

Appendix A

Methods for the Tun Mustapha Marine Park fishery model

Habitat maps

Habitat designation in the planning units for the Tun Mustapha Park (TMP) model was based on maps of eight key coral reef habitat types in the marine park region (Table A1). Habitat maps were available at a resolution of 100 m². We focused our analysis on the northern region of the proposed park and at a lower spatial resolution than the original habitat maps (1 km², Figs. A1 and A2). The choice to focus on a smaller area and at a lower resolution was made to balance the efficiency of computational optimizations with the minimal area of a single reserve and a biological relevant spatial scale (Sale 2002).

The subset of the park's region that was selected has gradients in fishing pressure and a broad variety of habitat types in a relatively small area. Spatial variability in habitats and fishing pressure was necessary for a meaningful comparison of static and dynamic models. The habitat maps were aggregated to a scale of 1 km² to create the habitat maps by planning units. Each planning unit was assigned the most common habitat type that occurred within its area. It is possible to have multiple habitat types per planning unit, however, this means optimizations of reserve design are more difficult to solve.

Dynamic fishery model

The Malaysian fishery model was a discrete space model. The fishery model was divided into two components: (1) a fish population model, and (2) a spatial fishing effort model.

The fish population model was a space and age-structured discrete time (annual time-step) model of Coral Trout (*Plectropomus leopardus*, Table A2).

Within each time-step, processes occurred in this order: (1) Distribution of fishing effort; (2) harvest, natural mortality, ageing and allometric growth; (3) spawning; (4) larval dispersal; (5) intra-cohort density dependent recruitment.

Ageing was annual and mortality was applied during the age transition. Numbers at age a , time t and in planning unit i were calculated:

$$\begin{aligned} N_{a-1,t-1,i} \exp(-M - F_{t,i}) \forall a > 1 \\ N_{1,t,i} = R_{t,i} \end{aligned} \quad (\text{Eqns. A.1})$$

Where M is the natural mortality rate (constant over space) and $F_{t,i}$ was the fishing mortality rate at time t in planning unit i . $R_{t,i}$ is the number of new recruits to the planning unit. There was total mortality of the few fish older than the maximum age, a_{max} .

Individual growth at age was modeled according to the von Bertalanffy growth equation (von Bertalanffy 1960):

$$L_a = L_\infty \left(1 - \exp(-k(a - a_0)) \right) \quad (\text{Eqn. A.2})$$

Where k is the intrinsic growth rate, L_∞ is the asymptotic length and a_0 is the age at size zero.

Individual weight at age was modeled as a power function of length:

$$W_a = cL_a^b \quad (\text{Eqn. A.3})$$

where c and b are the multiplicative and power coefficients for the weight-length conversion.

Larval production from a planning unit was a linear function of the weight of fish

$$S' = (\mathbf{N}(Wg)) / S_0 \quad (\text{Eqn. A.4})$$

Where \mathbf{N} is a 2 dimensional matrix of numbers in each planning unit, W is the vector of weights at age, \mathbf{m} is an indicator vector of mature ages ($g_a = 0$ for immature ages and $g_a = 1$

for mature ages) and s_0 is the maximum larval production with no fishing. Maturity was a knife edge function.

Larval dispersal was modelled

$$S = S' \mathbf{M} \quad (\text{Eqn. A.5})$$

Where \mathbf{M} is the matrix of dispersal probabilities. Dispersal probabilities from planning unit i to planning unit j were modeled as an exponential function:

$$M'_{i,j} = \exp(-m d_{i,j}) \quad (\text{Eqn. A.6})$$

and then normalized to sum to one from each donor planning unit:

$$M_{i,j} = M'_{i,j} / \sum_i M'_{i,j} \quad (\text{Eqn. A.7})$$

The parameter m is the rate larval dispersal probability declines with distance and $d_{i,j}$ is the distance between planning units i and j . We know of no existing models of larval connectivity for the region. Regardless, empirical studies of grouper (Almany et al. 2013) and other coral reef fish species (Jones et al. 2009) have shown that a large proportion of recruits are spawned from nearby reefs. We set the parameter m so that 55% of larvae settled in their natal planning unit. In additional analyses, we also considered two extreme dispersal scenarios where 96% of larvae settled in their natal planning unit or where larvae were equally distributed across the region (Table A2).

