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Ecosystem Service Trade-offs in Coral Reefs: A Global Analysis

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Abstract

Coral reefs are biodiversity hotspots that provide humans with extractive ecosystem services like fisheries and non-extractive services like tourism. Though no studies have examined these trade-offs among these services at a global scale, there are now large, open datasets that make this possible. Using these datasets, we asked whether spatial protection and the level of protection (World Database on Protected Areas) impacted coral reef tourism value (Atlas of Ocean Wealth). We also examined whether proximity to regions of high fishing effort, defined as the top 25th percentile of the average annual effort from Global Fishing Watch, impacted tourism values. Since our data were zero-inflated, we used a two-step modeling approach for statistical analysis, examining predictors of tourism presence/absence then predictors of tourism value magnitude. We found that protected coral reefs are more likely to have a tourism value than those that are not. Among reefs with tourism value, protected areas had a higher average value than unprotected areas. In addition, the level of protection influenced the likelihood and magnitude of tourism value, though in some instances in unexpected ways, possibly due to restrictions on tourism in strongly protected reefs. Coral reefs in closer proximity to high fishing effort had a lower chance of having tourism value, but in areas that did have tourism value, areas closer to high fishing effort had a higher average value. Using large, global datasets of ecosystem services, our study informs management trade-offs in coral reef ecosystems.

Keywords

ecosystem, services, marine protected areas

Disciplines

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Comments

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ABSTRACT

Coral reefs are biodiversity hotspots that provide humans with extractive ecosystem services like fisheries and non-extractive services like tourism. Though no studies have examined these trade-offs among these services at a global scale, there are now large, open datasets that make this possible. Using these datasets, we asked whether spatial protection and the level of protection (World Database on Protected Areas) impacted coral reef tourism value (Atlas of Ocean Wealth). We also examined whether proximity to regions of high fishing effort, defined as the top 25th percentile of the average annual effort from Global Fishing Watch, impacted tourism values. Since our data were zero-inflated, we used a two-step modeling approach for statistical analysis, examining predictors of tourism presence/absence then predictors of tourism value magnitude. We found that protected coral reefs are more likely to have a tourism value than those that are not. Among reefs with tourism value, protected areas had a higher average value than unprotected areas. In addition, the level of protection influenced the likelihood and magnitude of tourism value, though in some instances in unexpected ways, possibly due to restrictions on tourism in strongly protected reefs. Coral reefs in closer proximity to high fishing effort had a lower chance of having tourism value, but in areas that did have tourism value, areas closer to high fishing effort had a higher average value. Using large, global datasets of ecosystem services, our study informs management trade-offs in coral reef ecosystems.

INTRODUCTION

The oceans provide important ecosystem services to humans but are experiencing increased pressure from local to global anthropogenic stressors (Ghermandi and Nunes 2012; Lester et al. 2012). A 2014 study concluded that 66% of the ocean has experienced an increase in cumulative human impacts and that 97.7% of the ocean is impacted by multiple anthropogenic stressors (Halpern et al. 2015). The ecosystem services provided by the ocean can be extractive or non-extractive and range from a source of food to aesthetic value like tourism (Lester et al.

2012). In some cases, ecosystem services can't coexist without trade-offs, like high impact fishing and high-quality tourism (Lester et al. 2012). The surrounding community must make decisions about what ecosystem services are most important to them, thus impacting the management decisions in the area (Lester et al. 2012).

Tourism does have trade-offs with other ecosystem services that reduce biodiversity and, if not managed properly, can itself lead to ecosystem degradation. Previous studies have found that for every 1% increase in biodiversity is associated with a 0.87% increase in tourism (Chung et al. 2018). In addition, Chung et al. (2018) found that areas that were protected for biodiversity saw more tourists than those coral reef areas that were designated as mixed-use. In coastal regions, tourism and recreation have increased, increasing stress on coastal marine ecosystems (Ghermandi and Nunes 2012). The stressors include damage from sewage, pollution, and construction debris from development of residents for tourists, which can also ultimately reduce the attractiveness of that area to future tourists (Cowburn et al. 2018). Tourists may also trample coral while enjoying the beach or snorkeling and or wear sunscreen that damages reefs (Cowburn et al. 2018). However, tourism may also provide funding for marine conservation efforts like Marine Protected Areas (Atmodjo et al. 2017; Coke-Hamilton 2020).

MPAs are one of the most effective tools being used for marine conservation (Selig and Bruno 2010; Hargreaves-Allen et al. 2017). They have been shown to protect and restore local fish and coral populations, increase the resilience of the ecosystem, protect biodiversity, and provide socio-economic benefits (Selig and Bruno 2010; Hargreaves-Allen et al. 2017). The effectiveness of MPAs, however, is highly variable and only 2.7% of the oceans are considered to be in "highly to fully protected" zones (Marine Conservation Institute 2020). For MPAs to be the most effective, they must be larger than 100 km^2, have high strength of protection, and be actively managed and enforced (Edgar et al. 2014). For example, a meta-analysis of 10 studies on MPA effectiveness found that fish biomass was 343% greater in MPAs that did not allow fishing (no-take reserves) than in those with partial protection (Sala and Giakoumi 2018).

The strength of protection within MPAs can be categorized in two ways: 1) whether they are fully no-take (no fishing allowed), partially no-take (restrictions on fishing with some gear, in some areas, or during some parts of the year), or have no no-take area (open to fishing), or 2) on International Union for the Conservation of Nature (IUCN) protected area categories. For IUCN, the strictest category of protection is Ia a “strict nature reserve”, where impacts to the environment are strictly controlled and limited (IUCN 2020). The most lenient category of protection, VI or “protected areas with sustainable use of natural resources”, are the least strict and allow for low levels of non-industrial sustainable use of the land to take place (IUCN 2020).

MPAs play a critical role in fisheries management and protection of coral reef ecosystems (Selig and Bruno 2020), but only 2.5% of the world's coral reefs are actively protected (McClanahan 2020). Coral reefs represent some of the most diverse and productive ecosystems on earth, which makes them ecologically and economically valuable (Lachs and Onate-Casado 2020; Selig and Bruno 2010). While some of the ecosystem services coral reefs provide have minimal impacts on these ecosystems and can therefore be sustainable, others like destructive fishing practices can lead to rapid degradation of these systems (Buhl-Mortensen 2017). The loss of coral habitat can have a cascading effect on other organisms living in and around these ecosystems (Selig and Bruno 2010; Hargreaves-Allen et al. 2017) and therefore to the loss of tourism revenue. The health of coral reef ecosystems is a major driver in economic viability of tourism (Brown 2001). Poor management or environmental degradation can lead to the collapse of the tourism economy (Brown 2001).

The United Nations has dedicated this upcoming decade, from 2021 to 2030, as the “Decade of Ocean Science for Sustainable Development” (UNESCO). Current global initiatives support the creation of more MPAs as a key step towards improved conservation of marine ecosystems. The United Nation's Sustainable Development Goals (SDGs) are global goals that meet environmental, political, and economic challenges facing the world. Sustainable Development Goal 14 (SDG14) aims to “conserve and sustainably use the oceans, seas and marine resources for sustainable development" and Target 5 of this goal was to protect 10% of the ocean by 2020 (United Nations 2020). The Convention on Biological Diversity Aichi Targets, which focus on conserving and protecting biodiversity, similarly aimed to protect 10% of the ocean by 2020 (Aichi Target 11). Though we are behind on these goals, there is interest and demonstrated need to build on these initiatives by creating the goal of protecting 30% of the ocean by the end of 2030 (Ocean United 2020). In January, United States President Joe Biden issued an executive order to protect 30% of the nation's land and oceans by 2030 (White House, 2021).

