DEVELOPMENT AND TESTING OF A SPATIAL TOOL FOR THE ASSESSMENT ANTHROPOGENIC THREATS IN COASTAL AND MARINE HABITATS IN PUERTO RICO

By

Idelfonso Ruiz Valentín

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Approved by:

______________________________ ____________________
Francisco E. Pagan, Ph.D. Date
Member, Graduate Committee

______________________________ ____________________
Roy A. Armstrong, Ph.D. Date
Member, Graduate Committee

______________________________ ____________________
Manuel Valdés Pizzini, Ph.D. Date
Member, Graduate Committee

______________________________ ____________________
Richard S. Appeldoorn, Ph.D. Date
President, Graduate Committee

______________________________ ____________________
John M. Kubaryk, Ph.D. Date
Interim Director, Marine Sciences Department

______________________________ ____________________
Fernando Gilbes, Ph.D. Date
Representative, Graduate School

Abstract

Marine and coastal ecosystems are subject to a wide variety of stressors derived from anthropogenic activities. Because of the complexity of these threats and the difficulty in monitoring them in a synoptic manner, scientists and managers must generally rely on proxy approaches to model stressors in the assessment of habitat condition and threat levels. Most previously developed coastal and marine threat indices acknowledge that the use of socioeconomic variables would assist in defining threatened marine habitats. This study investigated the incorporation of anthropogenic stressors into a GIS databased and applied them to develop a spatially explicit index of stress for the coastal and marine environments in Puerto Rico. Spatial quantitative data were used to construct and map indices of several key threats, including an overall threat index, In this application, socio-economic data were combined with watershed characteristics and coastal forcing that impact stress level and distribution. Twelve variables were reduced to develop high-resolution (250x250 m) indices for four threats: i) turbidity, ii) eutrophication, iii) pollution and iv) overfishing. All variables were standardized on a scale of 0 = 1 and then reclassified on a scale of 1 to 5. Classification was conducted within ArcView using the natural breaks function. The spatial index was defined by using a hexagon grid, (2 km^2 or 898.6 m on a side), Means threats were merged within a grid cell, which resulted in 10,828 marine planning units for the region. Individual threat indices were combined to produce the overall index. Several problems were encountered in assessing anthropogenic stress using socio-economic vulnerability indicators due to the inherent difficulties involved in ranking socio-economic data on an interval scale. Temporal aspects also caused difficulties as socio-economic variables vary over time as coastal populations and land use change. Validation of results was assessed by comparing area-specific scores to field studies; these correlated well with expected outcomes. Cumulative threats generally were higher along the west and south provinces in comparison to the east and north provinces, suggesting that management of reserve areas must occur at a larger spatial scale, as many of the anthropogenic stressors may occur on the scale of the coast. Results of the combined threat index were classified and interpreted using a triage approach relative the management of marine protected areas.

Resumen

Los ecosistemas marinos y costeros están sujetos a una amplia variedad de factores de estrés derivados de las actividades antropogénicas. Debido a la complejidad de estas amenazas y la dificultad en el control de una manera sinóptica, los científicos y los manejadores generalmente se basan en una representación que se acerca a los factores estresantes para modelar y evaluación de la condición del hábitat y de los niveles de amenaza. Los índices de amenazas costeras y marinos más desarrollados anteriormente reconocen que la utilización de variables socioeconómicas ayudaría a definir los hábitats marinos amenazados. Este estudio investigó la incorporación de los factores de estrés antropogénico utilizando una base de datos en Sistema de Información Geográfica (GIS) y aplicados a desarrollar un índice espacial explícita de estrés para los entornos costeros y marinos en Puerto Rico. Datos cuantitativos espaciales se utilizaron para construir y asignar índices de varias amenazas clave, incluyendo un índice general de amenazas, en esta aplicación, los datos socio- económicos se combinaron con las características de las cuencas hidrográficas y los forzamientos costeros para medir nivel de estrés o impacto y su distribución espacial. Se utilizaron doce variables donde se desarrollaron mapas de alta resolución (250x250 m) para cuatro amenazas: i) la turbidez, ii) la eutrofización,
iii) la contaminación y iv) la sobrepesca. Todas las variables fueron normalizados en una escala de 0 = 1 y se reclasificaron en una escala de 1 a 5. La clasificación se realizó en ArcView utilizando la función de “natural breaks”. El índice espacial se define por medio de una rejilla hexagonal, (2 km^2 o 898,6 m de lado), las amenazas se fusionaron en una celda o cuadrícula, que se tradujo en 10.828 unidades de

