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# Hotspots for rockfishes, structural corals, and large-bodied sponges along the central coast of Pacific Canada

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## Abstract

Biological hotspots are places with outstanding biodiversity features, and their delineation is essential to the design of marine protected area networks (MPANs). For the Central Coast of Canada's Northern Shelf Bioregion, where a MPAN is being developed, we identified hotspots for structural corals and large-bodied sponges, which are foundation species vulnerable to bottom contact fisheries, and for Sebastidae, a fish family that includes long-lived (>100 years) and overexploited species with high trophic positions. Using 10 years of survey data that spanned from inland fjords to oceanic waters, we derived hotspot indices that accounted for species characteristics and abundances, and examined hotspot distributions across depths and oceanographic subregions. The results highlighted previously undocumented hotspot locations for each species group, thereby informing the placement of MPAs for which high levels of protection are warranted. Given the vulnerability of the species groups that we examined to cumulative fishery impacts, prospective MPAs derived from our data should be considered for interim protection measures during the protracted period between final network design and the enactment of MPA legislations. These recommendations reflect our scientific data, which are only one way of understanding the seascape. Our extensive surveys did not cover many locations known to Indigenous peoples as biologically important. Consequently, it is paramount that Indigenous knowledge also contributes substantially to the design of the MPAN.

## Introduction

Biodiversity loss affects all human societies<sup>1</sup>, yet its harm can be disproportionately greater for Indigenous peoples who derive food security and cultural identity from local ecosystems<sup>2,3</sup>. In the latter part of the 20<sup>th</sup> century, First Nations along the Central Coast of British Columbia, Canada, began to experience rapid declines in the abundance of marine species inherent to traditional foods, including Pacific salmon (*Oncorhynchus* spp.)<sup>4</sup>, eulachon (*Thaleichthys pacificus*)<sup>5</sup>, and yelloweye rockfish (*Sebastes ruberrimus*)<sup>6</sup>. These species have yet to recover. The rise of commercial and recreational fisheries, combined with anthropogenic climate change<sup>2,5,7</sup>, have influenced these negative trends, amplifying the challenge of cultural revitalization in the aftermath of colonialism<sup>2,8,9</sup>.

Many species of cultural significance play important ecosystem roles. They include upper-level predators (e.g. yelloweye rockfish<sup>10</sup>) that may indirectly benefit smaller organisms via trophic cascades<sup>11</sup>, anadromous species that transport nutrients from offshore areas to estuaries and riparian ecosystems (e.g. Pacific salmon<sup>12</sup>, eulachon<sup>13</sup>), and foundation species that create biogenic habitats (e.g. kelps<sup>14</sup>). Consequently, losses of biological and cultural diversity are inextricably linked.

Evidence from diverse regions of the world suggests that networks of Marine Protected Area (MPAs) can help reverse negative trends and support sustainable ecosystems, economies, and cultures<sup>15</sup>. For example, fish and invertebrates inside MPAs become more abundant and grow to greater size and age than in exploited areas<sup>16</sup>. Consequently, MPAs may increase the productivity of exploited species, promoting the export of larvae and adults to fished areas and resilience to climate change<sup>16,17</sup>. MPAs or other forms of spatial fishery closures can also protect foundation species vulnerable to bottom-contact fisheries, such as corals and sponges<sup>18,19</sup>.

MPAs, however, often have been established without involving Indigenous peoples, undermining their governance structures and curtailing their access to traditional harvest areas, thereby hampering cultural diversity. Accordingly, there is growing recognition that Indigenous peoples should lead their own spatial planning processes or, at the very least, be legitimate partners in MPA governance, research, design, and implementation<sup>20,21</sup>.

The effectiveness of an MPA network also depends on the extent to which the location and protection levels of individual MPAs prioritize conservation objectives over extractive activities<sup>22</sup>, and on the monitoring and enforcement of regulations limiting such activities<sup>23</sup>. Consequently, commercial fishers and other stakeholders may lose access to areas they used previously. If convinced of the conservation benefits, stakeholders may accept displacement and support spatial protections; if unconvinced, they may stymie implementation of the MPA network<sup>24</sup>.

The delineation of biological hotspots—places with outstanding biodiversity or ecological features—can help justify spatial protections<sup>25,26</sup>, potentially reducing stakeholder conflicts. Hotspot criteria include (but are not limited to) endemism<sup>26</sup>, localized prey aggregations or oceanographic processes that persist over time and support predators<sup>27</sup>, and aggregations of species that are ecologically important and vulnerable to extractive activities<sup>18,19,28</sup>.

The ongoing development of an MPA network for Canada's Northern Shelf Bioregion (NSB)<sup>21</sup> (Fig. 1) is a potential nexus for Indigenous governance and the protection of biological hotspots. The MPA process intends to honour Indigenous rights and title to their traditional territories, such that 17 First Nations and the federal government are governance partners responsible for network design and implementation<sup>21</sup>. The First Nations involved have used their traditional and local knowledge to identify areas of cultural, spiritual, and biological importance to be protected by the

MPA network. These Nations also support Western science as a knowledge system complementary to their own<sup>29</sup>.

The Wuikinuxv, Nuxalk, Heiltsuk and Kitasoo/Xai'xais First Nations live along the Central Coast subregion of the NSB (Fig. 1) and are among the Indigenous governance partners for the MPA network. Collaborating under the umbrella of the Central Coast Indigenous Resource Alliance (CCIRA), since 2013 they have been using fishery-independent methods (dive and towed video transects, hook and line sampling) to survey biodiversity features in their territories<sup>30–32</sup>. The surveys encompass oceanic and inland waters at depths of 5 m to 200 m. Additionally, research in 2018 included a collaboration with Fisheries and Oceans Canada (DFO: the federal aquatic ecosystem and resource management agency)—which contributed technical capacity and infrastructure (large vessel, crew, and the towed video camera described by Gale et al.<sup>33</sup>) to sample depths of 200 m to 500 m.

The surveys have targeted locations where fish of cultural significance, such as rockfish (*Sebastes* spp.), are expected on the basis of local Indigenous knowledge, yet have also documented foundation species, such as structural corals (i.e., taxa that are erect and branching, including the orders Antipatharia, Alcyonacea, and Anthoathecata) and large-bodied sponges (taxa that are erect and vase- or mound-shaped, including the classes Hexactinellidae and Demospongiae)<sup>30</sup>. The data span three distinct oceanographic areas, known as Upper Ocean Subregions (UOS)<sup>34</sup> (Fig. 2), for which we can identify biological hotspots in support of MPA network planning and implementation. Notably, our extensive surveys include the Mainland Fjords UOS (Fig. 2), where data gaps curtailed earlier analyses of biodiversity distributions<sup>35</sup>.

Rockfish are well-suited for hotspot delineation at small spatial scales. They include sedentary species with long maximum lifespans (>100 years)<sup>36</sup> and high trophic positions<sup>10</sup>, and therefore are fishery vulnerable<sup>37</sup> and ecologically important<sup>11</sup>. The genus has been an important food for coastal peoples for over 2500 years<sup>38</sup>. Marked declines in the abundance and body sizes of culturally-significant rockfishes began in the 1980s, concurrently with a surge in commercial fishery activity<sup>6,39,40</sup>. Body size declines appear to be ongoing<sup>41</sup>, likely signaling overexploitation and loss of population productivity<sup>17</sup>.

As sessile foundation species vulnerable to bottom contact fishing gear<sup>18,42</sup>, structural corals and large-bodied sponges also are suited for hotspot delineation at small spatial scales. In addition to forming biogenic habitats used by other species<sup>43</sup>, corals and sponges influence ecosystems through water filtration, carbon sequestration and basal support for food webs<sup>19,44–46</sup>.