Target 7 of SDG14 is also highly relevant to the management of coral reef ecosystems. This target aims to improve economic benefits from the ocean to Small Island Developing States (SIDS), whose EEZs contain many of the world's coral reef ecosystems, through sustainable fisheries, aquaculture, and tourism. There are, however, likely to be trade-offs between sustainable management of tourism and fisheries that further interact with goals for more widespread ocean protection through SDG 14 Target 5.

Small island developing states (SIDS) are highly and in many cases increasingly dependent upon tourism as a part of their economy (Lee et al. 2014). Previous to the international tourism boom, SIDS like the Seychelles had economies dominated by the agriculture sector (Lee et al. 2014). Now, tourism accounts for 86% of jobs in the Seychelles (Lee et al. 2014). A rise in tourism is also linked to a decrease in fish catch in Seychelles, from 4,211 tones in 2007 to 2,875 in 2011 (Lee et al. 2014).

Given these pressing goals, understanding the interactions between marine protection, tourism, and fishing are timely. Several recent scientific studies have focused on where MPAs are most needed. For example, an overlay analysis found that 55% of the ocean has been identified as a priority by at least one of 10 United Nations and non-governmental initiatives based on criteria such as uniqueness, productivity, threatened species, and biodiversity, and that many of these areas could be candidates for future protection (Gownaris et al. 2019). In other studies, areas that were heavily impacted or degraded by human impacts were identified as areas in need of stronger protections (Halpern et al. 2015). There has not yet been a global study, however, assessing trade-offs between human activities on coral reefs and the effectiveness of MPAs in supporting coral reef tourism.

OBJECTIVES

In this study, I will examine overlap among extractive and non-extractive uses of coastal ecosystems in marine priority regions across the globe using open datasets on fishing intensity, tourism value, and protected area coverage. I chose to focus on coral reefs because many of these habitats have high consensus as priority areas but low MPA coverage or strength of protections (Gownaris et al. 2019) and because these are key habitats in the economy of SIDs. Through examining overlap and identifying trade-offs, I hope to inform management of these marine biodiversity hotspots. I will test the following hypotheses:

1) Protected regions will have higher tourism values than non-protected regions and protected regions with stronger protection (IUCN category, no-take status) will be associated with higher tourism values.
2) Areas with a greater consensus as priority regions (as defined by Gownaris et al. 2019) will be associated with higher tourism values.
3) Coral reef regions that are closer to regions of high fishing intensity will have lower tourism values.

METHODOLOGICAL APPROACH

Data

I used coral reef polygons defined by the World Conservation Monitoring Centre (WCMC) and the UN Environment Programme as the spatial unit for all analyses (Figure 1). This dataset shows the global distribution of coral reefs and includes data as early as 1954 to 2009 and was created using a number of sources by the UNEP World Conservation Monitoring Centre and the WorldFish Centre, in collaboration with WRI (World Resources Institute) and TNC (The Nature Conservancy) (UNEP-WCMC 2021). Prior to use, this layer was repaired and dissolved without multi-part features; this resulted in 237,395 coral reef polygons ranging in size from less than 1 km^2 to 861.3 km^2.

For information on the protection status of these reef polygons, I used The World Database on Protected Areas (WDPA), which compiles information on marine and terrestrial protected areas into a singular spatial dataset (Figure 1). The WDPA dataset includes data as early as 1819 and is updated monthly by the United Nations Environment World Conservation Monitoring Centre based on national reporting (UNEP-WCMC and IUCN 2020). In addition to their location, size, and other characteristics, WDPA also includes the strength of protection within these areas based on the IUCN category and no-take status. Prior to analysis, I removed all protected areas that were “proposed” or “not reported”.

I also considered the extent to which coral reefs were deemed a marine "priority" by governmental and non-governmental organizations using the analysis by Gownaris et al. (2019) on priority consensus (Figure 1). Coral reef regions that were deemed a priority had the consensus of between one (a portion of the Pacific Ocean outside Hawaii’s EEZ) and seven (Galapagos Islands) priority initiatives. As mentioned above, these regions were identified based on criteria such as uniqueness, productivity, threatened species, and biodiversity, which may all impact tourism value (Gownaris et al. 2019).

For ecosystem service data, I used the Nature Conservancy's Atlas of Ocean Wealth (Spalding et al. 2016) (Figure 2). This database was created from previous studies and scientific reviews of tourism and recreation across the globe and includes information on services like carbon sequestration, storm protection, and recreation from 2005 to 2012. The tourism data itself comes from collected on-reef value from diving, snorkeling, and glass bottom boat tourism and from off-reef value from hotel room rates, beach views and advertisements (Spalding et al. 2017). Additional information was collected from World Travel and Tourism Council (Spalding et al. 2017). The dataset I will use provides information on the total annual tourism value per 500m^2 reef cell (Spalding et al. 2017) (Figure 1). The purpose of the database is designed to guide decision-making related to ecological and socio-economic risks.

For fishing, I will be using Global Fishing Watch, which uses vessel tracking data, including automatic identification systems, to determine where boats are traveling and fishing in real time at a spatial scale of 0.01° (Global Fishing Watch 2020) (Figure 2). It does this using computer algorithms to determine the type of ship, what type of fishing gear it is using, and locations of where the ship is fishing. Global Fishing Watch data are available from 2012-2016. For this analysis, I used the average fishing effort across these five years, as the impacts of high fishing intensity on coral reef ecosystems are likely to be cumulative and long-lasting rather than instantaneous.

Spatial Analysis

I ran all spatial analyses in ArcGIS Desktop 10.7. Before running spatial analysis, many of the datasets had to be cleaned and prepared. All datasets were projected into Eckert IV to ensure consistency, but distance calculations were run using geodesic methods.

The coral reef polygons were clipped and erased by the WDPA to create coral reef layers that were protected and unprotected, respectively; the resulting layers were then merged. Using WDPA dataset, I separated the data into coral reef polygons that were partly no take, polygons without any no take areas, polygons that were completely no take, and polygons with unreported or unknown no take areas. This process was repeated with the IUCN status based on WDPA, which was divided into 8 groups: Ia, Ib, II, III, IV, V, VI, and NA. Similarly, I clipped the coral reef polygons by layers representing each level of consensus, from level 1 to level 7, as defined by Gownaris et al. (2019).

To prepare the Global Fishing Watch (GFW) data, I used R statistical software to download annual datasets on fishing (2012-2013) and to create a raster that had aggregated data across all fishing fleets with a spatial scale of 0.1°. In ArcGIS, I replaced all no data cells with zeros then averaged the fishing effort across years to find the average value for each cell for this analysis I was interested in the distance to high fishing effort, which I defined as the top 75th

percentile of the data. I first excluded any values that had less than one hour of annual average fishing effort, then used quantile breaks to determine the 75^{th} (> 9.33 annual fishing hours) percentile cut-off. Once the high fishing effort cut-offs were generated, I used Euclidean distance to determine the distance between the coral reef centroids and the nearest cell that met the high fishing effort cut-off.

Once these spatial datasets were developed, I used the same approach to combine them with tourism value data. In these joins, reef polygons that contained information on no take status, current protection level, consensus level, IUCN status, and distance to high fishing effort were the target features. The global tourism raster data were spatially joined to these features. For each join, I added columns for the mean, maximum, minimum, sum, standard deviation, count, median, and mode, of the tourism values within the reef polygon (the target feature) or within a 100-meter distance of that polygon (to allow for small offsets in the data). All spatially joined tables were exported for further statistical analysis in R.

Statistical Analysis

Statistical analysis was conducted using RStudio version 3.6.0 with packages tidyverse (version 1.3.0), rstatix (version 0.7.0), and viridis (version 0.3.0). Since the data were zero-inflated, I used a two-step hurdle model for analysis; presence/absence of tourism value was modeled in the first step and magnitude of tourism value was modeled in the second step.

