

Modeling the distribution of Palos Verdes giant kelp and devil weed

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ABSTRACT- The recent invasion of the Asian macroalgae *Sargassum horneri* along the coastline of Palos Verdes Peninsula represents a potential challenge to the recovery of the native kelp forests of the giant kelp (*Macrocystis pyrifera*). Management of *S. horneri* to combat this challenge can be assisted by statistical and spatially-explicit data on the distributional patterns of the species. However, research of these kinds of data are lacking in the Palos Verdes locality. This study aims to provide insights about the habitat distribution of *M. pyrifera* and *S. horneri* in Palos Verdes through approaches combining statistical and modeling techniques to model and quantify the nature of potential areas where suitable habitats for both species may overlap. In the statistical approach, the correlation of seven relationships between either species and three abiotic seafloor aspects (bathymetry, rugosity, and slope) were tested using the non-parametric rank correlation Spearman's ρ to quantify the influence each variable has on the density of either species. In the modeling approach, suitable habitats for both species were modeled using the machine learning procedure of maximum entropy. The resulting maximum entropy models were analyzed to quantify geographical patterns within overlaps in the suitable habitats of *M. pyrifera* and *S. horneri*. Statistical results have found that the distribution of *S. horneri* has higher linkage with bathymetry and rugosity than does *M. pyrifera*. Maximum entropy modeling found that overlapping suitable habitats are generally characterized by seafloors with depths shallower than 12.5m, smoother surfaces, and gentle slopes. These results can be used to identify potential areas where *S. horneri* may appear, from which strategic management practices can be utilized to minimize any negative impacts the invasive species may inflict upon local ecosystems.

INTRODUCTION

The nutrient-rich temperate waters along the California coastline is the home of a wide distribution of the giant kelp, *Macrocystis pyrifera*. Attaining lengths of up to 50 meters (Hoek et al., 1995), it is among the largest of the marine producers. *M. pyrifera* also provides the largest single source of fixed carbon in Southern California and is a foundational species of large three-dimensional kelp forests of which its ecological importance is enormous. It provides valuable habitat for many species of juvenile fish and supports a complex system of food webs and fisheries that altogether can consist of over 700 species of marine animals including species of additional ecological and economic importance such as the kelp bass, California spiny lobster, rockfish, and sea urchins (Graham, 2004). This makes the kelp forests one of the most diverse and productive temperate ecosystems. Furthermore, the giant kelp plays a critically important part in California's ocean economy, which brings in over \$40 billion in revenue per year. *M. pyrifera* itself is harvested as an important resource for a wide range of human uses; the giant kelp-harvesting industry alone has been valued at over \$30 million annually (Bedford, 2003).

In recent times, giant kelp forests in Southern California have faced significant depletion. The main factor is overgrazing by sea urchins, whose populations in the area have exploded due to the decimation of keystone species by overfishing (House et al., 2018). Rising sea temperatures and disruption of

the cycling of vital nutrients like nitrate are also attributed as contributing factors (Schmid et al., 2020). A particularly affected area is the kelp forests off the coast of the Palos Verdes Peninsula, a region within the Santa Monica Bay. It is adjacent to the largest urban area in the western United States and its decline has thus been amplified by additional human factors including urban runoff, sedimentation, and pollution (Foster and Schiel, 2010; Burdick et al., 2015). Palos Verdes has seen a loss of over 75% of its original giant kelp forests (Burdick et al., 2015) and has historically been nearly extirpated in the 1960s and 70s (Wilson, 1982).

Efforts to restore the kelp forests of Palos Verdes were first attempted following *M. pyrifera*'s virtual extirpation in 1967 but were unsuccessful. Gradual improvements in the environmental quality of the area following advancements in waste management and other developments has led to some recovery of *M. pyrifera* forests to an extent, although populations remained much smaller than historical numbers (Wilson, 1982). One of the main factors currently inhibiting its full recovery is the continued high presence of urchin barrens across Palos Verdes, which continue to threaten *M. pyrifera*. This is currently being addressed with joint efforts led by The Bay Foundation to eliminate such urchin barrens over a multi-year period since 2013. This ongoing project has seen increases in *M. pyrifera* densities and regional biodiversity as urchin populations were manually stabilized (House et al., 2018). Despite these

successes, several challenges in restoring the giant kelp forests in Palos Verdes remain.