Recruitment was density dependent, with density dependence occurring between recruits to a planning unit (Hixon and Webster 2002, Jones and McCormick 2002). We modeled density-dependence using the Beverton-Holt recruitment function:

$$R_{t,i} = \alpha P_i / (1 + \beta_i P_i) \quad (\text{Eqn. A.8})$$

Where α is the recruit survival rate at low density and β_i controls the maximum number of recruits to a planning unit. We applied the methods in Walters et al. (2007) to calculate the stock-recruitment parameters α and β , based on a hypothesized compensation ratio and natural settlement rate (see also (Brown and Mumby 2014)). Coral trout abundance varies naturally across the eight coral reef habitat types, and is related to exposure (Kingsford 2009). We incorporated variation in habitat quality for coral trout through density dependent recruitment, by scaling α by habitat quality. (Walters et al. 2007). The net effect was that maximum recruitment was lower in habitats with lower quality for coral trout (Table A1).

Harvest weight was integrated over an annual time-step (Rassweiler et al. 2012)

$$H_{t,i} = \sum_{a=a_f}^{a_{\max}} \left[\left(N_{a,t,i} F_{t,i} \left(\frac{\alpha - \exp(-M - F_{t,i})}{M + F_{t,i}} \right) \right) W_{a,i} \right] \quad (\text{Eqn. A.9})$$

Here a_f is the age at first fishing.

Results are presented as profits rather than harvest, which is the difference between revenue and costs. For convenience, costs were modeled in the same units as harvest

$$P_{t,i} = uH_{t,i} - c_{t,i} / (1 + D)^t \quad (\text{Eqn. A.10})$$

Where D is the annual discount rate. The discount rate was zero for the equilibrium planning horizon and set to 0.1 for the short-term planning horizon. Private discount rates can exceed 0.1 in coral reef fisheries (Teh et al. 2013). However, we found higher discount rates had little additional affect on our results, so we present results using a value of 0.1.

The value per kilogram harvested was assumed to be set exogenously (it was fixed). Grouper species are commonly exported to high value markets, to which Tun Mustapha fisheries would only make a small contribution (e.g., McGilvray and Chan 2002), so this assumption is reasonable. Costs were modeled as a function of fishing effort, distance from port, u_i , and a

fixed cost of fishing (Abernethy et al. 2007). The units of effort are arbitrary, so we modeled effort as equal to fishing mortality (Rassweiler et al. 2012).

$$C_{t,i} = F_{t,i} \left(k_1 + (k_2 u_i) \right) \quad (\text{Eqn. A.11})$$

Where k_1 is the fixed cost per unit effort and k_2 is the distance cost per unit effort.

The cost parameters were set such that profits were zero in a planning unit the mean distance from port and when biomass was a fixed fraction, θ (fishing efficiency), of the unfished biomass (Rassweiler et al. 2012). Further, the parameter k_2 was set so that travel costs were a fixed proportion of total costs in planning unit that was the mean distance from port.

Fishing effort was distributed at each time-step according to a location choice model.

Location choice in artisanal fisheries is poorly understood and not adequately represented by economic models such as the Ideal Free Distribution (Abernethy et al. 2007). We allocated effort based on expected profits ($P'_{t,i}$) from a small amount of effort in a habitat patch.

Profits are non-linear function of effort, for high effort levels. However, the relationship between effort and profits is near linear for low effort levels, so it was appropriate here to model expected profits based on a small unit of effort (Rassweiler et al. 2012).

Fishing effort was aggregated on reefs where expected profits are high (e.g., Walters et al. 2007).

$$F_{t,i} = F_T \left(\left(P'_{t,i} x_i \right)^v / \sum_i \left(P'_{t,i} x_i \right)^v \right) \quad (\text{Eqn. A.12})$$

Where F_T is the total effort over all planning units, higher values of v mean greater levels of aggregation at planning units where expected profits are high and x_i indicates whether a planning unit is fished ($x_i = 1$) or a reserve ($x_i = 0$). If $v > 1$, then fishing effort tended to aggregate along the edge of reserves, where fish biomass and expected profits were higher.

This reflects patterns of fishing effort observed in the field (e.g., Goñi et al. 2010).

Population abundance and age structure began at a fished equilibrium for a given effort scenario.

A greedy algorithm was used to optimize the objective function for the dynamic approach (Possingham et al. 2000). Multiple random reserve designs were used as the starting point for the algorithm, to ensure convergence on the near optimal solution. A form of the greedy algorithm was also used to optimize the objective function for the static approach. The optimal reserve design was found by selecting the lowest opportunity cost planning units that ensured adequate protection of each habitat type.

The TMP models and its optimizations were performed using the Julia programming language (julialang.org), in Julia Studio (version 0.3.2, 13 March 2013). Parameters and variables are in Tables A2 and A3.