First, I separated my data into two groups, polygons with tourism value (USD > $0) and polygons without tourism value (USD = $0 or NA) to determine whether my predictors impacted the likelihood of tourism presence. I used a binomial generalized linear model with a logit link to determine whether the predictors of interest and/or coral reef polygon size impacted the likelihood of tourism value. The models I tested included the null model (no predictors), the predictor I was interested in (e.g., protected or not protected, distance to fishing), coral reef polygon size, and additive or multiplicative combinations of these terms. After running these models, I used the Akaike information criterion (AIC) to determine the best fitting model for each of the parameters. I defined the best model as the model with the lowest value of AIC; if two models were within 2 AIC of each other, I chose the simpler model.

Second, I created models to determine if any of my parameters of interest predicted the magnitude of tourism value in areas that had value. Since tourism value and coral reef area were right-skewed, I used a logarithmic transformation on the dataset to approximate a normal distribution prior to analysis. For this step, I used an ANOVA to test the significance of the predictors I was interested in (protection, level of protection, consensus), followed by a pairwise t-test when the results of the ANOVA were significant. For the protected vs. unprotected prediction, I just used a t-test, since there were only two groups.

Because distance to fishing was a continuous predictor rather than a categorical predictor, I used a regression to examine the relationship of distance to high fishing and tourism value magnitude using the gam function in the mgcv package (1.8-34). As with the binomial models, I ran all combinations of predictors. I also tested for linear and non-linear effects of distance to fishing by comparing the fit of regular linear models to those of linear models with smooth (non-linear) terms with gam. As with the binomial models, I used AIC to assess model fit.

Spatial Autocorrelation

I used GeoDa to determine if there was spatial autocorrelation impacting my analysis for average fishing effort and tourism value. The spatial autocorrection analysis was run the same way for both datasets. I started the analysis by constructing spatial weights, which contain information about the value of the target parameter for neighbors of each polygon. I used the number of nearest neighbors' method to run spatial autocorrelation analysis. While some polygons had several other polygons in the immediate vicinity, others did not have a close neighbor. We therefore ran the analysis above three times, using two, four, and eight neighbors, to see if the resulting spatial autocorrelation statistics were sensitive to number of neighbors. After selecting the number of nearest neighbors, GeoDa constructs spatially lagged variables to calculate the Moran's I statistic. Moran's I is a measure of spatial autocorrelation that is based on how similar the coral reef polygon's tourism or distance to fishing effort is compared to its immediate neighbors. To determine significance, I ran 999 permutations to find the distribution of the Moran's I statistic and compare our value to that distribution. The permutations act as the null hypothesis by calculating the Moran's I value distribution under the assumption that the data are spatially random. I also created Univariate LISA maps were then to see if Moran's I value changed over space. This process helps to identify whether spatial autocorrelation is widespread or confined to a few local clusters.

RESULTS

My analysis relied on 237,395 coral reef polygons across the globe with an average area of 0.63 km^2. The average tourism value was $24.42 annually ($24.42 ± $343.93 USD), with a range of $0 to $53,987 per 500-meter cell. The total annual fishing effort ranged from 0 to 27389.7 hours per year. The distance from coral reef polygons to high fishing effort ranged from 0 to 2,793,663.5 kilometers.

Protection Status

The model that best predicted the presence or absence of tourism included coral reef area and protection status without an interaction term (ΔAIC = 0.44). The odds of having tourism value were 1.22 times higher for protected coral reefs than those that were not protected. For both protected and unprotected reefs, the odds of having tourism value decreased by 1.27 for each 1 km^2 increase in reef size. Among areas with tourism value, coral reef areas that were protected (129.36 ± 26.83) had a higher average value than those that were not (108.59 ± 19.70) ($n1 = 3735$, $n2 = 1949$, $df = 3874$, $t = -7.84$, $p < 0.001$) (Figure 3).

Strength of Protection

The first indicator of strength of protection was the no-take status of the reef. When strength of protection was based on no-take status, the model that best predicted the presence or absence of tourism value was the model that included no-take status and the interaction between coral reef area and no-take status (ΔAIC = 5.47). Partly no-take reefs had the lowest likelihood of tourism value, while reefs without no-take area had the highest. When compared to those that were fully no-take, partially no-take reefs had 1.80 times higher odds of having tourism value and reefs without a reported no-take status ("No Data") had 1.58 times higher odds of having tourism value. Areas that were reported as having no no-take area had 1.79 times lower odds of having value than those that were fully no-take. Area did not impact the odds of having tourism value in fully, partially, or unreported no-take coral reefs. However, within reefs without no-take area, the odds of having tourism value increased by 1.01 with each 1 km^2 increase in total area.

Among areas with tourism value, no take status impacted the magnitude of value ($dfd = 2494$, $dfn = 3$, $F = 4.31$, $p = 0.005$). Areas that were fully no-take ($n1 = 257$, $n2 = 416$, $t = 2.79$, p

=0.005) or that were without no-take area ($n1 = 258$, $n2 = 416$, $t = 3.24$, $p = 0.001$) had a significantly higher average tourism value than partially no-take area (Figure 4).

I also used the IUCN protected area categorization system (seven categories, ranging from Ia, the strongest protection, to VI, the weakest protection) as an indicator of strength of protection. The best supported model was the full model, which included the interaction between coral reef area and IUCN level, but the model did not converge, so I chose to use the additive model, which was the next best supported model ($\Delta AIC = 113.01$). The IUCN status had a significant impact on tourism value ($p < 0.001$). With the exception of IUCN Ib, which had 5.07 times lower odds, all IUCN categories were more likely than IUCN Ia to have tourism value. The areas with the highest likelihood of value were IUCN II and IUCN III, which had 3.35- and 4.83-times higher odds, respectively, than IUCN Ia. I also found that across all IUCN categories, each 1 km^2 increase in area lead to 1.01 times higher odds of having tourism value (Figure 5).

Among coral reef areas that have tourism value, IUCN II had the highest average tourism value (70.55 ± 4.27) while IUCN VI had the lowest average value (24.73 ± 1.66). Average tourism value was significantly impacted by IUCN status. Coral reefs with a protection status of IUCN II had a significantly higher average tourism value than all other categories ($p < 0.001$ in all cases; Table 3), with the exception of IUCN level III, for which the difference was not significant ($n1 = 3168$, $n2 = 59$, $t = 1.58$, $p = 0.95$). Coral reefs with IUCN level VI protection had a significantly lower average tourism value than all other categories except for IUCN 1b ($n1 = 3805$, $n2 = 61$, $t = 1.57$, $p = 0.95$). The only other significant difference was between IUCN Ib and IUCN III; IUCN Ib had a significantly lower average tourism value than IUCN III ($n1 = 61$, $n2 = 59$, $t = -3.20$, $p = 0.032$) (Table 3).

Level of Priority Consensus

Coral reef consensus level varied from one to seven overlapping priority initiatives. The model that best predicted the presence or absence of tourism value included coral reef area, consensus level, and an interaction between area and consensus level. With the exception of consensus level 5 ($p = 0.44$), all consensus levels had a higher likelihood of having tourism value than did consensus level 1 reef areas. The highest was among consensus level 7 areas, which had 14.8 times higher odds of having tourism value than consensus level 1 areas. As compared to consensus level 1 areas, consensus level 2 areas had 1.21 times higher odds, consensus level 3 areas had 1.91 times higher odds, consensus level 4 areas had 1.89 times higher odds, and consensus level 6 areas had 1.52 times higher odds. Therefore, the odds increased between consensus levels 1 and 3, declined slightly, then increased again for consensus level 7. Larger reef areas had a higher likelihood of having tourism value; each 1 km^2 increase in area lead to an increase of 1.01 times higher odds of having value (Figure 6).

