UNIVERSITY OF CALIFORNIA

Santa Barbara

Management and design of marine reserves and rights-based systems in small-scale fisheries

A dissertation submitted in partial satisfaction of the requirements for the degree Doctor of Philosophy in Environmental Science and Management

by

Daniel Fadigas Viana

Committee in charge:

Professor Steven D. Gaines, Chair Professor Christopher Costello

Professor Benjamin Halpern

June 2018

ProQuest Number: 10815941

All rights reserved

INFORMATION TO ALL USERS The quality of this reproduction is dependent upon the quality of the copy submitted.

In the unlikely event that the author did not send a complete manuscript and there are missing pages, these will be noted. Also, if material had to be removed, a note will indicate the deletion.



ProQuest 10815941

Published by ProQuest LLC (2018). Copyright of the Dissertation is held by the Author.

All rights reserved. This work is protected against unauthorized copying under Title 17, United States Code Microform Edition © ProQuest LLC.

> ProQuest LLC. 789 East Eisenhower Parkway P.O. Box 1346 Ann Arbor, MI 48106 – 1346

The dissertation of Daniel Fadigas Viana is approved.

Christopher Costello

Benjamin Halpern

Steven D. Gaines, Committee Chair

May 2018

Management and design of marine reserves and rights-based systems in small-scale fisheries

Copyright © 2018

by

Daniel Fadigas Viana

ACKNOWLEDGEMENTS

I would first like to dedicate my dissertation to my family, Talita and Marina, and thank for for the unconditional love and support during my PhD. Second, I would like to thank Steve Gaines, my advisor, for all the incredible discussions and support throughout these years. Third, I would like to thank my committee members, Christopher Costello and Benjamin Halpern, for helping make my dissertation research as impactful as possible. Lastly, I would like to thank my friends and lab mates for tremendous feedback and discussions throughout the years.

VITA OF DANIEL FADIGAS VIANA May 2018

EDUCATION

Bachelor of Science in Fisheries Engineering, Federal Rural University of Pernambuco, Recife, Brazil, June 2011 (summa cum laude) Master of Science in Environmental Science and Management, University of California, Santa Barbara, June 2013 Doctor of Philosophy in Environmental Science and Management, University of California, Santa Barbara, May 2018

PUBLICATIONS

Viana D.F., Halpern B, Gaines S.D. 2017. Accounting for tourism benefits in marine reserve design. PLoS ONE. 12(12): e0190187"

Viana D.F., Gornik K., McDonald G., Lin C., Ng N.S.R., Quigley C., Potoski M. 2017. Recreational boaters value biodiversity: The case of the California Channel Islands National Marine Sanctuary. Marine Policy. 81: 91-97.

Ayres A, Degolia A, Fienup M, Kim Y, Sainz J, Urbisci L, Viana D, Wesolowski G, Plantinga AJ, Tague C. 2016. Social Science/Natural Science Perspectives on Wildfire and Climate Change. Geography Compass. 1;10(2):67-86.

Viana D.F. Hazin F.H., Oliveira P.G. 2016. Reproductive Biology of the lane snapper, *Lutjanus synagris*, caught in the coast of Pernambuco, Northeastern Brazil. Arquivos de Ciencias do Mar, 48(2):67-73.

Viana D.F., Hazin F.H, Andrade H. 2015. Fisheries in the St Peter and St Paul Archipelago: 13 years of monitoring, Brazil. Instituto de Pesca, 41: 239-248.

Viana D.F., Camargo E., Dutra G. 2015. Avaliação econômica da pesca do camarão setebarbas, *Xiphopenaeus kroyeri* (Keller, 1862), no município de Caravelas – BA, Brasil. Instituto de Pesca, 41(2): 419-428.

Mendonça, S. A., Macena, B. C. L., Creio, E., Viana, D. L. & Hazin, F. H. 2012. Record of a pregnant Mobula thurstoni and occurrence of Manta birostris (Myliobatiformes: Mobulidae) in the vicinity of Saint Peter and Saint Paul Archipelago (Equatorial Atlantic). Pan-American Journal of Aquatic Sciences, 7(1): 21-26.

AWARDS

CAPES fellowship, University of California, Santa Barbara, 2013

Dr. Daniel Vapnek Scholarship for Sustainable Fisheries Research, University of California, Santa Barbara, 2013

Latin American Fisheries Fellowship, University of California, Santa Barbara, 2012

FIELDS OF STUDY

Major Field: Fisheries management

ABSTRACT

Management and design of marine reserves and rights-based systems in small-scale fisheries

by

Daniel Fadigas Viana

Sustainable management of small-scale fisheries is one of the greatest challenges facing our ocean today. These fisheries have a collectively large ecological footprint and are key sources of food security, especially in developing countries. My dissertation explores

different pathways to provide incentives for small-scale fisheries conservation and management. For the first chapter, I explore how we can provide economic incentives for the establishment of marine reserves (areas where no fishing is allowed) in coastal communities. I develop a framework to incorporate both tourism and fisheries benefits in marine reserve design and apply this framework into a bioeconomic model simulation. Results show that

accounting for tourism benefits will ultimately motivate greater ocean protection. The findings from this chapter demonstrate that marine reserves are part of the optimal economic solution even in situations with optimal fisheries management and low tourism value relative to fisheries. The extent of optimal protection depends on specific location characteristics, such as tourism potential and other local amenities, and the species recreational divers care about. Additionally, as tourism value increases, optimal reserve area also increases. Finally, I

demonstrate how tradeoffs between the two services depend on location attributes and management of the fishery outside marine reserve borders. Understanding when unavoidable tradeoffs will arise helps identify those situations where communities must choose between competing interests. For the second chapter, I explore key design challenges and management incentives of Territorial Use Rights for Fisheries (or TURFs) from all over the globe. TURFs establish exclusive fishing zones for groups of stakeholders, which eliminates the race to fish with other groups. A key design challenge is setting the size of TURFs—too large and the number of stakeholders sharing them impedes collective action; too small and

the movement of target fish species in and out of the TURFs effectively removes the community's exclusive access. I found that about one third of the TURFs worldwide are not appropriately designed, thus hindering their potential for success. Results suggest that these fisheries, which target mobile species in densely populated regions, may need additional

interventions to be successful. For the third chapter, I use a bioeconomic model to investigate whether TURF networks have the potential to address design challenges of single

TURFs. I explore the cooperation incentives of TURFs within a network and ask how market-based strategies can improve system-wide outcomes. I found that without a market intervention, TURFs that are competing for the same resource have profit incentives to harvest above optimal levels for the entire system (i.e., non-cooperative behavior). I predict

that above a certain species mobility rate, incentives within TURFS will lead to noncooperative behavior. However, offering a price premium for cooperating TURFs has the potential to provide the incentives needed to achieve full cooperation. The price premium required to achieve optimal economic outcomes will depend on the mobility characteristics of the species being managed, but at maximum is 26% for highly mobile species. Finally, I explore how such market-based initiatives can be implemented and provide some insights on the local conditions that would best support this strategy. Overall, the insights from my dissertation suggest that providing the right incentives is key for sustainably managing small-scale fisheries.

viii

TABLE OF CONTENTS

I. Accounting for Tourism Benefits in Marine Reserve Design1			
A. Introduction1			
B. Material and Methods5			
1. Biological model5			
2. Economic model7			
C. Results14			
D. Discussion20			
E. Conclusion27			
II. Design Tradeoffs in rights-based management of small-scale fisheries29			
A. Introduction			
B. Material and Methods			
C. Results35			
1. Adult mobility35			
2. Group size			
3. Interaction between collective action and resource mobility39			
D. Discussion40			
III. Using TURF networks to manage mobile species: challenges and opportunities .47			
A. Introduction47			
B. Material and Methods50			
1. Biological model51			
2. Economic model			
C. Results55			

	D.	Discussion	59
	E.	Conclusion	64
Reference	s		64
Appendix			76

LIST OF FIGURES

- Figure 5. Theoretical relationships between TURF size and resource outcomes in response to resource mobility (A) and collective action (B). (C, E and F) represents scenarios where TURFs across a range of sizes have the enabling conditions to successfully address both problems simultaneously. By contrast, (D) represents a scenario where there is an inherent tradeoff between collective action and resource mobility problems. No TURF size in case D would likely have good performance without other interventions.32

- Figure 10. Predicted equilibrium cooperation level of a TURF network for different mobility levels across TURFs. This result does not include any market-based intervention.56
- Figure 12. (A) illustrates predicted equilibrium cooperation level of a TURF network for different price increase percentage (applied for cooperating TURFs only) for different mobility levels across TURFs. (B) illustrates the price increase necessary to achieve full cooperation within a TURF network for different mobility rates across TURFs.59

I. Accounting for Tourism Benefits in Marine Reserve Design

A. Introduction

Degradation of ocean ecosystems driven by human activities has led to an increased global interest in the establishment of ocean protected areas (Halpern et al. 2008, 2012). One type of protected area, where all forms of fishing are prohibited, is known as a "marine reserve" (Gaines et al. 2010a). Much of the interest in marine reserves is driven by their success in recovering important habitats and increasing species biomass and diversity within the reserve's boundaries (Lester & Halpern 2008). Although reserves can fail to reach their full potential because of the lack of resources for monitoring and enforcement (Edgar et al. 2014), they are a globally important conservation tool. In addition to these clear conservation benefits, the increases in species population size within reserves can also generate important economic benefits. For example, fisheries benefits can arise through the spillover of adults and/or the export of larvae to surrounding fished areas (Roberts et al. 2001; Gofii et al. 2010).

Several frameworks have now been developed to help capture these joint conservation and economic benefits in effective marine reserve designs (Foley et al. 2010; Gaines et al. 2010b; Rassweiler et al. 2014). One key limitation of the existing work, however, is that by focusing primarily on fisheries economic benefits it has ignored a potentially far larger source of added revenues – tourism [17]. Tourism gains can be obtained through diving operations within the marine reserve (Green & Donnelly 2003; Pascoe et al. 2014) and the consequent multiplier effects on local businesses related to tourism (e.g. hotels, restaurants). Collectively, these tourism benefits can be the main source

of economic gains from many marine reserves worldwide (McCook et al. 2010; Atmodjo et al. 2017). To date, there is no clear framework to maximize these potential benefits through effective reserve design. As a result, key questions remain, such as: will the range of conditions where marine reserves are profitable conservation tools grow when tourism is accounted for, and are there inherent economic tradeoffs between reserve benefits to fisheries versus tourism?

Marine reserve benefits to conservation, fisheries and tourism all depend on the buildup of biomass and diversity of species within their borders. Thus, many design elements (such as appropriate reserve size relative to scales of fish movement) might align regardless of reserve objectives, while others might be at odds with each other. For example, while fisheries benefits depend on the spillover of adults and/or larval export, tourism and conservation benefits may benefit from higher levels of local retention. This can have important implications in terms of edge location and size of the reserve (Gaines et al. 2010a). Additionally, optimal location of a marine reserve in relation to the coast might differ depending on the objective. Placing a reserve close to port may decrease costs for tourism operators and enforcement agencies while at the same time increase costs for fishers, since they will have to travel longer distances to reach their fishing grounds. Moreover, while conservation objectives require protection of all threatened species and habitats, reserves designed for tourism or fisheries objectives might require only protection of some key species and habitats. This distinction can have important design implications in relation to the location and size of reserves (Halpern & Warner 2003).

Studies have shown that divers and snorkelers consider ecosystem characteristics and other local amenities when deciding where to visit (Sala et al. 2013). Divers are attracted to conservation gains of marine reserves (Lester et al. 2009b) such as increases in the

abundance of fish, the diversity of species, iconic species, and coral reef conditions (Williams & Polunin 2000; Alban & Boncoeur 2006; Uyarra et al. 2009). Additionally, since divers are also tourists, other local amenities can also play an important role. Characteristics such as tourism infrastructure, local attractions, proximity to airports, and quality of restaurants and hotels can directly influence a diver's decision on where to visit (Weaver & Lawton 2007). Relative importance of local amenities versus ecosystem health depends on divers' preferences and availability of different habitats and species. For example, in the Great Barrier Reef, Australia, whales and dolphins were the preferred draws for divers followed by sharks and rays, overall species richness, turtles and large fish (Farr et al. 2014). In the western Caribbean islands, variety of fish, fish abundance and coral variety were the preferred attributes (Williams & Polunin 2000). In contrast, divers from Barbados listed terrestrial characteristics (beaches) and warm and clear water as their main reason for visiting the area followed by coral and fish diversity and abundance (Uyarra et al. 2005). Such differences in preferences show evidence of two categories of divers, one category that is driven by ocean biodiversity and another category that is mainly driven by other local amenities (Uyarra et al. 2005). The former group will likely be attracted by marine reserves, while the latter may be indifferent.

Benefits from tourism can in many cases be far greater than the opportunity cost of foregone fishing. For example, in the Great Barrier Reef annual revenue from tourism is 36 times greater than income from commercial fishing (McCook et al. 2010). In the Medes Islands Marine Reserve (Spain) annual revenue from tourism is about 20 times greater than fishing revenue (Merino et al. 2009). Potential tourism revenue from marine reserves can be generated directly through user fees (Green & Donnelly 2003) or by boosting the tourism economy in the region. Marine Reserves can potentially increase value of all business

associated with tourism (e.g. hotels, restaurants), especially those dependent on underwater activities (e.g. dive centers). These benefits depend on the location of the reserve as well as the biomass of fish in the water (Sala et al. 2013). Reserves located near coastal areas with intense tourism activity and other tourist attractions are likely to have high visitation rates quickly after reserve creation (Chae et al. 2012). In such situations, the marine reserve may not be the main draw to the area and often does not require high levels of biomass to attract divers. By contrast, locations where there are no other coastal attractions other than the marine reserve may only attract more experienced divers that are drawn by high levels of fish biomass and diversity (Aburto-Oropeza et al. 2011). These areas may need to be more spectacular and tied with marketing strategies to attract large numbers of divers, since the reserves will often be competing with diverse diving options around the globe.