One of these challenges includes the invasion of the non-native devil weed, *Sargassum horneri*, in the region. Similar to *M. pyrifera*, *S. horneri* establishes itself onto hard surfaces most commonly between depths of 3-15 m using a holdfast and utilizes gas-filled vesicles called pneumatocysts to stand upright for maximum photosynthesis (Marks, 2015). Originally native to temperate waters off the coasts of Korea and Japan, *S. horneri* was first identified in the Long Beach Harbor in 2003, likely carried across the Pacific via commercial vessels (Miller et al., 2007). It has since rapidly spread across the coastline and was first recorded in Palos Verdes by 2009 (Marks et al., 2015). Part of the rapid proliferation of *S. horneri* in Southern California lies within its ability to establish large populations very quickly with its high reproductive capacity, rapid annual growth cycle, and ability to drift long distances (Choi et al., 2008). It has been expressed that the chance of eradicating *S. horneri* is unlikely and that the next best option is to manage its spread (Marks et al., 2017).

The full ecological ramifications of *S. horneri* remain unclear. As an invasive species with an especially high capability for proliferation, early studies have initially described it as a potentially dangerous competitor to native algae (Marks et al., 2017; Ginther & Steele, 2018; Espreilla et al., 2019). However, recent studies suggest that the impact of *S. horneri* may not be as dire as previously anticipated, although negative impacts have still been identified. Ginther & Steele (2018) found that *S. horneri* inflicts a significant negative impact on kelp bass, which relies on safe habitat in the middle water column for recruitment that *M. pyrifera* provides but *S. horneri* cannot replicate. Although a follow-up study found that *S. horneri* otherwise had little impact on local fish assemblages, this demonstrates that fauna that highly rely on habitats exclusively provided by *M. pyrifera* may be under threat as *S. horneri* would be unable to replicate them (Ginther & Steele, 2020). The consequences on other organisms, such as other native algae and marine mammals, remain subject to further investigation. In the concern of competition with native algae, *S. horneri* appears to have limited success in environments where well-established native algae populations are abundant (Britton-Simmons, 2006; Caselle et al., 2018). Sullaway & Edwards (2020) suggested that *S. horneri* takes advantage of preexisting disturbances on native algae to dominate an area rather than engaging in direct competition. This was supported by Marks et al. (2020), which found that *S. horneri* did not typically compete with

native algae and instead occupied complementary niches, whose success was additionally strengthened by higher resistance against grazing from sea urchins. Thus, with the preexisting strain imposed on *M. pyrifera* in Palos Verdes, the invasion of *S. horneri* can represent a serious threat to the local kelp forests. The invasive algae can take advantage of the decline of *M. pyrifera* and dominate larger plots of the seafloor, which can further hinder the recovery of the kelp forests and exacerbate the well-being of communities that rely on the unique three-dimensional habitat the kelp forests provide.

Response against *S. horneri* in Palos Verdes began in 2015, and current practices in combatting the invasive algae consists of physical removal of individuals through slashing and vacuuming. The most recent effort was organized by the LA Waterkeeper and performed between 2016 and 2018, with conclusive results in the impact of such efforts still pending (Los Angeles Waterkeeper, 2020). While both slashing and vacuuming techniques were utilized, it has been reported that the slashing technique was more preferred due to its scalability when working with large populations. It is understood that *S. horneri*'s is incapable of asexual reproduction and as a result this prevents slashed individuals from regenerating. The proliferative capabilities of *S. horneri* will nevertheless allow it to quickly reoccupy areas of removal (Marks et al., 2017). However, given that *S. horneri* faces diminished success in areas with strong pre-existing native algae establishment, it has been suggested that if *S. horneri* was strategically removed in areas where native algae like *M. pyrifera* can re-establish themselves, they may be able to prevent the re-establishment of *S. horneri* (Marks et al., 2017; Espreilla et al., 2019).

To identify areas where management of *S. horneri* would have the most strategic effect, it is important to fully understand the invasive potential and patterns of the species. Combining spatially-explicit data through predictive distribution modeling (Espreilla et al., 2019) with statistical data can arguably produce a very effective tool in assisting in such identifications of strategic areas of management. The utilization of spatially-explicit predictive data was introduced into the field of *S. horneri* management by Espreilla et al. (2019), which utilized habitat classification and machine-learning based species distribution modeling to quantitatively analyze potential areas of overlapping suitable habitat for *M. pyrifera* and *S. horneri* in Santa Catalina Island. Thus, this study aims to provide preliminary data on the possible distribution patterns of *M. pyrifera* and *S. horneri* through two approaches that aims to gain insight on the statistical patterns of

the distribution of both species in relation to various environmental variables and produce and quantify spatially-explicit models predicting where overlaps in potential suitable habitats for *M. pyrifera* and *S. horneri* may occur.