Interestingly, the findings regarding the magnitude of tourism value were dissimilar to those regarding the presence or absence of tourism value. While areas with a consensus level of 1 were less likely to have any tourism value than others, among areas with tourism value they had among the highest average value. Consensus level 4 areas also had high average tourism values, while the lowest average values were seen among consensus level 5 and consensus level 6 areas (Figure 6).

Consensus level also impacted the magnitude of tourism value among areas with value ($dfd = 33366$, $dfn = 6$, $F = 197$, $p < 0.001$). All pairwise comparisons were significant ($p < 0.01$), with the exception of those involving consensus level 7 and the comparison of consensus levels 1 and 4 ($p = 1$) and 5 and 6 ($p = 1$). Consensus level 7 had a very small sample size ($n = 7$ reef polygons) and only contained reef areas in the Galapagos Islands.

Distance to Fishing

The model that best predicted presence or absence of tourism value based on distance to high fishing (top 75th percentile of annual fishing effort) was the additive model (ΔAIC = 1.99). Being farther away from high fishing effort increased the likelihood of having tourism value ($p < 0.001$), while being a larger coral reef area decreased the odds by 1.21 for every 1 km^2 increase in area. However, the odds ratio associated with distance to fishing was very small; the odds of having value increased 1.001 per every 1 km^2 increase in distance.

Among coral reef areas with tourism value, the best-supported model included coral reef area and a non-linear (smoothed) relationship with distance to fishing. Smaller coral reef polygons had an overall higher tourism value ($b = -0.23$, $p < 0.001$) (Figure 7). However, the relationship is non-linear as tourism value as distance to finish effort decreases until approximately 200 km, then the tourism value increases as the coral reef polygons gets farther away from high fishing effort.

Spatial Autocorrelation

It is important to note that there was spatial autocorrelation in the tourism data and fishing data (Table 1, Table 2). For both datasets, the spatial autocorrelation results were similar whether I used two, four, or six nearest neighbors to calculate weights. Distance to high fishing is inherently spatially autocorrelated (areas closer to each other will have a more similar distance to a high fishing pixel), and a result the Moran's I value was always 1. Therefore, there were both high-high and low-low clusters of spatial autocorrelations for the distance to fishing parameter. Tourism data were also spatially autocorrelated, with a Moran's I of 0.48-0.49 depending on the number of neighbors. This autocorrelation was driven by clusters of high-high tourism value.

DISCUSSION

Overall, the results supported my hypotheses that protection would increase tourism value, though some of the results were unexpected (e.g., tourism value initially was higher in areas that were closer to high fishing effort). In general, these findings agree with past studies that found 71% of MPAs positively influenced fish populations, but that the success of these protected areas varied with their management strategy (Gill et al. 2017; Edgar et al. 2014). It is also known that areas of high biodiversity attract tourism, because divers and other visitors are drawn to the diverse marine species in these areas (Chung et al. 2018; Giglio 2015). However, six million people worldwide depend on coral reef fisheries for their livelihood, and these fisheries provide an important source of protein for more than 400 million people (Cinner 2014). Though sustainable tourism jobs can replace some traditional marine-related jobs like fishing and aquaculture, fisheries will remain vital because they provide protein and nutrients that local communities may not get otherwise (Pham 2020, Cinner 2014). It is important to recognize the trade-offs present within MPAs when determining the best management policies, while also meeting the nutritional needs of local communities.

Protection Status

I found that coral reef regions that were protected had a higher chance of having tourism value and significantly higher tourism values than those that were not protected. This result supports previous studies that found protected areas provide ecosystem benefits, like fish and coral population restoration and resiliency, as well as socio-economic benefits, like tourism (Selig and Bruno 2010; Hargreaves-Allen et al. 2017). Tourism also plays a large role in generating a large portion of the funding needed to support local conservation efforts (Atmodjo et al. 2017; Coke-Hamilton 2020). Collectively, these studies suggest that that, when marine protected areas are managed correctly, they can provide other socio-economic benefits to support the local economy and offset the short-term loss of revenue due to reduced fishing.

When examining how the strength of protection impacts the likelihood and magnitude of coral reef tourism value, I found some results that deviated from what was expected and from previous studies. Previous studies have found that when MPAs are larger than 100 km^2, strictly protected (completely closed to fishing), and are being effectively managed they will provide the highest ecological and socio-economic benefits (Edgar et al. 2014). A recent study found that MPAs that were managed mainly for biodiversity conservation had almost 35% more visitors to their marine ecosystem than those that were managed for mixed-use (Chung et al. 2018).

However, I found that areas that were completely closed to fishing had the lowest likelihood of having tourism value. A possible explanation of the result is that some areas that do not allow fishing may also not allow tourism, since the level of protection is so strict. Some no-take areas may also be IUCN Ia or Ib MPAs, the other indicator of protection strength that I used in this study. IUCN Ia areas are defined as "strictly protected areas…where human visitation, use, and impacted are strictly controlled and limited" (Day et al. 2019). Often in these areas, the only allowable activities are scientific research or removal of invasive species (Day et al. 2019). Accordingly, IUCN Ia and Ib coral reefs were the least likely to have tourism value among all reef areas; IUCN III coral reefs, for example, were almost five and 25 times as likely to have value than were IUCN Ia and IUCN Ib reefs, respectively. As compared to IUCN Ia and Ib areas, IUCN III areas "should also provide visitation, non-extractive recreational activities, and natura tourism" and are explicitly defined as areas with "high visitor value".

It is important to note that, although all IUCN Ia and Ib areas should be fully no-take based on definition, not all fully no-take areas need be IUCN Ia or Ib areas. An example of an area that is fully no-take and has high tourism value is a large portion of the Great Barrier Reef in Australia, which is a world-renowned tourism location as well (UNEP – WCMC 2021).

Among areas with value, areas that were completely closed or completely open to fishing had comparable average tourism values that were higher than those in partly no-take protected reefs. Interestingly, this is in direct contrast with IUCN status, where the values that were in the intermediate levels on the protection scale had the highest tourism values.

The coral reef regions that had the highest level of tourism value were IUCN level II, which is a National Park Designation. As suggested above, the major objective of a National Park is to protect natural biodiversity and its ecological structure, but also to be a place for education and tourism to support the local economy (IUCN 2020). The next highest tourism values were associated with IUCN level V reefs, which are protected seascapes. A study on terrestrial ecosystems found that IUCN levels Ia, Ib, and V all had a relatively large positive ecosystem effects when compared to other IUCN categories (Coetzee et al. 2014); IUCN V may therefore have a similarly high positive impact in marine ecosystems without the strict restrictions on tourism in IUCN Ia and IUCN Ib areas. This also makes sense because the goal of the IUCN V designation is to encourage the sustainable interaction between people and nature, thus providing a place for recreation and tourism to occur (IUCN 2020).

These results are encouraging; areas that are set aside to conserve biodiversity while also encouraging sustainable human interaction with this biodiversity seem to be doing their job in terms of supporting tourism. Though there have been no studies on MPAs that explicitly examine biodiversity and IUCN category, some studies of terrestrial protected areas suggest that stricter IUCN categories have greater ecosystem benefits (Coetzee et al. 2014). In terrestrial areas it was found that being in an IUCN protected area of any level showed a statistically significant higher species richness and abundance when compared to areas that are not protected (Gray et al. 2016). Therefore, an effective global MPA network should include some stricter protections that maximize ecosystem benefits and intermediate IUCN category areas that maximize socioeconomic benefits like tourism.