In this study we identify hotspots for rockfish and the closely related shortspine thornyhead (*Sebastolobus alascanus*), structural corals, and large-bodied sponges in the Central Coast subregion of the NSB. We standardized and combined abundance data from different survey types conducted by CCIRA-member Nations and collaborating DFO scientists, and weighted species according to their ecological role, vulnerability to fisheries, and depletion level. For each species group, we then calculated hotspot indices that accounted for the spatial overlap and relative abundance of vulnerable and/or ecologically important species and examined hotspot distributions across different UOS while accounting for the maximum depths sampled. We interpret these results in terms of potential locations and protection levels for prospective MPAs in the NSB.

## Methods

The Wuikinuxv, Kitasoo/Xai'xais, Heiltsuk and Nuxalk First Nations hold Indigenous rights to their own territories, where all data were collected. Scientific staff who are members of these Nations, or

who work directly for them, had direct approvals from Indigenous rights holders and were exempt from other research permit requirements. Collaborating DFO scientists worked under federal research permits. Experiments with live animals were not conducted.

Data were collected by four fishery-independent methods—shallow diver transects, mid-depth video transects, deep video transects, and hook-and-line sampling—detailed in earlier publications<sup>30–33,47,48</sup> (Table 1). Throughout the sampling period (2006–2007, 2013–2020: Table 1), we focused on the geographic expansion of sampling locations, rather than on tracking temporal variability within sites. Accordingly, we assume that our data primarily reflect spatial variability in species distributions.

Data had a spatial resolution of  $\leq 130 \text{ m}^2$  but were aggregated into planning units,  $u$  (see equation 1), which were either  $16\text{-km}^2$  ( $N = 281$ ) and used to generate visual displays (First Nations of the Central Coast require this coarse scale for visual displays to protect sensitive locations), or  $1\text{-km}^2$  ( $N = 619$ ) and used for inferential analyses. Within a given spatial scale, for each  $u$  we calculated three hotspot indices:  $B_{SEB,u}$  for fish of the family Sebastidae (rockfishes and thornyheads: 89,316 individual records),  $B_{Cor,u}$  for structural corals (4,772 individual records), and  $B_{Sp,u}$  for large-bodied sponges (755 individuals counted by deep video transects, and thousands more recorded as percent covers by shallow diver transects and mid-depth video transects: Table 1). These indices did not consider cup corals, whip-like corals or encrusting corals or sponges.

As detailed below (equations 1, 2), each species of Sebastidae or genera of corals contributed to  $B_{SEB,u}$  or  $B_{Cor,u}$ , according to their abundance weighted by  $W_t$ : a variable depicting taxon characteristics. For the 26 species of Sebastidae that we observed,  $W_t$  equaled the sum of scores for fishery vulnerability (with maximum age as proxy<sup>37</sup>), depletion level (estimated from the ratio of recent biomass to unfished biomass), and ecological role (with trophic level as proxy) (Table 2; Appendix S1). For the 5 genera of structural corals analyzed (Appendix S2),  $W_t$  depended on maximum height (see Table 1 for estimation method using deep video transects), which correlates positively with vulnerability to physical damage from bottom-contact fishing gear (including longer time to recovery)<sup>49–51</sup> and with strength of ecological role (e.g. amount of biogenic habitat and carbon sequestration increases with height)<sup>42,52</sup> (Table 2, Appendix S2).  $W_t$  for corals did not include depletion level due to lack of data.

As detailed below (equations 1, 3), the hotspot index for large-bodied sponge,  $B_{Sp,u}$  did not differentiate between species characteristics and pooled the abundances of all observed species of Hexactinellidae (*Aphrocallistes vastus*, *Farrea occa*, *Heterochone calyx*, *Rhabdocalyptus dawsoni*, *Staurocalyptus dowlingi*) and Demospongiae (*Mycale cf loveni*). This approach is consistent with regional fishery bodies worldwide, which treat large-bodied sponges as a single functional group<sup>53</sup>.

To derive each hotspot index, we first estimated relative density,  $D$ , for taxa  $t$  observed with method  $i$  at planning unit  $u$ :

$$D_{t,i,u} = \frac{\sum_{l=1}^{n_{i,u}} \left( \frac{C_{t,i,l}}{E_{i,l}} \right)}{n_{i,u}} \quad (1)$$

where  $C_{t,i,l}$  is the method-specific count (or other abundance measure, see below) of  $t$  at sampling location  $l$ ,  $E_{i,l}$  is the method-specific survey effort (area or time, depending on  $i$ ), and  $n_{i,u}$  is the total number of locations sampled by that method in the planning unit. For Sebastidae and structural corals there are, respectively, 26  $t$  (Appendix S1) and 5  $t$  (Appendix S2). For large-bodied sponges there is a single  $t$  (i.e., counts aggregated all taxa). For most taxa and method combinations,  $C$  is a count of individual organisms. The exceptions are large-bodied sponges recorded with mid-depth

towed video or dive surveys, where  $C$  is a categorical variable for percent cover with integer values of 0 to 4 (0 = 0% cover, 1 = 1-25%, 2 = 26-50%, 3 = 51-75%, 4 = 76-100%).

Because survey methods differed in their maximum values and potential biases (e.g., field of view is greater for divers than for video cameras),  $D_{t,i,u}$  was rescaled as a min-max normalization,  $D'_{t,i,u}$  (i.e., difference between observed value and the minimum value across all  $u$ , divided by the range of values across all  $u$ ).

For Sebastidae or structural corals, the hotspot index was then calculated as:

$$B_{SEB,u} \text{ or } B_{COR,u} = \sum_t \left( \left[ \frac{\sum_i D'_{t,i,u}}{I_{t,u}} \right] \times W_t \right) \quad (2)$$

where  $W_t$  is the taxon-specific weighing factor (Table 2, Appendices S1, S2),  $I_{t,u}$  is the number of survey methods  $i$  that are valid for  $t$  and that occurred in the planning unit. For Sebastidae, if  $i$  detects  $t$ , then the method is valid, but if  $i$  does not detect  $t$  then the method is valid only if the 10<sup>th</sup> and 90<sup>th</sup> percentiles of sampled depths encompass the expected depth range of  $t$  (Appendix S3); also, for hook-line-sampling  $i$  is valid only if  $t$  is not a planktivore (*Sebastes diaconus*, *S. emphaeus*, or *S. jordani*) (Appendix S3). For corals, the only valid methods are mid-depth and deep video transects; hook-and-line gear cannot systematically sample sessile benthic organisms and dive surveys are shallower than the expected range of most structural corals.

The hotspot index for large-bodied sponges, which aggregates all taxa, is:

$$B_{Sp,u} = \frac{\sum_i D'_{t,i,u}}{I_{t,u}} \quad (3)$$

Because glass sponges can occupy shallow depths<sup>19</sup>, hook-and-line is the only survey method not valid.

The above hotspot indices generate continuous variables that range from zero to their maximum value, which we converted into ten ranks with integer values of 1 to 10. These ranks approximate deciles, except that—to account for the large proportions of planning units where a species group was unobserved (1-km<sup>2</sup> scale: 85% for corals, 54% for sponges, 11% for Sebastidae)—repeated values were assigned to their lowest common rank (i.e., all zeroes were assigned to the first rank). Consequently, sponges and corals lack ranks 2-5 and 2-8, respectively (Appendix S4). While recognizing that threshold-based definitions of hotspots are arbitrary and may vary between studies, we defined “hotspots” as planning units containing the top 2 ranks for one or more hotspot indices.