MATERIALS AND METHODS

Abiotic data of the Palos Verdes seafloor was accessed through regional surveys conducted by the Seafloor Mapping Lab at California State University Monterey Bay. The surveys were conducted between 2001 and 2002 using acoustic remote sensing and encompasses the coastline of Palos Verdes from Torrance Beach-adjacent waters to between Point Fermin and Cabrillo Beach between depths of approximately 3-58 m (Fig. 1). Products of these surveys include multiple digital elevation model (DEM) grids and detailed imagery in 2m resolution raster format, but for the scope of this study three DEM grids representing bathymetry, rugosity, and slope were used (Fig. 2). The raw DEM grids were not in a suitable format for use, so they were assembled and reformatted into GeoTIFF rasters of the WGS 84 (EPSG: 4326) projection using the Geographic Information System QGIS 3.10.0. To minimize distortion, the three DEM grids were aligned by rescaling the values of each DEM grid according to cell size with the bathymetry model being used as the reference layer.

Species occurrence data of both *M. pyrifera* and *S. horneri* were obtained from multi-annual surveys conducted by the Vantuna Research Group of the Occidental College, which were provided upon personal request. The surveys were compiled into a dataset that reports the densities (individuals per square meter) of *M. pyrifera* between 2004 and 2019 and *S. horneri* between 2013 and 2019 observed in multiple 30m x 2m transects at 39 dive sites (Fig. 3). I further organized the dataset into a total of 2289 records of *M. pyrifera* and 423 records of *S. horneri*. However, both species are very frequent in the low intertidal zone (Chelsea Williams, personal communication), where a littoral gap exists in the DEM grids. As a result, abiotic data is only available up to around 3m of depth. Over half of the records of *M. pyrifera* and *S. horneri* were found to lay outside the vicinity of the abiotic data and could not be utilized. Therefore, only 1635 records of *M. pyrifera* and 267 records of *S. horneri* were used.

Two approaches were taken in this study. The first approach utilizes the non-parametric rank correlation Spearman's ρ to analyze patterns between the occurrences of *M. pyrifera* and *S. horneri* and the three seafloor aspects to determine possible statistical relationships between both species and environmental



Figure 1: Map of Palos Verdes and the study area

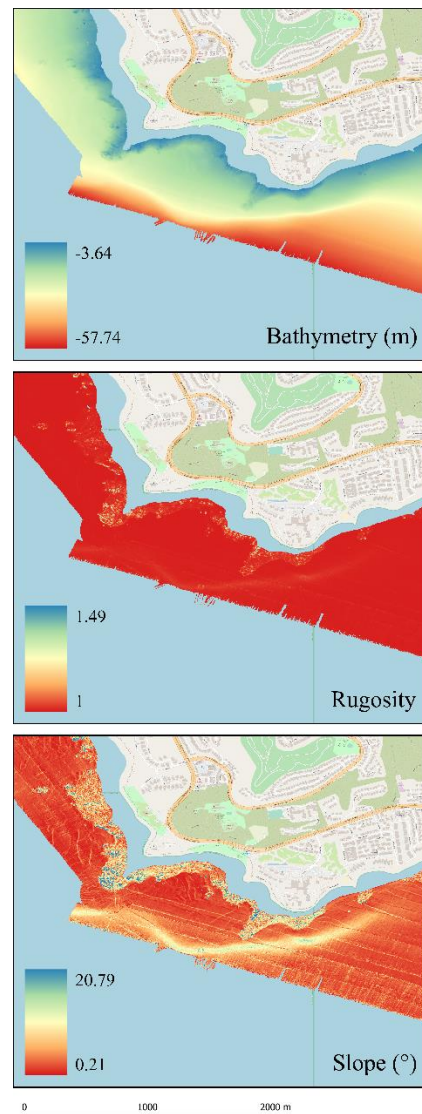


Figure 2: Digital elevation models representing the three abiotic seafloor aspects around the Point Vicente area.

factors. The goal of a Spearman's ρ test is to determine the correlational relationship between two given variables. In this study, the test is used to determine the correlation between the densities of either *M. pyrifera* or *S. horneri* in regards to differences in the values of an environmental aspect, which provides clues on how much influence such environmental aspect has on the occurrence of the species.

The mean densities of each record of both species were organized into seven relationships. The first six relationships tested the correlation between the density of either species and one of each abiotic seafloor aspect. Complementary data of each aspect specific to each record of *M. pyrifera* or *S. horneri* were obtained by generating the species records as individual points on the three DEM grids on QGIS 3.10.0. Then, data values of the individual pixels on each DEM grid from which each record point is located on were sampled. This generated the specific value of bathymetry, rugosity, or slope that each record occupies. When pitted in a correlational relationship between two coexisting species with ecological similarities, Spearman's ρ can also be used to determine possible direct ecological interaction. Specifically, if it is shown that a strong statistically significant correlation exists between the densities of *M. pyrifera* and *S. horneri*, then it indicates that the presence of one of the species specifically influences the presence of the other. As such, the seventh relationship tests the correlation between the densities of *M. pyrifera* and *S. horneri*. Only overlapping records can be utilized to test the relationship, and so only *M. pyrifera* records that coexist in date and transect with *S. horneri* records were used, which were 267 in total.

