

# **What's the Damage? Assessing the Costs of Spatial Closures to Protect Corals and Sponges from Bottom Trawling**

**by**

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## **Abstract**

Spatial closures are a widely used management tool for protecting corals and sponges from potential damage by bottom trawling. However, closures can be challenging to design and implement because they may reduce fishing opportunities and landings. To help address this challenge, I assess the costs of spatial closures that protect coral and sponge habitat predicted by species distribution models within Hecate Strait, British Columbia, Canada. Closures are selected using Marxan spatial optimization to target the most suitable coral and sponge habitat for protection, while minimizing losses in bottom trawl landings value, which is estimated using spatially explicit catch records from at-sea observer data. Results suggest negligible losses in landings for 0 to 70% protection of predicted coral and sponge habitat, revealing potential win-win options for spatial closures within Hecate Strait, BC, that protect the majority of predicted coral and sponge habitat while maintaining fisheries value.

**Keywords:** bottom trawling; corals and sponges; Marxan; spatial optimization; species distribution modeling; trade-off analysis

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# 1. Introduction

Corals and sponges are important components of seafloor ecosystems because they provide habitat structure, nursery areas, food resources, and refugia from predators (Turner et al. 1999; Krieger & Wing 2002; Boutillier et al. 2010; Baillon et al. 2012). These ecosystem services may contribute to fisheries productivity, yet they may also be impacted by fishing activities (Sainsbury et al. 1997; Hiddink et al. 2006). The towing of mobile fishing gear over the seafloor removes emergent epifauna (Watling & Norse 1998; Turner et al. 1999), causing long-term and potentially irreversible damage to corals and sponges (Kaiser et al. 2006; Williams et al. 2010). As a result of their sensitivity and ecological value, many coral and sponge habitats are designated as vulnerable marine ecosystems, which United Nations General Assembly resolution 61/105 calls upon nations to protect from destructive fishing practices.

Over the past decade, several bottom trawl fisheries have had regulations implemented to reduce impacts on benthic habitats, especially where those impacts affect corals and sponges (Shester & Ayers 2005; Brock et al. 2009; Penney et al. 2009; Bodtker et al. 2013). A critical component of these regulations is spatial closures that limit where bottom trawling is permitted. Closures are the most effective measure to ensure protection of coral and sponge habitats within an area, with the exception of outright bans on bottom-contact fishing (Auster et al. 2011). However, closures that overlap with fishing areas can lead to losses in fishing opportunities and landings, creating potential trade-offs between fisheries and habitat conservation (Clark & Dunn 2012). To minimize fisheries losses, many closures have been designed based on the distribution of past fishing effort, such that they allow bottom trawling within previously fished areas while

prohibiting expansion into new areas. This approach, known as freezing the fishery footprint, is advantageous because it does not adversely affect fisheries, or require information on the location of coral and sponge habitats (Hourigan 2009). However, because freezing the footprint is based on the distribution of past fishing effort, its effectiveness at protecting coral and sponge habitats is unknown, and they remain vulnerable if they occur within the fishery footprint (Heifetz et al. 2009; Rieser et al. 2013).

Designing spatial closures in consideration of the distribution of coral and sponge habitat could result in greater protection than closures based solely on freezing the fishery footprint (Penney & Guinotte 2013; Rieser et al. 2013), however, ignorance of the location of coral and sponge habitats is a major challenge. The seafloor ecosystems in which corals and sponges occur cover a vast area and are costly to survey, resulting in sparse sample coverage (Ross & Howell 2013). We can fill these gaps in sample coverage using species distribution models (SDMs), which estimate relationships between species occurrence records and environmental data, and then predict species distributions over broader areas where only environmental data is available (Franklin 2009). SDMs have become a common tool for predicting coral and sponge distributions because of their utility in data-sparse situations (Tittensor et al. 2009; Finney & Boutillier 2010; Knudby et al. 2013; Ross & Howell 2013).

In this study, we assess the fisheries costs of spatial closures to protect coral and sponge habitat predicted by SDMs within Hecate Strait, British Columbia (BC), Canada. By estimating costs over a range of management targets, we investigate potential trade-offs between fisheries and habitat conservation objectives. We also evaluate the conservation effectiveness of current closures (Fisheries and Oceans Canada 2012a) by determining their overlap with predicted coral and sponge habitat. Addressing these questions can inform the development of cost-effective measures to protect corals and sponges, and help determine whether current

closures provide sufficient protection (Penney & Guinotte 2013; Rieser et al. 2013). In addition, understanding the trade-offs associated with spatial closures can help balance fisheries and conservation objectives (Stewart & Possingham 2005; Klein et al. 2010), and lead to improvements in conservation outcomes (Gaines et al. 2010; Lester et al. 2013).

We use a spatial optimization approach to select closures that minimize losses in fisheries landings while protecting a pre-determined portion of predicted coral and sponge habitat. First, we create maximum entropy presence-only SDMs to predict habitat suitability for Hexactinellida sponges and Alcyonacea and Pennatulacea corals. We match these predictions with spatially explicit estimates of bottom trawl landings values based on at-sea observer data. Spatial optimization is then performed using Marxan (Ball et al. 2009) to select closures that protect the most suitable coral and sponge habitat while minimizing losses in landings value. We investigate trade-offs between fisheries value and protection of coral and sponge habitat by repeating the spatial optimization over a range of conservation targets to determine how landings value availability varies with the proportion of habitat protected.

## **2. Methods**

### **Study area**

Our study area consists of Hecate Strait and the nearby regions of northern Queen Charlotte Sound and Dixon Entrance, BC (Fig.1). This area is actively fished by the BC groundfish trawl fishery, and contains many coral and sponge occurrences from research surveys, museum records, and incidental catch in commercial fisheries (Finney & Boutillier 2010). The study area also contains a large, globally unique Hexactinellid sponge reef, which has been closed to bottom trawling since 2002 (Jamieson & Chew 2002). In 2012, additional closures were implemented that restricted bottom trawling from expanding to unfished areas, and decreased the fishery footprint by 20% coastwide (Bodtker et al. 2013).

We divided the study area into 31 032 grid cells of 1 km<sup>2</sup> for analysis. The size of grid cells was determined by the spatial scale of environmental predictors for species distribution modeling, the precision of coral and sponge occurrence locations, and the mean length of trawl tows.

### **Data inputs for distribution models**

Coral and sponge occurrence records originated from research surveys, personal collections and museum records. We obtained records from Fisheries and Oceans Canada's invertebrate general status report (Boutillier & Gillespie n.d.), and from catch of Fisheries and Oceans Canada's bottom trawl surveys within Hecate Strait and Queen Charlotte Sound (Olsen et al. 2009) (see Appendix 1 for additional information). These sources are

independent of the distribution of commercial fisheries effort, and therefore not confounded with estimates of landings value.

We selected ten physical and biological environmental variables as model predictors based on availability and relevance to coral and sponge biology. The predictors consisted of bottom salinity and temperature, non-tidal flow speeds, and bottom tidal currents adapted from a circulation model of the eastern North Pacific (Foreman et al. 2008), as well as chlorophyll *a* bloom frequency, depth and slope (see Appendix 1).

## **Coral and sponge distribution models**

Presence-only maximum entropy SDMs (version 3.3.3k; Phillips et al. 2006) predicted habitat suitability for the coral orders Alcyonacea (including Gorgonacea) and Pennatulacea, and the sponge class Hexactinellida. We selected these groups because occurrence records indicated they were the most abundant corals and sponges within our study area (Finney & Boutillier 2010). The Alcyonacea and Hexactinellida groups were modelled at their respective order and class levels, but we divided the Pennatulacea order into the Halipteridae and Pennatulidae families and created SDMs for each family because occurrence records were identified to a higher taxonomic resolution for this group. Model evaluation also revealed that splitting the Pennatulacea order increased predictive accuracy compared to a single SDM with both families.