Maximum depths sampled within planning units were deepest in the Mainland Fjord UOS and shallowest in the Aristazabal Banks Upwelling UOS (Appendix S5). Accordingly, we used multiple logistic regression<sup>54</sup> implemented with the glm function of R to estimate the probabilities of hotspot occurrence within 1-km<sup>2</sup> planning units in relation to UOS and maximum depth.

## Results

For all species groups, hotspots spanned from oceanic areas to inland waters at the head of fjords (Figs. 2a-c) but were distributed unevenly across Upper Ocean Subregions (UOS) and/or depths (Fig. 3; Appendix S6). For Sebastidae and structural corals, the probabilities of hotspot occurrence within 1-km<sup>2</sup> planning units increased with the maximum depth sampled. After accounting for depth, Sebastidae hotspots were more likely to occur at Eastern Queen Charlotte Sound (EQCS) than at the other two UOS, while hotspots for structural corals did not differ in probabilities of occurrence at Mainland Fjords and EQCS (Fig. 3; Table 3). Hotspots for large-bodied sponges were more likely to occur at Mainland Fjords than at EQCS, but their probability of occurrence declined

at greater depth, although support for these depth effects was weaker (Fig. 3; Table 3). We did not detect hotspots for corals or sponges at Aristazabal Upwelling Banks, the least sampled UOS (Fig. 3, Appendix S6).

Spatial congruence among hotspots for different species groups occurred primarily in the Mainland Fjords OUS (Fig. 2d). Of 33 1-km<sup>2</sup> planning units containing hotspots for structural corals, seven (21%) also contained Sebastidae hotspots (all Mainland Fjords) and an additional seven also contained hotspots for large-bodied sponges (6 in Mainland Fjords, 1 in EQCS). Similarly, of 104 1-km<sup>2</sup> planning units containing hotspots for large-bodied sponges, 20 (19%) also contained Sebastidae hotspots (17 in Mainland Fjords, 3 in EQCS). Hotspots for all species groups overlapped at single 1-km<sup>2</sup> planning unit, located in EQCS.

## Discussion

The pace of biodiversity loss is staggering<sup>55</sup>, and there is an urgent need to spatially protect biological hotspots. Towards that end, our research highlights previously undocumented hotspot distributions for long-lived fishes of the family Sebastidae, large-bodied sponges, and structural corals (Fig. 4) along the central portion of Canada's Northern Shelf Bioregion (NSB), particularly in the little-studied Mainland Fjords. The data are timely and are contributing to the design of the MPA network for the NSB, which is nearing its final stages<sup>21</sup>. Given that commercial and recreational fisheries remain open throughout most of the NSB, a well-designed MPA network could potentially ameliorate fishery impacts on the species groups that we examined.

The hotspots that we identified (i.e., planning units containing ranks 9 or 10 of hotspot indices, ≈20% of planning units) can inform the placement of MPAs for which high levels of protection are warranted (e.g., exclusion of commercial and recreational fisheries). Given the vulnerability of structural corals, large-bodied sponges, and long-lived rockfishes and thornyheads to cumulative fishery impacts<sup>18,19,56</sup>, prospective MPAs containing hotspots should be considered for interim protection measures during the protracted period between final network design and the enactment of MPA legislations. Additionally, planning units that did not qualify as hotspots but that contained important biological values (i.e., ranks 6-8 of hotspot indices) might inform the siting of MPAs with lesser protection levels (e.g., some types of fisheries permitted).

The above recommendations reflect our scientific data, which are only one way of understanding the seascape. Our surveys, extensive as they are, failed to cover many locations known to local Indigenous peoples as biologically important. For that reason, it is paramount that Indigenous knowledge contributes substantially to the design of the MPA network<sup>29</sup>.

We also acknowledge that our analyses did not account for spatial variation in historical exploitation rates. It is plausible that some non-hotspots locations containing structurally complex rocky habitats, where many rockfish species are known to thrive<sup>30,36</sup>, are former hotspots that are currently depleted but that could be potentially restored through spatial protections. The distribution of heterogeneous, high quality habitats, therefore, should also inform site selection for the MPA network<sup>57</sup>, especially where such habitats do not overlap with current hotspot that are species-based.

The positive effect of sampling depth on the probability of hotspot occurrence for Sebastidae and structural corals is unsurprising. Many Sebastidae species characterized by long lifespans (and, in some cases, high trophic positions) are more abundant at greater depth<sup>36</sup>, and these species were weighted more heavily in the hotspot index. Similarly, structural corals tend to form thickets at deeper depths but exist more often as scattered individuals at shallower locations<sup>58</sup>.

We found weak support for a negative relationship between the maximum depth sampled and the location of hotspots for large-bodied sponges, yet context for these results is warranted. Hotspots for sponges at shallow depths (≤35 m) occurred primarily on rocky walls of the Mainland Fjords UOS, where aggregations of *Aphrocallistes vastus* can be very dense (Fig. 4b). These shallow

aggregations are sponge “gardens,” not bioherms (i.e., reefs where live sponges grow on the remains of dead sponges)<sup>19,59</sup>. Our surveys did record likely bioherms at deeper depths (≈60-120 m) (Fig. 4a), but these records were too infrequent to generate a positive relationship between depth and the probability of hotspot occurrence. Given the tremendous ecological importance of bioherms<sup>19,44,60</sup>, future research should delineate separate hotspot distributions for bioherms and sponge gardens.

We found relatively low congruence in the spatial distributions of hotspots for different species groups. This result appears to be inconsistent with evidence that corals and sponges are foundation species that provide biogenic habitats to Sebastidae<sup>42,43</sup>, but our data might be equivocal on this matter. Our sampling at the deeper depths where sponge and coral reefs are expected (>100 m)<sup>58</sup> was limited, and the sponge gardens (Fig 4b) documented at shallow depths tended to be associated with benthopelagic species of rockfish that are relatively short-lived, and therefore weighted more lightly in hotspot index calculations. Future surveys that prioritize sampling at deeper depths are needed to better understand spatial overlap among hotspot distributions for Sebastidae and foundation species<sup>42,43</sup>.

The species group that we examined are either sessile (sponges and corals) or include long-lived demersal fishes with strong site fidelity (many rockfishes<sup>36</sup>). Thus, they are likely to benefit from spatial protection, both directly (i.e., no fishery removals or impacts from bottom-contact fishing gear) and indirectly (increased resilience to ocean warming and other environmental shifts)<sup>16</sup>. Species important to the culture of Central Coast First Nations, however, span beyond those that we examined, and include migratory fishes<sup>4,5,61</sup> that are more difficult to protect spatially (spawning aggregations excepted). The implication is that, alongside MPAs, improved fishery policies that extend beyond the narrow objectives of maximum sustained yield and that encompass broader ecosystem objectives also are needed to restore and protect biodiversity<sup>62</sup>.

### Data availability

Computer code and data used in our analyses, along with an explanatory document, are available at <https://zenodo.org/record/4677768#.YHTEteKhPY>.

