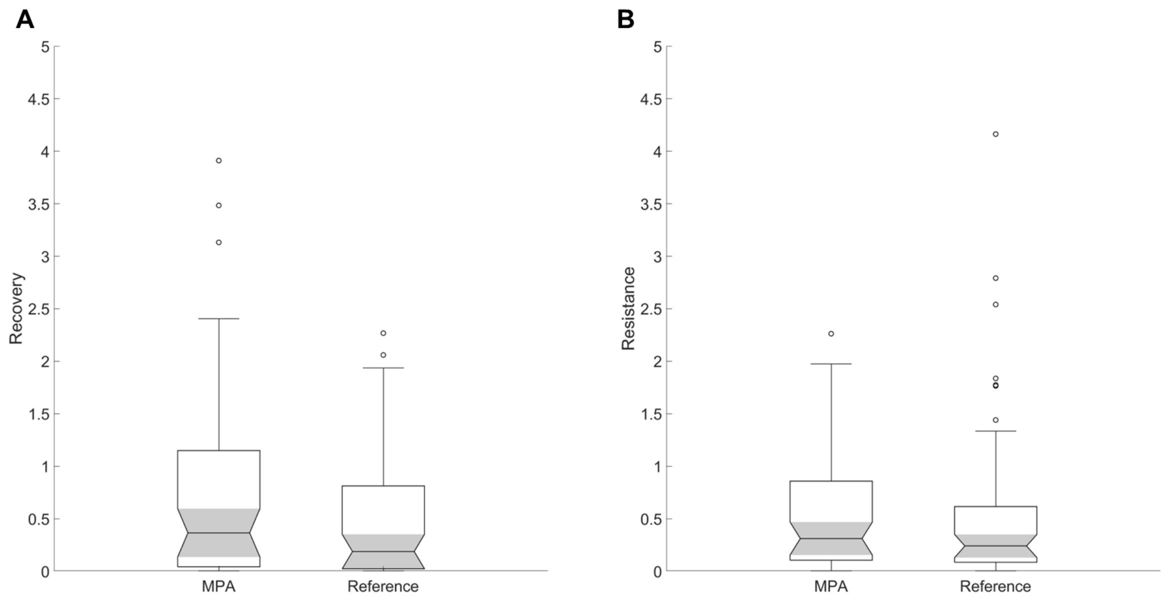

Fig. 4. Distribution of recovery and resistance metrics for MPAs and their respective reference site.

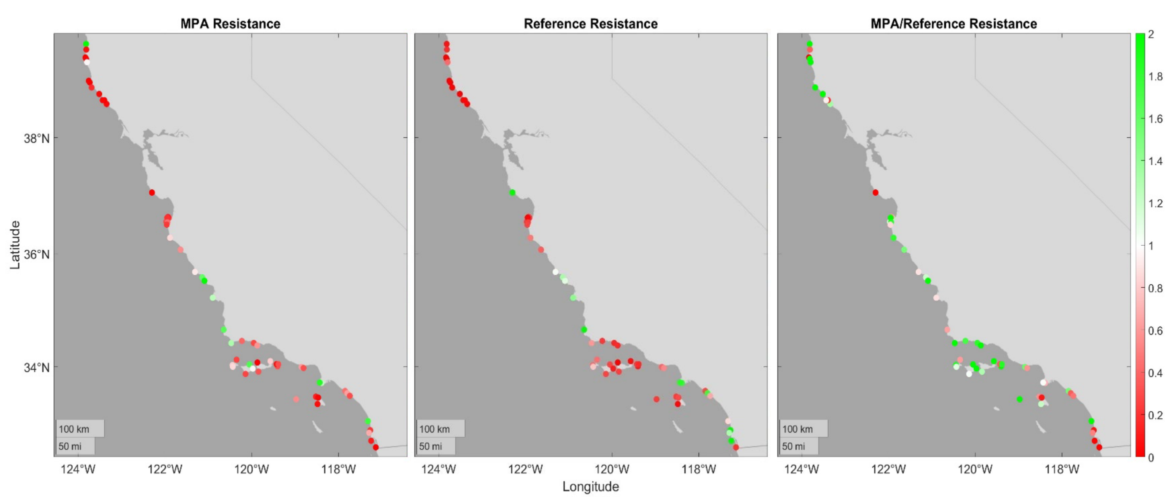

Fig. 5. Resistance metrics for kelp canopy biomass in MPAs and reference sites across California

for the 2014-2016 marine heat wave. MPA/Reference resistance shows the proportion between MPA resistance and resistance recovery points represent MPA sites and colors represent strength in resistance.