Models estimated a habitat suitability index (HSI) between 0 and 1 for every grid cell, representing a relative measure of occurrence probability. We applied thresholds that transformed low HSI values to zero to prioritize conservation towards areas that are more likely to contain corals and sponges. This is analogous to applying a presence-absence threshold, except we retained HSI values above the threshold because they may inform habitat suitability. We used the threshold value for each model that maximised the

sum of specificity plus sensitivity (SSS) (Jiménez-Valverde & Lobo 2007). The SSS threshold is robust, generally resulting in a low rate of false negative and false positive prediction errors (Jiménez-Valverde & Lobo 2007), however, other threshold methods were also tested (see Appendix 2). To obtain HSI values for the Pennatulacea order, separate SSS thresholds were applied to the model predictions for the Halipteridae and Pennatulidae families, and then HSI values in each grid cell were summed and rescaled between 0 and 1.

We evaluated SDM predictive accuracy by cross-validation and the area under the receiver operating characteristic curve (AUC). The AUC is a threshold-independent measure of the probability that a randomly selected occurrence location will have a higher predicted HSI than any randomly selected background location (Phillips et al. 2006). An AUC value of 1.0 indicates perfect predictive accuracy while a value of 0.5 indicates the model performs no better than random (Pearce & Ferrier 2000). We took the average AUC for each species group over ten cross-validation models, which we created by excluding ten percent of the occurrence records from model training, and then calculating AUC for these excluded occurrence records. Each of the ten models excludes a different partition of the occurrence records, providing estimates of predictive accuracy based on occurrence data that was not used in model creation. Final SDMs used to estimate trade-offs included all available occurrence records.

## **Fisheries landings value**

We estimated spatially explicit landings value (\$Cdn) for the BC groundfish bottom trawl fishery by multiplying landed weight times the price for each fish species. We obtained annual prices per kilogram for trawl landings from Fisheries and Oceans Canada's Pacific catch statistics database. Spatially explicit catch weights were obtained from the

PacHarvTrawl database and are based on at-sea observer estimates of location and composition for all trawl tows from 1996 to 2011 (Fisheries and Oceans Canada 2012b). The weights and prices represent the 23 groundfish species managed with annual quotas, including 11 rockfish (*Sebastes* and *Sebastes* spp.), 5 flatfish (family *Pleuronectidae*), 2 skates (*Raja* spp.), Spiny Dogfish (*Squalus acanthias*), Walleye Pollock (*Theragra chalcogramma*), Pacific Cod (*Gadus macrocephalus*), Lingcod (*Ophiodon elongatus*) and Sablefish (*Anoplopoma fimbria*).

Due to privacy restrictions, spatially explicit catch weights were only available for grid cell/year combinations that were fished by three or more vessels. Therefore, we obtained catch weights at the 4 km, 2 km, and 1 km resolutions and interpolated weights that were missing at 1 km from lower resolutions. Missing weights were first interpolated from 4 km to 2 km resolutions, then interpolation was repeated from 2 km to 1 km resolutions. Catch weights unavailable at 4 km resolution could not be interpolated and were assumed to be zero because of the low number of vessels (< 3) active within these cells. This assumption could not be avoided given data availability, however, losses in landings value due to spatial closures were underestimated as a consequence. From 1996 to 2011, the percent of landed weight within privacy restricted cells was on average 13% of the total landed weight, and varied by year between 9 to 21% of annual landed weight.

We summarized landings value by adding together the annual estimates of landings value for all catch species and then averaging these annual values across years. Thus, a single estimate for each grid cell represented the mean annual landings value for all groundfish quota species combined. We calculated this mean annual landings value using all years of available data (1996-2011) and the most recent half of years (2004-2011). These two ranges were selected to demonstrate how temporal changes in the distribution of landings value affect trade-off predictions.

## **Spatial optimization of closures**

We used Marxan reserve design software (Ball et al. 2009) to select optimal spatial closures that protect a target portion of coral and sponge habitat while minimizing losses in landings value. Spatial closures were selected using stochastic optimization to minimize an objective function based on: (1) the value of mean annual landings within closed cells, and (2) a penalty for species groups that fall below their conservation target, as defined by the proportion of predicted coral or sponge habitat included within closed areas. For example, a conservation target of 0.5 required the summed HSI value of grid cells within closed areas to be greater than or equal to 50% of the summed HSI value over all cells, otherwise a penalty was added to the objective function. The value of the penalty was adjusted using a species penalty factor so that the conservation target was reached under most optimization scenarios, and the proportion of HSI protected was never lower than the conservation target by more than 0.05. Optimal spatial closures were produced for conservation targets ranging from 0 to 0.99, with increments of 0.05, and applying the same conservation target to each coral and sponge group. To minimize the effects of stochasticity in the optimization procedure, we produced 500 near-optimal spatial closure solutions for all conservation targets and used the average over all solutions.

We used the spatial closure solutions to determine the proportion of total landings value remaining available to fishing for each conservation target. This trade-off relationship represents the minimum proportional reduction in landings value necessary to protect each target portion of coral and sponge habitat. We also examined this relationship for each of the 23 groundfish quota species to investigate if the area closures affected landings for individual species differently.

Spatial closure solutions were also selected with Pennatulacea corals excluded as a conservation target to examine how trade-off relationships



were affected. We hypothesized that excluding Pennatulacea would disproportionately increase available landings because Pennatulacea corals are associated with soft substrates that are easily trawled compared to rocky, hard substrates where Alcyonacea corals and Hexactinellida sponges are more likely to be found.

### **3. Results**

#### **Species distribution models**

The maximum entropy models for the Alcyonacea order, Hexactinellida class, and Halipteridae and Pennatulidae families had AUC values of 0.88, 0.88, 0.85 and 0.86 respectively. The high values and narrow range of AUC results indicates that all models are providing a good fit for the data and have similar predictive accuracy.

There was high overlap in suitable habitat for Alcyonacea and Hexactinellida (Fig. 2), as demonstrated by the proportion of grid cells sharing high HSI values. For example, 71% of grid cells above the SSS threshold overlapped between the Alcyonacea and Hexactinellida models. By contrast, only 19% of cells above the SSS threshold overlapped between Pennatulacea and Alcyonacea, while 32% overlapped between Pennatulacea and Hexactinellida.

#### **Fisheries landings value**

The total annual landings value for the BC groundfish bottom trawl fleet within our study area was estimated at \$6.6 million when averaged over all years (1996-2011), and \$4.7 million when averaged over recent years (2004-2011). This decrease in landings value is attributable to changes in prices and landed weights; the total annual weight of landings declined from 7,900 tonnes to 6,500 tonnes when averaged over all years compared to recent years, while the price per kilogram averaged over all species declined from \$1.21 to \$1.15.

A high proportion of landings value was concentrated over a small portion of the study area, especially when estimated using recent years (Fig.3). There were no fisheries landings in over 78% of grid cells when estimated using all years, and 90% of grid cells using recent years. Grid cells within the 99<sup>th</sup> percentile of landings value, an area of 3 103 km<sup>2</sup>, represented 35% of total landings value over all years and 54% of landings value over recent years.

The contribution of each groundfish species to mean annual landings value was highly variable. The six most valuable groundfish species over all years contributed 71% of the annual landings value and the six least valuable species contributed only 1.4% of landings value. The most valuable species were: Silvergray rockfish and Pacific Ocean perch (*Sebastes* spp.), Dover sole, Lemon sole and Rock sole (family *Pleuronectidae*), and Pacific cod (*Gadus macrocephalus*).