Despite growing evidence of economic benefits associated with tourism activities in marine reserves, most spatial planning models only take into account fisheries and/or conservation benefits but ignore tourism gains. To incorporate potential tourism benefits we develop a bioeconomic model to simulate different marine reserve designs and their predicted impacts on fisheries and tourism revenue. We model the potential benefits for both services under different tourism and fisheries management scenarios to ask under which conditions are marine reserves part of the optimal solution that maximizes total economic benefits. We then analyze the potential tradeoffs between fisheries and tourism economic benefits to understand the incentives stakeholders face and the situations where conflicts are likely to arise.

B. Material and Methods

We use a bioeconomic model to simulate different marine reserve designs and the potential economic benefits to fisheries and tourism over time. We divide a hypothetical coastline into 100 homogeneous linear patches where we track the biomass within each patch. Patches are wrapped to eliminate any boundary effect and to make sure all patches are homogeneous. Patches are connected through adult spillover. A fraction of the population emigrates from each patch to nearby patches with a probability that depends on the distance between the patches. A certain fraction of the biomass is also removed through fishing from each patch that is not a marine reserve, with the sum of discounted revenues over time representing the economic gains to fisheries. Larval dispersal is assumed to occur within each patch as population growth in a patch is only dependent on local population size. Although we acknowledge the important design implications driven by larval dispersal dynamics (Siegel et al. 2008; Pelc et al. 2010), we did not consider larval connectivity to simplify the model. Tourism benefits are associated with an increase in the demand for dives inside the marine reserve associated with increased fish density (Sala et al. 2013). We did not consider diving activities in fished areas since our source of revenues are the user fees charged to gain access to the marine reserves.

1. Biological model

We use a simple logistic model that tracks biomass of a given species in each patch over time:

$$B_{t,i} = B_{t-1,i} + g * B_{t-1,i} * \left(1 - \frac{B_{t-1,i}}{K_i}\right) - f_i * B_{t-1,i} - E_{t,i} + I_{t,i}$$
(1)

Where $B_{t,i}$ is the biomass in year *t* and patch *i*, *g* is the intrinsic growth rate, K_i is the carrying capacity, f_i is the harvest fraction, $E_{t,i}$ is the emigration from patch *i* and $I_{t,i}$ is the immigration to patch *i* from all other patches.

Harvest fraction in each patch, f_i , is calculated according to Hilborn et al. 2006, where the intensity of harvest is proportional to the biomass in each patch. We assume that total effort remains constant when a marine reserve is created. This translates to an increased fishing intensity in areas open to fishing as the size of marine reserves grows. The combination of a constant overall fishing effort and a resulting fixed fishing mortality rate in fished patches accounts for the displacement of effort caused by marine reserve placement and creates the fishing the line effect (Kellner et al. 2007) associated with higher catches in patches surrounding marine reserves. For well managed scenarios, total fishing effort is calculated as the amount that generates maximum sustainable yield at equilibrium when the entire area is open to fishing. For overfished scenarios, we assume a fishing effort that would drive fish biomass down to 10% of carrying capacity at equilibrium when all patches are open to fishing. This open access equilibrium biomass value was assumed according to (Costello et al. 2016a). Harvest fraction inside patches designated as marine reserves is zero. Initial biomass is assumed to be the equilibrium biomass under the different fisheries management scenarios (50% and 10% of carrying capacity for well managed vs. overfished, respectively).

Emigration from patch *i* (E_i) equals the biomass of fish in the previous year, $B_{t-1,i}$, times the movement fraction, represented by μ :

$$E_{t,i} = B_{t-1,i} * \mu$$
 (2)

Immigration to patch i (I_i) is the sum of the emigration contributions from all other patches j:

$$I_{i,t} = \sum_{j=1}^{100} E_{j,t} p_{ji}$$
(3)

where the proportion of emigrant fish moving from each patch j to patch i, p_{ji} is defined as (Sala et al. 2013):

$$p_{ji} = \exp(-d_{ji}) \tag{4}$$

where $d_{i,j}$ is the distance between patch *j* and patch *i*. Relative proportions are then normalized so that the proportions moving to all other patches sum to one.

2. Economic model

Fisheries value

Fisheries revenue (R_t) is the sum across all patches of the product of the harvest fraction (f_j), resource price (λ) and biomass ($B_{t,i}$) in year t.

$$R_t = \sum_{j=1}^{100} f_j * B_j * \lambda \tag{5}$$

Total net present value of fisheries revenue (FV) is then calculated by summing across all years and applying a discount rate:

$$FV = \sum_{t=1}^{50} R_t * \left(\frac{1}{1+\delta}\right)^t \tag{6}$$

Where δ is the discount rate.

Tourism value

Tourism value is assumed to be associated with the density of fish inside the marine reserve to reflect the underwater experience of divers. As described by Sala et al. 2013, we assume a dive's marginal value is directly influenced by the diver's underwater experience. Increased fish density inside the marine reserve will shift diver's demand outward, increasing potential revenue generated from the system (Sala et al. 2013). Additionally, we assume a congestion effect restricting the total number of divers per marine reserve area per unit time. This reflects the fact that divers prefer less crowded areas, and marine reserves often adopt a cap on the total number of dives per day per area of marine reserve to ensure conservation benefits. Such policy results in a diver carrying capacity inside the marine reserve area per unit time. The size of the reserve thus limits the potential number of dives per marine reserve area per unit time.

We assume that a subset of patches, denoted by M, is designated as a marine reserve. Thus, $f_{j \in M} = 0$, and the size of the marine reserve is denoted by x = card(M). Year t

biomass in reserve is just $\sum_{j \in M} B_{j,t}$ which is denoted by $B_{M,t}$. We used a modified version of

the equation described by Sala et al. 2013 to model the marginal value of additional dives:

$$P_t = \alpha_0 + f(B_{M,t}) - g(x)D_t \tag{7}$$

where P_t is the marginal value of dive $D_{r,t}$, α_0 is the intercept of the demand function, $f(B_{M,t})$ is the demand shifter reflecting fish abundance in the marine reserve, and g(x) changes the slope of demand to reflect congestion of divers in the marine reserve (this congestion effect will depend in reserve size, x). The fish abundance effect on demand, $f(B_{M,t})$, is increasing in fish biomass inside the reserve and the congestion effect, g(x), is decreasing in the size of the reserve (Figure 1). The function forms for $f(B_{M,t})$ and g(x) are given as follows:

$$g(x) = \frac{\alpha_1}{(\log_{100} x)^{\frac{1}{W}}}$$
(8)

Where α_1 is a location specific price elasticity, x is the reserve size and w controls the

slope of the logarithmic function. We assumed a logarithmic function because it allows different slopes to be modeled. The different slopes represent distinct levels of tourism potential, reflecting the fact that when there is a high number of possible divers, small reserves cannot capture all potential tourism revenue because of the congestion effect. This allows the model to account for crowding issues and diver carrying capacity, which limits the number of divers per area of reserve. We assume that the diver carrying capacity is set to prevent environmental degradation by divers so that tourism activities does not interfere with biomass buildup inside reserves. By setting a cap on the number of dives, reserve area will directly affect the total revenue that can be generated, especially in locations with high tourism potential. Under such conditions, tourism value is expected to increase as marine reserve size increases, since more divers will fit in a larger reserve. On the other hand, in

locations with low tourism potential, the crowding effect is less important. This is expected to happen, because all potential divers can fit in a relatively small area. Thus increasing reserve size does not imply a significant increase in the number of dives. Although maximum tourism values are scaled to one, revenues generated in locations with high tourism potential can be dramatically higher than locations with low tourism potential.



Number of Dives

Figure 1. Hypothetical illustration of the effects of congestion, g(x), and fish density, $f(B_{M,t})$, in divers' demand (equation 7). Dotted line illustrates equation 7 at higher fish density levels. Dashed line illustrates equation 7 at higher congestion levels.

The influence of fish density in the demand curve is represented by f(B), which shifts a dive's marginal value in a logistic manner:

$$f(B_{M,t}) = \frac{b}{1 + b * e^{-(c * \frac{B_{M,t}}{K_M})}}$$
(9)

Where $B_{MR,t}$ is the total marine reserve biomass, K_{MR} is the marine reserve carrying capacity, and b and c are the parameters for the logistic curve that regulate the relationship between density and tourism value. Parameter b represents the additional number of dives that can be obtained due to fish density improvements. Parameter c regulates the rate of increase and the minimum density level required for tourism value to begin increasing. We assume that the demand for dives in a marine reserve will shift outward through a logistic relationship with fish density. This assumption is meant to address the fact that marine reserves can achieve a certain threshold of fish density where their attraction to divers will grow far more rapidly (at least more than fish density in areas open to fishing) and after a certain point increasing density will attract few additional divers. This relationship is determined by the *c* parameter, with actual values representing different location conditions. In locations where the main draw to the area is not the marine reserve, fish density may not be as important to achieve a given level of tourism revenues. Under such conditions tourism revenues may start growing even with relatively low fish densities. An example of this scenario is Barbados, where divers reported that terrestrial characteristics are the main reason for visiting the area (Uyarra et al. 2005). In addition, such a pool of tourists is likely to have a higher fraction of less experienced divers, for whom the underwater experience is not as important. Conversely, in locations where the main tourism draw is the marine reserve itself, diving experience is more important and dive tourism value will likely increase at higher fish densities. An example of this scenario would be Cabo Pulmo, Mexico, an isolated community where the marine reserve is the primary tourism draw and tourism

revenues grew rapidly after a 400% increase in the biomass of targeted species (Aburto-Oropeza et al. 2011).

Equation 7 can be used to calculate the number of dives in a given patch for any given price and biomass level. The optimal price (OP_t) that maximizes total revenue can also be calculated by taking the derivative of the product of the fee per dive and the number of dives in the reserve and setting the equation equal to zero:

$$OP_t = \alpha_0 + \left(\frac{\alpha_0 + f(B_{M,t})}{-2}\right) + f(B_{M,t})$$
(10)

Tourism revenue $(TR_{r,t})$ is calculated by multiplying the number of dives in the reserve by the optimal price per dive (OP_t) :

$$TR_{r,t} = OP_t \left(\frac{f(B) + \alpha_0 - OP_t}{g(x)} \right)$$
(11)

Equilibrium tourism revenue is calculated as the tourism revenue generated in year 50. Total net present value of tourism revenue (TV) is calculated by summing the predicted revenue across all years and applying a discount rate:

$$TV = \sum_{t}^{50} TR_{r,t} * \left(\frac{1}{1+\delta}\right)^t \tag{12}$$

Where δ is the discount rate.

To obtain general results, we normalize potential tourism and fisheries revenue to each be between 0 and 1, as actual revenue is context dependent. Assuming a 0 to 1 value

allows us to test the influence of different relative values from fisheries and tourism on the optimal marine reserve design. Additionally, this assumption does not affect the shape of the tradeoff between these two services, as relative values will only help choose along the tradeoff curve the marine reserve design that provides highest economic returns. We further explore the implication of different relative tourism and fisheries values by demonstrating how actual values can alter optimal marine reserve size. Two metrics are used to determine the value of these services: normalized net present value (NPV) and equilibrium revenue. Net present value of tourism and fisheries services considers the time required for such benefits to be realized. Since future revenues are discounted, timing of benefits becomes a crucial factor. Characteristics such as low initial biomass or slow population growth rates increase the time required for benefits to be realized and therefore negatively affects the NPV. For this metric, a value of one represent the maximum possible NPV that can be achieved for fisheries and tourism services given all possible design and fisheries management options. Equilibrium revenue of fisheries and tourism services does not consider the time component. This would be important for stakeholders that have a longterm vision, without time consideration. For this metric, initial biomass or growth rate are not as important. A value of one represent the normalized maximum equilibrium tourism or fisheries revenue that can be achieved by the system. When considering total revenues, optimal marine reserve design is calculated for different relative values of fisheries and tourism services. Optimal marine reserve size is defined as the design that maximizes total economic value of the system (tourism + fisheries) for every given relative worth of both services. The timing component of the model is also explored more explicitly by calculating the number of years required for particular relative tourism values to be realized under different management scenarios. For default values, we assume a movement fraction (μ) of

0.2, an intrinsic growth rate (r) of 0.2 and a 5% discount rate. For the tourism model we assume a moderate dependence of the revenue on fish density (c=15) and a moderate crowding effect (w=1). Sensitivity analysis of all model parameters are shown in the supplementary material.

C. Results

Expected tradeoffs between fisheries and tourism services vary according to different management scenarios and metrics (Figure 2). In well managed scenarios, maximum fisheries revenue is achieved with no marine reserves. Fisheries revenues decrease as marine reserve size increases. In such cases, if tourism value is ignored, marine reserves are not part of the optimal economic solution. Thus, with perfect fisheries management, accounting for tourism benefits will be crucial for marine reserves to be part of the optimal economic solution. By contrast, in the overfished scenario higher fisheries value can be achieved with marine reserve implementation. Consequently, even if tourism value is ignored, marine reserves will be part of the optimal solution when resources are overfished. When considering the net present value of fisheries and tourism services (Figure 2 A and B), overfished areas can only obtain a fraction of the total NPV from well managed systems because of the difference in the initial biomass values and harvest levels. In contrast, when considering equilibrium revenues of tourism and fisheries services, overfished scenarios can achieve much higher values relative to well managed systems. This happens, because equilibrium values do not account for the time required for biomass recovery. Thus, since equilibrium values do not consider discount rate, initial biomass is not as important. Additionally, tourism benefits have a maximum value of one in both cases (well managed and overfished), because closing the entire area to fishing does not affect equilibrium values.



Figure 2. Tradeoffs between fisheries and tourism services for well managed (A and C) and overfished (B and D) scenarios. (A) and (B) demonstrate results in terms of net present value and (C) and (D) demonstrate results in terms of the equilibrium revenue. Colors represent the percent of the area designated as marine reserve.