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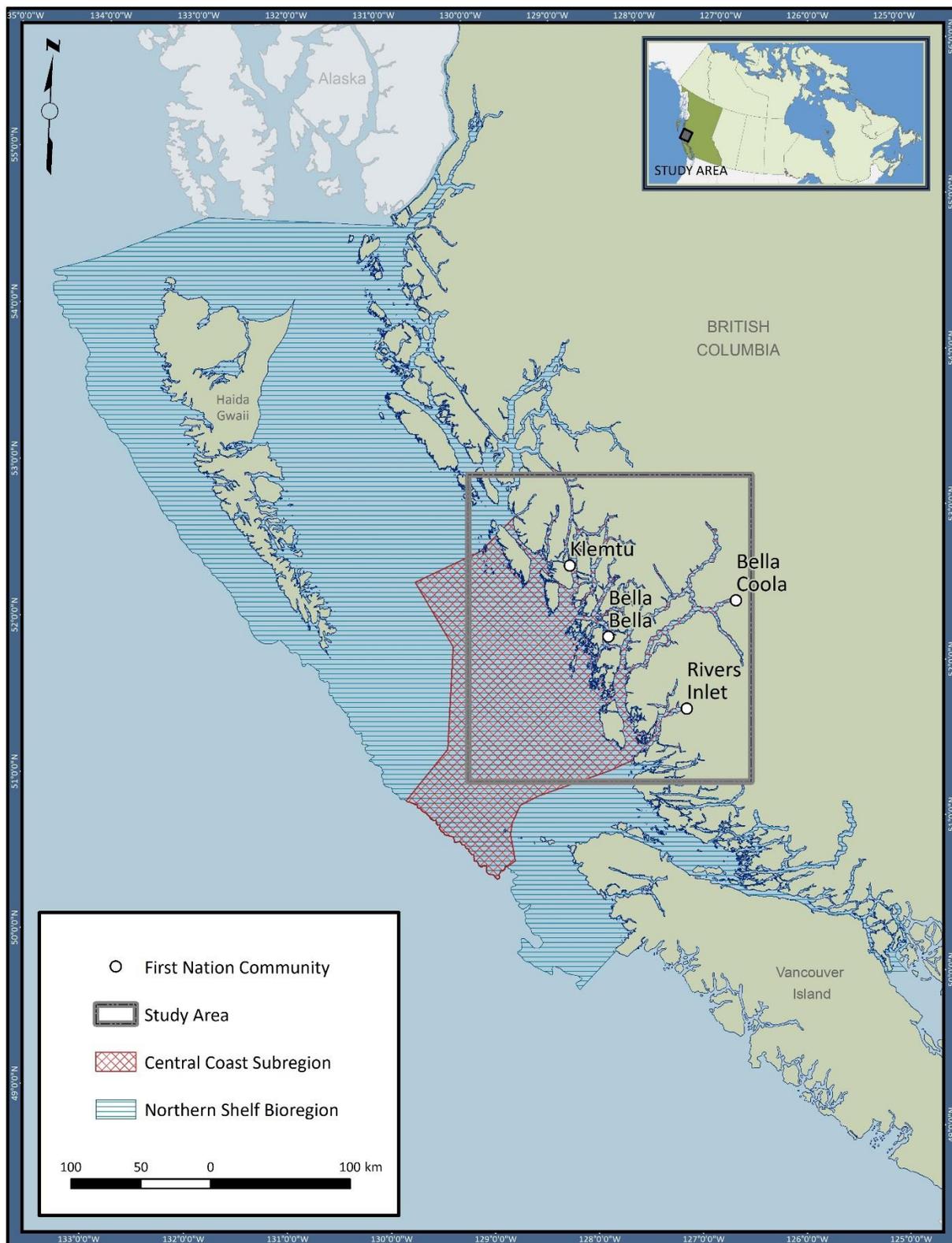
Carpenter, Richard Reid, Davie Wilson, Julie Carpenter, Doug Neasloss, Vern Brown, Derek Van Maanen, Kyle Hall, Andrew McCurdy, Jarred O'Connell, Wayne Jacob, Lily Burke, Natalie Ban, Lauren Eckert, John Volpe, Courtney Edwards, Aaron Heidt, James Pegg, Diana Chan, Mike Reid, Barry Edgar, and Twyla Frid. Jennifer Long and Darienne Lancaster contributed to video annotation. For analysis advice, we thank Katie Gale and Emily Rubidge. Julie Beaumont and Hannah Kobluk provided technical support in figure preparation. For contributions to species identifications, we thank Merlin Best (corals), Andy Lamb, and Milton Love (rockfishes).

**Author contributions**

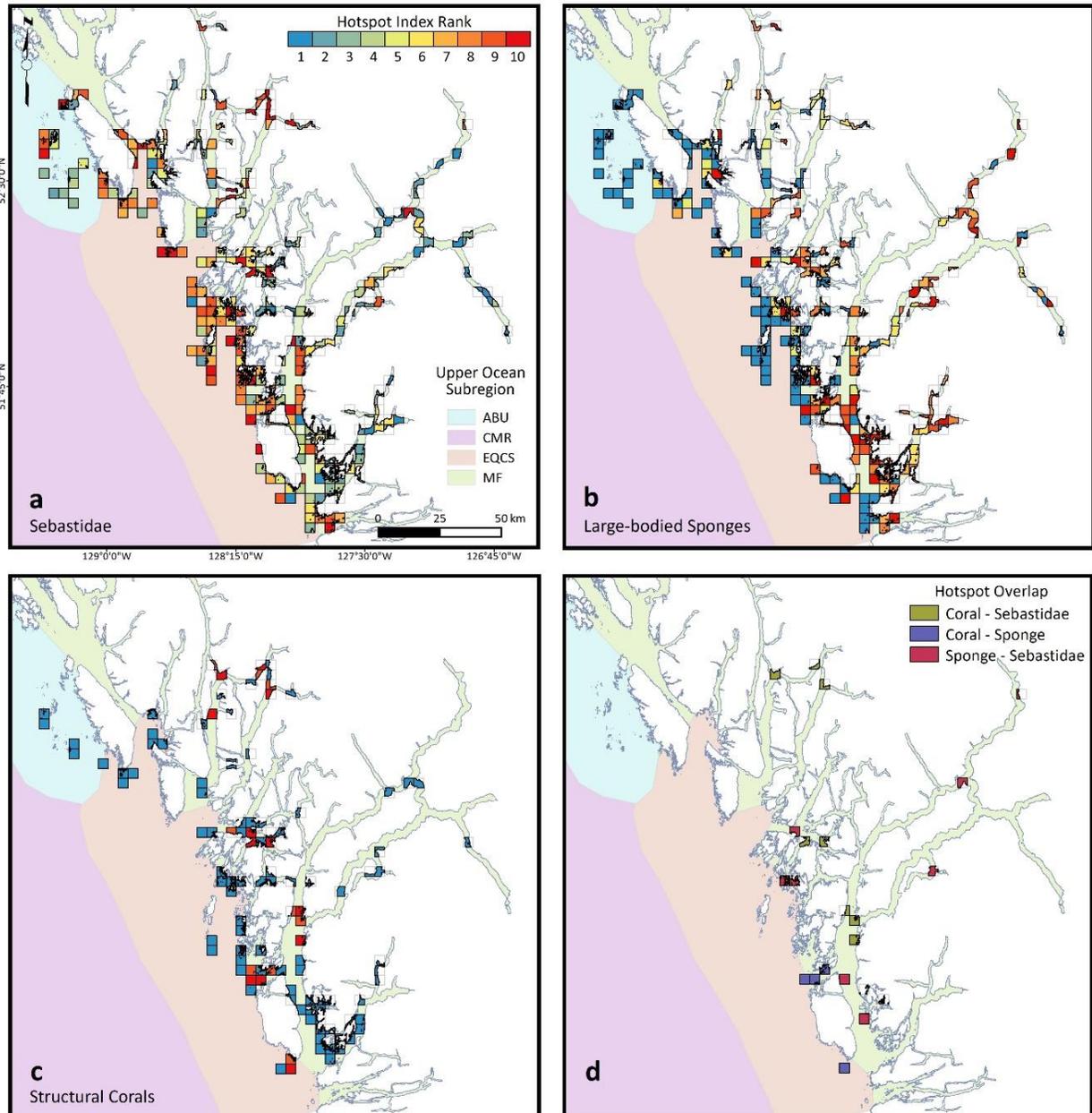
AF led the study and was the primary manuscript writer. MM, AF analyzed the data. AF, MM, CD, TB, TN contributed to study design and data collection. All authors reviewed and edited manuscript drafts.