## **Occurrence records within spatial closures**

The proportion of coral and sponge occurrence locations within optimized spatial closures was above or equal to the conservation target for the Alcyonacea and Hexactinellida groups, and slightly below or nearly equal to the conservation target for the Pennatulacea group (Fig. 4). We calculated this proportion to measure the effectiveness of closures at protecting corals and sponges; proportions should be near or above the conservation target if spatial closures are accomplishing their intended conservation objective, and assuming that occurrence records are representative of true distribution patterns.

The proportion of the Hexactinellid sponge reef closure overlapping with optimized spatial closures was also above or equal to the conservation target, and nearly equal to the proportion of Hexactinellida occurrence records within closures (Fig. 4). The Hexactinellid sponge reef closure

encompasses the entire Hexactinellid sponge reef including a small buffer (Fisheries and Oceans Canada 2012a). Therefore, the proportion of overlap between the optimized closures and the Hexactinellid sponge reef closure can help assess the effectiveness of closures at protecting Hexactinellida sponges, and we would expect closures to overlap with the sponge reef by a proportion nearly equal to the conservation target.

## **Trade-offs between fisheries landings and habitat protection**

Trade-offs described in this section are based on spatial closures using landings value estimated over all years (1996-2011), and including Pennatulacea as a conservation target, unless otherwise noted.

Spatial closures did not overlap with areas of landings value for conservation targets from 0 to 0.7 (Fig. 5). Consequently, there were no reductions in available landings value over this range of targets (Fig. 6a). At conservation targets of 0.7 and above, available landings value decreased most rapidly at the highest conservation targets. For example, available landings value was 94.8% of total value with a conservation target of 0.8, decreasing to 85.4% and 63.0% of total landings with conservation targets of 0.9 and 0.99, respectively.

The proportion of available landings value at higher conservation targets differed among groundfish species (Fig. 6). Rockfish (*Sebastes* spp.) landings had the most overlap with spatial closures, while landings for most flatfish (family *Pleuronectidae*) had low overlap. Consequently, the proportion of available rockfish landings decreased by approximately twice as much as landings value averaged all species, and more than three times landings value for flatfish. For example, with a conservation target of 0.9, available landings value for rockfish was 70.2% of total rockfish landings compared to 92.6% of flatfish landings remaining available.

Excluding Pennatulacea as a conservation target had a negligible effect on available landings value, except at conservation targets from 0.9 to 0.99 (Fig. 6b). With a conservation target of 0.99 and excluding Pennatulacea, available landings value was 74.3% of total landings, compared to 63.0% available with Pennatulacea included as a target. Increases in available landings from excluding Pennatulacea was higher among flatfish as well as Pacific cod (*Gadus macrocephalus*), than it was for rockfish (Fig. 6b).

We estimated a higher proportion of available landings value when spatial closures were selected using landings value estimated from recent years (2004-2011) instead of all years of available data (Fig. 6c). Spatial closures did not reduce available landings up to a conservation target of 0.9 when using recent years of landings (Fig. 6c). With the highest conservation target of 0.99, available landings decreased to 89.6% of total landings.

### **Predicted habitat within current bottom trawl closures**

The current spatial closure for the BC groundfish bottom trawl fishery covers 65% of grid cells within the study area and overlaps with suitable habitat equivalent to conservation targets of 0.47, 0.51 and 0.57 for Alcyonacea, Hexactinellida, and Pennatulacea respectively. The proportions of occurrence records within the closures are 0.60, 0.70 and 0.55 for Alcyonacea, Hexactinellida, and Pennatulacea. The current closures reduced available landings value to 97.5% of the total value estimated from 1996 to 2011. This small loss was partly a result of the Hexactinellid sponge reef closure, as well as closure of a few areas with low landings value at the edges of the fishery footprint. By comparison, spatial optimization found closure solutions up to a conservation target of 0.7 with available landings value remaining at over 99% of its total.

## 4. Discussion

Spatial closures may be opposed by commercial fisheries if they reduce fishing opportunities, yet they are key components of regulations to protect corals and sponges from bottom trawling. We estimated the costs of spatial closures for predicted coral and sponge habitat to better understand potential trade-offs between fisheries landings and habitat protection. Our results indicate that up to 70% protection of suitable coral and sponge habitat can be achieved within Hecate Strait, BC, with negligible losses in bottom trawl landings. This suggests potential win-win options for spatial closures that achieve a high level of coral and sponge protection while maintaining fisheries value.

Our results should be interpreted in consideration of error and uncertainty in SDM predictions. Estimated AUC values indicate our models have good predictive power, however, this metric frequently underestimates error in presence-only SDMs, especially if sampling effort is not evenly distributed throughout the study area (Yackulic et al. 2013). The trawl surveys and other sources of occurrence records covered an extensive portion of our study area, but some bias in the distribution of sampling effort is likely. Our study area also lacked some environmental predictors that may be important for corals and sponges, such as dissolved oxygen and bottom type (Tittensor et al. 2009; Knudby et al. 2013). As a consequence of model error, the actual proportion of coral and sponge habitat protected by spatial closures may not be accurately represented by the conservation targets in our analysis. For management purposes, the amount of predicted habitat that is protected could be adjusted upwards of conservation objectives based on the level of uncertainty, according to the precautionary approach (Auster

2001). Accurately quantifying uncertainty and its implications for spatial closures (eg. Tulloch et al. 2013) is therefore an important focus for further research. Model predictions could also be ground-truthed with in situ observations to further assess their reliability and improve accuracy.

Habitat suitability predicted in this study is more likely to represent the potential distribution of corals and sponges (where they could occur) rather than their realized distribution (where they actually occur) because predictions of realized distributions require absence records and non-environmental factors as model predictors (Jiménez-Valverde et al. 2008). Non-environmental factors may be important for the realized distribution of corals and sponges because areas with a history of trawl exposure are less likely to contain intact coral and sponge habitats (Turner et al. 1999; Kaiser et al. 2000). Discounting factors that reduce predicted habitat suitability based on trawl exposure can be applied to SDM predictions to better represent realized distributions (Penney & Guinotte 2013). However, discounting implicitly assumes that recovery of degraded habitats is not an objective of spatial closures, so distinguishing between potential and realized distributions is important when determining conservation objectives.

Our spatial optimization procedure exclusively targeted the major coral and sponge groups within Hecate Strait, however, additional conservation objectives could be considered to address concerns about bottom trawl impacts on other species and habitat types. For example, closures could target soft-sediment habitats that experience low natural disturbance because they may take years to recover from bottom trawling (Kaiser et al. 2006). Alternatively, predictions of benthic habitat sensitivity could be developed based on natural disturbance regimes and biological characteristics (Kostylev & Hannah 2007; Hiddink et al. 2007), then used to develop spatial closures that minimize trawling impacts on benthic productivity (Jennings et al. 2012). Given sufficient biological and

environmental information, additional conservation objectives could be incorporated into the spatial optimization procedure in this study.

Our results suggest optimized spatial closures can protect more coral and sponge habitat in Hecate Strait than current closures for the BC bottom trawl fishery, without any additional losses in landings value. However, these differences may be partly explained by privacy restrictions on landings data as well as operational requirements for closures. To more directly represent trade-offs, we did not consider the patchiness of spatial closures or minimum size requirements for trawl operations, but these practical constraints would reduce the efficiency of closures (Bodtker et al. 2013).

We also found that differences in conservation efficiency between current closures and optimized closures were particularly pronounced when landings value was estimated using recent years (2004-2011) instead of all years (1996-2011) of available landings data, which resulted in approximately 20% greater protection of coral and sponge habitat without affecting available fisheries landings. This suggests that the time frame used to determine fisheries landings or the fishery footprint is an important component of closure decisions, yet the rationale for selecting a particular time frame is rarely given, and in many cases seems to be determined by data availability (Shester & Ayers 2005; Jennings et al. 2012). Since the spatiotemporal distribution of fishing effort is not static and may be influenced by many factors including market conditions, changes in regulations, area closures and resource availability (Béné 1996; Murawski et al. 2005), its influence on closure decisions should be considered whenever possible.