Despite inherent tradeoffs between tourism and fisheries services, relatively high values of both services can be achieved simultaneously. For example, for all scenarios where maximum tourism can be achieved (Figure 2 A, C and D), both services can simultaneously achieve about 80% of their maximum value. This is the point along the tradeoff curve that maximizes the sum of both normalized values. Interestingly, when considering equilibrium revenues, a reserve of about 40% is desired to maximize the sum of both values (tourism + fisheries), independent of the management scenario. If for economic or social reasons

revenues higher than 80% are desired for one of the two services, it will lead to significant costs to the other. For example, for all scenarios where maximum tourism can be achieved (Figure 2 A, C and D), achieving 90% of tourism benefits will reduce fisheries revenue to about 40% of its maximum value. On the other hand, achieving 90% of fisheries value in well managed scenarios (Figure 2 A and C) will reduce tourism revenue to about 60% of its maximum value.

Sensitivity analysis of crowding (w) and fish density (c) effects on the tradeoffs between tourism and fisheries services show that the shape of the tradeoff is sensitive to these parameters. In locations where diving is not the main driver of tourism benefits (high cvalue), a small marine reserve might be enough to generate the density of fish needed to attract divers. Locations where the marine reserve is the main tourism driver (low c value) larger areas are necessary to create the density needed for tourism benefits to be realized. Additionally, strength of the crowding effect will affect the optimal marine reserve design. Since an area can only fit a certain number of divers at any given time, locations with high tourism potential (low w value) will require more protection to achieve full benefits. Conversely, in locations with low tourism potential (high w value), crowding is not significant. Thus, small marine reserves can accommodate all divers. Simulation of these scenarios shows that although high fish densities can often be achieved with small marine reserves, larger areas may be necessary to capture all potential tourism benefits.

If the planning objective is to maximize overall revenues from both fisheries and tourism services, actual economic values will be crucial to determine the optimal design. Figure 3 shows the influence of relative tourism and fisheries values on optimal marine reserve size under different fisheries management scenarios and outcome metrics. Generally, optimal marine reserve size increases as relative tourism value rises, eventually reaching

100% of the area. For overfished scenarios, marine reserves are always part of the optimal solution, even with relatively low tourism values. The optimal marine reserve size for overfished scenarios when tourism value is extremely low is about 30% of the area. As the relative revenues from tourism and fisheries reach a value close to one, optimal marine reserve size increases rapidly, eventually reaching 100% of the area (Figure 3). In well managed situations, where marine reserves are not part of the optimal solution for fisheries alone, including tourism value changes the outcome even when tourism revenues are well below fisheries values. This happens, because even tiny reserves (1-2%) can bring larger tourism value than the corresponding losses to fisheries value. As tourism value increases relative to fisheries, the optimal marine reserve size grows, eventually reaching 100% of the area. When considering equilibrium revenues (Figure 3B), optimal marine reserve area is very similar for both management scenarios when tourism value is about 10 times fisheries value. Sensitivity analysis to crowding and fish density effects show that optimal marine reserve size for different relative values of tourism and fisheries services can be quite different depending on these parameters values (S4 Fig). Generally, as tourism potential and crowding effect decreases (high w values), higher relative tourism value is needed for high levels of protection. Since higher relative tourism versus fisheries values are harder to achieve when there is low tourism potential, closing big portions of the area becomes less likely. Additionally, with low tourism potential, fish density effect plays an important role. As dependence on fish density increases (locations where the main draw is the marine reserve) greater protection is desired.



Figure 3. Optimal marine reserve size for different relative net present values of tourism and fisheries services. The two figures represent different outcome metrics, where (A) is in terms of net present value and (B) is in terms of equilibrium revenue.

Timing of benefits is an important factor to consider when creating a marine reserve expecting tourism gains. Figure 4 demonstrates the number of years required for tourism revenues to be generated under different marine reserve designs. Because of different starting points and intensities of fishing in open areas, the timing of benefits varies significantly. In well managed scenarios tourism benefits can happen relatively quickly, because stocks inside the marine reserve start at higher values. By contrast, overfished scenarios can take much longer for marine reserve densities to reach peak values. Additionally, the larger the area protected the quicker benefits will be realized, because fewer fish leave the boundaries of the reserve where they can be caught. For example, it takes 40 years to achieve 0.5 of the maximum tourism revenue in overfished scenarios for a marine reserve size of 25%. The more tourism revenue is dependent on fish density the longer these benefits take to be realized.





The benefits of reserves are sensitive to growth and movement characteristics of the target species. Generally, species that have high movement rates and low growth rates will require larger reserves to achieve tourism benefits. This is consistent with the literature on biological responses of marine reserves, where species that move more require larger areas to be protected (Gaines et al. 2010b). On the other hand, if movement rate is close to zero, relatively small areas will be sufficient to produce fish densities that attract divers, and tourism value will depend mostly on the strength of the crowding effect. Additionally, effects of growth and movement rates on optimal design is greater in overfished relative to

well managed scenarios because of the difference in fishing mortality of the fish that spill over from reserves.

D. Discussion

Accounting for tourism benefits can significantly influence optimal marine reserve design. Results from our model show how considering tourism objectives can be crucial for marine reserves to be part of the optimal economic solution, regardless of the state of the fishery. This result challenges previous findings that marine reserves are not part of the optimal economic solution when the fishery is well managed (Buxton et al. 2014). Our findings demonstrate how marine reserves should be implemented even in situations with optimal fisheries management and low tourism value relative to fisheries. In such situations, a relatively small reserve can generate more benefits from spillover and tourism development than foregone fisheries value. As tourism value increases relative to fisheries revenues, larger areas should be protected to maximize economic outcomes.

Conflicts are likely to be highest when there is clear preference for one service over the other. Optimal marine reserve design choice will be greatly influenced by the relative social and economic value of tourism and fisheries services. Different stakeholders can have distinct social tradeoffs, which can be related to higher relative profits, social motives such as employment, or cultural reasons such as local traditions and customs. Therefore, some stakeholders might care more about one service than the other, influencing the optimal marine reserve design and inherent tradeoffs between fisheries and tourism services. Stakeholders that depend solely on resource extraction, such as fishers, might only value fisheries and not care about tourism benefits. Consequently, optimal design will be the point along the tradeoff curve that maximizes fisheries value – i.e., the location where a horizontal

line reflecting a pure preference for fishing is tangent to the tradeoff curve. Generally, greater preferences for fisheries services will lead to lower tourism values and less area being protected. In well managed scenarios, the optimal solution that maximize fisheries services is to open the entire area to fishing and manage the fishery well. This will lead to zero tourism value, since our model assumes that revenue is generated through the collection of user fees that depend on marine reserve establishment. In overfished scenarios, marine reserves are required to maximize fisheries value. This creates a win-win situation with tourism services, where maximizing the value of one service also generates value to the other service. Stakeholders that rely only on tourism activities (e.g dive operators) might have a high preference for tourism benefits and may not care about fisheries services. When there is a pure preference for tourism services optimal marine reserve design is where tourism value is maximized. This happens where a vertical line is tangent to the tradeoff curve, which for all scenarios is where 100% of the area is set as marine reserve. The greater the preference for tourism services the bigger the compromise to fisheries revenue, leading to a strong tradeoff between the two services. Stakeholders that value both services equally for economic or social reasons (e.g., government managers) might not have a preference for one or the other service. Thus, following general economic theory, optimal design will be the point along the tradeoff curve that maximizes the sum of both relative values. In this case, optimal design is where a 45 degree line is tangent to the tradeoff curve. For all scenarios, this equal weighting point has relatively high values of both services and relatively low tradeoffs.

Revenues generated through user fees can be used in many ways to offset potential costs associated with the marine reserve. Revenues can be used for direct compensation to fishers, investment in better management of fisheries, creation of alternative livelihoods,

community infrastructure, and/or monitoring and enforcement. Direct compensation to fishers can be used to compensate for losses associated with reduced fishing grounds. One example of such a scheme is in the Great Barrier Reef Marine Park, where the Australian government provided compensation for commercial fishers adversely affected by the reserves (Olsson et al. 2008). Such schemes might be useful to obtain support from key stakeholders, but can create perverse incentives to overharvest areas that are still open to fishing. Utilizing revenues to invest in better management for adjacent areas open to fisheries can be an alternative to achieve long term sustainability of the fisheries. Although such investment would not address short term costs, it can help to ensure spillover benefits from marine reserves to affected fishers and achieve better fisheries profits in the future. For example, in the Galapagos National Park, revenue from fees are used by the government to manage the fisheries around the islands (Castrejo & Charles 2013). An alternative to investing in the fisheries sector would be to invest in alternative livelihoods such as aquaculture or tourism. Such alternatives can have many positive effects by increasing resilience of the system through income diversification. Additionally, it can decrease problems associated with the displacement of effort to outside areas by converting some of those fishers into tourism operators or aquaculture farmers. For example, in the Raja Ampat Marine Reserve system located in Indonesia, 30% of the user fees are directed to communities in the region for projects related to tourism development (Atmodjo et al. 2017). The remaining revenue is used for managing the marine reserves, including costs related with monitoring and enforcement. In many cases, benefits generated through user fees are entirely used by government agencies or NGOs for monitoring and enforcement of the area (Reid-Grant & Bhat 2009; Thur 2010). Using all revenue for reserve management can help enforcement of the area but does not address the root of the problem. In such cases, fishers

typically bear all costs and do not have any secure benefits from the reserves, which can lead to strong opposition to any marine reserve creation. Therefore, uncertainty and timing of fisheries benefits might lead to increased illegal activities and enforcement costs, which is one reason for many "paper parks" worldwide (Edgar et al. 2014).

Short term costs to fisheries (Ovando et al. 2016) and long term maintenance costs of marine reserves can be a strong deterrent to their success (Edgar et al. 2014). As fisheries and tourism benefits are related to the density of fish inside closed areas, such benefits can take a long time to be realized depending on reserve size, species characteristics and the fishing pressure before and after reserve creation (Babcock et al. 2010). Results from our model demonstrate how tourism revenues generated through user fees can take many years to be realized, especially when the fishery is overfished prior to reserve creation. With such benefits occurring in the future, innovative market strategies might be needed to compensate for short term fisheries losses. Such market-based strategies can be a promising solution to use future tourism benefits to offset short-term fisheries losses (Ovando et al. 2016). For example, in areas with high tourism potential, significant revenues are expected in the future. Thus, agreements between the tourism industry and fishers can be established to ensure fishers are guaranteed a share of future tourism benefits. Although this alternative does not address short term losses it ensures future benefits to fishers, which might be sufficient to gain their support. The level of support might in turn depend on the timing of such benefits and discount rate of the fishers. If their discount rate is high, future benefits can be insignificant compared to short term losses. Another market-based alternative might be to acquire a loan with banks or philanthropic organizations to compensate short-term losses, with payments from future tourism benefits. Philanthropic organizations interested in marine conservation might offer lower discount rates than banks and are usually more
willing to take the risks related to an uncertain benefit. The magnitude of uncertainty on future tourism benefits will likely depend on the characteristics of the area related to their tourism potential. For example, in areas where there are no other major attractions other than the marine reserve, marketing campaigns and tourism infrastructure need to be fomented to create a reputation of the area among the diving community and provide minimal conditions for tourists. Otherwise, there is a chance that tourism benefits are going to take too long or will not happen at all, especially if diving experience is not spectacular enough to compete with other marine reserves from around the world.

Our model assumes that tourism revenue is associated with the density of fish inside the marine reserve. Although fish density is one of the main ecosystem attributes preferred by divers (Williams & Polunin 2000), other characteristics can also be important. For example, diversity and size of fish and corals can be an important factor for divers (Uyarra et al. 2009). Although we don't explicitly account for these characteristics, such attributes are generally correlated with increases in density inside marine reserves (Lester et al. 2009a). Additionally, we assumed that divers are driven by only one species of fish, while in reality there will undoubtedly be far more than one important species. Optimal marine reserve design will vary depending on the biological characteristics of the species and focusing on only one may not be sufficient to increase the biomass of the other. One approach would be to focus on the species that have the greatest movement rates to ensure positive growth of all species. Focusing on species with high mobility would mean having to close a relatively large area, which might be challenging depending on the context. For example, locations with low tourism potential and high tradeoffs with fisheries services, protecting large areas might not be viable. On the other hand, for locations with high tourism potential that depend on mobile species for diving activities, benefits from protecting a large area likely outweighs

potential costs to the fisheries sector. Another important assumption of our model is that fisheries target the same species that divers care about. In cases where the main draw for divers are charismatic species not targeted by fisheries (e.g. dolphins, whales, turtles), such an assumption might not hold true. Although such species are not expected to be directly affected by protection as much as species targeted by the fisheries, marine reserves can provide indirect benefits through increased food availability (Scott et al. 2012). Additionally, even though there might not be any significant increases in density, marine reserves still create an instrument to collect revenue that can be invested in the region.

Tourism activities inside marine reserves can have positive and negative effects for marine conservation. Since tourism activities are dependent on marine conservation, high synergies between tourism and conservation services can be expected. Additionally, having a regular presence of divers in the reserves can help with monitoring and enforcement of the area as it can discourage poachers and facilitate detection of illegal fishing activities. On the other hand, inexperienced divers can cause significant habitat degradation and alter important fish behaviors (Hawkins et al. 1999). Many studies have pointed out the damage caused by divers in sensitive coral reef areas [45,46]. Prevention of damage can be achieved by setting a maximum number of divers for a given area (Davis & Tisdell 1995) and providing proper training and education to dive masters and recreational divers about best practices and potential harms associated with this activity. Several marine reserves around the world have been using a diving carrying capacity to minimize environmental damage caused by divers. For example, the Mendes Islands Marine Reserve has established a maximum of 450 dives per day (Sala et al. 2013). Protecting large portions of the ocean can also help decrease diver density and increase potential conservation benefits. Such methods can significantly decrease adverse tourism effects and increase synergies between

conservation and tourism services. In our model, we assume that a maximum number of dives per reserve area is set to prevent environmental degradation by divers. Thus, diving activities does not interfere with biomass buildup inside reserves. Future research can relax that assumption and explore how environmental impacts by divers interfere with design outcomes.