**Competing interests**

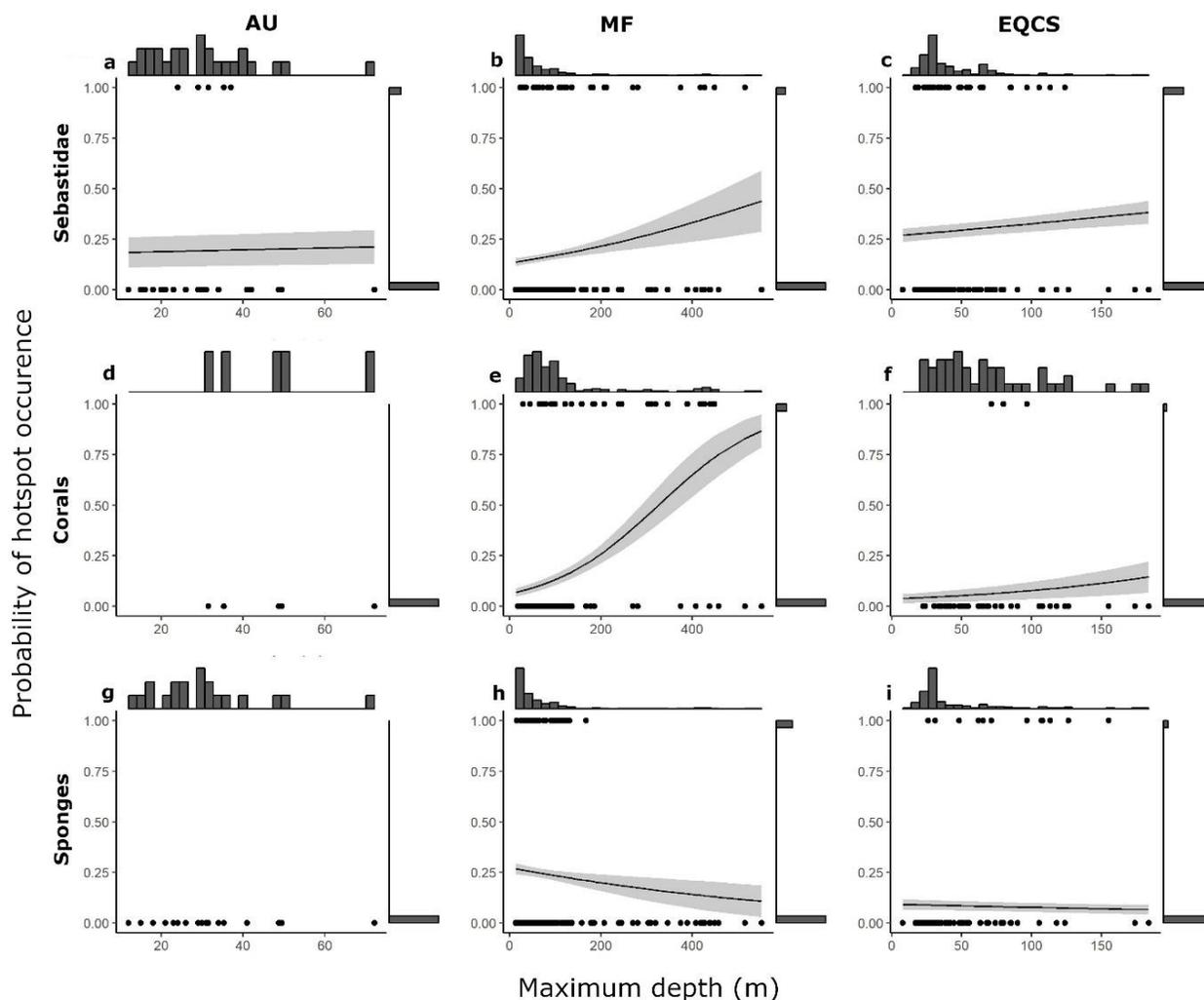
The authors declare no competing interests.



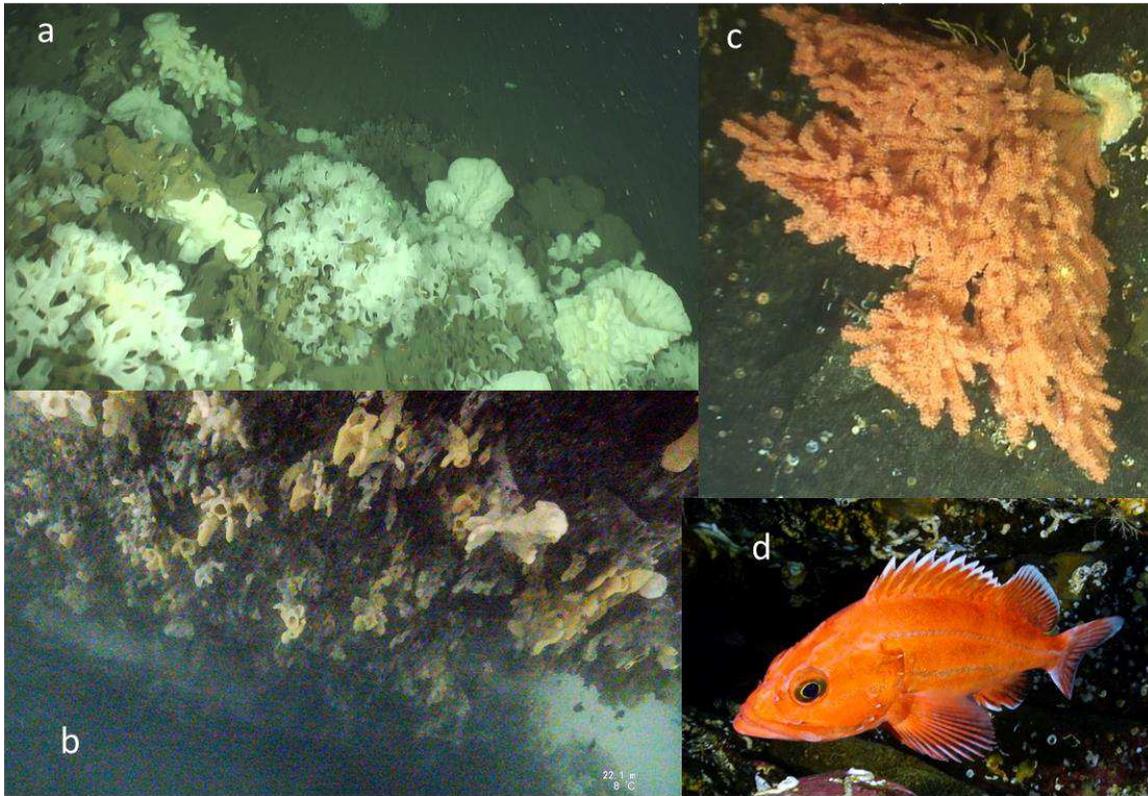
**Figure 1.** Map of the study area in the context of the Northern Shelf Bioregion.