There are potential limitations in using past landings value to represent fisheries preferences for spatial closures because landings value oversimplifies fisheries objectives and may not accurately represent costs. Estimated losses in landings value do not consider the potential positive

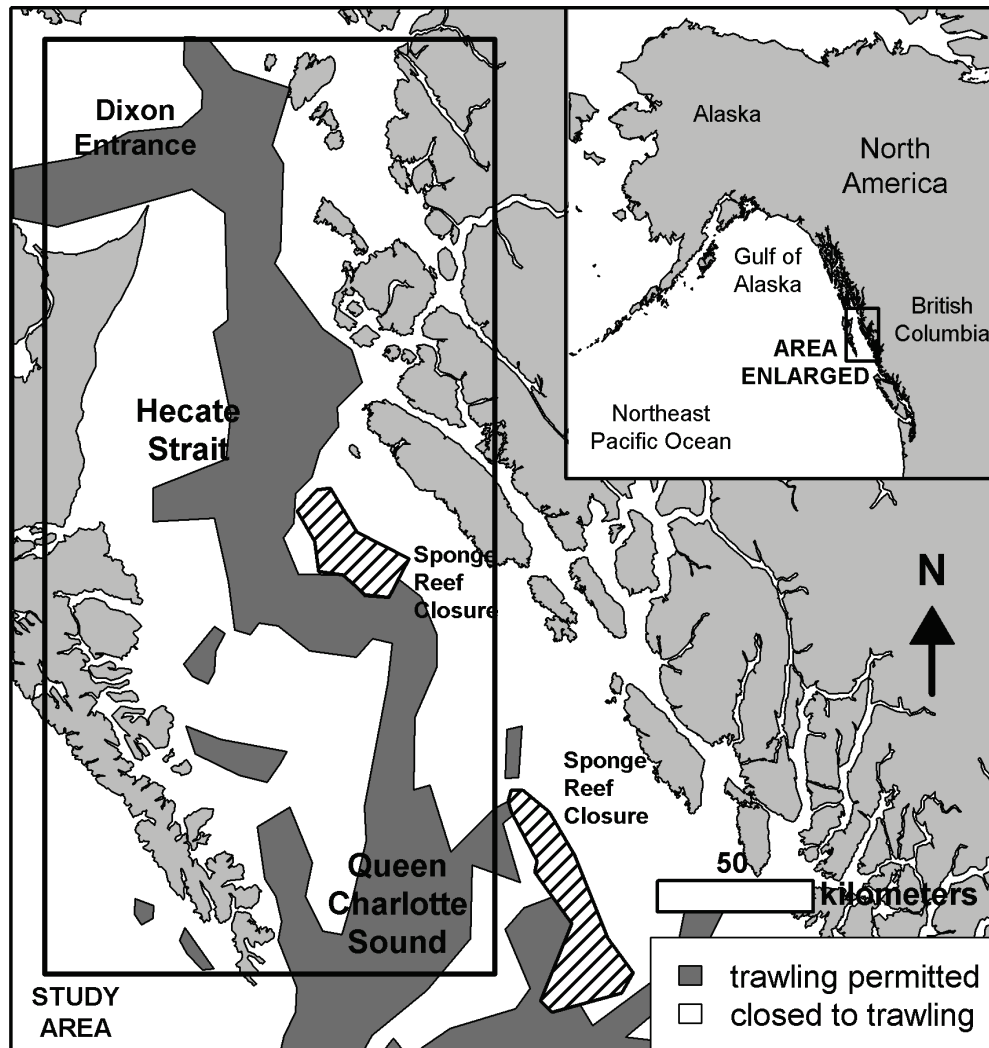


effects of habitat protection, such as increased productivity and spill-over into adjacent fishing grounds, which could increase fisheries yields and offset some losses in landings (Sainsbury et al. 1997; Gaines et al. 2010). Bottom trawl vessels may also mitigate losses by redistributing effort to areas that remain open (Klein et al. 2010), or selling individual transferable quota to other groundfish sectors, such as hook and line or longline vessels (Fisheries and Oceans Canada 2012a). On the other hand, past landings value does not include potential opportunity costs associated with spatial closures, or differences in losses among individual vessels. Furthermore, we found that losses in landings value were unevenly distributed among groundfish species, with rockfish landings being particularly sensitive to spatial closures, likely because they are associated with coral and sponge habitat (Malecha & Heifetz 2005; Stone 2006). Many vessels within the BC groundfish fleet are effective at targeting specific species based on available quota (Branch & Hilborn 2008), however, uneven overlap of spatial closures among groundfish species would affect the ability of vessels to fill certain quotas, or adjust to changes in allowable catch. This could have a greater impact on fisheries profitability than suggested by available landings value over all groundfish species combined.

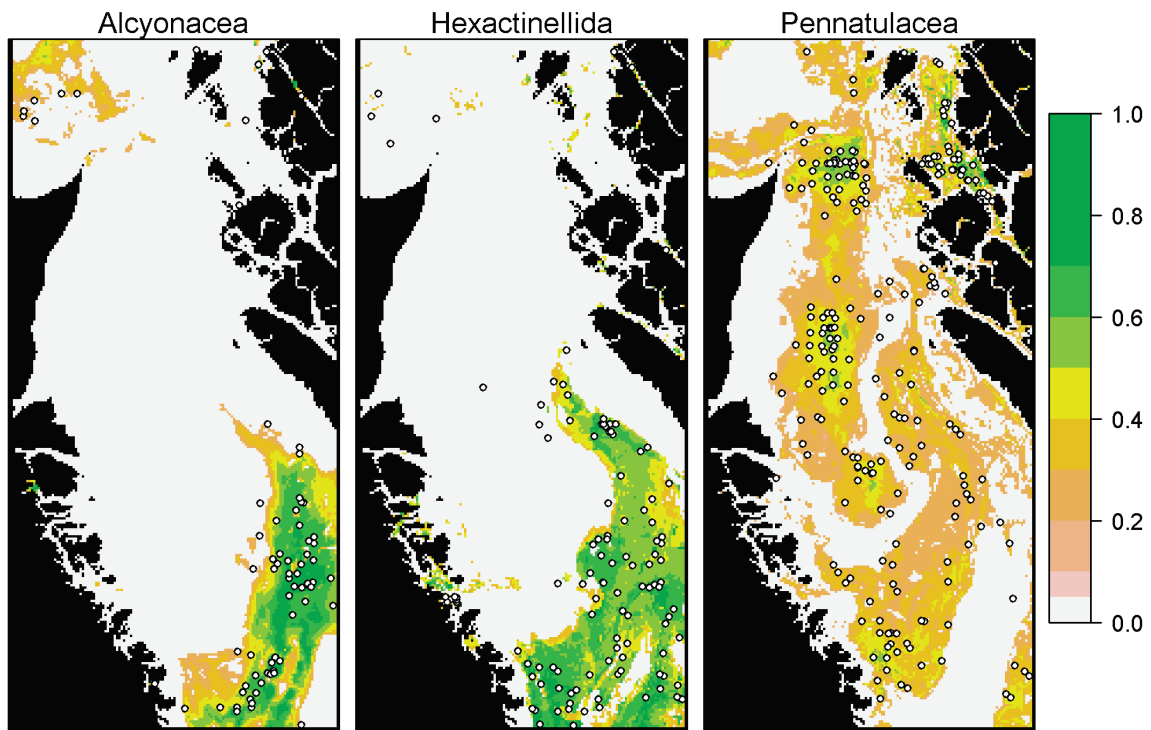
Further research could improve the optimization procedure for spatial closures by better representing fisheries preferences, and increasing the reliability of predicted distributions for corals and sponges. Nonetheless, our study demonstrates how such an approach can assess the trade-offs and costs associated with protecting coral and sponge habitats, which can inform the development of cost-effective spatial closures. By applying explicit conservation targets to spatial closure decisions, our approach also allows management options to be evaluated and compared as part of a systematic conservation planning framework (Margules & Sarkar 2007). This can increase the credibility and transparency of protection measures for corals, sponges and other components of vulnerable marine ecosystems (Agardy et

al. 2003; Rieser et al. 2013), and allow closures to be improved adaptively as new information becomes available.