We use a conservative model in terms of the benefits that can be generated to fisheries. First, our model only considers adult spillover as benefit source. It does not account for potential recruitment increases through larval and egg spillover which in many cases can be the main source of benefit (Siegel et al. 2003; Botsford et al. 2009). We did not include larval dispersal dynamics in our model in order to obtain simplified but conservative results. In our model, when adult movement rate is zero, there is no possible source of benefit to the fishery. This is not true in many cases where marine reserves can be an important source of eggs and larvae to fished areas thus increasing recruitment and growth rate of the fished population. This can have important design implications in terms of reserve location and the expected recruitment benefits to fished areas (Siegel et al. 2008). Second, we assume that effort is going to remain constant through time, being redistributed into fished areas after reserve creation (Hilborn et al. 2006). This causes an increase in fishing mortality in the outside areas as marine reserve increases and the fishing the line effect (Kellner et al. 2007). This assumption can be true in many situations with weak management outside reserve boundaries. If the fisheries are optimally managed, fishing effort is expected to adjust in order to provide optimal economic returns. As marine reserves increase, overall effort in outside areas should be decreased and concentrated near reserve borders to optimize economic returns (Hilborn et al. 2006; Rassweiler et al. 2014). Effort reduction can be facilitated with increased tourism activities as it can create alternative livelihoods for the

local community. Thus, even though we conservatively assumed that effort is going to remain constant, tourism activities in the reserve might in reality decrease fishing effort in outside areas. For example, in Raja Ampat – Indonesia, many locals that used to depend on fishing as their main source of revenue are transitioning to the tourism sector using user fee revenues to invest in local tourism infrastructure (Atmodjo et al. 2017). Additionally, increased tourism activities might influence local consumption of sustainable seafood and increase the price of locally harvested products, allowing reductions in catch without compromising total revenue generated.

E. Conclusion

Our model provides the first attempt to incorporate future tourism revenue in the design of marine reserves. Tourism is a way to capture benefits from conservation and turn it to a monetary value, which is crucial when comparing with fisheries value. We provide significant insights on the importance of the specific location characteristics in the prediction of future tourism benefits. Previous tourism infrastructure and other local attractions can play a critical role in determining the expected benefits and their relationship with fish density. Tourism potential of each area can also have significant implications to marine reserve design because of congestion effects. In all scenarios tested, marine reserves were part of the optimal design when considering both tourism and fisheries benefits, even when the fishery is well managed outside. The amount of area to be protected will greatly depend on the value of tourism relative to fisheries. As relative tourism value is orders of magnitude greater then fisheries value, it would be optimal to close the entire area to fishing. Therefore, accounting for tourism benefits can be crucial to optimally design marine

reserves. Additionally, the use of revenues generated through user fees to offset potential costs associated with reserve creation can be crucial to gain support of local stakeholders and increase conservation effectiveness.

II. Design Tradeoffs in rights-based management of small-scale fisheries

A. Introduction

Mismanagement of small-scale fisheries is one of the largest challenges facing our ocean today. One solution that has been widely advocated as a solution to overfishing problems of small-scale fisheries are the Territorial Use Rights for Fisheries (or TURFs). TURFs allocate exclusive rights to a group of fishers to use all or part of the resources in a particular area of the sea (Wilen et al. 2012). National governments from several countries have turned to such local-level governance institutions because of the potential benefits this strategy can provide to small-scale fishing communities (Agrawal 2005; Aceves-bueno et al. 2017; Nguyen et al. 2017). TURFs recognize fishers as an integral and indispensable part of contemporary efforts to conserve environmental resources, especially when there are weak regulatory institutions. Unlike traditional management strategies, TURFs change overharvesting incentives prevalent in open-access systems by allocating exclusive and secure access to marine resources (Costello 2012). Such rights motivate more sustainable management actions by TURF users, because they ensure that future benefits from those actions are secured for TURF owners. The logic is that once a group of fishers has secure rights to a fishery they will act as sole owners and manage the resource to obtain maximum long term economic gains (Costello & Kaffine 2008).

In practice, TURFs will only achieve these goals if they are well designed. There is a growing body of literature exploring the design factors that affect the success of selforganized resource regimes (Agrawal 2001; Ostrom 2009). For TURFs, one of the most

basic design challenges is TURF size, which can affect performance via two distinct modes: collective action and resource dispersal.

Collective action is generally compromised as group size increases (Olson 1965), suggesting smaller TURFs with fewer fishers may provide management benefits. This happens because the number of users within the system can influence many variables that affect self-organization (Agrawal 2002) and can affect incentives to free ride (users that enjoy resource benefits without paying for costs). First, as groups become larger, the perception of individual contributions tends to decrease and transaction costs (communication, enforcement) tend to increase (Poteete & Ostrom 2004). This leads to greater incentives to free ride and diminishes the capacity of users to enforce regulations and punish defectors (Ostrom 2010). Second, as group size increases, the capacity to devise appropriate and legitimate management rules diminishes (Olson 1965), since larger groups tend to have greater heterogeneity of users (social, cultural, economic) (Poteete & Ostrom 2004) and diminished communication opportunities (Lopez & Villamayor-Tomas 2017). Overall, increases in TURF size create larger groups, which accentuates challenges for collective action and may dwarf the capacity of self-organizing systems to achieve optimal outcomes.

By contrast, movement of target species beyond the boundary of the TURF can create incentives to overharvest before fish leave the TURF (White & Costello 2011), suggesting larger TURFs may provide management benefits. Successful resource management depends on the size of TURFs relative to the natural spatial scales of dispersal (Janmaat 2005; White & Costello 2011). When fish swim or drift out of the bounds of a TURF, they become available to fishers outside. Boats lining the boundary of a TURF provide clear visible evidence of the loss of resources to others, which incentivizes TURF owners to harvest

above sustainable levels rather than let fish leave. Additionally, resources with high mobility can be more unpredictable, which affects the ability of users to set appropriate harvest rules. Overall, TURFs that are small relative to dispersal scales do not provide the correct biological incentives to optimally manage the resource.

These opposing effects can pose challenges, especially in cases where TURF sizes that would be small enough to avoid collective action problems would not be large enough to avoid spillover problems created by species movement (Figure 5). Thus, for fisheries targeting highly mobile species in regions with dense coastal human populations, TURFs may be ineffective unless additional interventions are made to overcome either spillover problems or the collective action problems. Here, using a combination of theory and a review of 137 TURFs worldwide, we assess the prevalence of fisheries facing this challenge, and we discuss possible solutions.



Population Density

Figure 5. Theoretical relationships between TURF size and resource outcomes in response to resource mobility (A) and collective action (B). (C, E and F) represents scenarios where TURFs across a range of sizes have the enabling conditions to successfully address both problems simultaneously. By contrast, (D) represents a scenario where there is an inherent tradeoff between collective action and resource mobility problems. No TURF size in case D would likely have good performance without other interventions.

B. Material and Methods

We assembled a global database from peer-reviewed literature, governmental and non-governmental reports, masters and PhD theses, and interviews of local stakeholders to assess where existing TURFs lie with respect to these conceptual size guidelines. We compiled general data on TURFs from 30 countries. For 19 of these countries, we were able to assemble a complete data set on a total of 137 TURFs where we obtained the requisite biological and social data to forecast their expected performance, including information on TURF size, primary species harvested and group size. We constrained the number of TURFs from any given country in our database (maximum of 27 TURFs from Chile) to avoid bias related to any country-specific design guidelines. From the 137 TURFs used in this study, 113 had information on all aspects and 24 had incomplete information on one or two aspects. For example, for some TURFs in Vanuatu, there is only information available on the main species harvested but no information on group size.

To calculate the predicted yield due to adult movement, we use a simple gametheoretic bioeconomic fisheries model developed by White and Costello (2011) (White & Costello 2011). This model considers the effect of TURF size relative to the scale of adult fish movement on potential yields. This two-patch model simulates the behavior of noncooperative TURFs acting to maximize their yield and computes the expected Nash equilibrium of this competitive behavior. It calculates the potential loss in yield due to the dispersal of adult fish relative to a perfectly designed TURF (i.e., with no adult dispersal) that maximizes its yield. Absolute yields clearly can increase with TURF size, but we scale all evaluations of TURF performance relative to the maximum sustainable yield for the TURF. We use species home range as a proxy for movement. This information was primarily extracted from the peer-reviewed literature. When data were not available in the

literature, we used either values from a species in the same family with similar characteristics or calculated the estimated home range from Kramer and Chapman (1999) (Kramer & Chapman 1999). This method estimates the home range of coastal species based on the species' maximum length. To simplify our model, we did not consider larval dispersal in our analysis. Uncertainty on population source/sink dynamics and data limitations can constrain managers' ability to properly align TURF spatial scale with scales of larval dispersal. Therefore, adult mobility is often the most important component driving management incentives of TURF owners.

To estimate predicted yield due to the number of users in a TURF, we assumed a negative logistic relationship to reflect the fact that groups above a certain size are expected to have performance similar to open access systems. The shape and predicted yield values are derived from the literature and are context dependent. Several studies show how collective action outcomes decrease sharply with groups larger than a few hundred members (Dunbar 1998; Agrawal & Goyal 2001; Yang et al. 2013). Here, we conservatively assume that group sizes above 200 fishers will have a sharp decrease in performance, reaching yield levels expected in equilibrium open access fisheries (Costello et al. 2016a) at groups of 400 or more fishers. We assumed that TURFs with large groups will have performance similar to open access systems to simplify the model. Given that the validity of these assumed values can be context dependent, and that there is no consensus among scholars on forecasting the ideal group size to achieve optimal collective action outcomes (Yang et al. 2013) in specific cases, we also explored the sensitivity of conclusions to these presumed values.

Our model only considers effects of group size on TURF success. Collective action problems created by large group sizes can be overcome through strong leadership (Gutiérrez et al. 2011) or institutional support (Poteete & Ostrom 2004), as we outline in our discussion. The objective of our model is thus to identify cases in which such additional governance interventions are needed.

All TURFs within the database were assigned to one of three categories according to their predicted performance with respect to collective action and resource mobility: *optimally sized, resizing needed* and *additional support needed*. TURFs considered *optimally sized* are those that have predicted performance in or above the 0.75 quantile of predicted yields from both group size and resource mobility effects. TURFs with *resizing needed* could potentially have high performance (0.75 quantile) on both dimensions with an appropriate change in TURF size. TURFs in the *additional support needed* category cannot achieve high performance simultaneously with respect to group size and resource mobility solely from changes in TURF size.

C. Results

We estimate there are approximately 3,700 TURFs worldwide, from which we gathered detailed information on 137. These TURFs have an average size of 367 km² (Table S1). The number of fishers varies greatly across TURFs (mean=1995, median=180, min=11, max=32,000). TURFs are managing for species that differ greatly in adult mobility relative to TURF size, with average predicted yield ranging from 33% to 100%. The effect of group size on average predicted yield from TURFs also varied greatly among TURFs, with projected values ranging from 22% to 100%.

1. Adult mobility

With respect to species mobility, the predicted yield for 137 TURFs worldwide follow the generally expected trend. Large TURFs have consistently high predicted yields, while small TURFs have a wide range of predicted outcomes ranging from very high yields to yields near 20% of maximum sustainable yield (MSY) (Figure 6). Although some TURFs are managing for species with high mobility relative to TURF size, most TURFs are managing for relatively sedentary species such as bivalves and crustaceans. In such cases, TURFs can be relatively small without creating overharvest incentives. At the other extreme, several TURFs are managing for species that have extensive adult movement relative to the size of the TURF resulting in low predicted yields relative to MSY. Countries such as Brazil and Vanuatu are in many cases managing for highly mobile species such as tunas, sharks and sardines. In such cases, it is certain that the species will regularly move outside TURF boundaries, thus creating incentives for fishers to overharvest the resource.



Figure 6. Predicted yield relative to maximum sustainable yield (MSY) due to resource mobility and along shore TURF length for 137 TURFs worldwide.

One important characteristic of several TURFs worldwide is that they manage multiple species that exhibit a wide range of biological characteristics. Consequently, in the same TURF there can be species with high and low mobility, leading to different management incentives within the same area. From our database, 57% of the TURFs are managing for only one species, with the remaining TURFs managing for two or more species. Based on the examples in our database, single species TURFs generally focus on sedentary resources, while multiple species TURFs commonly harvest both mobile and sedentary species.

By examining only the species mobility aspect of TURF design, one solution for increasing predicted yield relative to MSY is to increase TURF size (Figure 6). As indicated in the figure, larger TURFs have lower predicted yield loss relative to smaller TURFs. For smaller TURFs, the predicted performance varies greatly, reflecting the wide variability in the biology of the species being harvested. Thus, many of the existing TURFs within the small end of the TURF size spectrum would likely benefit from increasing TURF size. For example, some TURFs in Brazil have as their main resource species that migrate large distances along the coastline. In this case, increasing TURF size to cover the entire home range of the species would increase predicted yield. However, such increases in TURF size would undoubtedly also increase the number of users within the TURF, negatively affecting collective action outcomes.