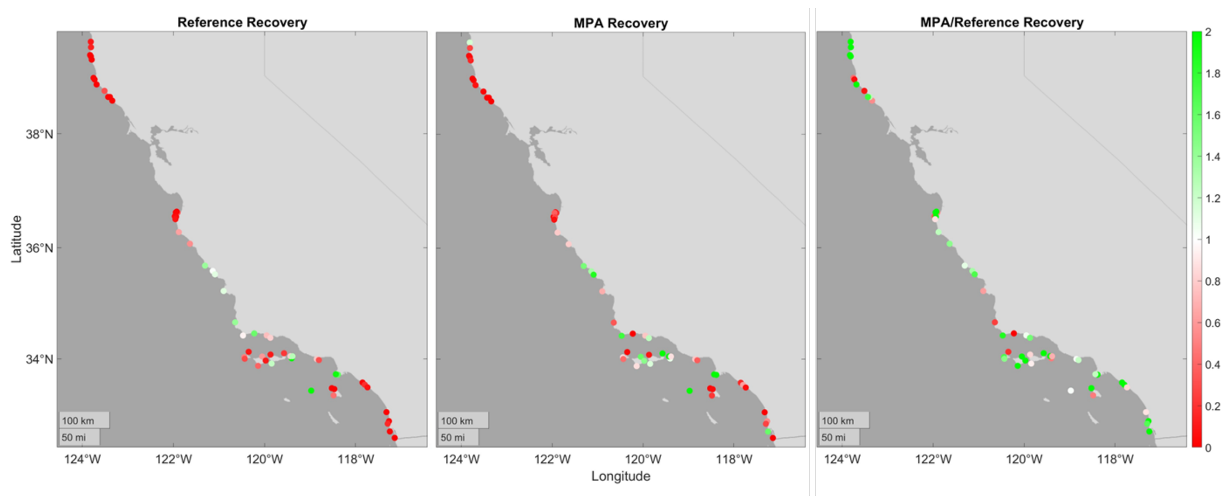

Fig. 6 Recovery metrics for kelp canopy biomass in MPAs and reference sites across California for the 2014-2016 marine heat wave. MPA/Reference recovery shows the proportion between MPA recovery and reference recovery. Points represent MPA sites and colors represent strength in recovery. A value of 1 represents no difference in recovery after the heatwave.

4. Discussion

In this study, we employed a BACI analysis to examine the effects of MPA implementation on kelp abundance. Our findings revealed that there was no significant increase in average kelp abundance in MPAs as compared to reference sites. One potential explanation for this lack of effect may be the high natural variability in kelp abundance swamped any MPA effect signal. It can be difficult to detect impacts using BACI approaches for species that experience large fluctuations in abundance through time, such as giant and bull kelp (Rassweiler et al. 2021). Malakhoff and Miller (2021) also did not find indirect effects of MPAs on kelp abundance in southern California and pointed to the possibility that predators had not yet sufficiently recovered to a level to support trophic cascades. This highlights the significance of considering the necessary time frame for conservation policies to be effective. Also, MPAs are explicitly designed to influence populations beyond their borders, and if both MPAs and reference sites exhibit similar ecological dynamics, it’s still possible they may demonstrate different rates of recovery (Caselle et al. 2015; Margaret A. Palmer, Joy B. Zedler 2003).

While we did not see an overall increase in average kelp abundance in MPAs, we did find some evidence that kelp in MPAs recovered more quickly following declines in kelp abundance that were linked to the 2014-2016 marine heatwave and increased urchin grazing. Specifically, kelp recovery was higher in MPAs than in their paired reference sites (Fig.4). The combined disturbance of the marine heatwave and loss of seastars, an important urchin predator, may have created a situation where indirect MPA effects on kelp abundance were easier to detect. For example, increased abundance of other urchin predators in MPAs may have lowered the probability that large urchin barrens would form after declines in kelp associated with the heatwave. In Norway, a study provided evidence of this phenomenon, where the recovery of kelp became more apparent in regions where environmental conditions favored the proliferation of alternative urchin predators. This helped facilitate a reverse shift in kelp recovery (Christie et al. 2019). Previous studies have found increased abundance of urchin predators such as California sheephead (Semi-cossyphus pulcher) and California spiny lobster (Panulirus interruptus) inside California MPAs (Eisaguirre et al. 2020; Selden et al. 2017). Therefore, the observed recovery in kelp abundance could be due to the concurrent recovery of other urchin predators, which appears to be size-dependent and influenced by limited fishing pressure (Selden et al. 2017). Again, these impacts may have been swamped by natural variability prior to the disturbance. This disturbance effect could help explain the apparent contradiction between the findings of Eisaguirre et al.

(2020) and Malakhoff and Miller (2021). Both studies examined MPA impacts on kelp abundance in the northern Channel Islands. While Malakhoff and Miller (2021) did not find evidence of increased kelp abundance in MPAs using a BACI analysis between 1991 and 2019, Eisaguirre et al. (2020) did find higher algal abundance inside MPAs after 2014, which was when seastars were extirpated from this region.