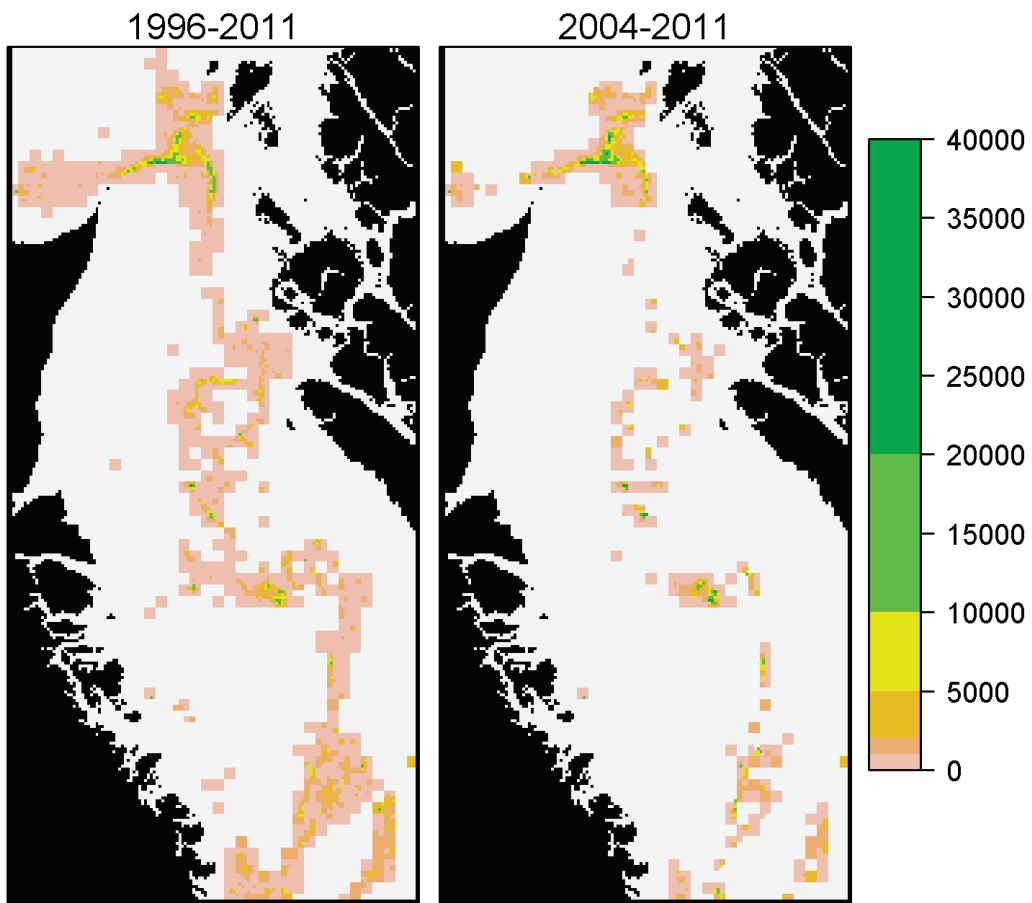
## Figures



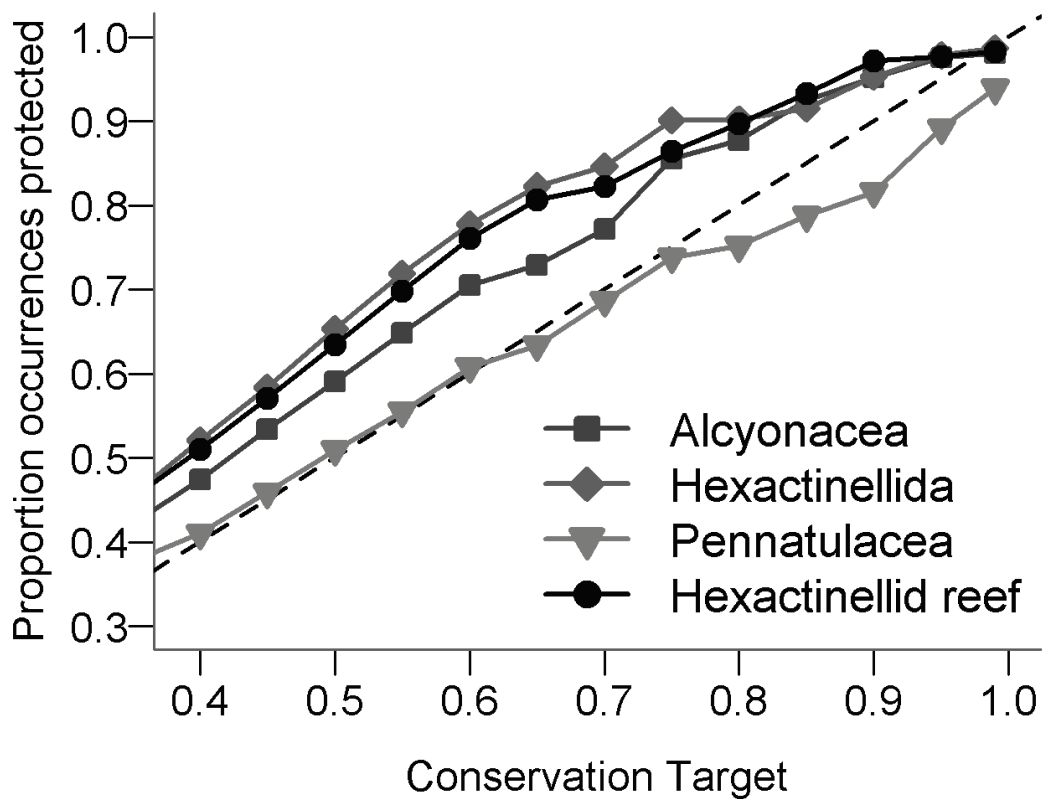
**Figure 1.** The study area of Hecate Strait, northern Queen Charlotte Sound, and Dixon Entrance, along with current spatial closures for the British Columbia bottom trawl groundfish fishery. Spatial closures that restricted bottom trawling to dark grey areas were implemented in 2012. The sponge reef closures (hatched areas) were implemented in 2002 to protect large *Hexactinellid* sponge reefs.