The resulting seven tests were outputted with a ρ value and 2-sided p-value. The ρ value represents the correlation coefficient for a nonlinear function, but it can also be interpreted as a percentage of linkage. In other words, a ρ -value of 0.32 in a two-variable relationship can be interpreted as 32% of the observed variation in one variable being linked with the other; inversely, it can also be interpreted as the remaining 68% of the observed variation in one variable is linked to unknown variables (Ariel Levi Simons, pers. com.). The latter application is adopted in the scope of this study. Because multiple relationships were tested, the raw p-values are inaccurate. To resolve this, the Benjamini-Hochberg adjustment procedure, which adjusts each statistical outcome to be accurate with multiple relationships, was implemented to produce corrected p-values and critical values for all seven relationships.

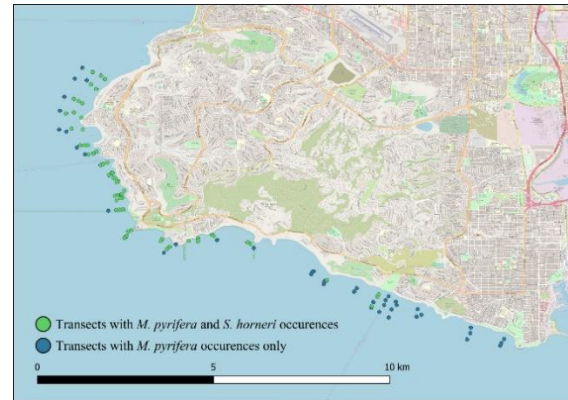


Figure 3: Examined transects between 2004 and 2019 along Palos Verdes. There are no transects containing only records of *S. horneri* occurrences in the dataset.

The second approach utilizes spatially-explicit modeling of the potential suitable habitats of *M. pyrifera* and *S. horneri* to predict possible overlapping distributions and quantify the nature of such areas. To do so, the machine learning procedure of maximum entropy was used to generate a pair of species distribution models of both species. The procedure was performed using the module MaxEnt v3.4.1 through the statistical computing system R v3.6.1. Presence and background sample data were used to train the machine-learning module. All 1635 records of *M. pyrifera* and 267 records of *S. horneri* were used as the presence data for their individual models.

Background data was represented as pseudoabsence points, which were generated in QGIS 3.10.0. There, the species records were again generated as individual points, from which a buffer with distances of 0.001 geographic degrees around each point was generated. Within the buffer, the pseudoabsence points were generated as random points with a density of one thousand and minimum distance of 0.0001 geographic degrees between each random point. The points were then compiled into the background dataset. The predictor data was provided as a raster stack of the three DEM grids. The two species distribution models produced from the maximum entropy procedure were cross-validated through a k-fold procedure five times each. A jackknife test, which measures the importance a variable has in the generation of a maximum entropy model, was performed on each model to measure the percentages of contribution each abiotic seafloor aspect had, which can be compared with the Spearman's ρ tests of the first approach.

The resulting species distribution models were exported in presence-absence form as GeoTIFFS. In this form, the models were re-illustrated to contain a

Bathymetry level	Designated values (m)
Inner	3-7.5
Middle	7.6-12.5
Outer	12.6-17.5
Deep	17.6-25
Deepest	25.1+
Rugosity level	Designated values
Low	1-1.22
Moderate	1.23-1.44
High	1.45-1.66
Slope level	Designated values (°)
Low	0-16.83
Moderate	16.84-33.65
High	33.66-50.48

Figure 4: Classification scheme of designated values of each defined level of bathymetry, rugosity, and slope that was used in this study.

presence variable (represented by all areas where suitability exceeds 0.5) and an absence variable (where suitability is below 0.5). The two models were polygonised into shapefile vectors, overlaid, and clipped to isolate the areas where both models overlap. This was done in the earlier version QGIS 2.8.3, as it had a better ability in processing manipulations of very complex vectors than QGIS 3.10.0 in the case of this study. However, QGIS 3.10.0 was still used in tasks that do not require the manipulation of vectors. To identify and quantify the common environmental patterns of the overlapping habitats, an informal classification scheme defining specific levels of bathymetry, rugosity, and slope was devised. The scheme divides the range of each seafloor aspect values into different levels (Fig. 4). Bathymetry levels were classified into five zones. They are the “inner zone” (3-7.5m), “middle zone” (7.6-12.5), “outer zone”, (12.6-17.5m), “deep zone” (17.6-25m), and “deepest zone” (all depths below the “deep zone”). Rugosity and slope levels were classified into low, moderate, and high levels. The value range of each levels represent a third of the entire range of rugosity and slope values identified in their respective DEM grids. Eleven models isolating each seafloor aspect level were generated by individually masking each DEM grid to the respective value ranges in QGIS 3.10.0. These masks were polygonised into shapefile vectors in QGIS 2.8.3. The environmental patterns of the overlapping habitats of *M. pyrifera* and *S. horneri* were quantified by isolating its occurrences within each seafloor aspect level by individually clipping the