We performed several supplemental analyses with the TMP model. First, we calculated the mean opportunity cost of reserves, comparing the realistic static and dynamic outcomes (Fig. A3). The opportunity cost was calculated as the summed value of all cells that were designated as reserves, at the time of designation. We also compared the realistic static and dynamic outcomes when larval dispersal was assumed to be either further, or shorter than in the main scenario (Fig. A4).

Methods for the Southern Californian Bight models

The Southern Californian Bight (SCB) red sea urchin population and fishery model follows closely that described by Rassweiler et al. (2012). It has 135 habitat patches (11 with existing marine reserves), and, like the TMP model, is age-structured and in discrete time (annual time-steps) (eqn. A.1). Urchin growth was modeled using the von Bertallanfy function (eqn. A.2). Maturation was knife edge, and upon maturation larval production scaled with biomass (determined via eqn. A.3). Larval dispersal was based on numerical output from a simulation of Lagrangian particles (“virtual larvae”) with red sea urchin life history traits (spawning

season and planktonic larval duration) in a Regional Ocean Circulation Modeling System (ROMS) hydrodynamic model of the SCB (Mitarai et al. 2009, Rassweiler et al. 2012). The output, set as \mathbf{M} , indicated the estimated fraction of transport of urchin larvae between all pairs of patches in the study system. Recruitment of larvae was density limited by an intra-cohort Beverton-Holt function (eqn. A.8). Recruit density depended on available habitat at the settlement location. Mortality of post-recruit urchins was density independent. Model equations also can be found in White et al. (2012b) and Carden et al. (2013), and parameter values to the equations can be found in Carden et al. (2013) and in Table A4.

In the dynamic model fishery harvest (yield) is calculated in relation to fishing effort, as described by eqn. A.9. Effort is regulated in the aggregate by an exogenously determined total allowable fishing effort; in theory, total allowable effort could be controlled by a variety of mechanisms, such as limiting the number of fishermen and days fished, as is the case for the red sea urchin fishery in the SCB (www.dfg.ca.gov). The total allowable fishing effort is allocated across the patches according to Ideal Free Distribution so that average profit per unit effort is uniform among all fishable patches (fleet model: Rassweiler et al. 2012, White et al. 2012b). Given these conditions and a reserve design, total allowable fishing effort is set in the dynamic model to the level that maximizes total fishery profit across the study domain.

In the dynamic model, fishery profit in a patch is a function of fishery revenue less fishing cost in that patch. Revenue is yield multiplied by market price. Cost of harvest is the integral of marginal cost from initial stock density to escapement density in a patch in a year. We assumed marginal cost depended inversely on local harvestable urchin population density (Clark 1990), and for marginal profit of harvesting to decline to zero (i.e., marginal revenue, or price = marginal cost) when the population density is reduced to 10% its carrying capacity (White et al. 2008, Rassweiler et al. 2012, White et al. 2012a). Beyond that point, it is no longer profitable to continue fishing in that year.

Static estimates of profit and biomass in each patch were determined from equilibrium conditions of the dynamic model under status-quo conditions (i.e., given optimal total allowable effort, the 11 no-take marine reserves in the system, and the fleet model). The static profit in a patch that was fished was its status quo profit. The loss in profit when a patch was designated as a reserve as that patch's value (i.e., opportunity cost). The static estimate of wild fish biomass in a patch that is fished was its status quo biomass. The increase in the biomass for designating a patch as a reserve was calculated using the 11 existing reserves as a reference. A function was fit to the relationship between biomass and habitat area in each of the 11 existing reserves under status-quo conditions. The function was then used to estimate unfished (if it were converted to a reserve) biomass in a focal patch in relation to its habitat area (Fig. A5).

$$B_{i,T} = \bar{C}_a(x_i) \quad (\text{Eqn. A.13})$$

Here, $\bar{C}_a(x_i)$ is a function for the expected biomass of a habitat patch with area a and a binary variable for reserve status, x_i (Fig. A5). Thus, the static model estimated potential biomass increase in a patch due to its protection based on the observed biomass and habitat area in the already-protected reserves in the study domain. That is, it uses the existing reserves to predict the effect of closing other patches. If not converted to a reserve, the patch biomass is $\bar{C}_a(x = 0) = B_{i,0}$, which was the equilibrium biomass with status-quo fishing effort in that patch prior to the reserve siting process.

The optimal solutions to equation 1 for the static and dynamic models were found using a Genetic Algorithm, implemented in program MATLAB (ga.m; R2012b; The MathWorks, Inc; Natick, MA, USA) and set to accept 0 (reserve) or 1 (fishable) integer values for each patch of the binary variable x_i . To compare how the designs differed using the static or

dynamic model, we plot the total area of the reserve networks against the conservation weighting (Fig. A6).