planificación marina de la región. Luego los Índices de amenazas individuales se combinaron para producir el índice general. Se encontraron varios problemas en la evaluación de las presiones antropogénicas utilizando indicadores de vulnerabilidad socio- económicas debido a las dificultades inherentes a la clasificación de datos socio-económicos en una escala de intervalo. Aspectos temporales también causaron dificultades como datos de poblacionales y el cambio de uso del suelo. La validación de los resultados se evaluó mediante la comparación de resultados específicos del área de estudios de campo, los cuales correlacionaron bien con los resultados esperados. Amenazas acumulativas general fueron superiores a lo largo del oeste y la provincia del sur, en comparación con las provincias del este y norte de Puerto Rico, lo que sugiere que la gestión de las áreas de reserva debe realizarse a una escala espacial más grande, ya que muchos de los factores de estrés antropogénico puede ocurrir a una escala mayor al del litoral costero. Los resultados del índice de amenaza combinada fueron clasificados e interpretados utilizando un enfoque de “triage” al manejo de las áreas marinas protegidas.

Dedication

To my wife Katherine Q. Kortright Rivera who provided me support during all this years, without her assistance this work could not have been possible; to my daughter Kaideliz M. Ruiz Kortright who provide me with unique ideas; and to my father Idelfonso Ruiz Garcia who always had time for conversation.

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Chapter 1. Introduction

Coastal and marine resources in Puerto Rico have been influenced by anthropogenic activities since pre-Colombian times. Taino populations relied heavily on protected estuaries, harbors and rivers mouths, where they could establish fish weirs or set traps (Griffith and Valdés Pizzini, 2002). However, it was not until after 1940 that Puerto Rico`s rapidly increasing population placed significant pressure on its coastal resources (Griffith and Valdés Pizzini, 2002). Consequently, the exploitation of pristine and fertile watersheds becomes a major threat. These social activities accelerate sediment delivery to coastal waters and causes stress on coastal ecosystems (Morelock and Taggart, 1988). Problems arise when these sediments contain elevated nutrient levels that promote eutrophication (leading, e.g., to increased algal overgrowth of coral reefs and increase coral susceptibility to high temperatures and resulting bleaching), or possess high concentration of heavy metals with detrimental effects on aquatic life, and consequently degradation of coral (Rogers, 1983; Micheli, 1999; Jackson et al., 2001; Linton and Warner, 2003; Warner et al., 2005). In general, marine resources in the Puerto Rico have been overexploited for decades (Appeldoorn and Meyers, 1993; Matos-Caraballo et al., 2005).

Worldwide, key questions remain as to which coastal and marine habitats are most vulnerable to natural and anthropogenic stressors and which are most likely to cause negative impacts on coastal and marine ecosystems. Until now there has been little success in mapping threat regimes or in incorporating such regimes in designing networks of marine protected areas or in coastal and marine spatial planning in general.