overlapping vector model with each of the eleven vector models. The areas of each isolated vectors were then compared with the area of the overlapping habitats to determine the percentages of each seafloor aspect level making up the overlapping habitats. Additionally, the percentages of overlap were calculated between the suitable habitats of both species within the Abalone Cove and Point Vicente State Marine Conservation Areas. These areas can be of particular interest due to their statuses as protected marine sanctuaries. The percentages were calculated by clipping the maximum entropy models of both species to within the legal borders of each conservation area and then comparing the areas of the suitable habitat for both species and their overlapping areas.

RESULTS

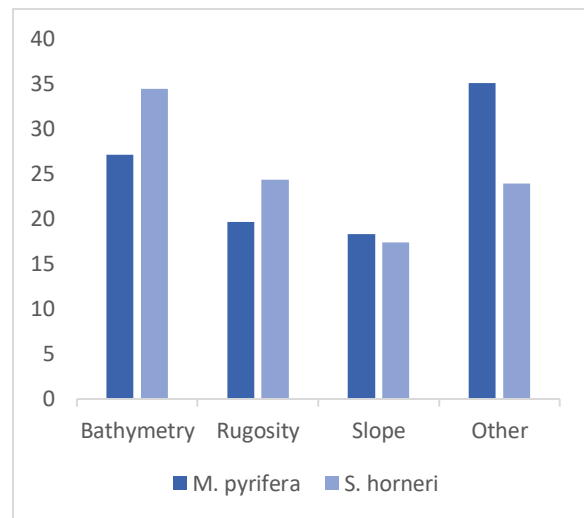


Figure 5: Percentages of linkage based on Spearman's ρ of each seafloor aspect to the observed variation in the density of *M. pyrifera* and *S. horneri*

All six correlational relationships between a species and abiotic seafloor aspect tested using Spearman's ρ with Benjamini-Hochberg corrections are statistically significant (Fig. 6). Using the interpretation of percentage of linkage, the ρ value results of these relationships show that the variation in the density of *M. pyrifera* in Palos Verdes is 27.06% linked with bathymetry, 19.63% linked with rugosity, and 18.29% linked with slope. The variation in the density of *S. horneri* in Palos Verdes was found to be 34.41% linked with bathymetry, 24.32% linked with rugosity, and 17.38% linked with slope (Fig. 5).

The correlational relationship between the densities of

Species	Variable	Spearman's ρ	p-value	BH-adjusted p-value	BH-adjusted critical value
<i>M. pyrifera</i>	Bathymetry	0.2706	8.0390e-29	5.6273e-28	0.007
<i>M. pyrifera</i>	Rugosity	0.1963	1.1682e-15	4.0887e-15	0.0143
<i>M. pyrifera</i>	Slope	0.1829	9.1980e-14	2.1462e-13	0.0214
<i>M. pyrifera</i>	<i>S. horneri</i>	0.0734	0.2322	0.2322	0.0500
<i>S. horneri</i>	Bathymetry	0.3441	7.7608e-09	1.3581e-08	0.0286
<i>S. horneri</i>	Rugosity	0.2432	5.9374e-05	8.3124e-05	0.0357
<i>S. horneri</i>	Slope	0.2738	0.1738	0.0044	0.0429

Figure 6: Raw values of Spearman's ρ analysis with Benjamini-Hochberg adjustments