Consensus

Next, I looked at consensus level, or the number of organizations agreeing that a marine region should be considered a conservation priority, as an indicator of the ecological value of these areas. The commonly shared criterion that the UN and NGO initiatives considered were threatened species, vulnerability, biodiversity, and productivity, with eight, six, five and four of ten initiatives, respectively (Gownaris et al. 2019). In theory, these are factors that are also likely to influence tourism value, so I expected that higher tourism values would be associated with higher consensus. On the other hand, an area with a high percentage of threatened species and vulnerability, as accounted for in most of the UN and NGO initiatives, could imply a degraded habitat which is known to have negative repercussions for the tourism value (Uyarra 2005).

The relationships between priority consensus and ecosystem services are therefore likely to be complex. I found that the highest tourism values were seen among consensus level 1 and 4 areas and the lowest were seen among consensus level 5 and 6 areas. This complexity may explain why I found that higher consensus areas more likely to have value but were not necessarily associated with higher tourism value.

The areas of consensus that I used in this study have also been suggested as possible candidates for future protection. It is therefore important to also understand the socioeconomic benefits provided by these regions in terms of tourism. For example, a global study found that biodiversity and tourism values were positively correlated and that, with proper management, maintaining both high biodiversity and high tourism value is possible (Chung et al. 2018; Gray et al. 2016).

Fishing

In addition to examining spatial protection as an indicator of reduced extractive threats, I also examined how proximity to areas of high fishing intensity influenced tourism value. Fishing can have negative impacts on fish communities, including the reduction of larger and older fish in the population and loss of marine megafauna like sharks, and can reduce the aesthetic value of reef and other complex habitats (McCook et al. 2010). Larger fish species and individuals are also those sought after by recreational divers, and therefore abundance of these species is likely to increase tourist satisfaction (Giglio et al. 2015).

I found that reefs that were closer to high fishing effort were less likely to have tourism value. In coral reef polygons that had tourism value, the relationship between value and proximity to high fishing effort was non-linear with a parabolic relationship. Reefs very close to high fishing effort had high tourism value, but tourism value then decreased until the point of inflection, around a distance of 200 km. After this point, tourism value and distance to high fishing were positively related.

Previous studies have found that primary production is related to global fisheries catches, and areas with high productivity can be a driver for higher tourism values since density of fish is found to attract more tourism (Chassot et al. 2010; Viana et al. 2017). This could explain the initial correlation of close proximity to high fishing effort and an increase in overall tourism value. However, it has been argued that, if done correctly, tourism could have higher and more sustained economic value than fisheries, furthering the demand for proper management plans and conservation measures (Viana et al. 2017). Local communities have also benefitted when individuals are working in both the local fisheries and tourism sectors but relying on tourism alone is not guaranteed to increase an individual's income (Pham 2014).

Caveats and Future Studies

Although this study provides evidence for socioeconomic trade-offs associated with coral reef ecosystem services, there are several limitations to the datasets used. In addition, future work would be needed to consider factors not integrated in this study (e.g., other impacts, spatial autocorrelation) and to inform management at the local scale.

Although I used the highest-quality datasets that exist at a global level for the ecosystem services examined, these datasets still have limitations. The time span covered by the tourism values and fishing effort data did not match up completely. The tourism data were collected for 2008-2013 for 220 coastal countries and territories, whereas the global fishing watch data was from 2013 to 2016. Global fishing watch collects movement patterns using satellite data from fishing vessels in real time to calculate the fishing effort. The dataset does not capture all fishing, as some fishing vessels turn off their satellite tracker while fishing illegally (Global Fishing Watch 2020). Artisanal or small-scale fishing also may not be fully captured in the dataset.

Due to the availability of open-source datasets, this study can be replicated to look at other ecosystem service trade-offs or look deeper into the services that were examined in this study. There are additional important non-extractive ecosystem services provided by coral reefs, like storm protection, and I only looked at tourism value. Similarly, coral reef ecosystems are subject to many threats that may reduce their tourism value (e.g., bleaching, pollution), and I only examined fishing. Coral bleaching and other coral reef degradation are likely to reduce tourists' interest in returning to a region and therefore loss of tourism value (Uyarra et al. 2005).

I was also unable to consider the effectiveness of the marine protected areas. One of the major predictors of the success of an MPA is staff and budget capacity (Gill et al. 2017). Another essential piece of establishing the most successful MPAs are the age of the protected areas. MPAs that have been in place for longer than 10 years have been found to be more successful in meeting ecological and socio-economic goals than areas that have just been recently designated as protected (Edgar et al. 2014). In addition, it is important to understand the fishing gear types have disproportional effects on marine ecosystems, and thus on tourism values. Future studies could include type of fishing effort in their models,

Additional research into the management of current MPAs would give insight on whether they are providing the maximum ecological and socio-economic goals as management plans will vary due to local needs. Not all MPAs are used to generate economic benefits from tourism. These trade-offs need to be further studied locally, to decide what is important to the people who will be utilizing these areas. This study is the first step in researching the interactions between ecosystem services that will be helpful in choosing what areas we need to protect in the future and what types of protection will most benefit local communities, thereby working towards SDG 14 Target 7.

We found that there was spatial autocorrelation between the tourism value and distance to high fishing effort. This means that if coral reef polygons that are closer together, they have a higher chance of having similar tourism values and distances to higher fishing effort. In the example of distance to high fishing effort, neighbors will inherently have similar distances because they are located in similar spatial location. To improve on this work, future studies should account for this spatial autocorrelation in its statistical models. Additionally, correlations between fishing and tourism may be driven more by confounding variables than by trade-offs. For example, marine areas that are biodiversity hotspots or have high productivity may influence both tourism and fishing value (Chassot et al. 2010). Similarly, I found marine protected areas can improve tourism value, but other studies have found that they also lead to higher fishing effort due to spillover effects (Lenihan et al. 2021). More insight will be needed to determine why these trends have emerged and to account for these confounding variables.

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FIGURES

Table 1. Spatial Autocorrelation results of the tourism data from Atlas of Ocean Wealth data.

Number of Nearest Neighbors	Moran's I	Significance
2	0.484	$p < 0.001$
4	0.496	$p < 0.001$
6	0.482	$p < 0.001$

Table 2. Spatial Autocorrelation results of distance to high fishing effort from Global Fishing Watch Data.

Number of Nearest Neighbors	Moran's I	Significance
2	1.000	$p < 0.001$
4	1.000	$p < 0.001$
6	1.000	$p < 0.001$

Table 3. The no take pairwise t-test results that shows significant differences between the different level pairs.

Group 1	Group 2	n1	n2	statistic	df	Significance
All	NoData	257	1567	1.97	365.95	$p = 0.05$
All	Part	257	416	2.79	545.15	$p = 0.03$
NoData	None	1567	258	-2.50	365.27	$p = 0.05$
None	Part	258	416	2.24	543.13	$p = 0.01$

Table 4. The IUCN level pairwise t-test results that show significant differences between two different level pairs.

Group 1	Group 2	n1	n2	statistic	df	Significance
Ia	II	636	3168	-7.22	1015.13	$p < 0.001$
Ia	IV	636	3805	4.44	932.96	$p < 0.001$
Ib	II	61	3168	-5.34	62.85	$p < 0.001$
Ib	III	61	59	-3.20	113.60	$p = 0.032$
II	IV	3168	1699	11.36	3571.34	$p < 0.001$
II	V	3168	1892	9.98	4040.27	$p < 0.001$
II	VI	3168	3805	18.61	6680.99	$p < 0.001$
IV	VI	1699	3805	3.83	3285.48	$p = 0.003$
V	VI	1892	3805	5.68	3758.19	$p < 0.001$

Table 5. The consensus level pairwise t-test results that show significant differences between two different level pairs.