2. Group size

The number of users varies greatly within and across countries, with a median of 180 fishers per TURF (Figure 7), but a range that goes from 11 to 32,000. Although many

TURFs have thousands of fishers, 70% of existing TURFs have fewer than 200 fishers. Therefore, most TURFs have group sizes that are small enough to facilitate collective action. For the other third of global TURFs, however, group sizes can be enormous. In areas with high population density, even a relatively small TURF can have thousands of users. For example, Brazil has TURFs with up to 32,000 fishers in an area spanning about 50 km of shoreline. On the other hand, TURFs can be quite large and still have relatively few users. For example, Mexico has TURFs that stretch about 200 km along shore. Yet, they still have fewer than 200 fishers. In the case of Mexico, there is considerable scope for expanding TURF size to match species mobility if necessary, due to the relatively low user density. By contrast, in Brazil the large population densities along the coastline compromise that option.



Figure 7. Number of fishers (A) and predicted yield (B) for 137 TURFs worldwide as a function of their respective along shore length.

When the number of fishers is used to predict TURF performance, most TURFs have predicted yield in the top quantile (Figure 7). However, many TURFs have low predicted yields because of large group sizes. TURFs of similar size across the entire range of observed TURF sizes can have distinctly different predicted performance because of enormous variation in human population densities. Even relatively small TURFs located in areas with high population densities can have low predicted performance. On the contrary, TURFs located in areas with low population densities can have relatively large TURFs and still maintain high predicted yield.

To address the consequences of uncertainty regarding the appropriate group sizes for collective action, we tested a range of inflection points to assess how such changes affect our conclusions. Because the majority of TURFs that are predicted to have poor performance have thousands of users, categorization of most TURFs in our database does not change across a wide range of alternative assumptions about the size of groups that limit collective action. Consequently, the broad trends in predicted performance are relatively insensitive to the current uncertainty surrounding group size impacts on collective action.

3. Interaction between collective action and resource mobility

From all TURFs in our database with complete information (N=113), 65% have all species in the *optimally sized* category, 18% have at least one species in the *resizing needed* category, and 30% have at least one species in the *additional support needed* category (Figure 8). Therefore, the majority of TURFs from around the world have sizes that are simultaneously appropriate with respect to both collective action and resource mobility (*optimally sized* category). However, more than one third of the examined TURFs do not provide the enabling conditions for success. Of these TURFs, a small fraction is predicted to achieve high performance on both dimensions solely from a change in TURF size (*resizing needed* category). The remaining TURFs are in the *additional support needed* category and would need to compensate for at least one driver of low predicted yield with other management solutions.



Figure 8. Interaction between predicted yield relative to maximum sustainable yield as functions of both adult mobility and group size. Different colors represent TURF categories, where: *Optimally sized* – both dimensions with predicted yield in the 0.75 quantile; *Resizing needed* – TURF size can be adjusted to achieve high performance (above 0.75 quantile) along both dimensions; *Additional support needed* – High performance on both dimensions cannot be achieved solely with a change in TURF size.

D. Discussion

The twin challenges of species mobility and collective action among large groups pose challenges to sizing TURFs. Despite these challenges, a majority of TURFs from our database have sizes that we project will foster their success. About two thirds of the TURFs have high predicted yields that are uncompromised by either collective action or resource mobility (*optimally sized* category). Therefore, such TURFs have the enabling success conditions and are predicted to have high performance indicators. TURFs in this category are mostly managing for sedentary species such as bivalves, mollusks or crustaceans, allowing establishment of small TURFs with low number of users. Another class of TURFs in this category are located in areas with low population density, allowing TURFs to be large enough to retain even relatively mobile species while still maintaining small groups of TURF owners.

We project that about 18% of TURFs have at least one species with overharvesting incentives that could be improved solely by changing their size (*resizing needed* category). Because of low population densities or low adult mobility, there will be a range of TURF sizes that can have small enough groups while at the same time being large enough to retain adult mobility. When TURFs are managing for sedentary species, their area can be reduced to decrease group size and still maintain incentives for sustainable management. For example, in Brazil some TURFs are managing species with low mobility (e.g. crabs, oysters) and have group sizes of thousands of fishers. In such cases, it might be more efficient to subdivide TURFs into many smaller TURFs to manage these sedentary species more sustainably. Smaller TURFs would reduce free riders in communities that manage resources with low mobility, thus incentivizing more effective and sustainable management. On the other hand, areas with low population densities can have relatively large TURFs to deal with more mobile species while simultaneously maintaining small group sizes. For example, in Mexico some TURFs can have relatively large areas with small groups. Thus, adjusting TURF size to solve one problem does not inevitably compromise the other. Changing TURF size can be a relatively simple fix to provide the proper management incentives.

For the remaining TURFs (~30%), we project that adjusting TURF size alone cannot address both the collective action problem and the species mobility problem simultaneously (*additional support needed* category). Therefore, more sophisticated and complementary management alternatives are needed (illustrated in Figure 9).



Figure 9. Illustration of the potential effects that different types of management alternatives might have on collective action factors and species mobility.

The first set of potential solutions to these challenges involves introducing new institutions that reduce the challenges posed by species spillover without increasing the TURF size. For example, creating a TURF network where multiple TURFs coordinate their management could produce an outcome that effectively functions as a larger TURF without the problems of collective action within each TURF. In such cases, TURF networks could maximize overall yield by assigning quotas to each individual TURF or by sharing profits across TURFs. Thus, fish will still move out of each individual TURF, but the resulting overharvesting incentives will be reduced or eliminated. However, when fish frequently move out of each individual TURF, the success of this strategy depends on successful coordination among all TURFs within the network. Such coordination might be challenged by heterogeneity in the quality of different TURFs in the network, insufficient communication or coordination among individual TURFs, or increased risk of cheaters. Enforcement across TURFs can also be challenging, since each TURF will have different owners.

Despite these potential challenges to TURF networks, effective coordination has been observed in TURF systems in Japan and Mexico (Wilen et al. 2012; Mccay et al. 2014) where species mobility would otherwise be expected to incentivize overharvest. These systems developed sophisticated management schemes to coordinate management across the network that incentivizes cooperation and maximizes outcomes. For example, the Sakuraebi shrimp TURF network in Japan created a profit sharing system (Wilen et al. 2012). Under this management scheme, harvest is pooled across all TURFs in the network, and net returns are redistributed according to prearranged rules (Uchida & Baba 2008; Wilen et al. 2012). This management system decreases the incentive to overharvest, since fish that leave one TURF, but are caught elsewhere in the network still benefit all TURF owners. Therefore, when there is a system in place to promote cooperation, TURF networks have the potential to alter negative consequences of fish mobility and maximize potential outcomes.

The second set of solutions addresses the collective action challenges within large user groups when reducing TURF size is not possible. Improving the collective action capacity of TURFs is complex and requires an enhancement of community organization within the TURF. A deep knowledge of the social system is required to identify what

elements of collective action need improvement. There are a number of ways large groups have overcome coordination problems, and we see examples in TURF systems as well. The first, and perhaps most widespread in the literature, is through effective leadership and a nested governance structure (Olson 1965; Agrawal 2006). As group size increases, an effective leader can bring different communities together and enhance the likelihood of shared goals (Olson 1965). To produce such leaders, institutions could invest in leadership training. Additionally, the governance structure inside the TURF can be designed to facilitate communication across different communities (Olson 1965). Effective communication is key to building trust among users and to designing effective and fair rules (Olson 1965). For example, a country's Exclusive Economic Zone can be viewed as essentially a large TURF, where each country has exclusive rights to exploit the natural resources within 200 miles from the coast. Because of the large area and high number of users, management is usually designated to states or municipalities that are then supervised by or coordinated with the federal government. Through this analogy, TURF systems could be nested into several communities where the leaders from each area collectively form a single central body where the decisions are made. For example, the TURFs in Brazil have up to 32,000 users, spread over up to 50 communities. One potential path forward would be for these TURFs to develop a governance structure that promotes more effective communication through nested tiers of social organization (Zhou et al. 2005).

When TURFs have rights to fish multiple species, management alternatives to address design problems may vary according to the resource. This situation is expected when TURFs are managing species with different mobility characteristics. For example, some TURFs in Brazil have thousands of fishers and are managing for both sedentary (such as crab) and mobile (such as mullet) species. While sedentary species allow TURF size

reduction to decrease group size challenges (*resizing needed* category), this change in TURF size would exacerbate the problem for more mobile species. In such cases, a combination of smaller TURFs that coordinate across TURFs for more mobile species could provide better management incentives to all species. Alternatively, TURFs could prioritize management alternatives based on the economic importance of the different resources they manage.

Other important factors influencing collective action can be affected by TURF size and are independent of the number of users. Factors such as heterogeneity of users, face-toface/repeated interactions and enforcement costs can be affected when small groups are spread over large areas (Poteete & Ostrom 2004). Under such conditions, increasing TURF size may increase the distance between communities, thus increasing interaction costs and the chance of having different social, cultural and/or economic characteristics regardless of group size. Increasing heterogeneity among users can increase transaction costs and potential conflicts of distribution of benefits and costs. Decreasing the frequency of face-to-face interactions among TURF users can significantly decrease their trust level, in turn diminishing the likelihood that individuals keep their promises to cooperate. Additionally, enforcement costs often have a direct relationship with area, with larger TURFs having much higher costs than smaller TURFs (Davis et al. 2014). Increasing such costs can decrease the ability of TURF owners to exclude other users, thus decreasing management incentives and resource outcomes. Therefore, even though such factors were not considered in this study, they can be affected by TURF size and will only reinforce the group size effect.

Percentage of TURFs in any given category may change if other countries are included in our analysis. Although we controlled for potential biases in our database by constraining the number of TURFs from any given country, including TURFs from other countries might change the worldwide percentages of TURFs within each design category.

For example, Korea has about 300 TURFs but we do not have detailed information on any of them. However, regardless of actual worldwide percentages, our database demonstrates that there will be TURFs in all categories, with a significant number of TURFs needing additional interventions to provide the enabling conditions for success.

The results presented here are theoretical predictions about performance, not empirical estimates of actual TURF performance. To date, there are very few empirical studies that have assessed actual TURF performance (González et al. 2006; Gelcich et al. 2012; Aceves-bueno et al. 2017). As a result, formal comparisons of predicted and actual performance are not currently possible. Collecting empirical evidence on the performance of TURFs worldwide will allow us to assess whether fisheries performance varies predictably with TURF size or whether the management alternatives discussed here for cases with inherent tradeoffs with TURF size are already effectively addressing the opposing challenges of fish mobility and collective action. Empirical analyses may also identify other innovative solutions that have been successful in addressing these potential tradeoffs. While we await such empirical evaluations, these theoretical predictions provide a useful framework for designing new TURFs, and prioritizing additional interventions in existing TURFs, to avoid the negative impacts of too much fish movement or too little collective action.

III. Using TURF networks to manage mobile species: challenges and opportunities

A. Introduction

The sustainability of global fisheries is a major concern worldwide due to severe overfishing of many fish stocks (Worm et al. 2009). Such decline of fished populations threatens food security and livelihoods of millions of people around the globe, especially small-scale coastal fishers in the developing world (Costello et al. 2012). Such communities are also the most affected by overfishing because of their high dependence on seafood as a major protein source, reaching 50% of average per capita intake in some small island nations (FAO 2014). Such overfishing problems faced by small-scale fisheries around the globe is often attributed to the tragedy of the commons derived from open access systems (Hardin 1968). When countries lack strong regulatory institutions, the absence of secure rights can lead to significant overfishing of natural resources. Management systems that provide some kind of rights over the resource can incentivize better resource management (Gutiérrez et al. 2011). Territorial Use Rights for Fisheries (TURFs) are an example of a rights based management system that allocates exclusive spatial rights to groups of fishers to motivate better outcomes (Wilen 2012).

Although TURFs can be a promising solution for overfishing problems in small-scale fisheries, their success is affected by their design, among other factors (Aceves-bueno et al. 2017). TURF design can play a major role in defining and structuring the incentives fishers experience to manage their resource. One of the most challenging design elements is TURF size. Two contrasting effects depend on TURF size – the extent of collective action among TURF owners and the effect of target species mobility on TURF performance. On the one hand, collective action benefits from smaller groups (Olson 1965), such that smaller TURFs function better. On the other hand, species mobility can lead to target species swimming out of a TURF, creating strong incentives to overharvest (White & Costello 2011), such that larger TURFs are better. These contrasting effects can create an inherent tradeoff, particularly in areas with high human densities that are managing species with high mobility. Under these conditions, no single TURF can have the appropriate fisher group size while also retaining fish within TURF borders.

When faced with such tradeoffs, innovative solutions that go beyond solely adjusting TURF size are required. There are two classes of potential solutions: alter the consequences of fish movement or take actions to succeed in large groups. To alter the consequences of fish movement, TURF owners need to make sure that resources are not going to be overharvested after leaving TURF boundaries. To succeed in large groups, TURFs need to develop sophisticated governance systems and have strong leaders (Olson 1965). Here, we explore one solution that has the potential to solve both collective action and species mobility problems - TURF networks (White & Costello 2011). A system comprised of several small TURFs increases the management scope over the target species while maintaining small groups that individually have greater likelihood of successful collective action outcomes. However, for such systems to eliminate overharvesting incentives, high levels of cooperation among the TURFs within the network is required. Thus, it is critical to understand under which conditions TURF networks will cooperate to achieve optimal outcomes. Previous studies suggest that TURFs have incentives to compete over the shared resource, leading to low cooperation levels and suboptimal outcomes (Kaffine & Costello 2011). However, this question deserves further investigation regarding how specific mobility

characteristics affect cooperation incentives and what are the costs derived from noncooperation.

Theory suggests that TURF networks managing for highly mobile species will likely benefit from outside interventions to promote cooperation (Kaffine & Costello 2011). TURFs targeting mobile resources share larger portions of the fished population across the network resulting in greater dissipation of potential benefits. This dissipation of benefit makes free-riding (users that receive benefits without paying for costs) more profitable and incentivizes non-cooperation. When TURF networks are predicted to have low cooperation incentives, the challenge then is how to motivate multiple TURFs to cooperate. One potential strategy to incentivize cooperation is through the use of market-based initiatives (Pirard 2012). Such initiatives can be designed to provide economic incentives for TURFs to cooperate by rewarding those that follow common rules. Such economic incentives can be derived from payments for ecosystem services and/or fish price increases through ecolabeling or access to markets (Pérez-Ramírez et al. 2012).