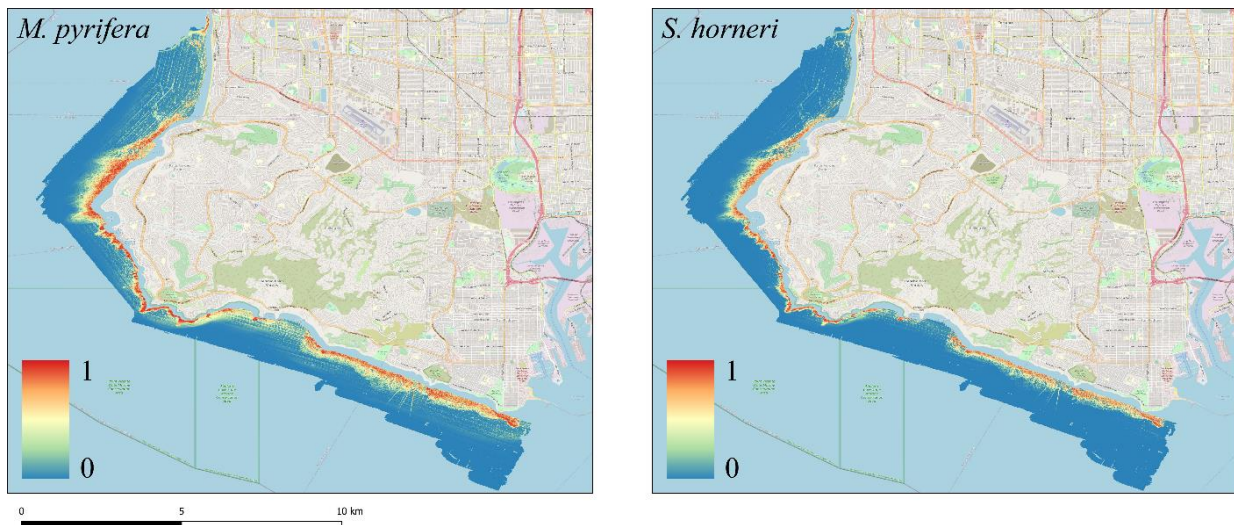


Figure 7: Maximum entropy models identifying habitat suitability for *M. pyrifera* and *S. horneri*. Values approaching zero represent least suitability while values approaching one represent most suitability.