Group 1	Group 2	n1	n2	statistic	df	Significance
1	2	11462	13997	19.66	24290.16	$p < 0.001$
1	3	11462	5448	7.51	10744.76	$p < 0.001$
1	5	11462	670	18.69	866.15	$p < 0.001$
1	6	11462	398	16.95	487.19	$p < 0.001$
2	3	13997	5448	-7.72	9793.76	$p < 0.001$
2	4	13997	1391	-7.97	1630.72	$p < 0.001$
2	5	13997	670	9.65	822.53	$p < 0.001$
2	6	13997	398	8.84	467.36	$p < 0.001$
3	4	5448	1391	-3.66	2047.27	$p = 0.002$
3	5	5448	670	13.24	1100.32	$p < 0.001$
3	6	5448	398	12.20	593.95	$p < 0.001$
4	5	1391	670	12.94	1953.99	$p < 0.001$
4	6	1391	398	12.37	1248.30	$p < 0.001$

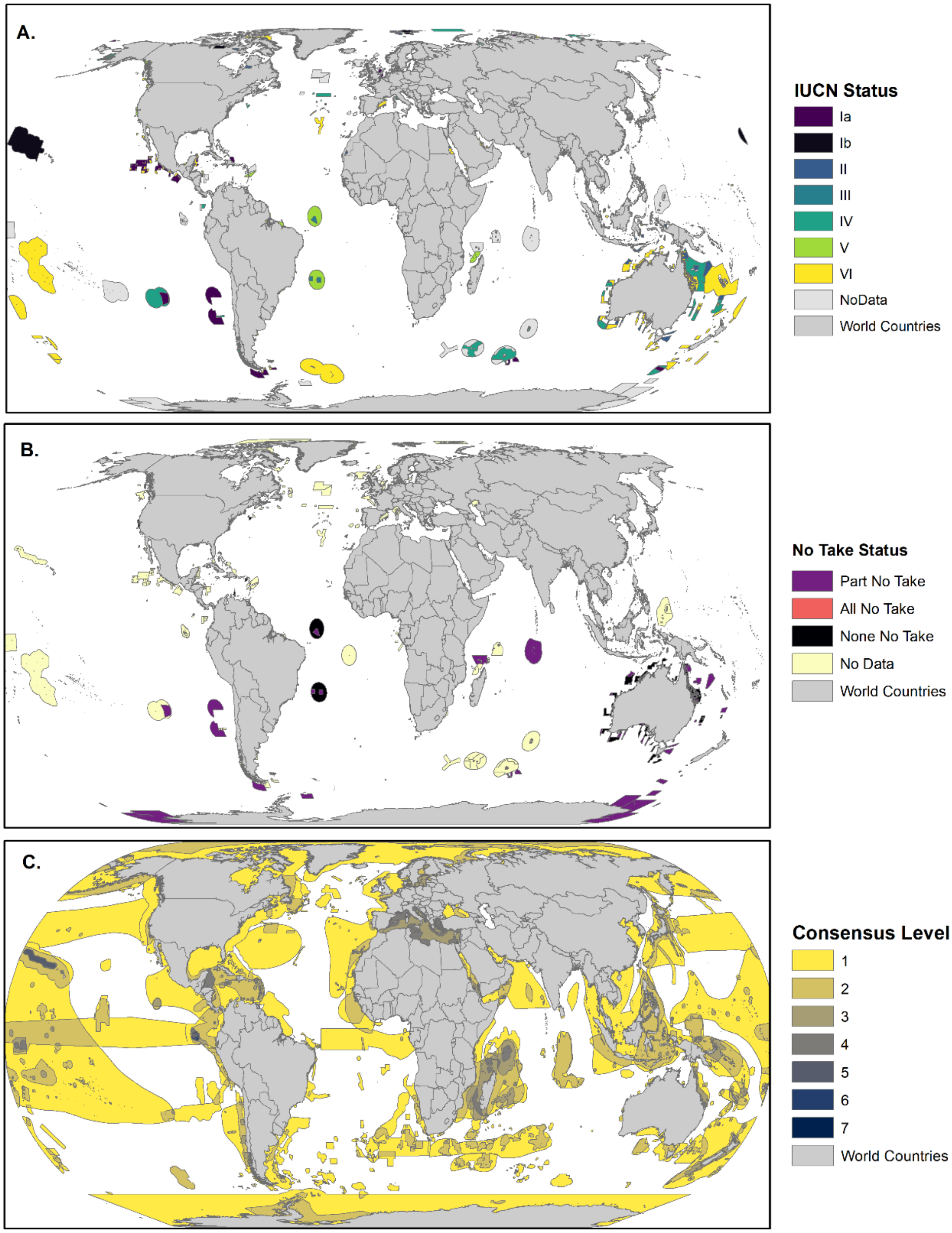

Figure 1. World Database on Protected Areas dataset as information on protection status from 1819 and is updated monthly by the United Nations Environment World Conservation Monitoring Centre based on national reporting (UNEP-WCMC and IUCN 2020). WDPA has information on IUCN Status (**A**) and no-take status (**B**). **C**. Coral reefs that were deemed a marine "priority" region by 10 governmental and non-governmental organizations using the analysis by Gownaris et al. (2019).