TURF networks can provide the appropriate conditions for market interventions to take place because of the large scale compared to individual TURFs or communities, especially within the small-scale fisheries context. For example, certification schemes such as the Marine Stewardship Council (MSC) are costly and require assessment of the entire stock (Pérez-Ramírez et al. 2016). For species with high mobility, one single TURF might not be sufficient to ensure sustainability of the resource and might not have the capacity and resources for a full stock assessment. Therefore, such initiatives have greater chances of success when working at the scale of a TURF network where the combination of many small TURFs have greater human capital and resources for such initiatives. Additionally, because of the production scales of a TURF network, cooperatives can have access to markets that

individual TURFs could not. Access to large markets can increase prices paid to fishers and guarantee high quality standards for the seafood caught (Sampson et al. 2015). Creating a system where cooperating TURFs have market advantages over non-cooperating TURFs have the potential to provide the incentives needed to properly manage their shared resource.

Therefore, the objective of this paper is to explore cooperation incentives in TURF networks and assess the viability of market-based initiatives in providing the necessary economic incentives for TURF networks to cooperate. We use a bioeconomic model to simulate the dynamics of a TURF network and evaluate how changes in price within cooperating TURFs can lead to improved system-wide outcomes. We demonstrate the economic and conservation benefits of cooperation and explore what is the price premium needed to achieve full cooperation. Finally, we explore how such interventions can work in practice and under which conditions such market-based initiatives are likely to work best.

B. Material and Methods

We use a bioeconomic model to simulate TURF network dynamics and cooperation incentives. We divide a hypothetical coastline into 100 homogeneous linear patches where we track the biomass of fish within each patch for 20 years. Patches represent individual TURFs within the network and are wrapped to eliminate any boundary effects and to ensure all patches are homogeneous. Although TURF systems would likely have heterogeneous patches, networks with well-designed incentives could account for this heterogeneity and make sure that benefits are homogeneously distributed. Patches are connected through adult movement. A fraction of the population emigrates from each patch to nearby patches with a probability that depends on the distance between the patches. Although the dispersal of larvae beyond TURF boundaries can also connect patches and alter incentives for

sustainable harvest, here we focus solely on the effects of adult movement. A certain fraction of the biomass in each TURF is also removed through fishing, with profits over time representing potential economic gains. Harvest fraction is assumed to be constant over time and does not respond to changes in biomass.

1. Biological model

We use a simple logistic model to track biomass of a given species in each patch over time:

$$B_{t,i} = B_{t-1,i} + g * B_{t-1,i} * \left(1 - \frac{B_{t-1,i}}{K_i}\right) - f_i * B_{t-1,i} - E_{t,i} + I_{t,i}$$
(8)

where $B_{t,i}$ is the biomass in year *t* and patch *i*, *g* is the intrinsic growth rate, K_i is the carrying capacity, f_i is the harvest fraction, $E_{t,i}$ is the emigration from patch *i* and $I_{t,i}$ is the immigration to patch *i* from all other patches. We assume density dependence in the growth rate (*g*) of the population, which ensures that our population does not exceed the carrying capacity of the system. We assume an intrinsic growth rate (*g*) of 0.2, which represents a species with moderate population growth. Additionally, we assume initial biomass at the level expected to achieve maximum sustainable yield (50% of carrying capacity). We assumed this value because it is the abundance when fishers from different TURFs have the lowest incentives to cooperate. When the stock is depleted, there are higher incentives for cooperation to rebuild the stock. However, once the stock is rebuilt, cooperation incentives are predicted to drop again.

Emigration from patch *i* (E_i) equals the biomass of fish in the previous year, $B_{t-1,i}$, times the escapement fraction, represented by μ :

$$E_{t,i} = B_{t-1,i} * \mu (9)$$

Immigration to patch i (I_i) is the sum of the emigration contributions from all other patches j:

$$I_{i,t} = \sum_{j=1}^{100} E_{j,t} p_{ji}$$
(10)

where the proportion of emigrant fish moving from each patch j to patch i, p_{ji} , is defined as (Sala et al. 2013):

$$p_{ji} = \exp(-d_{ji}) \tag{11}$$

/11

)

where $d_{i,j}$ is the distance between patch *j* and patch *i*. Relative proportions are then normalized so that the proportions moving to all other patches sum to one.

Species movement rate (μ) is a critical parameter of the model. This parameter represents the fraction of the population that emigrates from each TURF every year. μ depends on species biological characteristics (such as home range) and individual TURF size. As TURF size increases relative to the species' home range (Kramer & Chapman 1999), fewer individuals move to other TURFs. Conversely, TURFs that are small relative to species home range are expected to experience high mobility values

2. Economic model

Net present value of fisheries profit (NPV_{π}) is the sum across all patches of the discounted profits, defined as the difference between fisheries revenue and predicted costs:

$$NPV_{\pi} = \sum_{t=1}^{t} (pY_i - f_i c) * (\frac{1}{1+\delta})^t$$
(5)

where *p* is the price per kilogram of fish, Y_i is the yield of patch *i*, *c* is the cost per harvest unit, *t* is time in years and δ is the discount rate. Yield is calculated as the product of

the harvest fraction (f_i) and the biomass in each patch (B_i). Cost per unit of effort (c) is calculated assuming that profit is zero when biomass is at 10% of carrying capacity (steady state open access biomass value is derived from (Costello et al. 2016b)) and that cost scales linearly with catch. With these assumptions it is possible to derive the steady state equilibrium cost, which ultimately depends on the biomass, carrying capacity and fish price (Costello et al. 2016a). For model simplification, we assumed a default price (p) of one and a discount rate (δ) of 0.05.

We assume that each TURF will apply a harvest fraction that maximizes long-term profits. This can be accomplished by either complying with predetermined rules aimed to maximize system-wide outcomes (cooperative patches) or by ignoring the rules and setting a harvest level that maximizes individual TURF profit (non-cooperative patches). TURFs comply with predetermined rules (cooperative behavior) only when this action maximizes their profits. When TURFs can obtain greater profits by setting a higher harvest fraction, rules will be ignored (non-cooperative behavior). Therefore, TURFs can choose either to cooperate (comply with rules) or to display non-cooperative behavior (ignore rules) depending on the option that maximizes their individual discounted profits.

The management rule adopted by cooperating TURFs sets a harvest fraction, f_i , which maximizes NPV π across all TURFs (*i*) within the system:

$$\max_{f} \sum_{t=1}^{t} \sum_{i=1}^{N} (pY_i - f_i c) * (\frac{1}{1+\delta})^t$$
(6)

Non-cooperative harvest fraction in each patch, f_i , is calculated as the nash equilibrium harvest level that maximizes profit within each individual TURF, given complete knowledge of stock densities in neighboring patches and movement patterns of the target species. Each TURF sets a single harvest fraction for all years and harvest simultaneously to compute the nash equilibrium of non-cooperative patches, defined as the harvest level where one TURF cannot increase its fishing pressure and obtain higher profits than any other TURF (Fudenberg & Tirole 1991):

$$\max_{f} \sum_{t=1}^{t} (pY_i - f_i c) * (\frac{1}{1+\delta})^t$$
(7)

Predicted equilibrium cooperation level is defined as the number of TURFs that are projected to cooperate (or comply with the rule) within the network. Equilibrium cooperation is calculated as the cooperation level where cooperative and non-cooperative TURFs are predicted to have equal net present value profits. At such equilibrium cooperation level, any additional non-cooperators will be worse off than cooperating patches, thus the incentives for non-cooperation are eliminated. Cooperation outcomes vary from zero (non-cooperators are always better off) to 100 % (cooperators are always better off). Intermediate cooperation values are expected when cooperation (or non-cooperation) is beneficial to a few TURFs.

To explore the implications of market-based initiatives on cooperation incentives of TURF networks, we incorporate a price premium for cooperating TURFs (TURFs that follow the rules). We calculate equilibrium cooperation levels for a range of price increases and movement rates. These simulations allow us to determine the implications of different levels of price increase to cooperation outcomes. Additionally, it allows us to determine the price increase necessary to obtain full cooperation (where all TURFs are better off following the rules) for different mobility rates.

C. Results

Simulations that do not include a price premium intervention show that as movement rate increases, there is a sharp decrease in the predicted equilibrium cooperation among TURFs (Figure 10). When resources are completely sedentary (μ =0) equilibrium cooperation is predicted to be 100%, since TURFs are predicted to apply a harvest fraction identical to what is optimal to maximize economic value of the entire network. As movement rate increases, predicted cooperation levels drop rapidly, reaching 0% cooperation when 23% of the fish emigrate from each individual TURF every year (μ =0.23). Thus, without any strategy to prevent overfishing or incentivize cooperation, TURFs are predicted to apply harvest levels above what is optimal for the entire system, eventually reaching yields similar to open access equilibrium levels.



Figure 10. Predicted equilibrium cooperation level of a TURF network for different mobility levels across TURFs. This result does not include any market-based intervention.

Our results reinforce the importance of obtaining high levels of cooperation for all but the most sedentary species. Results from our analysis show that cooperation levels should be in many cases above 90% for networks to provide significant economic and conservation benefits. Additionally, achieving high levels of cooperation becomes more important as mobility rate increases. For example, when 100% (μ =1) of the fish move out of each single TURF every year, 90% cooperation is predicted to provide only 30% of the potential profit from the system. The benefits of cooperation rise rapidly to 100% of potential profits with 100% cooperation (Figure 11). On the other hand, with 20% mobility, 75% of potential maximum profits can be obtained with 90% cooperation. Predicted conservation outcomes also similarly depend on equilibrium cooperation levels and resource mobility (Figure 11). Systems that do not cooperate and are competing with each other for resources because of high mobility across TURFs are predicted to have biomass levels close to 20% of maximum sustainable yield levels (B_{msy}). These are similar to open access systems (Costello et al. 2016b). As cooperation levels increase, equilibrium biomass also increases, achieving B_{msy} at 100% cooperation. Resource mobility also plays an important role in determining conservation outcomes. Considering the same equilibrium cooperation levels, high mobility species have significantly lower conservation benefits than low mobility species. For example, for a 75% equilibrium cooperation level, 40% of B_{msy} is expected for highly mobile species (100% mobility) while 75% of B_{msy} is expected for low mobility species (10% mobility – Figure 11).





Applying a price premium to cooperating TURFs changes expected equilibrium cooperation levels (Figure 12). Offering market incentives to cooperating TURFs has the potential to overcome incentives for non-cooperation to maximize potential outcomes. Actual price premiums required to achieve full cooperation will ultimately depend on

mobility rates across TURFs, ranging from 0% to 26% (Figure 12). The higher the mobility rate, the higher the price premium required to achieve full cooperation. For example, if 80% of the fish emigrate from each TURF every year (μ =0.8), a 22% price increase is needed to achieve full cooperation. By contrast, species with low mobility rates require are lower price premiums to achieve full cooperation. For example, for a mobility rate of 0.1, a price increase of only 4% is needed to achieve full cooperation (Figure 12). The functional relationship between price increase and expected cooperation is typically quite steep. For all moderate to high mobility rates, equilibrium cooperation levels transition relatively abruptly from no cooperation to full cooperation once the critical price increase is achieved.



Figure 12. (A) illustrates predicted equilibrium cooperation level of a TURF network for different price increase percentage (applied for cooperating TURFs only) for different mobility levels across TURFs. (B) illustrates the price increase necessary to achieve full cooperation within a TURF network for different mobility rates across TURFs.

D. Discussion

Market-based initiatives offer a promising strategy to improve cooperation incentives faced by TURF networks. Lacking such interventions, TURFs with shared resources have profit incentives to harvest at intensities that are above of what is optimal for the entire system. Such non-cooperative behavior can lead to system-wide economic and conservation outcomes similar to open access systems. Offering a price premium for cooperating TURFs has the potential to improve cooperation incentives dramatically and to achieve better outcomes. The price premium required to achieve high cooperation levels depends on mobility rates across TURFs. Species that are more mobile will require higher price increases. When full cooperation is achieved, TURFs can optimize economic and conservation benefits from the entire system, and all TURFs are better off.

Model simulations show that above a certain species mobility rate, and without a price incentive, TURFs within the network are predicted to fish well above from what is optimal for the entire system. Results from our model show that when as little as 23% of the resource leaves each TURF on average over an entire year, no cooperation among TURFs is expected. TURFs that do not cooperate and harvest at rates above the sustainable average will have higher net present value profits over 20 years than TURFs that cooperate. Such
incentives are derived from the fact that when species move among TURFs, users that are cooperating create benefits that can be overexploited by non-cooperators. This allows high harvest levels to be profitable for non-cooperating TURFs and costs associated with limiting catch to sustain future stocks by cooperating TURFs are not compensated by future benefits. Therefore, when TURFs are driven by profit maximization there is a high economic incentive to overharvest whenever there is moderate to high fish movement. Although there are large financial gains to be garnered if the network can achieve high levels of cooperation (>90%), the motivations for individual TURFs make these collective benefits unlikely without other incentives to motivate cooperation.

One potential strategy to incentivize cooperation is through the use of market-based initiatives. Results from our model show that applying a price premium to cooperating TURFs has the potential to significantly improve cooperation incentives. Above a critical price premium, TURFs are predicted to achieve full cooperation and thereby maximize joint economic outcomes. It is important to note that we assumed TURFs have only profit incentives to cooperate. This assumption is important to find an upper bound of the price premium needed to provide sufficient management incentives. However, other social objectives and altruistic behavior can also promote cooperation thus reducing necessary price premium (Gintis et al. 2003). Regardless of the magnitude of price premium needed, it is important to highlight that the success of this strategy relies on price premiums paid to fishers. Thus, it is critical to find ways to increase ex-vessel prices for appropriate management incentives.