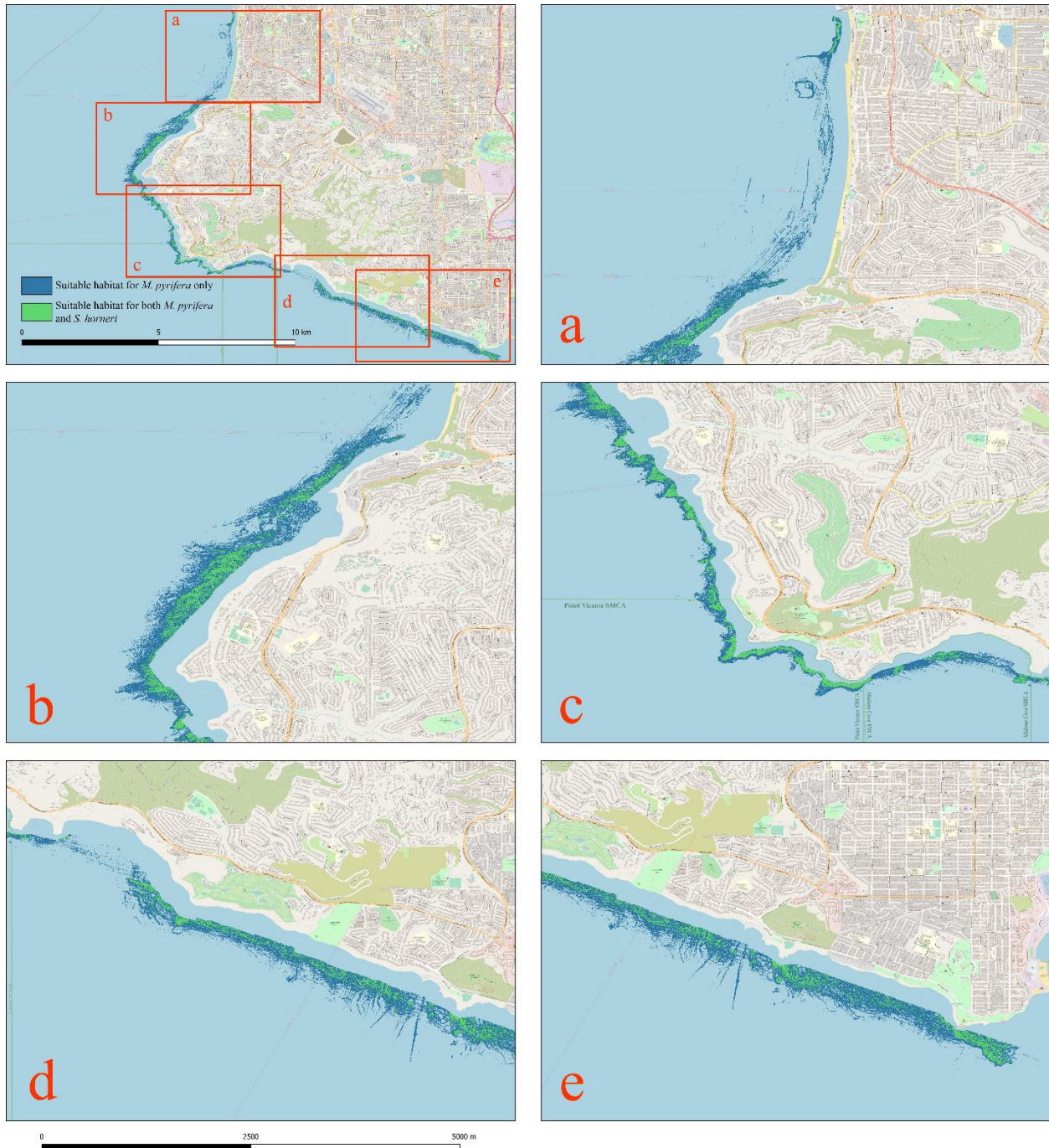


Figure 8: Areas of overlapping suitable habitats of *M. pyrifera* and *S. horneri* as represented by presence-absence maximum entropy models. Diagrams a-e are closeups as represented by the red boxes in the uppermost left diagram.

M. pyrifera and *S. horneri*, which was tested to determine the possibility of environmental interaction, was not statistically significant (Fig. 6). This indicates that neither species are directly influencing the presence of the other. This supports the previous notion that *S. horneri* proliferates by occupying complementary niches rather than directly competing

with native species like *M. pyrifera*. An alternative explanation for the outcome may be in that the sample size represented by the number of species records utilized may have simply been too small to provide a statistically significant outcome. However, this is contradicted by the statistically significant outcomes of the other *S. horneri* relationships, which have the

same sample size. A retest of the correlational relationship using a larger dataset may help confirm the interpretation in the future.

Bathymetry level	Occurrence
Inner	31.840%
Middle	66.592%
Outer	1.519%
Deep	0%
Deepest	0%
Rugosity level	Occurrence
Low	99.282%
Moderate	0.649%
High	0.0713%
Slope level	Occurrence
Low	91.482%
Moderate	7.889%
High	0.629%

Figure 9: Percentages of the overlapping suitable habitats that are made up by the respective designated levels of bathymetry, rugosity, and slope

Locality	Percentage of overlap
Entire study area	24.83%
Point Vicente SMCA	43.81%
Abalone Cove SMCA	19.38%

Figure 10: Percentages of suitable habitat for *M. pyrifera* overlapping with suitable habitat for *S. horneri* within the entire study area and local State Marine Conservation Areas

Maximum entropy modeling has generated models that predict areas that may hold the most suitable habitat for both species (Fig. 7). The presence-absence form of these models, which were used to examine areas of overlapping suitable habitats, can be closely examined to identify any strategic areas for management practices (Fig. 8).

Analysis of the maximum entropy models found that 24.83% of suitable habitat for *M. pyrifera* overlaps with suitable habitat for *S. horneri* (Fig. 10). All suitable habitats for *S. horneri* overlap with *M. pyrifera*, and there are no areas that were suitable only for the former. However, this is likely due to the nature of the dataset given that all records of *S. horneri* coexisted with *M. pyrifera*.

Nearly a third (31.84%) of the overlapping habitats were in the designated inner bathymetry level while roughly two-thirds (66.59%) were within the middle level. Only 1.52% was in the outer level (12.6-17.5m)

while no overlaps in depths below 17.5m were present. Nearly the entire area (99.28%) was characterized by low rugosity, with only minute portions having moderate (0.65%) or high (0.07%) rugosity. A large majority of the area was also characterized by low slope (91.48%), while a small percentage (7.89%) contained moderate slope and very little (0.63%) had high slope (Fig. 9). Jackknife tests found that bathymetry contributed to 54.7%, rugosity to 45.1%, and slope to 0.2% of the maximum entropy model of *M. pyrifera*. These percent contributions are also very similar in the model for *S. horneri*, where bathymetry contributed 54.2%, rugosity 45.8%, and slope 0%. However, this trend can be possibly attributed to the nature of the dataset.

Separate analyses specific to the Point Vicente and Abalone Cove State Marine Conservation Areas (Fig. 8-c) found that 19.38% of suitable habitat of *M. pyrifera* within the Abalone Cove SMCA overlaps with *S. horneri*. In the Point Vicente SMCA, there is a 43.81% overlap of *M. pyrifera* with *S. horneri* (Fig. 10).