Recent research in the Gulf of California (Sala et al., 2002) attempted to incorporate social factors into the design of marine reserve networks (MRNs) by integrating multiple levels of information on biodiversity, ecological processes and socio-economic factors. For example incorporating fishing pressure (quantified as the density of small fishing boats), reduces social conflicts by minimizing the overlap between reserves and heavily fished areas. Another implementation of social-economic factors was conducted for the Channel Islands, California (Airamé et al., 2003), which are threatened by anthropogenic activities such as oil exploration, drilling, production, transport, spills and other environmental events. Based on these threats, they incorporated social-economic data into the mathematical algorithms to produce a model that reduces these threats in a reserve network. It allows for the potential impacts of catastrophic events by increasing the percentage of area in a reserve. Hence they multiplied the planning unit by an ‘‘insurance factor’’ that takes into account the frequency of severe disturbance to the environment (Allison et al. 2003). As a consequence, the resulting area within the MRN needed to be increased 36–54%. In fact, the use of socioeconomic variables in addition to biodiversity data is particularly important because in marine systems, where fishing is a major threat, ecological criteria and socioeconomic measures are not independent process (Castilla, 1999; Sala et al., 2002).

Nevertheless different indices have been used in coastal areas to relate global warming (Gornitz et al., 1993) and human impact (McLaughlin et al., 2002) on coastal zones (Linton and Waner, 2003; Halpern et al., 2008). However, these indices do not examine socio-economic variables or “stressors” (Cooper and McLaughlin, 1998; McLaughlin et al., 2002), and they can lack efficiency as they often prescribe unrealistic scenarios. There is an urgent need to incorporate the socio-economic stressors in the design of coastal and marine geographic management's strategies. This strategy will assist local environmental regulatory agencies in the identification of potential fishing grounds and marine habitat areas more threaten by theses anthropogenic drivers.

This study will use available socioeconomic and natural data to develop stress indices linked to the known threats to coastal ecosystems, especially those associated with coral reefs, resulting from turbidity, land-based sources of pollution, eutrophication and overfishing. Specifically, the objectives of this study are to: (i) develop spatially explicit anthropogenic, socio-economic and natural stressor metrics for Puerto Rico, (ii) map individually predicted threat scores and cumulative impact scores within the Puerto Rico shelf, and (iii) map the coastal and marine spatial distribution of threats among the geographic provinces in Puerto Rico.

The analysis will consist of mapping the accumulated factors and spatial distributions of theses stressors, as a prerequisite for the design of coastal and marine zone management considerations. This, in turn, as part of this study will be developed in a database compatible with Geographical Information Systems (GIS) to account for multiple sources of stress.

Chapter 2. General concepts

The problems arising from coastal and marine threats involve all aspects of the natural and anthropogenic activities and often lead to difficult decisions in order to avoid dramatic consequences on ecosystems health. Among the most important coastal pressures are land cover change, pollutant loads, and introduction of invasive species. These pressures can lead to loss of marine biodiversity and a threat to human health. However, there are many potential indicators of natural and socioeconomic stressors that could contribute to coastal and marine ecosystem threats. Furthermore, in deciding which stressors to include, the desirability of including a parameter must be balanced against the availability of up-to-date data that is in a useable format. Therefore, the stressors chosen for inclusion in this study were those for which data could be deemed to be of relevance to fisheries and marine habitat resources (Fig. 2.1).

In the last several years there has been a substantial increase in the number of anthropogenic (World Resources Institute, 2004: Halpern et al., 2008) and climate change (Gornitz et al., 1993) threat indices developed for specific coastal and marine areas. The problem with the application of these approaches is that their results are dependent upon the scale of the analysis.

For example, Reefs at Risk in the Caribbean (World Resources Institute, 2004), the most detailed analysis of the Caribbean to date, was dependent upon spatial data sets that were Caribbean wide. At this broad scale, spatial data may reflect imperfect representations of the environment at the local scale (country wide) because of inaccuracies or mistakes in remotely sensed data. Thus, at the local scale results would introduce bias and a loss of spatial resolution in developing the individual threat layers.