Appendix Tables

TABLE A1. Habitat types in the Tun Mustapha model.

Reef type name	No. of planning units	Max. relative recruit density
Fringing, exposed	166	1
Patch semi-sheltered	108	0.5
Fringing, semi-sheltered	186	0.5
Fringing, very sheltered	98	0.5
Patch, sheltered	69	0.5
Patch, exposed	264	1
Limestone, sheltered	14	0.5
Limestone, exposed	24	1

TABLE A2. Parameters in the Tun Mustapha model for coral trout.

Parameter	Values	Units	Reference
<i>Population</i>			
Natural mortality, M	0.45	year ⁻¹	Little et al. (2009)
<i>Growth</i>			
Intrinsic growth, k	0.35	year ⁻¹	Ferreira and Russ (1994)
Asymptotic length, L_{∞}	522	mm	Ferreira and Russ (1994)
Age at size zero, a_0	0	years	Assumed
Maximum age, a_{max}	18	years	Little et al. (2009)
Coef. for weight to length conversion, c	7.9×10^{-5}	g/mm	Ferreira and Russ (1994)
Coef. for weight to length conversion, b	3.157	N/A	Ferreira and Russ (1994)
<i>Reproduction</i>			
Compensation ratio, G	4		Assumed (e.g. Little et al. 2009) uses a steepness parameter of 0.5 which is equivalent to the value used here.
Age at maturity, A_{mat}	3	years	Little et al. (2009)
Larval movement, m	0.5 (2)	PU ⁻¹	Calculated from Jones et al. (2009)
<i>Geographic</i>			
No. planning units with habitat, n	929	N/A	
Habitat quality of a PU, Q_i	{0.5, 1}	relative	
<i>Fishery</i>			
Age fishing begins, A_f	2	years	Ferreira and Russ (1994)
Effort aggregation parameter, v	1	N/A	Assumed

Fishing efficiency	0.01	Proportion of unfished biomass	Assumed
Prop. travel costs of total	$\frac{1}{2}$	relative	Assumed
Discount rate, D	0, 0.1	yr ⁻¹	Assumed

TABLE A3. Variables in the Tun Mustapha model for coral trout.

Variable	Units
Time, t	years
Planning units, i	N/A
Age, a	years
Abundance at age, $N_{a,t,i}$	numbers
Length at age, L_a	milometres
Weight at age, W_a	kilograms
Larval production matrix, S'_{ij}	proportion of p_0
Larval settlement matrix, S_{ij}	proportion of p_0
Maximum larval production, s_0	Kilograms
Over-water distance between PUs, $d_{i,j}$	Metres
Harvest, $H_{t,i}$	Kilograms
Costs, $C_{t,i}$	\$
Profits, $P_{t,i}$	\$
Distance to port, u_i	kilometres
Total fishing effort, F_T	years ⁻¹
Fishing effort, $F_{t,i}$	years ⁻¹
Planning unit reserve status, x_i	N/A

TABLE A4. SCB parameters and their values with justification.

Parameter	Values	Units	Reference
<i>Population</i>			
Natural mortality, M	0.08	year ⁻¹	Rassweiler et al. (2012)
<i>Growth</i>			
Intrinsic growth, k	0.22	year ⁻¹	Rassweiler et al. (2012)
Asymptotic length, L_{∞}	11	cm	Rassweiler et al. (2012)
Age at size zero, a_0	0	years	Rassweiler et al. (2012)
Maximum age, a_{max}	20	years	Rassweiler et al. (2012)
Coef. for weight to length conversion, c	6.7e-4	kg/cm	Rassweiler et al. (2012)
Coef. for weight to length conversion, b	2.68	N/A	Rassweiler et al. (2012)
<i>Reproduction</i>			
Fecundity	Scales with reproductive adult biomass		
Compensation ratio, G	4		Rassweiler et al. (2012)
Age at maturity, A_{mat}	4	years	Rassweiler et al. (2012)
Pelagic larval duration	29	days	Rassweiler et al. (2012)
<i>Geographic</i>			
No. planning units, n	135	N/A	
Habitat area of a PU, Q_i	0-43.6383	km ²	Rassweiler et al. (2012)
<i>Fishery</i>			
Age fishing begins, A_f	8	years	Rassweiler et al. (2012)
Price	0.36	\$/kg	MLPA Initiative (2008)
Discount rate, D	5%	yr ⁻¹	Assumed