The most widespread market-based initiative used around the globe to create price premiums for sustainable actions is seafood certification. Certifying agencies such as the Marine Stewardship Council (MSC) have grown exponentially in recent decades (Bellchambers et al. 2016). This increase has been driven mostly by increases in demand for sustainable seafood products by both consumers (Jaffry et al. 2004) and large seafood retailers (Deighan & Jenkins 2014). Evidence from the literature suggests that consumers are willing to pay higher prices for eco-labeled seafood products, especially in developed countries. For example, a study in the UK shows that eco-labeled seafood products have on average 14.2% higher prices than non-labeled products (Roheim et al. 2011). However, these price increases are at the retail level, and there is mixed evidence on whether this price premium is ultimately passed down to fishers. For example, a recent study from Sweden found no significant change in prices obtained by fishers after MSC certification (Blomquist et al. 2015). A different study found mixed effects of MSC certification on ex-vessel price of Alaskan salmon fisheries (Stemle et al. 2016). While prices at the dock for pink and chum salmon have increased relative to other non-certified fisheries, negative or no effects where found for ex-vessel prices of sockeye, chinook and coho salmon.

Another potential driver of price premiums for fishers can be increased efficiency of the supply chain because of certification. Because of increased interest of large retailers in certified products, certifying a fishery can shorten the supply chain, thus increasing ex-vessel prices. Such breaks in the market structure have been observed as the main benefit of MSC certification of a flounder fishery in Japan (Wakamatsu n.d.), which increases market efficiency and decreases price influence from other larger regional markets.

Building partnerships with large seafood companies can be an alternative to seafood certification for increasing ex-vessel seafood prices. Such partnerships can shorten the supply chain and promote active participation of large retailers in initiatives that improve management. An important example of how large seafood companies are incentivizing fisheries sustainability is through Fisheries Improvement Projects (FIPs) (Sampson et al.

61

2015). To meet global demand for sustainable seafood, important retailers include seafood sourced from FIPs and reward market access for fishing communities that are making progress towards better fisheries management. Fishing communities that are part of FIPs can potentially increase seafood price while improving the conditions of the stock. However, most FIP fisheries are gaining improved market access without delivering significant management improvements (Deighan & Jenkins 2014). This challenge is mainly attributed to the lack of secure rights over the resources (Sampson et al. 2015). Thus, establishing FIP initiatives in TURF networks offers a promising strategy to ensure progress towards better management and achieve the price increase required for high cooperation levels. By establishing FIPs in TURF networks, fishers have the right incentives to deliver required improvements in fisheries management while having the adequate scale desired by large seafood retailers.

Another set of solutions to achieve the required price premium is through market investments at the community level. For example, investments to create a community-based facility where all harvest from the network can be pooled and sold can potentially shorten the supply chain and provide access to larger markets. TURF networks have resources that individual TURFs do not, which can allow for a greater investment in seafood processing and eco-labeling. Seafood processing companies retain from 17% to 27% of seafood retail value (Gundmundsson et al. 2006). Therefore, investing in a community-based seafood processing facility has the potential to increase seafood value to fishers. Combining a processing facility with eco-labeling or FIPs has the potential to improve access to high-end seafood markets, shortening the supply chain and providing a steady demand for seafood products year-round. Therefore, using one or a combination of these strategies can potentially increase seafood prices for cooperating TURFs, incentivizing cooperation in a

62

TURF network. Funding could come from the TURF network itself or from philanthropic or government investments.

Globally, some of the most successful TURF networks have some kind of market incentive to cooperate. A survey of cooperatives from around the globe show that 39% are engaged in some kind of market-based strategy (Ovando et al. 2013). For example, the FEDECOOP TURF network located in the Gulf of Mexico has MSC certification for the lobster fishery and offers a better and steady price for cooperating TURFs. In this system, all yield is sold by the cooperative. Because of the eco-label, over 90% of their catch is sold to international markets (Mccay et al. 2014). Another prominent example of a successful TURF network is the Sakuraebi shrimp TURF system in Japan (Uchida & Baba 2008). This system has a sophisticated management strategy to deal with the high mobility of the shrimp across TURF boundaries. Daily quotas are set based on shrimp prices. All harvest is pooled into one common facility, where it is stored and sold to achieve the highest price possible. After the shrimp is sold, profits are divided among all TURFs within the network, regardless of who actually caught the shrimp (Uchida et al. 2012). These systems show how markets can provide an important incentive to cooperate and when combined with the right management strategies can help achieve optimal economic outcomes in TURF networks.

It is important to note that there are other mechanisms to induce cooperation that were not considered in our model, such as governance (e.g. rules and enforcement) and social norms (e.g. shaming). First, while here we have not considered potential punishments to defectors (through enforcement), it can potentially influence cooperation outcomes. Systems where TURFs have a high probability of being punished for defecting from predetermined rules might induce higher levels of cooperation. Second, social norms inside the communities can also influence cooperation incentives. Communities with a history of

63

collective action might have high social pressure towards following pre-determined rules. Therefore, it is important to consider these other cooperation strategies when predicting cooperation incentives of TURF networks.

E. Conclusion

TURF networks have the potential to change overharvesting incentives derived from open access systems. However, when TURFs are managing species with high mobility rates across the network, TURFs compete for the same resource and have economic incentives to fish above what is optimal for the entire system. Market-based strategies have the potential to provide the required financial incentives to achieve full cooperation and optimal economic outcomes. By providing a price premium for cooperating TURFs full cooperation can be achieved. Effective price premiums will depend on mobility rates – greater mobility rates require higher price increases – and the feasibility of creating situations that enable sufficiently high premiums.

References

- Aburto-Oropeza O, Erisman B, Galland GR, Mascareñas-Osorio I, Sala E, Ezcurra E. 2011. Large recovery of fish biomass in a no-take marine reserve. PLoS ONE **6**.
- Aceves-bueno E, Cornejo-donoso J, Miller SJ, Gaines SD. 2017. Are Territorial Use Rights in Fisheries (TURFs) su ffi ciently large? Marine Policy 78:189–195. Elsevier Ltd. Available from http://dx.doi.org/10.1016/j.marpol.2017.01.024.
- Agrawal A. 2001. Common Property Institutions and Sustainable Governance of Resources 29.

Agrawal A. 2002. Common Resources and Institutional Sustainability. Page 521 in E.

Ostrom, T. Dietz, N. Dolsak, P. C. Stern, S. Stonich, and Elke U. Weber, editors. The drama of the commons. National Academy Press, Washington, DC.

- Agrawal A. 2005. Decentralization and Participation : The Governance of Common Pool Resources in Nepal 's Terai **33**:1101–1114.
- Agrawal A. 2006. Explaining Success on the Commons : Community Forest Governance in the Indian Himalaya. World Development **34**:149–166.
- Agrawal A, Goyal S. 2001. GROUP SIZE AND COLLECTIVE ACTION Third-Party Monitoring in Common-Pool Resources **34**.
- Alban F, Boncoeur J. 2006. ASSESSING THE IMPACT OF MARINE PROTECTED AREAS ON RECREATIONAL USES OF A MARINE ECOSYSTEM: THE CASE OF SCUBA DIVING. Portsmouth Proceedings:1–9.
- Atmodjo E, Lamers M, Mol A. 2017. Financing marine conservation tourism : Governing entrance fees in Raja. Marine Policy 78:181–188. Elsevier. Available from http://dx.doi.org/10.1016/j.marpol.2017.01.023.
- Babcock RC, Shears NT, Alcala AC, Barrett NS, Edgar GJ, Lafferty KD, McClanahan TR, Russ GR. 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. Proceedings of the National Academy of Sciences of the United States of America 107:18256–18261.

00000aab0f01&acdnat=1499718638_a5b976e3e7c42a3fba66a20628f3e9a6 (accessed

Bellchambers LM, Phillips BF, Pérez-Ramírez M. 2016. From certification to recertification the benefits and challenges of the Marine Stewardship Council (MSC): A case study using lobsters. Fisheries Research 182:88–97. Available from http://ac.elscdn.com/S0165783615300758/1-s2.0-S0165783615300758-main.pdf?_tid=3750021a-65ae-11e7-a5ed-

July 10, 2017).

- Blomquist J, Bartolino V, Waldo S. 2015. Price Premiums for Providing Eco-labelled Seafood: Evidence from MSC-certified Cod in Sweden. Journal of Agricultural Economics 66:690–704.
- Botsford LW, Brumbaugh DR, Grimes C, Kellner JB, Largier J, O'Farrell MR, Ralston S, Soulanille E, Wespestad V. 2009. Connectivity, sustainability, and yield: Bridging the gap between conventional fisheries management and marine protected areas. Reviews in Fish Biology and Fisheries **19**:69–95.
- Buxton CD, Hartmann K, Kearney R, Gardner C. 2014. When Is Spillover from Marine Reserves Likely to Benefit Fisheries? PLoS ONE 9:e107032. Available from http://dx.plos.org/10.1371/journal.pone.0107032.
- Castrejo M, Charles A. 2013. Improving fisheries co-management through ecosystem-based spatial management : The Galapagos Marine Reserve **38**:235–245.
- Chae DR, Wattage P, Pascoe S. 2012. Recreational benefits from a marine protected area: A travel cost analysis of Lundy. Tourism Management **33**:971–977. Elsevier. Available from http://dx.doi.org/10.1016/j.tourman.2011.10.008.
- Costello C. 2012. Introduction to the symposium on rights-based fisheries management. Review of Environmental Economics and Policy **6**:212–216.
- Costello C, Ovando D, Clavelle T, Strauss CK, Hilborn R, Melnychuk MC. 2016a. Global fishery prospects under contrasting management regimes. Proceedings of the National Academy of Sciences:1–5.
- Costello C, Ovando D, Clavelle T, Strauss CK, Hilborn R, Melnychuk MC. 2016b. Global fishery prospects under contrasting management regimes:1–5.

Costello C, Ovando D, Hilborn R, Gaines SD, Deschenes O, Lester SE. 2012. Status and

Solutions for the World's Unassessed Fisheries. Science 338:517–520.

- Costello CJ, Kaffine D. 2008. Natural resource use with limited-tenure property rights **55**:20–36.
- Davis D, Tisdell C. 1995. Recreational scuba-diving and carrying capacity in marine
 protected areas Derrin Davis a & Clem Tisdell b. Ocean & Coastal Management 26:19–
 40.
- Davis K, Kragt M, Gelcich S, Schilizzi S, Pannell D. 2014. Accounting for enforcement costs in the spatial allocation of marine zones. Conservation Biology **29**:226–237.
- Deighan LK, Jenkins LD. 2014. Fishing for recognition: Understanding the use of NGO guidelines in fishery improvement projects. Available from http://ac.elscdn.com/S0308597X14002711/1-s2.0-S0308597X14002711-main.pdf?_tid=12e8583c-709e-11e7-93e7-00000aab0f6c&acdnat=1500921168_1f5bd227f447416a38172e08d6678162 (accessed

July 24, 2017).

Dunbar RIM. 1998. The social brain hypothesis. Evolutionary Anthropology: Issues, News, and Reviews **6**:178–190. Available from http://doi.wiley.com/10.1002/%28SICI%291520-

6505%281998%296%3A5%3C178%3A%3AAID-EVAN5%3E3.3.CO%3B2-P.

- Edgar GJ et al. 2014. Global conservation outcomes depend on marine protected areas with five key features. Nature **506**:216–20. Available from http://www.ncbi.nlm.nih.gov/pubmed/24499817.
- FAO. 2014. The State of World Fisheries and Aquaculture. Available from http://www.fao.org/3/a-i3720e.pdf (accessed September 7, 2017).

Farr M, Stoeckl N, Alam Beg R. 2014. The non-consumptive (tourism) "value" of marine

species in the Northern section of the Great Barrier Reef. Marine Policy **43**:89–103. Elsevier. Available from http://dx.doi.org/10.1016/j.marpol.2013.05.002.

Foley MM et al. 2010. Guiding ecological principles for marine spatial planning. Marine Policy **34**:955–966.

Fudenberg D, Tirole J. 1991. Game theory. Cambridge, MA.

- Gaines SD, White C, Carr MH, Palumbi SR. 2010a. Designing marine reserve networks for both conservation and fisheries management. Proceedings of the National Academy of Sciences of the United States of America 107:18286–18293.
- Gaines SD, White C, Carr MH, Palumbi SR. 2010b. Designing marine reserve networks for both conservation and fisheries management. Proceedings of the National Academy of Sciences of the United States of America 107:18286–18293.
- Gelcich S, Godoy N, Canepa A, A MF, Prado L, Castilla JC. 2012. Territorial User Rights for Fisheries as Ancillary Instruments for Marine Coastal Conservation in Chile 26:1005–1015.
- Gintis H, Bowles S, Boyd R, Fehr E. 2003. Explaining altruistic behavior in humans **24**:153–172.
- Goñi R, Hilborn R, Díaz D, Mallol S, Adlerstein S. 2010. Net contribution of spillover from a marine reserve to fishery catches. Marine Ecology Progress Series **400**:233–243.
- González J, Stotz W, Garrido J, Orensanz JM, Parma AM, Tapia C, Zuleta A. 2006. The Chilean turf system: How is it performing in the case of the loco fishery? Bulletin of Marine Science **78**:499–527.
- Green E, Donnelly R. 2003. Recreational scuba diving in Caribbean marine protected areas: do the users pay? Ambio **32**:140–144.
- Gundmundsson E, Asche F, Nielsen M. 2006. Revenue distribution through seafood value

chain. FAO Fisheries Circular 1019:42. Rome. Available from

ftp://ftp.fao.org/docrep/fao/009/a0564e/a0564e00.pdf (accessed August 15, 2017).