A result of that analysis, for example, was that over 80 percent of Puerto Rico was classified as high risk with respect to the overfishing threat putting it on a par with the most threatened areas: Jamaica and Hispañola. The same issue arises when combining across threats to get an overall threat score. Overall, the threat score for Puerto Rico was very high-high relative to the rest of the Caribbean. Such regional efforts may effectively relate threat levels over broad areas, but they cannot efficiently differentiate the levels of stress on the scale of Puerto Rico that would be useful for management purposes by the local responsible agencies.

On the local scale, threats indices need to be developed that include a range of anthropogenic and socioeconomic perturbations related to coastal and marine habitat degradation. These need to be expressed within units of relatively small spatial scale and in ways that allow for threat standardization and aggregation. These classifications can then assist in the implementation of management strategies in marine areas shown to be at risk.

This study examines four main threats to the coastal and marine environment: turbidity, eutrophication, pollution and overfishing (Table 2.1). In general the data used to model these coastal ecosystem threats and to explain their geographic distribution were: i) fishery landings, ii) fishing effort, iii) fishing traps or pots, iv) piers, marinas and ports, v) watershed, vi) humans populations densities and vii) sewage and outfall pipes and viii) land use. Table 2.2, gives more detail on these data and Figure 2.1 shows how these data sets were combined to develop the four threat indices.

In examining the data available, it is evident that there is wide variety of alternative data sets that potentially could be used to develop these and other threat indicators. Nevertheless, the data for inclusion here had to meet several criteria. First, the data set had to be comprehensive; that is, it needed cover the full geographic range of the study. For example, other data sets on potential stressors that could not be included were hypoxic zones, aquaculture, disease, invasive species, for all of these stressors data exist for one or more regions, but none have full area coverage. Second, the data had to have geographic specificity. For example, a seemingly obvious data set for recreational fishing pressure would be the location of registered non-commercial vessels. However, in Puerto Rico, location of registration (physical address of the owner) has little relation to the coastal area where a boat would actually be used.

Table 2.1 Summary of the threat analysis methodology each threat category.

Threat selection and description	Coastal Pressure
Turbidity: the degree of turbidity and changes in turbidity levels in coastal and estuarine waters affect light penetration and can have direct and indirect effects on fish (Bash et al. 2001, Bejarano and Appeldoorn 2012), and reduce coral and submersed macrophyte cover, resulting in habitat deteriorations (Goldsborough & Kemp 1988).	Turbidity problems develop in response to land use practices, which result in soil erosion and other point and non-point source runoff as river plume and watershed outflows.
Pollution: healthy coastal marine ecosystems can be affected by potently toxic substances that enter the marine environment as a result of agricultural runoff and industrial and residential sewage discharges.	Land-based sources of pollution are classified as point and non-point, with the biggest sources of pressure being the latter. Nonpoint source pollution pressures include coastal development (classified as residential or industrial), marinas, coastal population density, agriculture practices and runoff. In contrast, point source pollution occurs when there is a single, identifiable, and localized source of the pollution. An example is directly discharging sewage through an outfall.
Eutrophication: nutrient enrichment can have a negative impact on the marine and coastal environment (Nixon 1995), causing predictable increases in the biomass of algae. Phytoplankton blooms cause shading and may enhance oxygen consumption and depletion resulting in the death of benthic animals and fish. Benthic macroalgae and cyanobacteria may overgrow coral reefs; macroalgae may harbor pathogens and fine sediments (sustaining turbidity)	Eutrophication responds to changes in traditional land-use practice and wetland loss through drainage, development of marinas, ports and other anthropogenic recreational activities. The regenerative capacity of the marine ecosystem will be conditional on the capacity of humans to integrate coastal area management.
Overfishing: alteration of trophic pathways and depletion of key functional groups of reef species (Hughes 1994), can result in loss of resiliency, leading to cascading impacts on coral reef habitats (Jackson et al. 2001).	Ultimate pressures are population growth, especially when coupled with increasing affluence and tourism development. Proximal pressure indicators are total landing, fishing effort, resource condition, number pots, location and density of commercial and recreational marinas per town.