Appendix figures

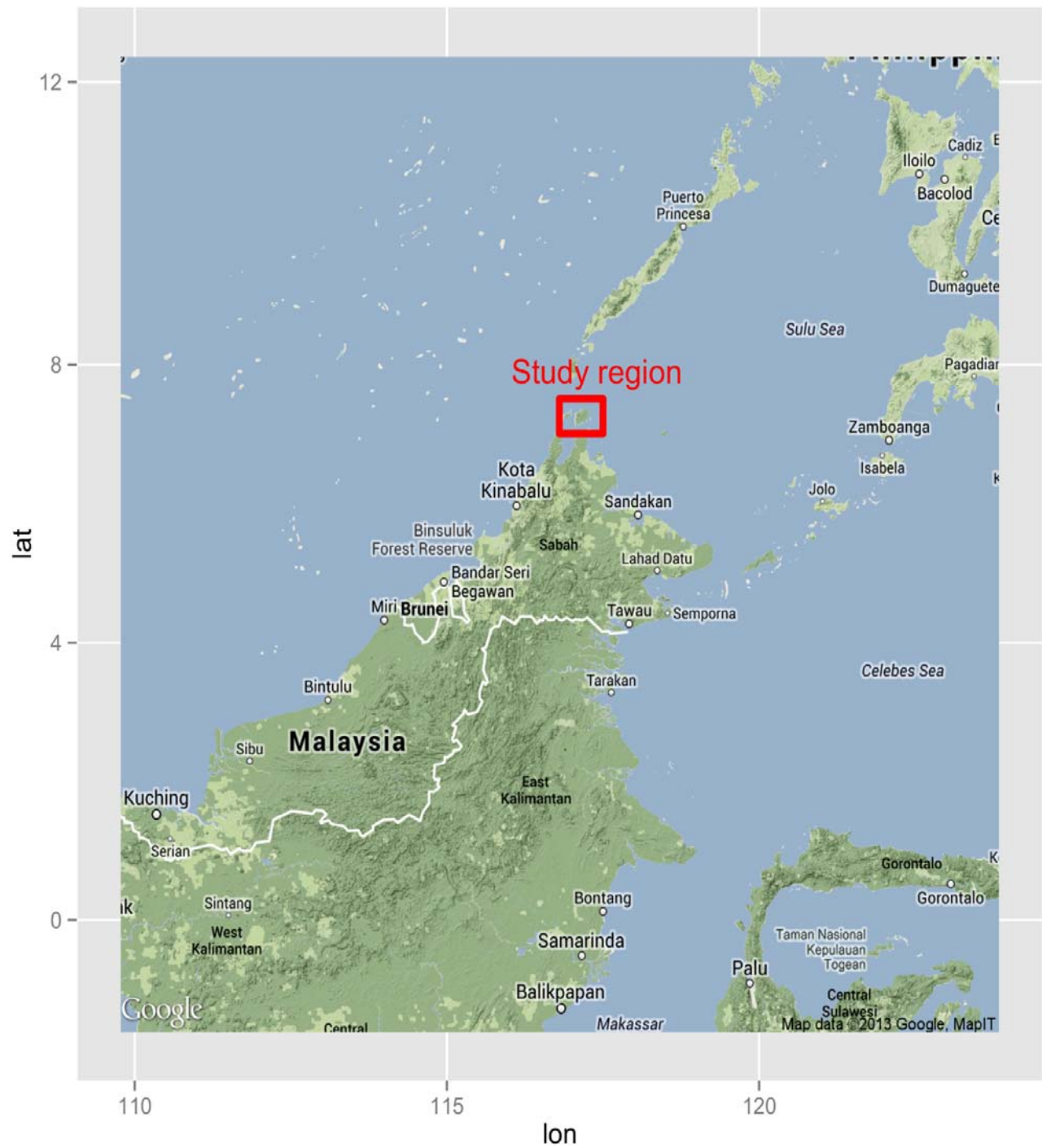


FIG. A1. Map of the Malaysia model study area (source: Google Maps)