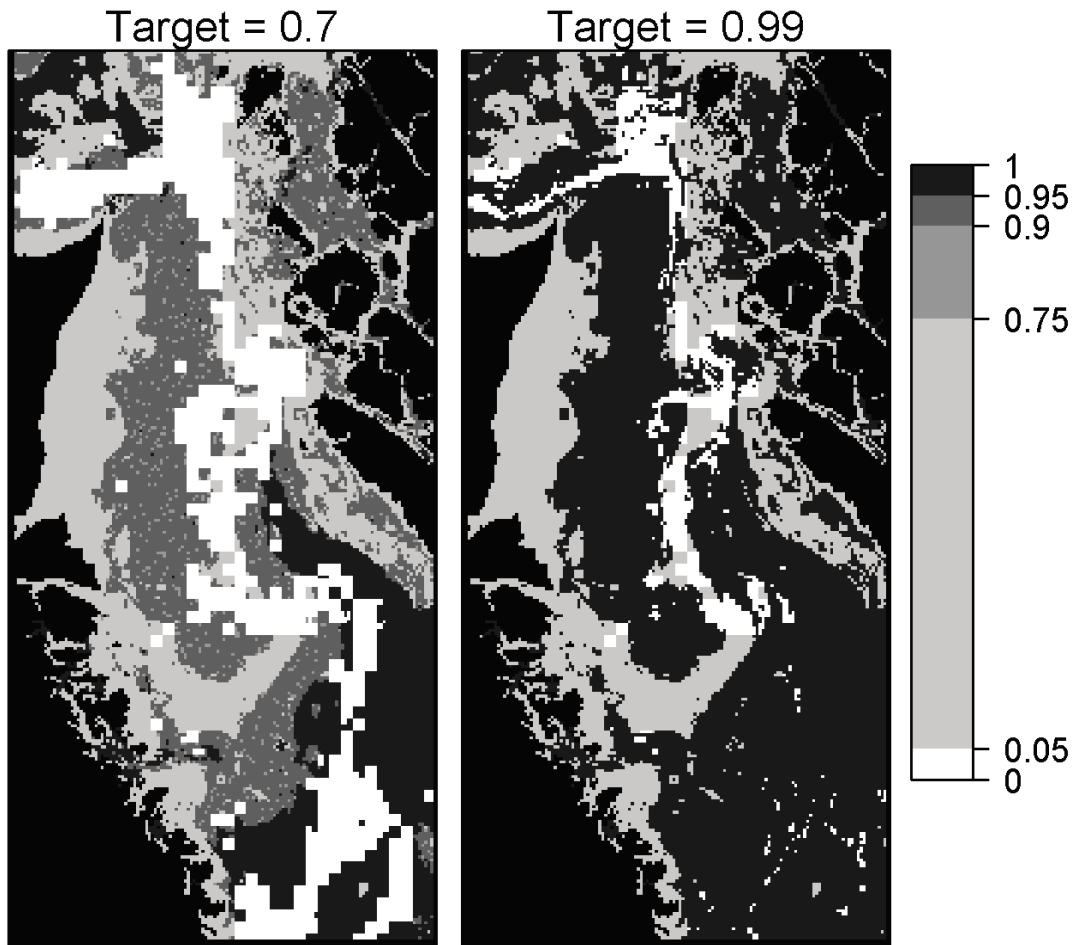
**Figure 2.** Maximum entropy model predictions of habitat suitability for the coral orders *Alcyonacea* and *Pennatulacea*, and the sponge class *Hexactinellida*, after applying the maximum sum of specificity plus sensitivity threshold. Presence records used in model fitting are indicated by white circles. The habitat suitability for the *Pennatulacea* order represents the summed model predictions for two families, *Halipteridae* and *Pennatulidae*, rescaled to a maximum value of 1.



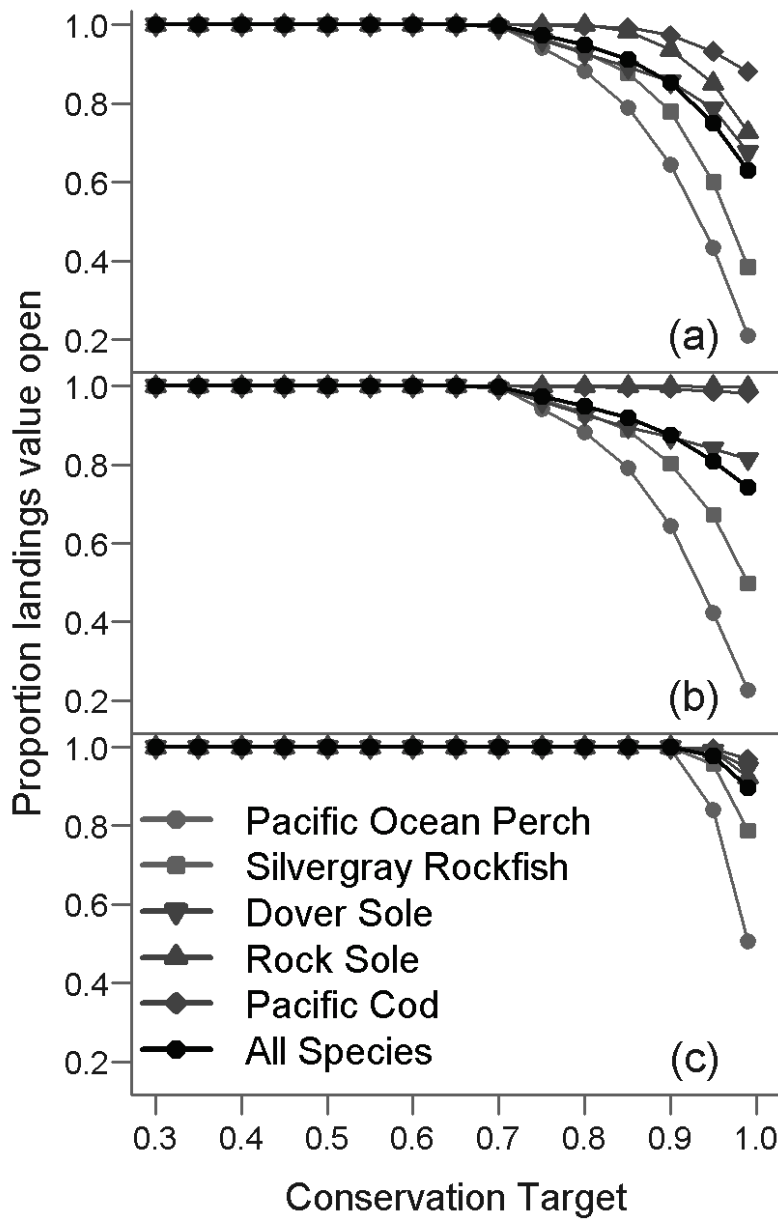
**Figure 3.** Estimates of mean annual landings value (\$Cdn) for all 23 groundfish species managed with quotas, averaged from 1996-2011 (left panel) and 2004-2011 (right panel).



**Figure 4.** Proportion of occurrence records within Marxan optimized spatial closures over the upper range of conservation targets and using landings value estimated from 1996-2011. The Hexactinellid reef line (black with circles) shows the proportion of the *Hexactinellid* sponge reef closure that overlaps with optimized spatial closures. The dotted line shows where the proportion of occurrences protected equals the conservation target.



**Figure 5.** Shaded frequencies representing the proportion of times each grid cell was closed out of the 500 spatial closure solutions produced by Marxan using conservation targets of 0.7 and 0.99. White spaces were closed by less than 5% of the spatial closure solutions and represent areas of significant landings value. The frequencies shown here estimate landings value by averaging over all years of data (1996-2011), and include the *Pennatulacea* order as a conservation target.



**Figure 6.** Proportion of the total landings value remaining open to trawling as a function of the conservation target for protection of coral and sponge habitat. Landings value is shown for all groundfish species added together and for the 5 most valuable species within the study area. Panel (a) is the default scenario where landing value is estimated using all years of landings data (1996-2011), and *Pennatulacea* is included as a conservation target. In panel (b), *Pennatulacea* is excluded as a conservation target. In panel (c), landings value is estimated using recent years of landings data (2004-2011) and *Pennatulacea* is included as a conservation target.