Instead, our findings demonstrate that kelp recovery becomes apparent when viewed from a broader spatial perspective. However, it is important to acknowledge that kelp recovery varies across regions, which aligns with Eisaguirre's perspective that indirect effects of MPA protection might only manifest after prolonged periods. Also, the time it takes for an MPA to recover may also be impacted by area size and proximity to anthropogenic stressors (i.e, population size, river mouths, water treatment plants (Bhat 2003; Brander, Van Beukering, and Cesar 2007; Lester and Halpern 2008). Investigating the influence of reserve size and other stressors on kelp recovery is an area that warrants further exploration and conducting a BACI analysis with this variable may yield valuable insights. Nevertheless, obtaining a suitable dataset for a BACI analysis can always be a challenge when working with species of high temporal variability (Caselle et al. 2015; Caselle, Davis, and Marks 2018; Aaron M Eger et al. 2020). The rate and speed at which these responses occur are largely unknown, which follows greater uncertainty in expectations of outcomes to MPA implementation (Nickols et al. 2019).

While we found higher recovery in MPAs after 2016, the patterns of recovery varied a great deal across our study area. In southern California, kelp demonstrated higher resistance to marine heatwaves, with favorable conditions for regrowth and recruitment after 2015 (Cavanaugh et al. 2019). Concurrently, urchin abundances in southern California were lower compared to their predator populations. A previous study (Nichols et al. 2009) showed that in southern coast MPAs, urchin predators were more abundant and urchins exhibited lower abundances (Nichols, Segui, and Hovel 2015). Considering this established pattern, along with the resistant distribution of kelp habitat following the heatwave and the observed densities of both sea urchins and predators, we can better understand why the Central and South coast regions exhibited greater recovery compared to the North coast regions, which showed contrasting ecological conditions.

The Central Coast and Southern Coast regions of our study sites had more MPAs combined, 45 sites, compared to the Northern Coast and Northern Coast regions, which only includes 12 sites. Our method of selecting MPAs and corresponding reference sites was dependent on regions where kelp was present in the historical Landsat dataset. During the marine heatwave period, the Landsat dataset did not capture sufficient kelp presence in the North Coast region, rendering it challenging to include all corresponding MPAs in our analysis. Consequently, a greater number of North Coast sites had to be excluded from our study since the reference finder method employed could not identify suitable habitats for inclusion. The limited number of sites made it difficult to identify a significant difference between MPA and reference sites. For example, sites like Kashayit SMCA were removed for having low abundance of kelp in the time series, and therefore output no values in the BACI and recovery calculations.

Satellite imagery is advantageous for detecting kelp abundance across a vast study area, but it does have limitations when applied to regions where kelp canopy is less prevalent. In this study, Landsat satellites provided the most spatially and temporally comprehensive dataset to identify reference sites after the intervention occurred. Long-term data offers more power to identify delayed effects of MPAs and capture impacts of rare disturbances. For example, we were able to successfully compare recovery and resistance of kelp canopy in each MPA using the MHW time period. However, using Landsat imagery comes with its drawbacks in regions where kelp canopy coverage is low, especially along North Coast regions. In this region bull kelp forms small, sparse canopies that are close to shore, and so difficult to detect with 30 m resolution imagery (Cavanaugh et al. 2023). There are opportunities for disentangling environmental drivers that impact kelp abundance in MPAs using higher resolution imagery, or via field surveys to capture low kelp canopy (Cavanaugh, et al. 2023, Rogers-Bennett and Catton 2019b). Another limitation of satellite data is that it only offers information on kelp canopy cover, so it cannot capture understory growth or finer-scale refugia. Nonetheless, this tool can continue to be useful in understanding other future disturbances, or to evaluate the power of an MPA designation over a particular region.

5. Conclusion

In 1999, California established one of the largest networks of Marine Protected Areas (MPAs) along its North Pacific coast, encompassing the state's coastline and Channel Islands. This move prompted researchers and conservationists to investigate the indirect effects of MPAs, such as trophic cascades, both inside and outside the protected areas. This study aimed to assess the potential of using long-term satellite-derived data on kelp canopy abundance as a proxy for selecting reference sites and evaluating the impact of MPAs on kelp abundance following the marine heat wave that occurred between 2014 and 2016.

Our findings revealed that, immediately after the implementation of MPAs, there was no significant average change in kelp abundance. However, kelp exhibited a faster recovery within MPAs compared to their corresponding reference sites. Kelp is an ecologically and economically important species, so understanding the impact of protection on its recovery and resistance becomes crucial as climate change and fishing pressure continue to pose challenges to kelp forests. This study highlights the importance of MPAs in supporting the recovery of kelp populations and how monitoring long-term kelp canopy abundance from satellite data can be a useful tool for assessing the effectiveness of conservation efforts in the face of environmental disturbances. This can inform future conservation strategies to ensure the recovery and sustainability of kelp ecosystems in the presence of ongoing climate and human-induced pressures.

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