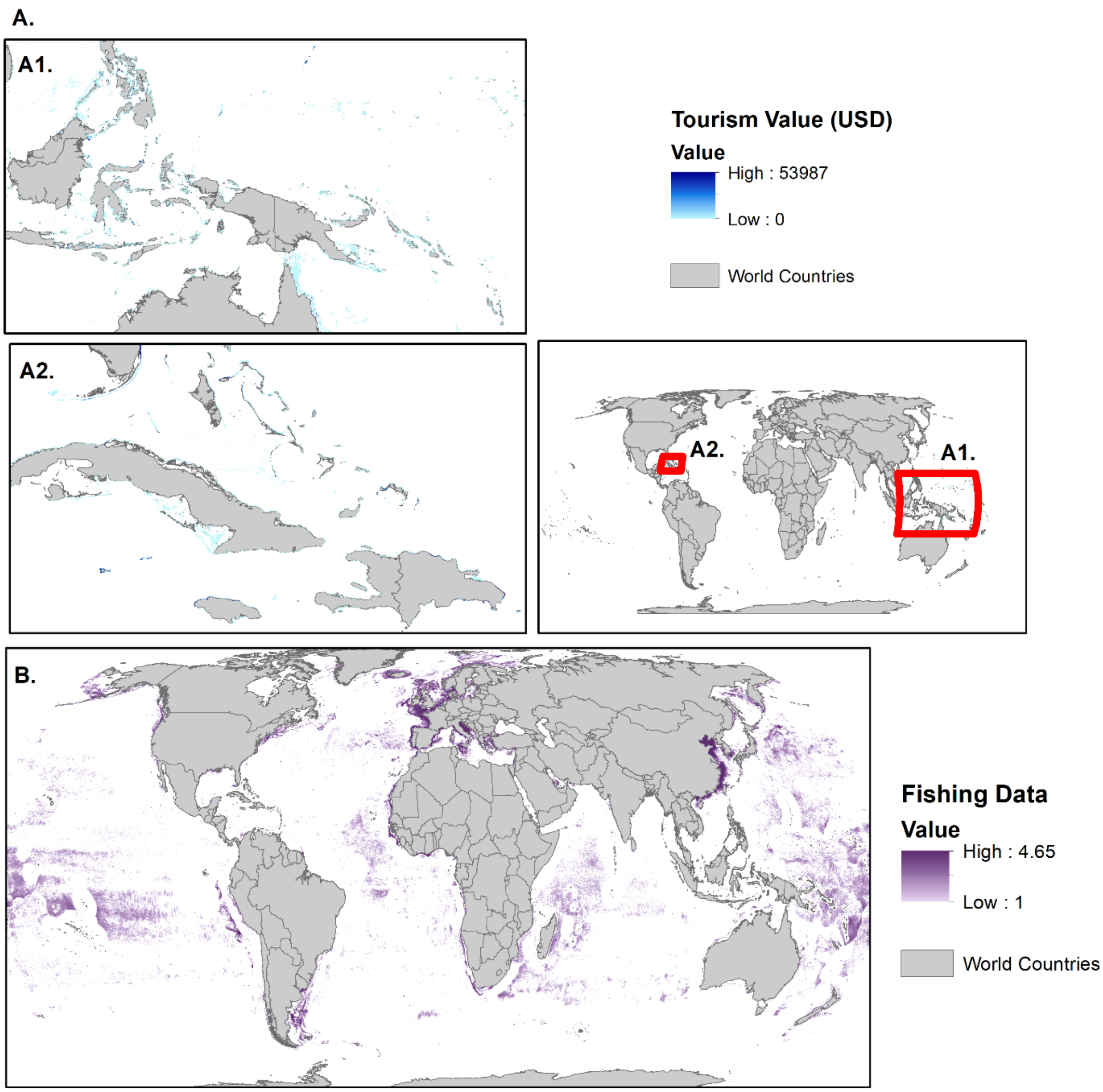

Figure 2. A. Nature Conservancy's Atlas of Ocean Wealth created from previous studies and scientific reviews of tourism and recreation across the globe and includes information on services like carbon sequestration, storm protection, and recreation from 2005 to 2012 (Spalding et al. 2016). **B**. Global Fishing Watch vessel tracking data from 2012 to 2016, including automatic identification systems, to determine where boats are traveling and fishing in real time at a spatial scale of 0.01° (Global Fishing Watch 2020). GFW uses computer algorithms to determine the type of ship, what type of fishing gear it is using, and locations of where the ship is fishing.

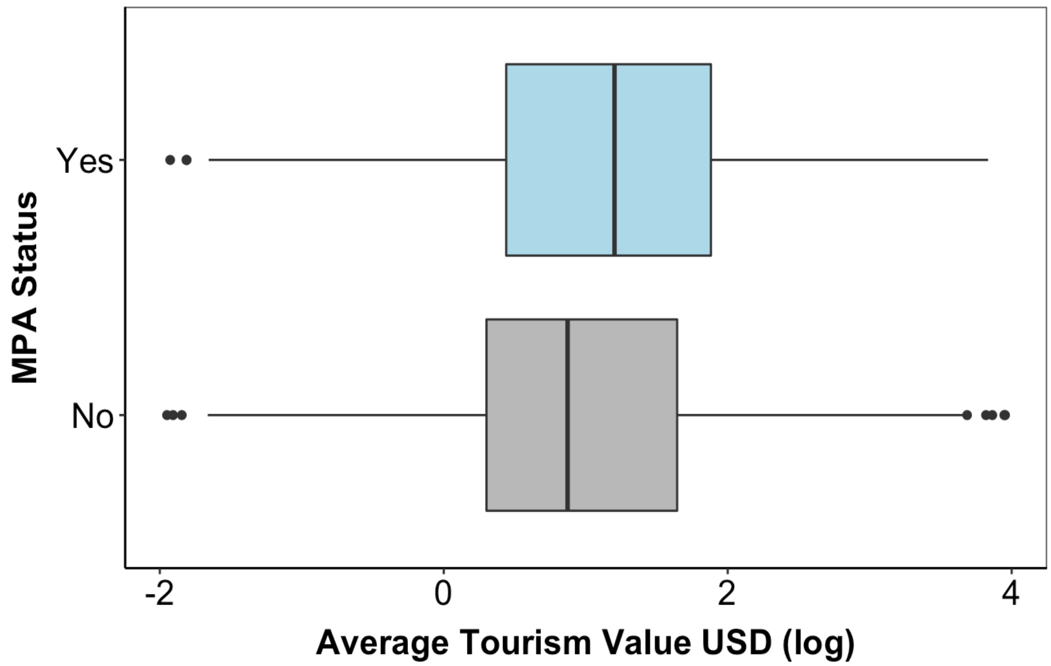

Figure 3. Log-transformed average tourism value (USD) was higher in coral reefs in a marine protected area (n = 3735) than those that were not protected (n = 1949) (t = 7.84). Protection was based on the World Database on Protected Areas and included only designated marine protected areas (UNEP-WCMC and IUCN 2020). Tourism value was based on the Atlas of Ocean Wealth (Spalding et al. 2016). Regions without tourism value were removed prior to analysis.

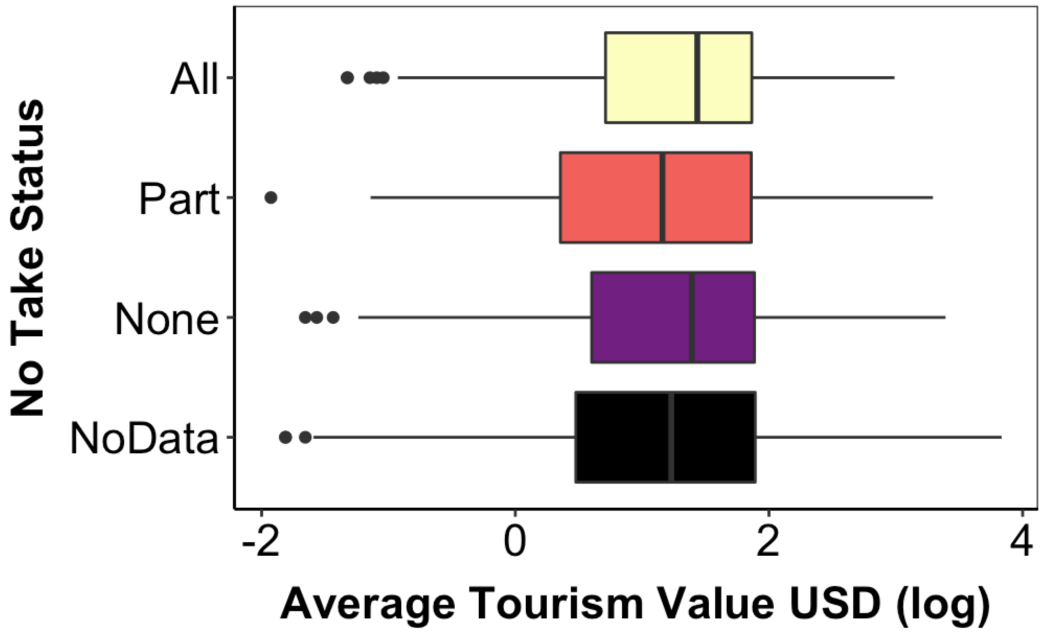

Figure 4. Log-transformed average tourism value (USD) was higher in coral reefs in all no take (n = 257) and none no take coral reef polygons (n = 258) than those that were part no take (n = 416). No take status was based on the World Database on Protected Areas and included only designated marine protected areas (UNEP-WCMC and IUCN 2020). Tourism value was based on the Atlas of Ocean Wealth (Spalding et al. 2016). Regions without tourism value were removed prior to analysis.