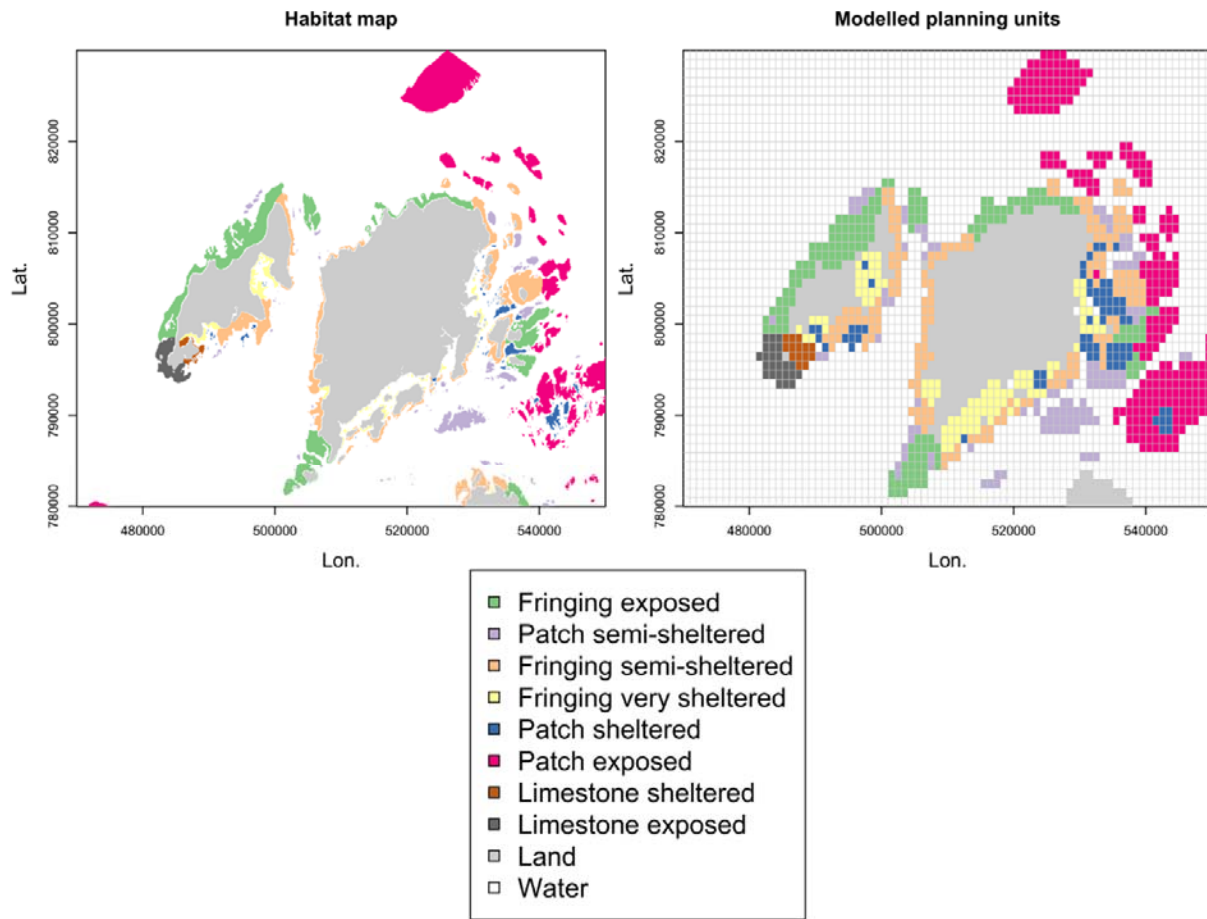


FIG. A2. Habitat maps for the northern Tun Mustapha Marine park (A) and their conversion to planning units for the static and dynamic models (B). ‘Exposed’ refers to exposure to wind/waves.

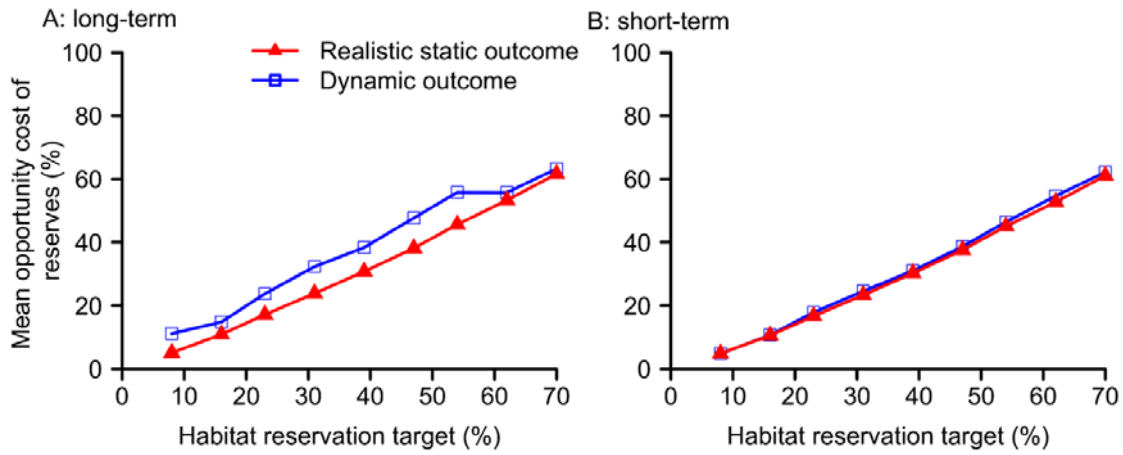


FIG. A3. Comparison of reserve placement in areas of high and low opportunity cost (value of planning units to the fishery before any reserves were implemented), when reserves are designed using the static and dynamic models for the Malaysia model with overfishing, for the long-term planning horizon (A) and the short-term planning horizon (B).