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## **Appendix A.**

### **Data inputs for maximum entropy species distribution modelling**

#### **Occurrence records**

The occurrence records of corals and sponges used in this study were obtained from catch records of the Fisheries and Oceans Canada (DFO) groundfish trawl surveys (Olsen et al. 2009), and from the DFO invertebrate general status report (Boutillier & Gillespie n.d.). The location of catch records from trawl surveys were based upon the start point of trawl tows with a target bottom time of 20 minutes (Olsen et al. 2009). The DFO invertebrate general status report is compiled from 37 different sources that include other DFO research surveys (eg. tanner crab survey and shrimp trawl survey), private collections and museum records (eg. Royal Ontario Museum, Royal British Columbia Museum). The Alcyonacea model had the least number of occurrence records ( $n = 74$ ), while the Hexactinellida model had the most occurrence records available for model fitting ( $n = 154$ ) (Table A1.1). Records dated as far back as the 1960s, although the majority of occurrences were recorded since the year 2000.

#### **Environmental predictors**

The environmental predictors were selected among all available, relevant predictors within our study area (Fig. A1.1, Table A1.2). We began with 19 environmental predictors and removed highly correlated predictors until 10 were selected for inclusion in final models. We assessed correlation using Pearson's correlation coefficient ( $r$ ) and removed predictors with  $|r| > 0.8$ . All the removed predictors were related in terms of the environmental variable that was represented. For example, maximum temperature was highly correlated with summer temperature. We retained maximum, minimum or ranges of environmental variables, and discarded seasonal variables.

A number of physical oceanographic variables were based on a tidal circulation model of the eastern North Pacific (Foreman et al. 2008). These variables included seasonal, maximum, and minimum bottom temperature and salinity, bottom tidal speed, and summer and winter non-tidal currents. These predictors were calculated at variable resolutions from 100 m in coastal channels up to 70 km in the deep ocean. Temperatures and salinities were estimated from conductivity-temperature depth, bottle expendable bathy-thermograph and Argo data from NOAA, Marine Environmental Data Service and Institute of Ocean Sciences archives (Foreman et al. 2008).

Average summer and winter non-tidal currents are based on monthly mean values from the National Centers for Environmental Predictions (NCEP).

Depth was obtained from an integrated 100 x 100 m<sup>2</sup> bathymetric grid (Gregr, unpublished data) derived from Fisheries and Oceans Canada and National Oceanic and Atmospheric Administration source data. Slope was calculated from depth using the terrain() function from the “raster” package in R (Hijmans & van Etten 2013; R Core Team 2013), which uses the eight neighbouring cells to calculate slope in degrees.

Chlorophyll a bloom frequency was calculated using the algal\_1 band from the Medium Resolution Imaging Spectrometer (MERIS) of the European Space Agency Environmental Satellite (ENVISAT) for spring 2007 to 2011 (Gregr, unpublished data). This variable represents the number of months in which each grid cell was classified as undergoing chlorophyll a blooms ([chl<sub>a</sub>] > 2.0 mg/m<sup>3</sup>) from March 18th to June 21st in 2007 to 2011. Therefore, values range from 0 to 20, with a value of 20 representing blooms occurring every month.

### Maximum entropy model settings

The Maxent models were created using the default settings of the Maxent species distribution modeling software, version 3.3.3k (Philips et al. 2006). These settings included: a) number of randomly selected background points = 10,000; b) default prevalence = 0.5; c) output = logistic; d) feature selection = auto; e) regularization parameter = 1. These default settings were shown to be appropriate for a similar set of species distribution models for coral groups in the Pacific region of British Columbia (Finney 2009).

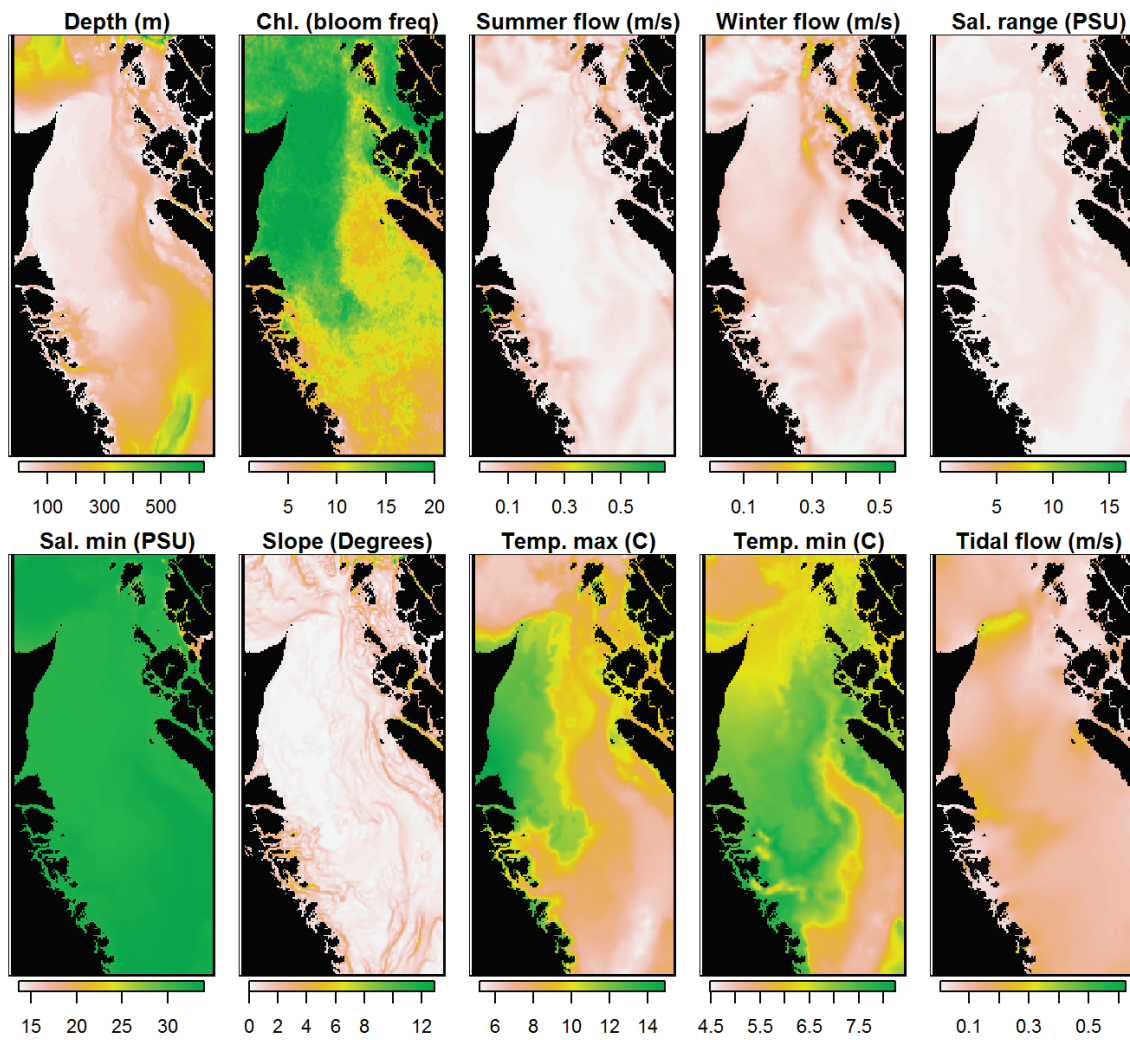
**Table A1.1.** Presence record data inputs for the Alcyonacea, Hexactinellida, Pennatulidae and Halipteridae species distribution models.

Species	Source	Year range	Number Occurrences
Alcyonacea	DFO Groundfish trawl survey	1987-2011	63
	DFO Invertebrate database	1960-2006	11
Hexactinellida	DFO Groundfish trawl survey	2005-2011	89
	DFO Invertebrate database	1976-2000	65
Pennatulidae	DFO Groundfish trawl survey	2000-2011	62
	DFO Invertebrate database	1960-2011	60
Halipteridae	DFO Groundfish trawl survey	2003-2011	62
	DFO Invertebrate database	1966-2008	52



**Table A1.2.** Environmental predictors that were available within the study area. I removed highly correlated predictors ( $r > 0.8$ ) from the final species distribution models.