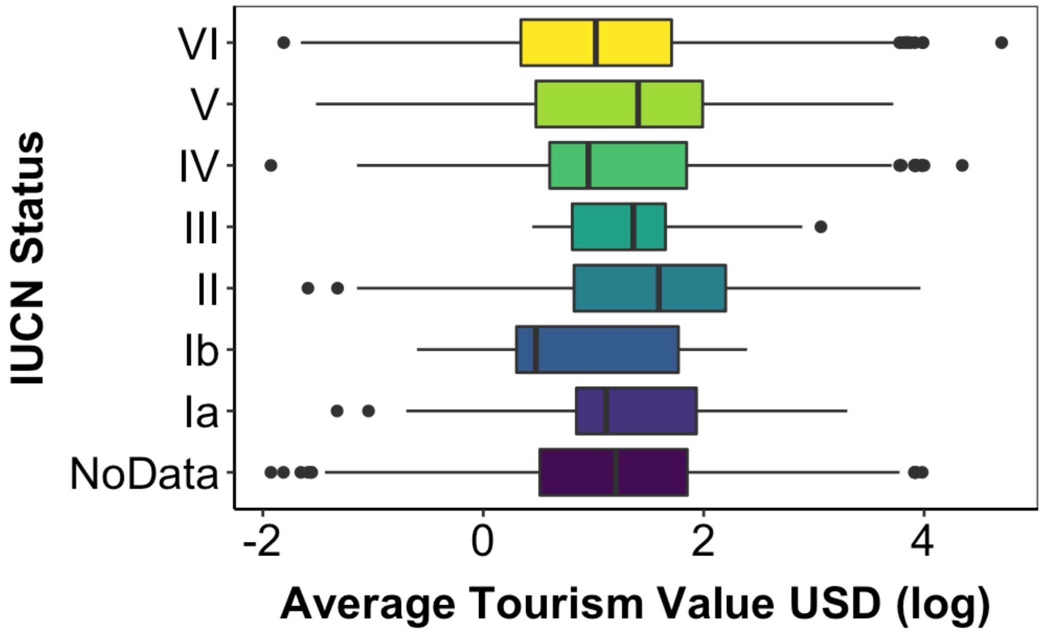

Figure 5. Log-transformed average tourism value (USD) was higher in coral reefs with IUCN levels in the intermediate range, than those with an IUCN status on the extremes. IUCN status was based on the World Database on Protected Areas and included only designated marine protected areas (UNEP-WCMC and IUCN 2020). Tourism value was based on the Atlas of Ocean Wealth (Spalding et al. 2016). Regions without tourism value were removed prior to analysis.

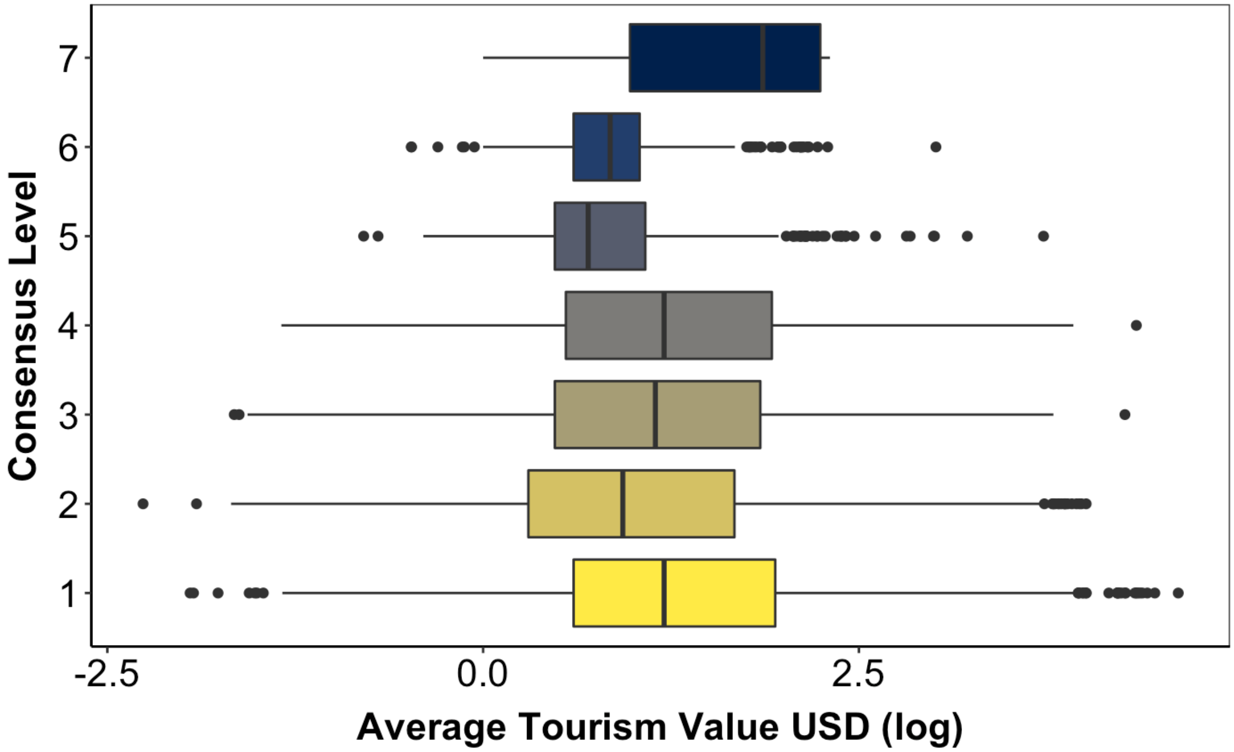

Figure 6. Log-transformed average tourism value (USD) was higher in coral reefs with consensus levels 1-4 and 7 than those with consensus levels 5 and 6. Consensus level was defined as overlap of marine priority areas among 10 UN and NGO initiatives (Gownaris et al. 2019). Tourism value was based on the Atlas of Ocean Wealth (Spalding et al. 2016). Regions without tourism value were removed prior to analysis.

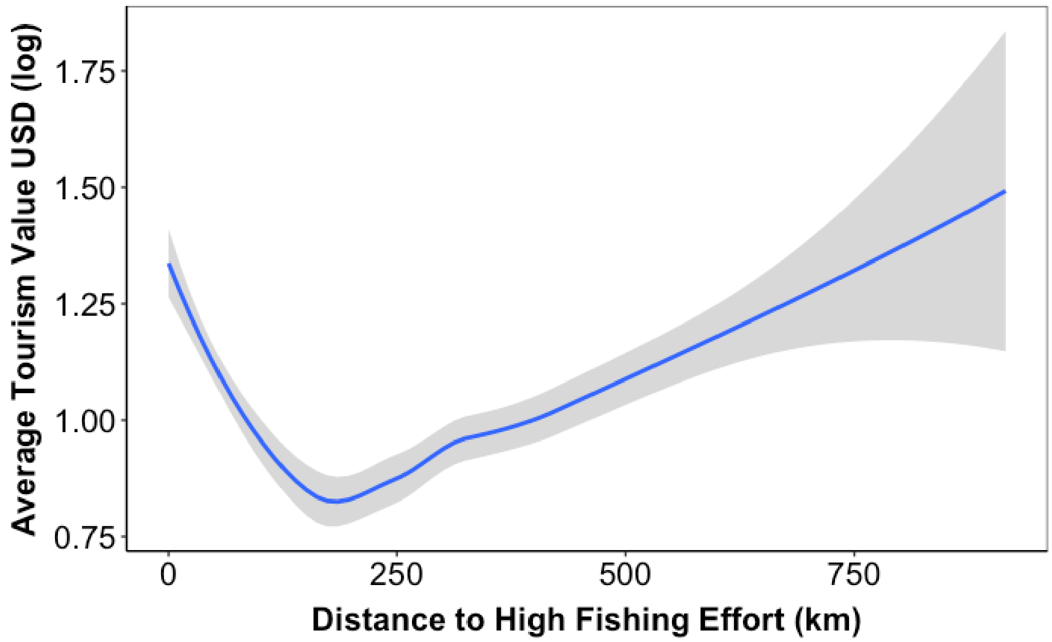

Figure 7. Log-transformed average tourism value (USD) was higher in coral reef areas that were closer to high fishing effort, then decreased until approximately 200 kilometers, and then increased with distance to high fishing areas. High fishing effort was defined by the top 75th percentile of annual fishing effort by global fishing watch (Global Fishing Watch 2020). Tourism value was based on the Atlas of Ocean Wealth (Spalding et al. 2016). Regions without tourism value were removed prior to analysis.