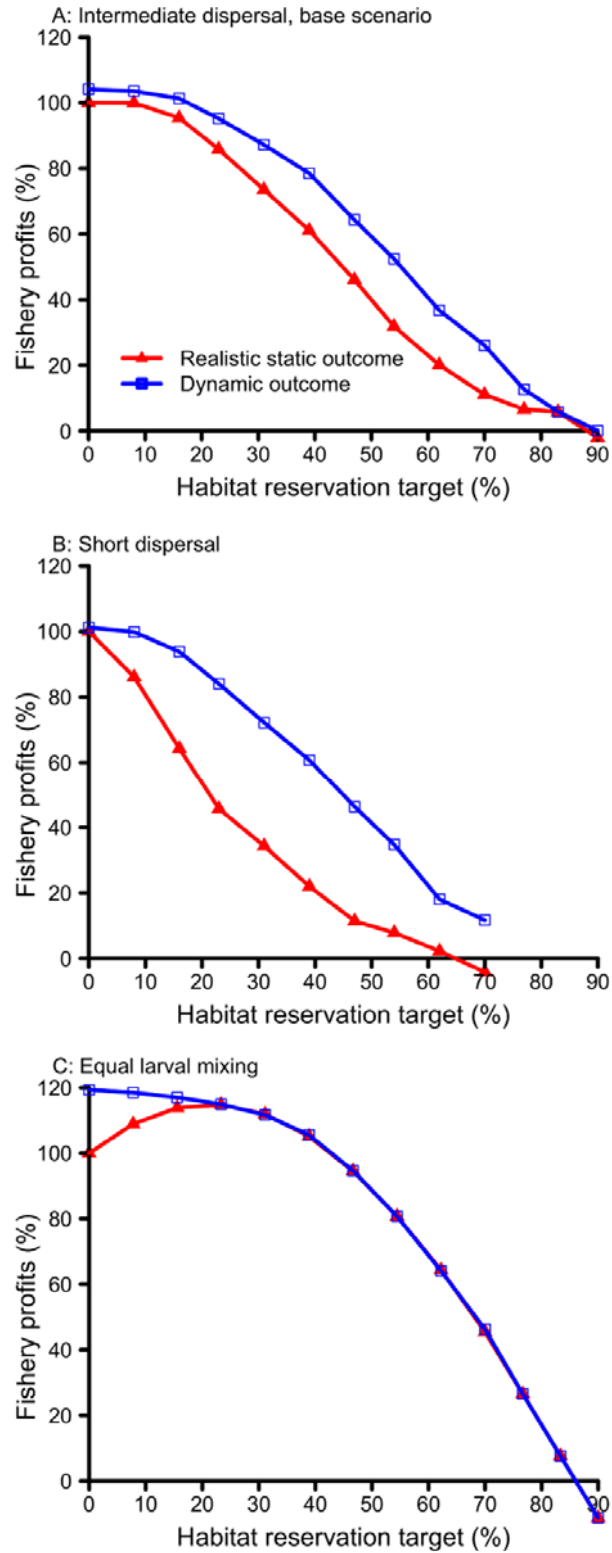


FIG. A4. Comparison of naïve static and realistic static scenarios for the Malaysia model in the long-term for overfishing with an intermediate larval dispersal distance (A, same as fig. 1A), short larval dispersal distance (B) and equal mixing of larvae across all patches (C).