**Figure 2.** Spatial distribution of hotspot index ranks within 16-km<sup>2</sup> planning units (squares, except where faded over land), by species group: (a) Sebastidae, (b) large-bodied sponges, (c) structural corals, and by Upper Ocean Subregions (ABU = Aristazabal Upwelling, CSTM=Cape Scott Tidal Mixing, EQCS = Eastern Queen Charlotte Sound, MF = Mainland Fjords). Panel (d) shows the subset of 16-km<sup>2</sup> planning units containing hotspots for more than one species group. (The 3 species groups overlapped at a single 1-km<sup>2</sup> planning unit; see text.) Although inferential analyses were conducted at the scale of 1-km<sup>2</sup>, First Nations of the Central Coast require this coarse scale for visual displays to protect sensitive locations.



**Figure 3.** Probabilities of hotspot occurrence within 1-km<sup>2</sup> planning units for (a-c) Sebastidae, (d-f) structural corals, and (g-i) large-bodied sponges, in relation to maximum depth sampled and Upper Ocean Subregion (ABU = Aristazabal Upwelling, EQCS = Eastern Queen Charlotte Sound, MF = Mainland Fjords). Circles are raw data (points overlap) and bars show their relative frequencies along each axis. Lines and shading are, respectively, logistic regression estimates with 95% confidence intervals (Table 3). Note that depth ranges sampled differ between ocean subregions.



**Figure 4.** Examples from each species group observed during the study (a) large-bodied sponge bioherm reef; (b): large-bodied sponge garden on a rocky wall); (c) a structural coral, *Primnoa pacifica*, (d), a Sebastidae, *Sebastes ruberrimus*. (All photographs are video stills obtained during data collection, except (a), which DFO scientist Anya Dunham obtained while-resampling one of our research sites).

Survey method	Sampling years	Depth, m (mean)	Key characteristics	Data used in current analyses.	Notes
Shallow diver transects <sup>30,31</sup>	2013, 2015–2020	5-35 (21)	Belt transects (30 m × 4 m × 4 m, or 480 m <sup>3</sup> ), along depth contours.	<ol style="list-style-type: none"> <li>1. Relative density (count/480 m<sup>3</sup>) of fish, by species.</li> <li>2. Percent cover category of large-bodied sponges, aggregated for all Hexactinellidae and Demospongiae.</li> </ol>	Depths were too shallow to survey larger, older rockfishes <sup>36</sup> and corals. Consistent with earlier publications <sup>31</sup> , analyses excluded fish <10 cm-long. Sponge cover categories: 0 = 0%; 1 = 1- 25%; 2 = 26-50%; 3 = 51-75%; 4 = 76-100%.
Mid-depth video transects <sup>30,47,48</sup>	2015-2018	15-200 (67)	Belt transects of variable size were divided into bins covering 75-130 m <sup>2</sup> (mean = 116 m <sup>2</sup> ) to reduce depth and habitat variability within spatial units. (Bins <75 m <sup>2</sup> are end cuts and bins >130 m <sup>2</sup> reflect GPS data gaps; analyses exclude both.)	<ol style="list-style-type: none"> <li>1. Relative density (count/m<sup>2</sup>) for each taxa of fish and structural coral.</li> <li>2. Percent cover category of large-bodied sponges, aggregated for all Hexactinellidae and Demospongiae(see dive transects for category values).</li> </ol>	Fish counts were corrected for species detection biases (i.e., attraction to laser beams) <sup>47,48</sup> . Camera lacks panning/tilting ability and depth capacity of BOOTS camera (see below). The lower bound for bin size in earlier analyses <sup>30</sup> was 100 m <sup>2</sup> , which we lowered to 75 m <sup>2</sup> to not exclude coral-rich areas.
Deep video transects (BOOTS) <sup>33</sup>	2018	100-500 (253)	Belt transects varied widely in area but were divided into similar size bins, as described for mid-depth video transects.	<ol style="list-style-type: none"> <li>1. Relative density (count/m<sup>2</sup>) for each taxon of fish, structural coral, and large-bodied sponge.</li> <li>2. Mean height of coral colony, by taxon.</li> </ol>	Fish counts were corrected for species detection biases <sup>47,48</sup> . Coral heights (distance from base to highest branch tip) were measured from 20 randomly selected images per taxon in which “flat” aspect of the corals were perpendicular to the camera.
Hook-and-line <sup>32</sup>	2006-2007; 2013-2015	15-205 (57)	Standardized gear fished the bottom for 15-min or 30-min sampling sessions.	Relative density (count/min) for each fish species.	The 2006-2007 data were collected by the Heiltsuk Nation prior to CCIRA’s inception.

**Table 1.** Survey methods used for data collection. For full description of each method and its suite of data, see references in first column.

<b>Taxonomic group</b>	<b>Criteria</b>	<b>Proxy variable</b>	<b>Value</b>	<b><math>W_t</math></b>
Sebastidae (rockfish and thornyheads)	Vulnerability	Max. Age (yrs)	>110	4
			91 to 110	3
			71 to 90	2
			50 to 70	1
			<50	0
	Depletion level	$B_y/B_0$ (median estimate of female spawning biomass during year $y$ , divided by the “unfished” biomass).	<0.13	4
			0.13 to 0.23	3
			0.24 to 0.35	2
			0.36 to 0.45	1
			>0.45	0
			Unassessed	1
	Ecological role	Trophic Level	>4.2	2
			4.0 to 4.2	1.5
			3.8 to 3.9	1
			3.4 to 3.7	0.5
			<3.4	0
Structural corals	Vulnerability and ecological role	Mean height, $h$ (cm)	$h$ (range 3 to 42 cm; Appendix S2)	$\left[ \frac{h}{h_{max}} \times 10 \right]$

**Table 2.** Criteria used to calculate  $W_t$  for Sebastidae and structural corals. The maximum weight of ecological role is half of that used for other criteria because trophic level varies ontogenetically with body size<sup>10</sup>, but for most species only mean estimates are available. Coral heights were estimated empirically; see Table 1.

<b>Response variable</b>	<b>Predictor</b>	<b>Estimate</b>	<b>SE</b>	<b>z</b>	<b>Prob.</b>
Sebastidae hotspot (Deviance= 620.05, DF = 618)	Maximum depth sampled	2.97E-03	1.28E-03	2.32	0.02
	ABU	0.36	0.52	0.70	0.49
	EQCS	0.86	0.22	3.98	7.00E-05
Coral hotspot (Deviance = 185.67, DF = 218 )	Maximum depth sampled	0.01	0.00	4.91	9.10E-07
	EQCS	-0.59	0.65	-0.90	0.37
Sponge hotspot (Deviance = 517.72, DF = 513)	Maximum depth sampled	-2.06E-03	1.65E-03	-1.25	0.21
	EQCS	-1.31	0.33	-4.00	6.42E-05

**Table 3.** Logistic regression results examining probabilities of hotspot occurrence within 1-km<sup>2</sup> planning units. Effects of Upper Ocean Subregions (ABU = Aristazabal Banks Upwelling, EQCS = Eastern Queen Charlotte Sound), use Mainland Fjords as the reference.