Table 2.2 The models components used to explain the geographic distribution of coastal and marine pressure threats.

Model components	General concepts	Supporting literature
Fishery landings	Composite of measurement of different factors that help determine the quality of the economic and natural environments. Expect that fish landings allow a better understanding of local marine resource dependence.	Griffith et al. 2007, Matos-Caraballo et al. 2008
Fishing effort	Factor influencing fishing effort is consumer demand. Consumer demand indirectly affects the marine ecosystem (affecting benthic habitat of the various species of marine biota) as well as intensive fishing of targeted species. Consequently leads to marked declines in catch per unit effort and size of individuals captured.	Cinner et al. 2006, Wilson et al. 2010, 2012, Russ et al. 2005, Jennings et al. 1996
Fishing traps or pots	Both techniques are effective and economically important multi-species fishing gear used widely for harvesting. Of most concern is the derelict or live fish trap, which represents a continuing threat to fish species and benthic habitat. Moreover, traps confined to inshore may also damage benthic habitat through snagging and wave driven movements.	Appeldoorn 2000; Renchen, 2011
Ports, marinas and recreation structures as pier	In recent years many anthropogenic structures have been created along shorelines (jetties, breakwaters and armored shorelines). These alter water flow regimes (limit the dispersal of propagules, modifying patterns of connectivity among populations) and sediments dynamics. Coastline and marine environment can also be adversely affected by groundings, associated operations during cargo handling operations (e.g., oil, fertilizer, antifouling paint) and illegal sanitary and bilgewater discharges (eutrophication, invasive species).	Pister 2009, Airoldi & Beck 2007, Martin et al. 2005, Bulleri & Chapman 2010
Human population density	Human population density, technological efficiency and market pressure have been cited as probable causes of overfishing and drivers of coastal development, including ports and marinas, land-use changes and increased sewage and nutrient loads.	Cinner et al. 2006, Brewer et al. 2012

Table 2. (Continued)

Watersheds & river basin	The impacts of heavy sediment loads and freshwater discharges into the Caribbean from the watershed and river basin to marine environment contribute to increased turbidity in the water column and consequently the partial inhibition of the coral reef zooxanthellae as well the reduction in abundance of seagrass beds.	Restrepo et al. 2006, Loya 1976, Morelock et al. 2001
Sewage systems & outfall pipes	Environmental impact of sewage effluent discharged into the marine environment contains a variety of harmful substances including virus, bacteria and heavy metals. The problem increases when wastewater treatment plants fail to disinfect bacteria and virus from effluent properly. Discharges have the potential to cause eutrophication of coastal waters due to chronic inputs of nutrients and organic matter. Long-term exposure can promote the degradation of sensitive environments such as coral communities and seagrass meadows by chronic exposure to sewage effluent.	Hartel et al. 2007, Islam & Tanaka 2004
Land use	Land-based activities increase turbidity, pollutants and nutrient losses in runoff. Accompany in eroded soils can be pesticides, fecal coliform bacteria, oil and grease and heavy metals, all delivered into coastal water and watershed basin. Moreover, increased use of land for agriculture or urban development can increase erosion of river channel beds and banks, consequently altering the hydrology pattern and increasing non-point source pollution.	Halpern et al. 2008, Tang et al. 2005, Gove et al. 2001, Doyle et al. 2000