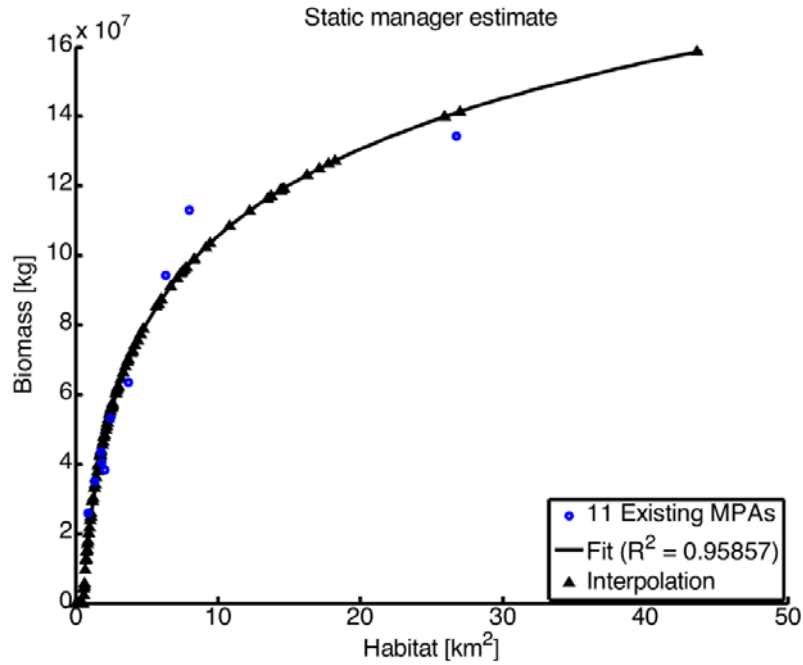


FIG. A5. Static model estimate of biomass in a patch if it were closed to fishing, based on habitat area in that patch and the relationship between habitat area and biomass in the 11 existing no-take RESERVEs at the Northern Channel Islands. Interpolation was calculated from fitting the existing reserve data to the model, $\text{Biomass} = a \cdot \ln(\text{Habitat} + 1) + \text{intercept}$, with the intercept set to zero to represent zero biomass in patches with zero habitat. The best-fit of the model determined $a = 4.294 \times 10^7$ (see legend for coefficient of determination).

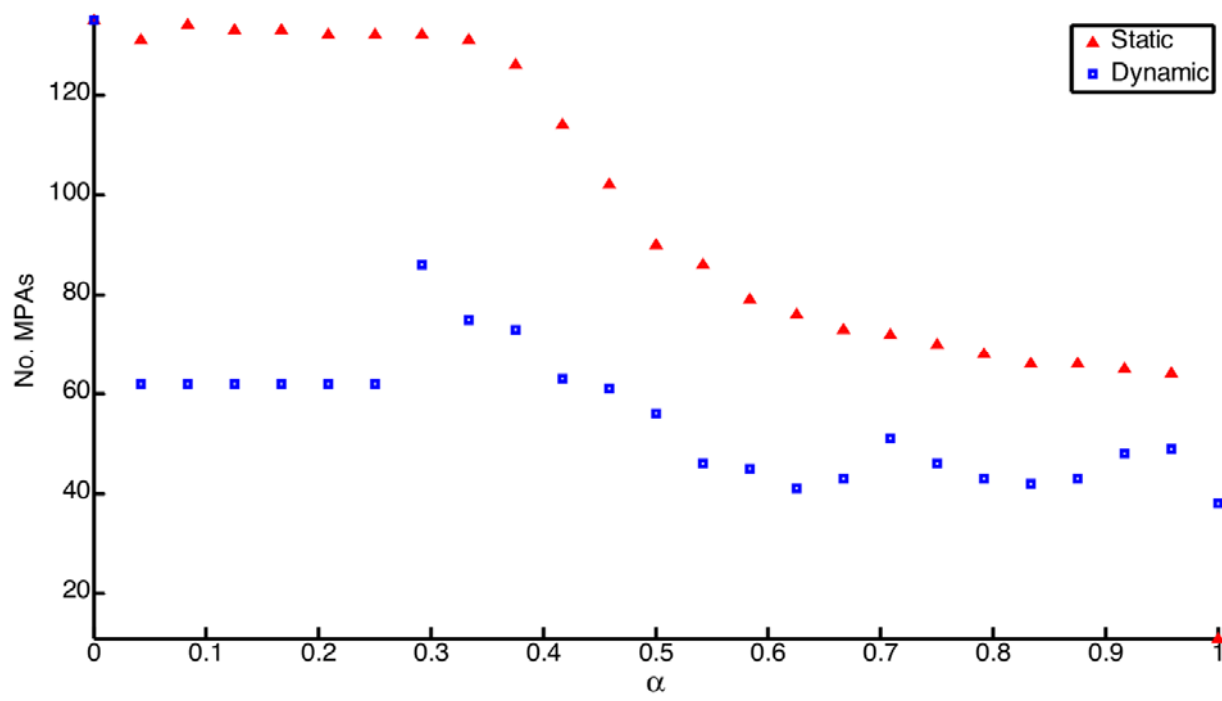


FIG. A6. Comparison of reserves designed using the static and dynamic models for the Californian, for the long-term planning horizon.

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