# Figures

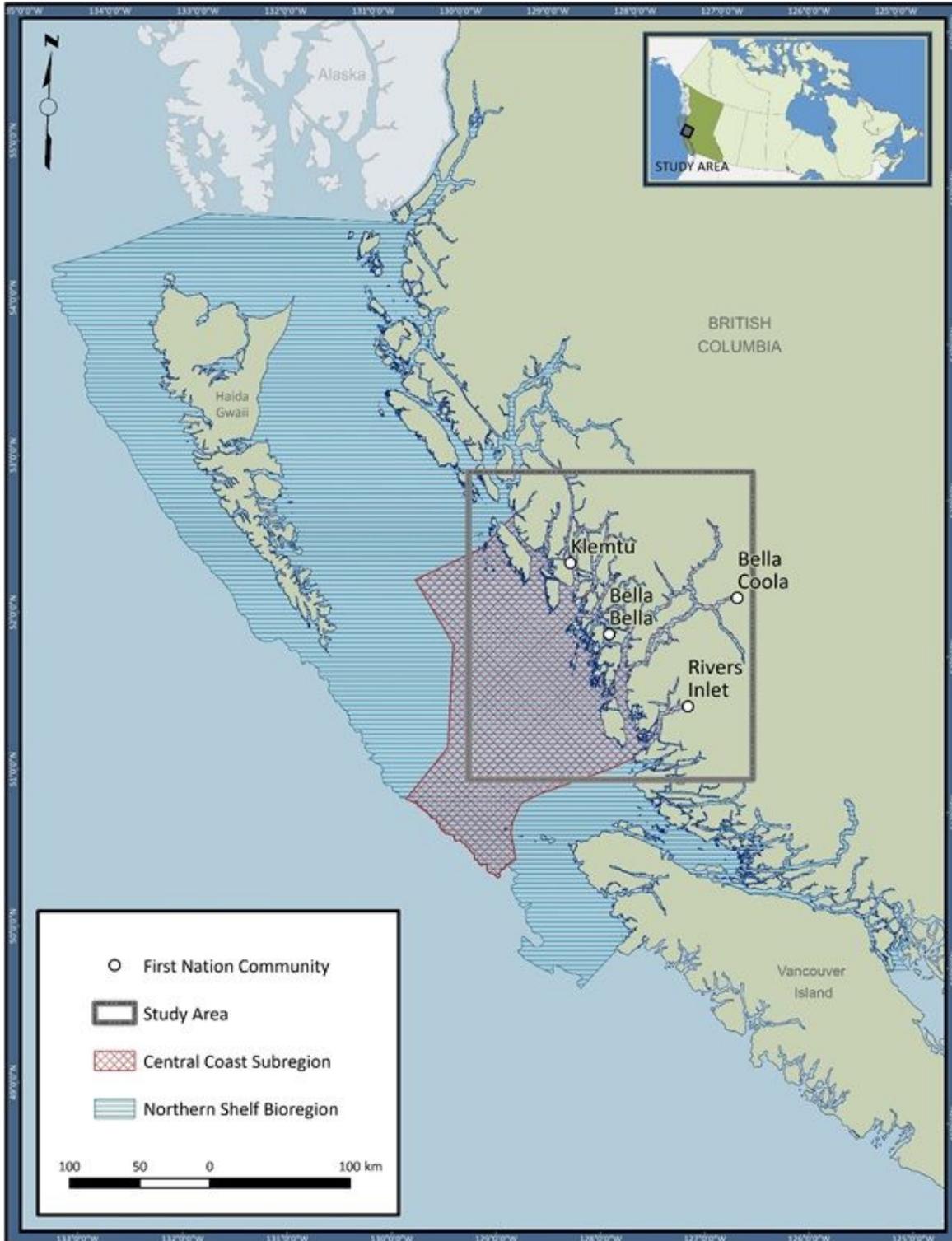
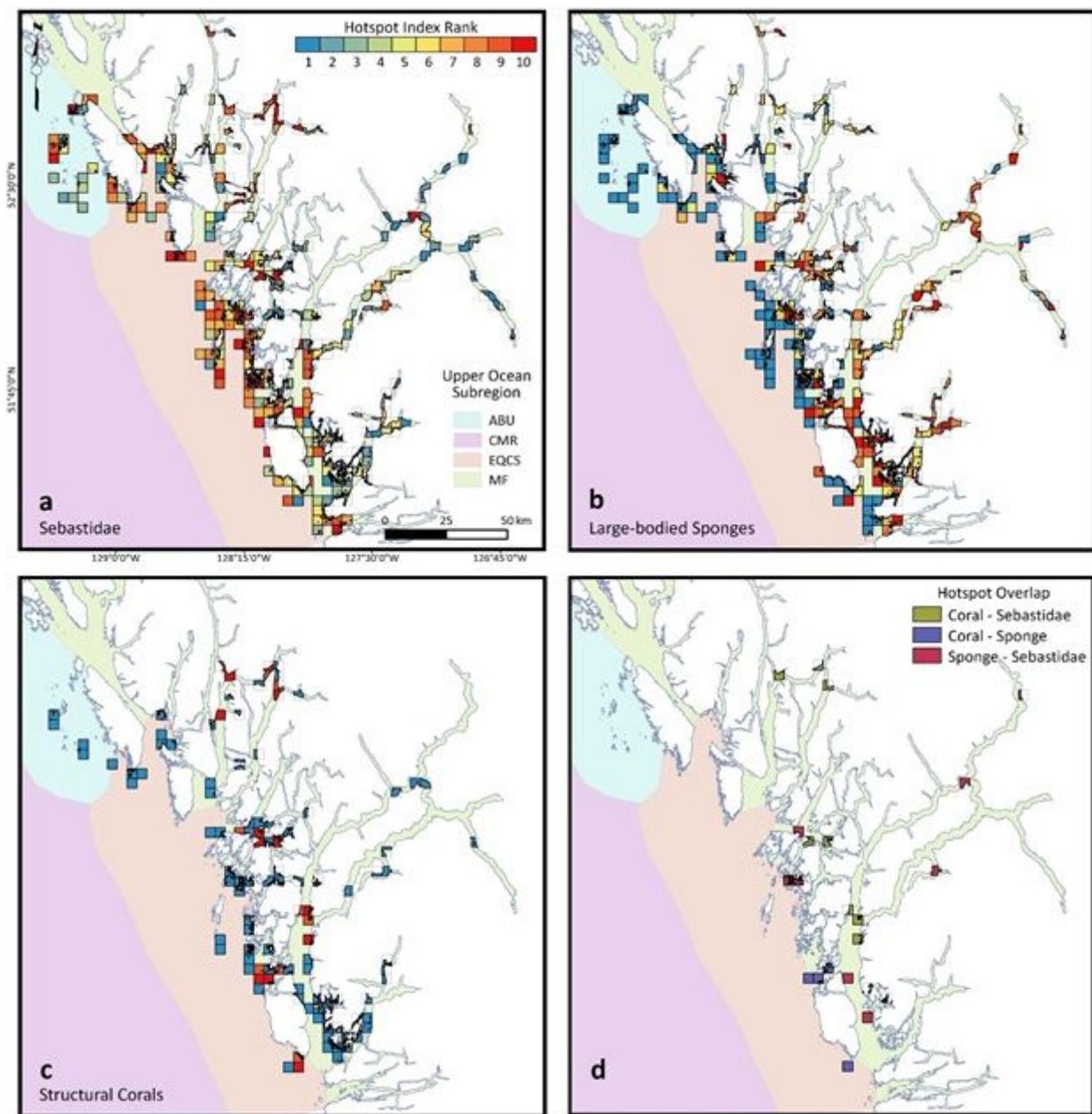