Environmental Layer	Units	Source	In final model
Depth	Meters	Fisheries and Oceans Canada and National Oceanic and Atmospheric Administration bathymetry	Yes
Slope	Degrees	Calculated from depth using terrain() function in R raster package	Yes
Chlorophyll-a bloom frequency	Frequency of blooms during spring months from 0-20	Medium Resolution Imaging Spectrometer (MERIS)	Yes
Flow – summer (non-tidal)	Meters/second	Foreman et al. 2008	Yes
Flow –winter (non-tidal)	Meters/second	Foreman et al. 2008	Yes
Tidal flow	Meters/second	Foreman et al. 2008	Yes
Salinity range	PSU	Foreman et al. 2008	Yes
Salinity summer	PSU	Foreman et al. 2008	No
Salinity winter	PSU	Foreman et al. 2008	No
Salinity fall	PSU	Foreman et al. 2008	No
Salinity spring	PSU	Foreman et al. 2008	No
Salinity max	PSU	Foreman et al. 2008	No
Salinity min	PSU	Foreman et al. 2008	Yes
Temperature range	Celsius	Foreman et al. 2008	No
Temperature summer	Celsius	Foreman et al. 2008	No
Temperature winter	Celsius	Foreman et al. 2008	No
Temperature fall	Celsius	Foreman et al. 2008	No
Temperature spring	Celsius	Foreman et al. 2008	No
Temperature max	Celsius	Foreman et al. 2008	Yes
Temperature min	Celsius	Foreman et al. 2008	Yes



**Fig. A1.1.** Environmental predictors used for species distribution modeling.

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## **Appendix B.**

### **Effects of alternative thresholds for species distribution models on predicted trade-offs**

#### **Methods**

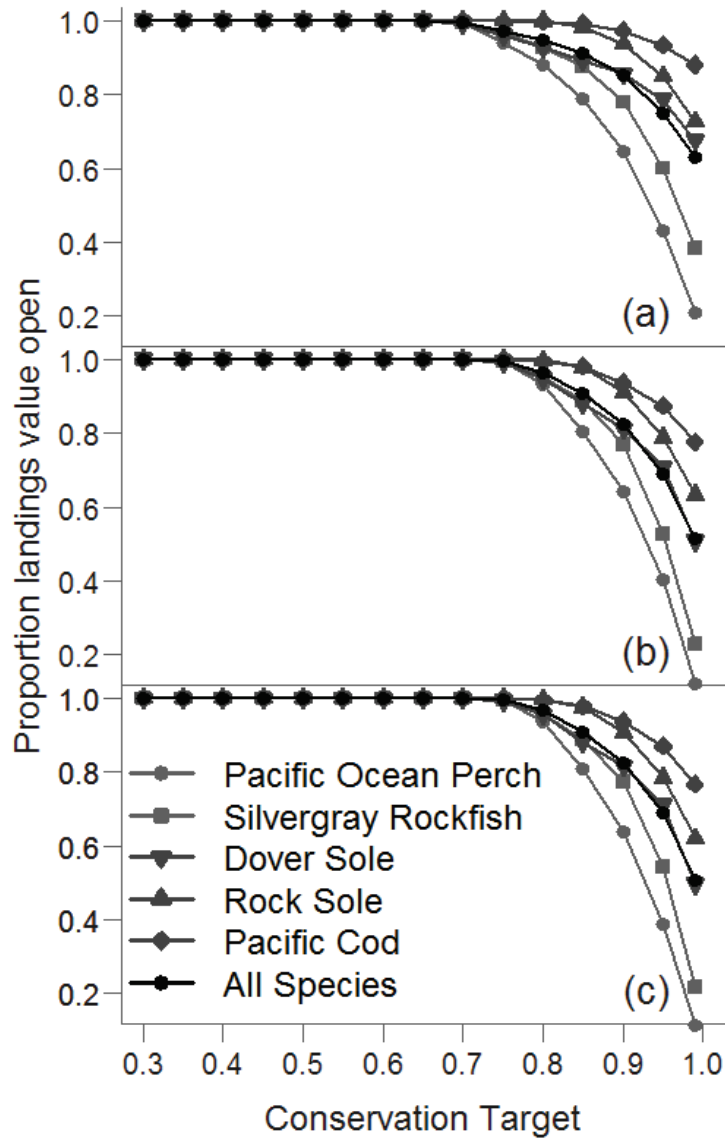
The selection of a threshold for presence-absence conversion of species distribution models is an important modeling decision that determines the extent of predicted presence locations (Jiménez-Valverde & Lobo 2007; Liu et al. 2013). For our analysis, we did not convert to binary presence-absence values, but instead applied a threshold that transformed low habitat suitability index (HSI) values to zero in order to prioritize conservation towards areas that are most likely to contain corals and sponges. If this threshold is set too high, then areas containing corals and sponges may have the HSI transformed to zero, resulting in false negative prediction errors. Therefore, we examined how alternative threshold methods, including not applying any threshold, affected the predicted trade-offs between landings value and protection of coral and sponge habitat.

We compared three threshold methods: the maximised sum of specificity plus sensitivity (SSS) threshold, the least presence (LP) threshold, and no threshold. The LP threshold is equal to the lowest HSI where a presence records occurs, and is therefore a conservative threshold without false negative prediction errors, but with a high probability of false positive errors (Liu et al. 2013). The SSS threshold is the value that maximises the sum of specificity and sensitivity, which are the proportions of pseudo-absence records and presence records that are correctly predicted, respectively (Jiménez-Valverde & Lobo 2007). The SSS threshold may result in false negative errors, but is more robust than LP because the rate of false positive errors is lower (Jiménez-Valverde & Lobo 2007; Liu et al. 2013). Thresholds were applied separately to the model predictions for the Halipteridae and Pennatulidae families, and then HSI values in each grid cell were summed and rescaled between 0 and 1. See Table A2.1 for the threshold values obtained for each model.

#### **Results**

There were no reductions in landings value up to conservation targets of 0.7 regardless of the threshold method used (Fig A2.1). Applying the LP instead of the SSS threshold resulted in greater reductions in landings value for conservation targets above 0.8, but slightly lower reductions for conservation targets of 0.75 and 0.8. The LP threshold also closed a larger area over the entire range of conservation targets because the predicted extent of coral and sponge occurrence was greater. For example, using a conservation

target of 0.99 with the LP threshold, 95.8% of the study area was closed and landings value was reduced to 51.6% of the total compared to 77.2% of the study area closed and 63.0% of landings value open using the SSS threshold. Using no threshold resulted in similar trade-off predictions to the LP threshold; reductions in landings value did not differ by more than 1% over the entire range of conservation targets using no threshold or the LP threshold.



**Figure. A2.1.** Proportion of the total landings value within areas open to trawling as a function of the conservation target for protection of coral and sponge habitat. Panel (a) uses the sum of specificity and sensitivity (SSS) threshold to convert low habitat suitability indices to zero; panel (b) uses the least presence (LP) threshold; panel (c) does not use any threshold.

**Table A2.1.** Threshold values, mean and median HSI for the species distribution models.

Model	Sum of sensitivity plus specificity (SSS) threshold	Least presence (LP) threshold	Mean HSI value	Median HSI value
Alcyonacea	0.256	0.039	0.130	0.0405
Hexactinellida	0.336	0.014	0.176	0.0739
Pennatulidae	0.284	0.043	0.149	0.0630
Halopteridae	0.295	0.044	0.190	0.0959

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