DISCUSSION

The Spearman's ρ tests found that the density of *S. horneri* has a higher linkage with the three tested abiotic seafloor aspects than *M. pyrifera* does, especially in bathymetry where it represents the highest influence on the density of *S. horneri*. In pure quantity, the areas of overlap between the suitable habitats of *M. pyrifera* and *S. horneri* from the maximum entropy models are typically characterized by depths up to around 12.5m, "low" rugosity, and "low slope". Classification of levels of bathymetry, rugosity, and slope were informally created for convenience. As a result, the definition of what range of each seafloor aspect is high or low can be ultimately subjective and so each level should be best interpreted by their objective value ranges (Fig. 4).

Recent research suggests that *S. horneri* does not typically compete with native algae and instead takes advantage of pre-existing disturbances affecting them (Marks et al., 2020; Sullaway & Edwards, 2020). Nevertheless, given the aggressive proliferative ability of *S. horneri*, it is still important to continue cautious management of the alga as the full consequences of *S. horneri* in the local environment, especially in long-term effects, still require further research (Ginther & Steele, 2020). The statistical data on density patterns and spatially-explicit models from this study can serve as preliminary data that can help predict potential areas of invasion by *S. horneri* in a previously understudied locality that can also be occupied or is already occupied by *M. pyrifera*. When paired with

current and future observations, they can assist in *S. horneri* management efforts by helping guide the identification of specific areas where removal efforts may be most effective and addressing what abiotic factors *S. horneri* are relying on to spread.

Strong parallels can be drawn with the second approach of this study and that of Espriella et al., (2019). Both utilized maximum entropy models to identify overlaps in suitable habitats of both *M. pyrifera* and *S. horneri* and DEM grids with the purpose of assisting in management efforts. Like in Espriella et al., (2019), this study was limited by a littoral gap in shallower depths. Depths shallower than approximately 3m were not available despite a large portion of *M. pyrifera* and *S. horneri* records being located within such gap. Littoral gaps, or “white zones”, are a limitation that remote sensing seafloor surveys can face. This is often due the inability of the survey vessel to enter shallower waters or interference from shallow-water organisms. In the case of the Palos Verdes seafloor survey, interference from a high presence of kelp in the littoral gap is ironically a factor (Chel. Wil., pers. com.). The littoral gap can be filled by bathymetric lidar data, which provides data at shallower depths (Copeland et al., 2011), but the technological requirements of accurately complementing the two abiotic datasets was beyond the ability of this study and was thus not attempted. However, given that nearly half of the dataset of *M. pyrifera* and *S. horneri* were within the littoral gap, complementation with lidar data would have been beneficial and is something that should be considered in future studies.

Another limitation of this study is in the narrow quantity of variables used in this study. Ultimately, only the relationships of five variables (bathymetry, rugosity, slope, species occurrence, and species density) were tested. This leaves out a multitude of other factors that may serve an important role in influencing the distribution of both *M. pyrifera* and *S. horneri* such as oceanic temperature, nutrient concentration, and the presence of other organisms such as sea urchins. The maximum entropy models produced in this study are already more limited than those in Espriella et al., (2019), as data on seafloor aspects such as topographic position indices and direction of slope were not available for Palos Verdes. Fortunately, tests like Spearman’s ρ partially account for the lack of important variables by indirectly finding the influence of unknown factors (Ari. Lev. Sim., pers. com.). The inclusion of quantified data of additional factors can greatly increase the accuracy and precision of predictions in future studies. Additionally, updated seafloor maps examining

additional abiotic aspects can greatly expand the accuracy and detail of the data provided by this study.

CONCLUSION

The recovery of the *M. pyrifera* kelp forests along the Palos Verdes coastline may be hindered by the invasion of *S. horneri*, which can negatively affect the well-being of local communities that rely on the unique habitat the former provides. Thus, recovery of the kelp forests can be better secured with the management of *S. horneri*. This study has successfully modeled and quantified potential areas along the Palos Verdes coastline where suitable habitats for both *M. pyrifera* and *S. horneri* may overlap, as well as statistically identify the influences of three abiotic seafloor features towards the presence of both species. The data from these results can be used to identify areas where management of *S. horneri* may be best effective.

As more research is conducted on the invasion of *S. horneri* and its consequences on important local ecosystems, future studies can build on this study to further optimize our understandings and abilities to manage the invasive species in Palos Verdes waters. The nature of how organisms are distributed can be a complicated subject potentially attributable to a broad spectrum of factors. As a result, incorporation of a broader variety of variables (such as additional climatic, oceanographic, and ecological factors) and the use of solutions such as lidar data to address the littoral gap issue can greatly increase the accuracy and informativeness of such future studies. Additionally, future research should test questions that may have arisen from the results of this study, such as retesting the correlational relationship between densities of *M. pyrifera* and *S. horneri* using a larger dataset to confirm its statistical insignificance, which can provide additional insight on the consequences of the interactions between the two species.

STATEMENT OF PUBLICATION

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REFERENCE

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