- Gutiérrez NL, Hilborn R, Defeo O. 2011. Leadership, social capital and incentives promote successful fisheries. Nature **470**:386–389.
- Halpern BS et al. 2008. A global map of human impact on marine ecosystems. Science (New York, N.Y.) **319**:948–952.
- Halpern BS et al. 2012. An index to assess the health and benefits of the global ocean. Nature 488:615–620. Nature Publishing Group. Available from http://dx.doi.org/10.1038/nature11397.
- Halpern BS, Warner RR. 2003. Matching marine reserve design to reserve objectives.Proceedings. Biological sciences / The Royal Society 270:1871–1878.
- Hardin G. 1968. The Tragedy of the commons. Science 162. Available from http://science.sciencemag.org/content/sci/162/3859/1243.full.pdf (accessed August 11, 2017).
- Hawkins JP, Roberts CM, Van'T Hof T, De Meyer K, Tratalos J, Aldam C. 1999. Effects of Recreational Scuba Diving on Caribbean Coral and Fish Communities\rEfectos del Buceo Recreacional en Comunidades de Coral y Peces del Caribe. Conservation Biology 13:888–897. Available from http://dx.doi.org/10.1046/j.1523-1739.1999.97447.x.
- Hilborn R, Micheli F, De Leo G a. 2006. Integrating marine protected areas with catch regulation. Canadian Journal of Fisheries and Aquatic Sciences **63**:642–649.
- Jaffry S, Pickering H, Ghulam Y, Whitmarsh D, Wattage P. 2004. Consumer choices for quality and sustainability labelled seafood products in the UK Motivation and background. Food Policy **29**:215–228. Available from

www.elsevier.com/locate/foodpol (accessed June 21, 2017).

Janmaat JA. 2005. Sharing clams : tragedy of an incomplete commons 49:26–51.

- Kaffine DT, Costello C. 2011. Unitization of Spatially Connected Renewable Resources
 Unitization of Spatially Connected Renewable. The B.E. Journal of Economic Analysis
 & Policy 11:1–31.
- Kellner JB, Tetreault I, Gaines SD, Nisbet RM. 2007. Fishing the line near marine reserves in single and multispecies fisheries. Ecological Applications **17**:1039–1054.
- Kramer DL, Chapman MR. 1999. Implications of fish home range size and relocation for marine reserve function. Environmental Biology of Fishes 55:65–79. Available from http://dx.doi.org/10.1023/A:1007481206399.
- Lester S, Halpern B, Grorud-Colvert K, Lubchenco J, Ruttenberg B, Gaines S, Airamé S, Warner R. 2009a. Biological effects within no-take marine reserves: a global synthesis. Marine Ecology Progress Series 384:33–46. Available from http://www.intres.com/abstracts/meps/v384/p33-46/.
- Lester SE, Halpern BS. 2008. Biological responses in marine no-take reserves versus partially protected areas. Marine Ecology Progress Series **367**:49–56.
- Lester SE, Halpern BS, Grorud-Colvert K, Lubchenco J, Ruttenberg BI, Gaines SD, Airamé S, Warner RR. 2009b. Biological effects within no-take marine reserves: A global synthesis. Marine Ecology Progress Series **384**:33–46.

Lopez MC, Villamayor-Tomas S. 2017. Understanding the black box of communication in a common-pool resource field experiment. Available from http://ac.elscdn.com/S1462901116308826/1-s2.0-S1462901116308826-main.pdf?_tid=b40b8024-9711-11e7-abaa-

00000aacb362&acdnat=1505148975_9a158a9149e2ad43c898a10a1ded4c76 (accessed

September 11, 2017).

- Mccay BJ, Micheli F, Ponce-díaz G, Murray G, Shester G, Ramirez-sanchez S, Weisman W.
 2014. Cooperatives , concessions , and co-management on the Pacific coast of Mexico.
 Marine Policy 44:49–59. Elsevier. Available from http://dx.doi.org/10.1016/j.marpol.2013.08.001.
- McCook LJ et al. 2010. Adaptive management of the Great Barrier Reef: a globally significant demonstration of the benefits of networks of marine reserves. Proceedings of the National Academy of Sciences of the United States of America **107**:18278–18285.
- Merino G, Maynou F, Boncoeur J. 2009. Bioeconomic model for a three-zone marine protected area: A case study of Medes Islands (northwest Mediterranean). ICES Journal of Marine Science **66**:147–154.
- Milazzo M, Chemello R, Badalamenti F, Riggio RC, Riggio S. 2002. The impact of human recreational activities in marine protected areas: What lessons should be learnt in the Mediterranean sea? Marine Ecology **23**:280–290.
- Nguyen C, Quynh T, Schilizzi S, Hailu A, Iftekhar S. 2017. Territorial Use Rights for Fisheries (TURFs): State of the art and the road ahead. Marine Policy **75**:41–52. Elsevier. Available from http://dx.doi.org/10.1016/j.marpol.2016.10.004.

Olson M. 1965. The Logic of Collective Action. University Harvard Press, Cambridge, MA.

- Olsson P, Folke C, Hughes TP. 2008. Navigating the transition to ecosystem-based management of the Great Barrier Reef , Australia **105**.
- Ostrom E. 2009. A General Framework for Analyzing Sustainability of Social-Ecological Systems. Science **325**.
- Ostrom E. 2010. Analyzing collective action. Agricultural Economics 41.
- Ovando D a. et al. 2013. Conservation incentives and collective choices in cooperative

fisheries. Marine Policy **37**:132–140. Elsevier. Available from http://dx.doi.org/10.1016/j.marpol.2012.03.012.

- Ovando D, Dougherty D, Wilson JR. 2016. Market and design solutions to the short-term economic impacts of marine reserves. Fish and Fisheries:1–16.
- Pascoe S et al. 2014. Estimating the potential impact of entry fees for marine parks on dive tourism in South East Asia. Marine Policy 47:147–152. Elsevier. Available from http://dx.doi.org/10.1016/j.marpol.2014.02.017.
- Pelc R a, Warner RR, Gaines SD, Paris CB. 2010. Detecting larval export from marine reserves. Proceedings of the National Academy of Sciences of the United States of America 107:18266–18271.
- Pérez-Ramírez M, Castrejón M, Gutiérrez NL, Defeo O. 2016. The Marine Stewardship Council certification in Latin America and the Caribbean: A review of experiences, potentials and pitfalls. Fisheries Research **182**:50–58. Available from http://ac.elscdn.com/S0165783615301302/1-s2.0-S0165783615301302-main.pdf?_tid=4d0c8104-65af-11e7-a4d0-

00000aab0f6c&acdnat=1499719104_26da03f3446bf35cda7bdae86b13bc2a (accessed July 10, 2017).

- Pérez-Ramírez M, Ponce-Díaz G, Lluch-Cota S. 2012. The role of MSC certification in the empowerment of fishing cooperatives in Mexico: The case of red rock lobster comanaged fishery. Ocean and Coastal Management 63:24–29. Elsevier Ltd. Available from http://dx.doi.org/10.1016/j.ocecoaman.2012.03.009.
- Pirard R. 2012. Market-based instruments for biodiversity and ecosystem services : A lexicon. Environmental Science and Policy 19–20:59–68. Elsevier Ltd. Available from http://dx.doi.org/10.1016/j.envsci.2012.02.001.

- Poteete AR, Ostrom E. 2004. Heterogeneity, Group Size and Collective Action : The Role of Institutions in Forest Management **35**:435–461.
- Rassweiler A, Costello C, Hilborn R, Siegel D a. 2014. Integrating scientific guidance into marine spatial planning Integrating scientific guidance into marine spatial planning:1–5.
- Reid-Grant K, Bhat MG. 2009. Financing marine protected areas in Jamaica: An exploratory study. Marine Policy 33:128–136.
- Roberts CM, Bohnsack J a, Gell F, Hawkins JP, Goodridge R. 2001. Effects of marine reserves on adjacent fisheries. Science (New York, N.Y.) **294**:1920–1923.
- Roheim CA, Asche F, Santos JI. 2011. The elusive price premium for ecolabelled products: Evidence from seafood in the UK market. Journal of Agricultural Economics **62**:655–668.
- Sala E, Costello C, Dougherty D, Heal G, Kelleher K, Murray JH, Rosenberg A a., SumailaR. 2013. A General Business Model for Marine Reserves. PLoS ONE 8:1–9.
- Sampson GS, Sanchirico JN, Roheim CA, Bush SR, Anderson JL, Ban NC, Fujita R, Jupiter S, Wilson JR. 2015. Secure sustainable seafood from developing countries. Science 348:504–506. Available from www.faa.gov/airports/airport_safety/wildlife/ (accessed July 31, 2017).
- Scott R et al. 2012. Global analysis of satellite tracking data shows that adult green turtles are significantly aggregated in Marine:1053–1061.
- Siegel D a, Mitarai S, Costello C, Gaines SD, Kendall BE, Warner RR, Winters KB. 2008. The stochastic nature of larval connectivity among nearshore marine populations.
 Proceedings of the National Academy of Sciences of the United States of America 105:8974–8979.
- Siegel DA, Kinlan BP, Gaylord B, Gaines SD. 2003. Lagrangian descriptions of marine

larval dispersion. Marine Ecology Progress Series 260:83–96.

Stemle A, Uchida H, Roheim CA. 2016. Have dockside prices improved after MSC certification? analysis of multiple fisheries. Fisheries Research 182:116–123. Available from http://ac.els-cdn.com/S016578361530031X/1-s2.0-S016578361530031X-main.pdf?_tid=bc36bc94-65ae-11e7-9f4e-00000aab0f01&acdnat=1499718861_30500fe68604d6866b6df582a7f00031 (accessed)

July 10, 2017).

- Thur SM. 2010. User fees as sustainable financing mechanisms for marine protected areas: An application to the Bonaire National Marine Park. Marine Policy **34**:63–69. Elsevier. Available from http://dx.doi.org/10.1016/j.marpol.2009.04.008.
- Tratalos J a., Austin TJ. 2001. Impacts of recreational SCUBA diving on coral communities of the Caribbean island of Grand Cayman. Biological Conservation **102**:67–75.
- Uchida E, Uchida H, Lee J-S, Ryu J-G, Kim D-Y. 2012. TURFs and clubs : empirical evidence of the effect of self-governance on profitability in South Korea 's inshore (maul) fisheries. Environment and Development Economics:41–65.
- Uchida H, Baba O. 2008. Fishery management and the pooling arrangement in the Sakuraebi Fishery in Japan. Pages 175–189in R.Townsend, R. Shotton, and H. Uchida, editors.Case studies on fisheries self-governence, 504th edition. Food and Agriculture Organization, Rome.
- Uyarra MC, Côté IM, Gill J a., Tinch RRT, Viner D, Watkinson AR. 2005. Island-specific preferences of tourists for environmental features: implications of climate change for tourism-dependent states. Environmental Conservation **32**:11–19.
- Uyarra MC, Watkinson AR, Côté IM. 2009. Managing dive tourism for the sustainable use of coral reefs: Validating diver perceptions of attractive site features. Environmental

Management **43**:1–16.

- Wakamatsu H. (n.d.). The Impact of MSC Certification on a Japanese Certified Fishery. Available from http://www.bioone.org/doi/full/10.1086/676287 (accessed March 9, 2018).
- Weaver DB, Lawton LJ. 2007. Twenty years on : The state of contemporary ecotourism research. Tourism Management **28**:1168–1179.
- White C, Costello C. 2011. Matching spatial property rights fisheries with scales of fish dispersal. Ecological Applications **21**:350–362.
- Wilen JE. 2012. The Economics of Territorial Use Rights Fisheries, or TURFs TURFs and ITQs : Collective vs . Individual Decision Making University of Rhode Island.
- Wilen JE, Cancino J, Uchida H. 2012. The economics of territorial use rights fisheries, or turfs. Review of Environmental Economics and Policy **6**:237–257.
- Williams ID, Polunin NVC. 2000. Differences between protected and unprotected reefs of the western Caribbean in attributes preferred by dive tourists. Environmental Conservation 27:382–391.
- Worm B et al. 2009. Rebuilding Global Fisheries. Science **325**:578–585. Available from http://www.sciencemag.org/cgi/doi/10.1126/science.1173146.
- Yang W, Liu W, Viña A, Tuanmu M, He G, Dietz T, Liu J. 2013. Nonlinear effects of group size on collective action and resource outcomes **110**:10916–10921.
- Zhou W-X, Sornette D, Hill RA, Dunbar RIM. 2005. Discrete hierarchical organization of social group sizes. Proc. R. Soc. B 272:439–444. Available from http://rspb.royalsocietypublishing.org/content/royprsb/272/1561/439.full.pdf (accessed December 7, 2017).

Appendix



Figure S1 – Relationship between congestion effect, f(x), and marine reserve size for

different tourism potential scenarios (represented by w).



Figure S2 – Relationship between the effect of biomass on potential tourism value, f(B), and fish density inside the marine reserve. Different *c* values represent distinct location

characteristics.



Figure S3 – Sensitivity analysis of the tradeoffs between fisheries and tourism services for well managed and overfished scenarios under different crowding (w) and fish density (b) effects.



Figure S4 – Sensitivity analysis of the optimal marine reserve size for different relative values of tourism and fisheries services under different fisheries management scenarios, and fish density (*b*) and crowding effects (*w*) on tourism net present value.



Figure S5 – Sensitivity analysis of timing of tourism revenue for different marine reserve sizes to crowding effects (w) and fish density effect (c) for well managed scenarios.



Figure S6 – Sensitivity analysis of timing of tourism revenue for different marine reserve sizes to crowding effects (w) and fish density effect (c) for well managed scenarios.



Figure S7 – Sensitivity of the optimal marine reserve size to growth and movement rates in a 1:1 relative fisheries and tourism value for overfished and well managed scenarios.



Figure S8 – Predicted reduction in yield due to adult movement for different TURF sizes and species home range.



Figure S9 – Theoretical prediction of the predicted yield relative to maximum sustainable yield (MSY) for different group size values.