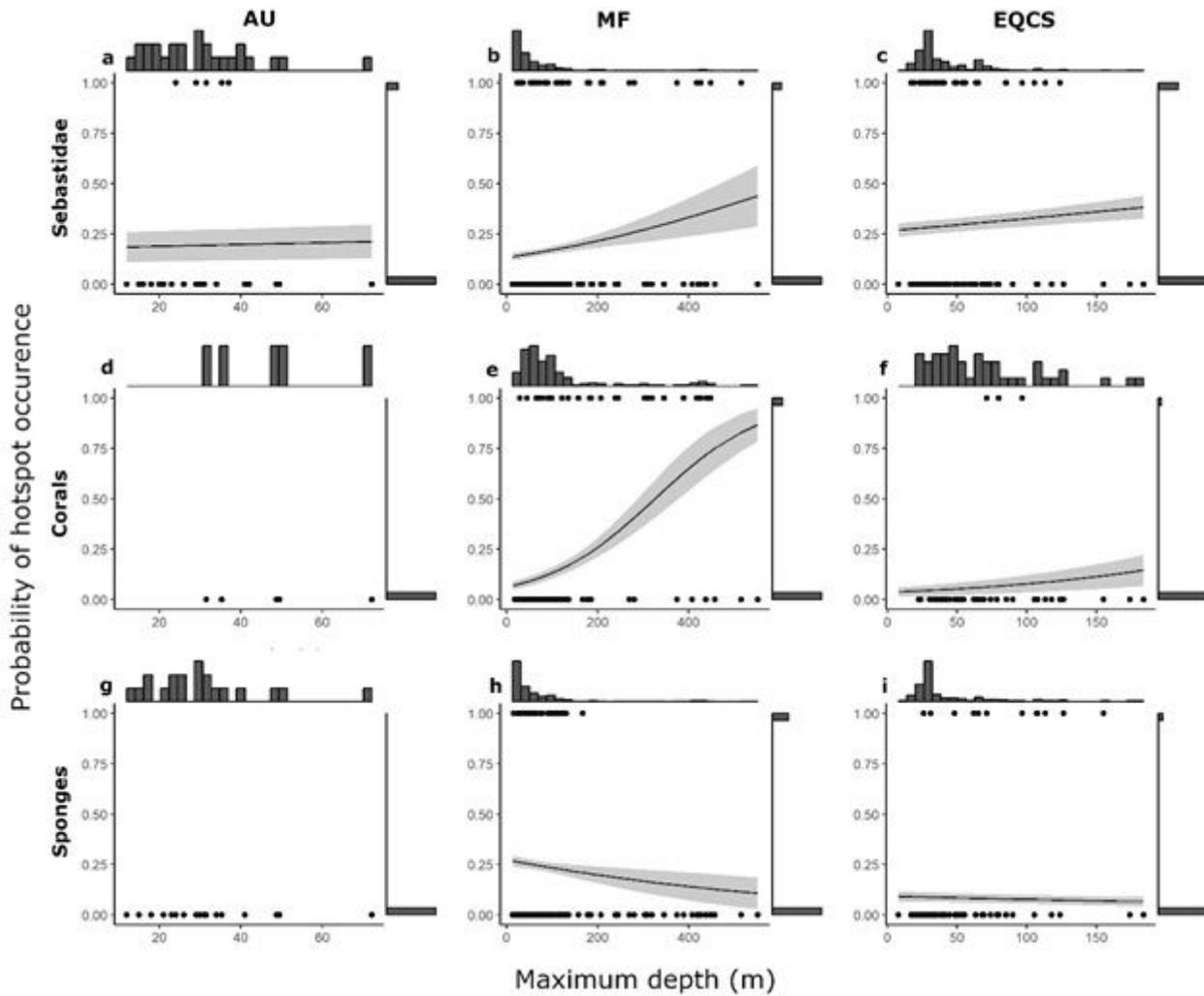
Figure 1

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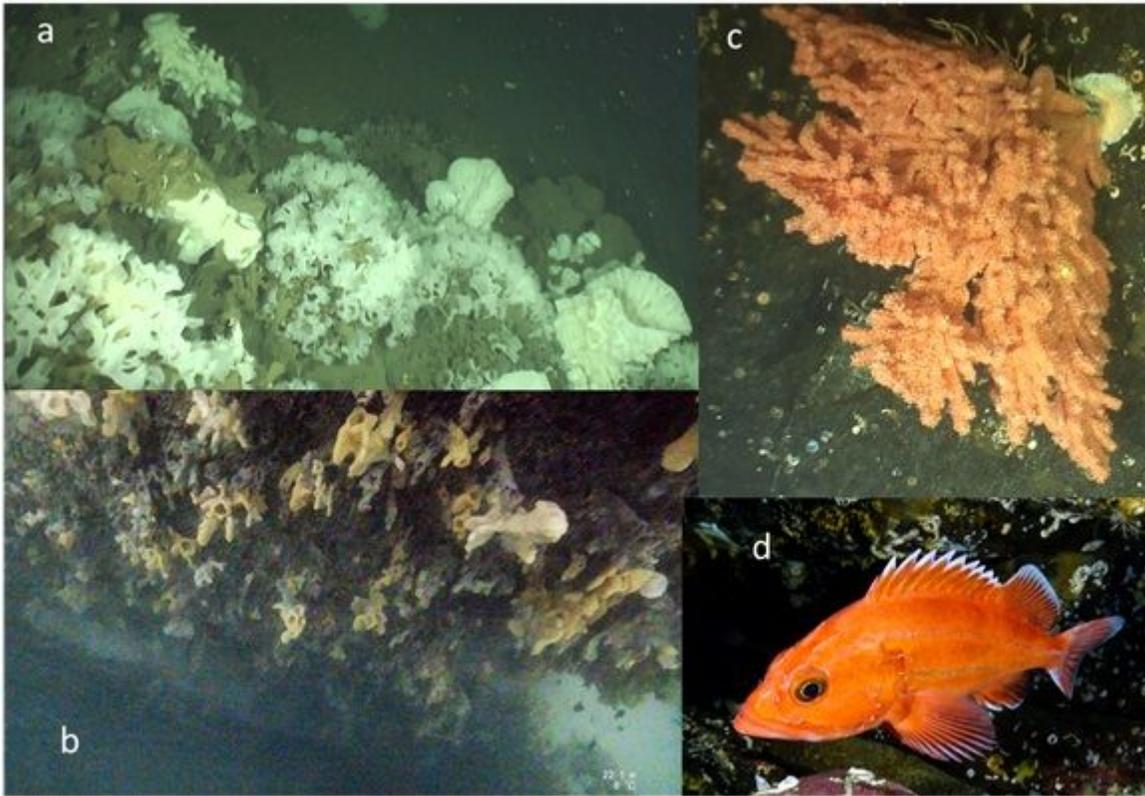
**Figure 2**

Spatial distribution of hotspot index ranks within 16-km<sup>2</sup> planning units (squares, except where faded over land), by species group: (a) Sebastidae, (b) large-bodied sponges, (c) structural corals, and by Upper Ocean Subregions (ABU = Aristazabal Upwelling, CSTM=Cape Scott Tidal Mixing, EQCS = Eastern Queen Charlotte Sound, MF = Mainland Fjords). Panel (d) shows the subset of 16-km<sup>2</sup> planning units containing hotspots for more than one species group. (The 3 species groups overlapped at a single 1-km<sup>2</sup> planning unit; see text.) Although inferential analyses were conducted at the scale of 1-km<sup>2</sup>, First Nations of the Central Coast require this coarse scale for visual displays to protect sensitive locations.



**Figure 3**

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