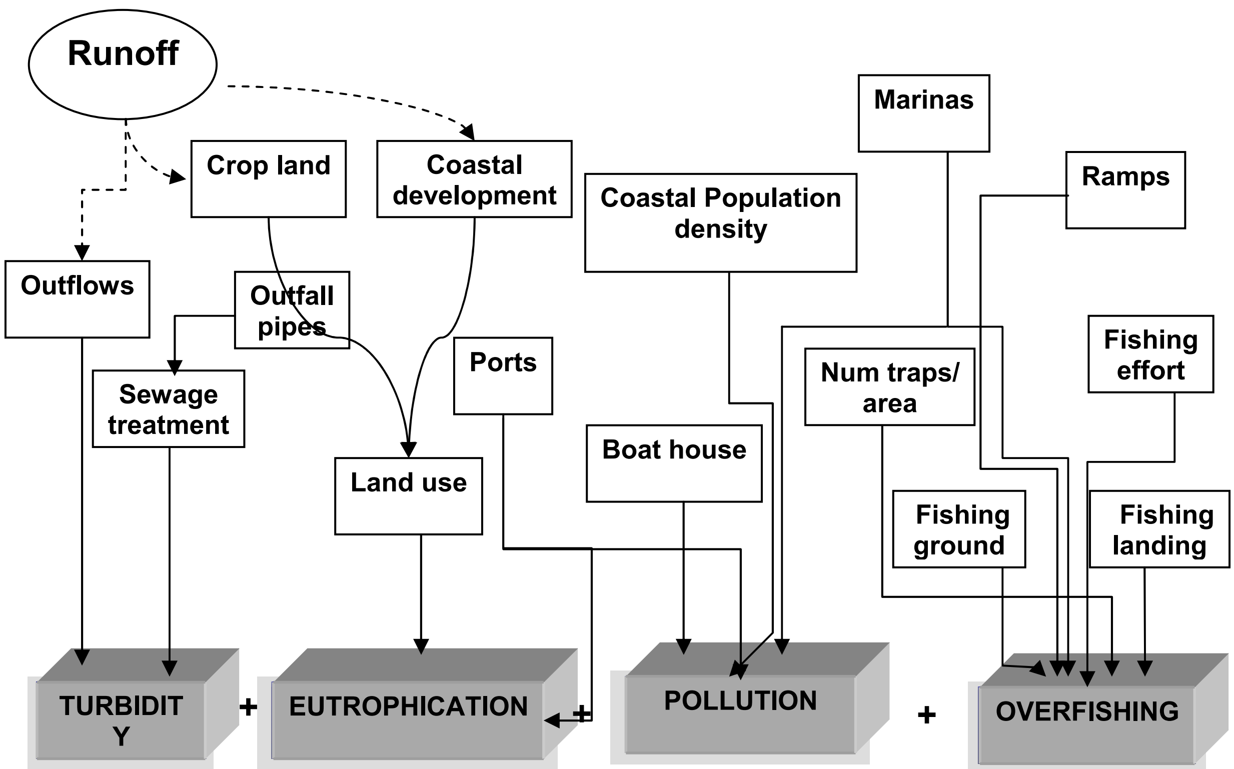


Figure 2.1 General model, significant anthropogenic and socio-economic stressors contributing to coastal and marine threats. Data on number of registered vessels were excluded from this analysis because there is little spatial relation to the coastal area where vessels are used.

Chapter 3. Methods

3.1 Study site

Puerto Rico (Fig. 3.1) is the smallest and eastern most island of the Greater Antilles and contains approximately 501 km of coastal line. In addition to the principal island, Puerto Rico includes the outlying islands of Vieques, Culebra, Culebrita, Mona, Monito and various others keys and isolated islands. To the east is the island of St. Thomas, USVI, which shares the same insular shelf, while to the west across the Mona Channel lies the island of Hispaniola.

For the purpose of analysis, a Coastal Zone delimitation of 1 km inland from the coast was adopted from the Costal Zone Management Program of the Puerto Rico Department Natural and Environmental Resources (PR-DNER). The outer marine extent of the study was defined as the territorial limits of Puerto Rico (9 nautical miles from the coast) except on the insular platform off the west coast where it was extended to included all areas shallower than 100 m. For purposes of analysis, the study area was divided into four geographical provinces: west, north, east, and south, which vary in their geomorphologies, watershed extents and oceanographic conditions.

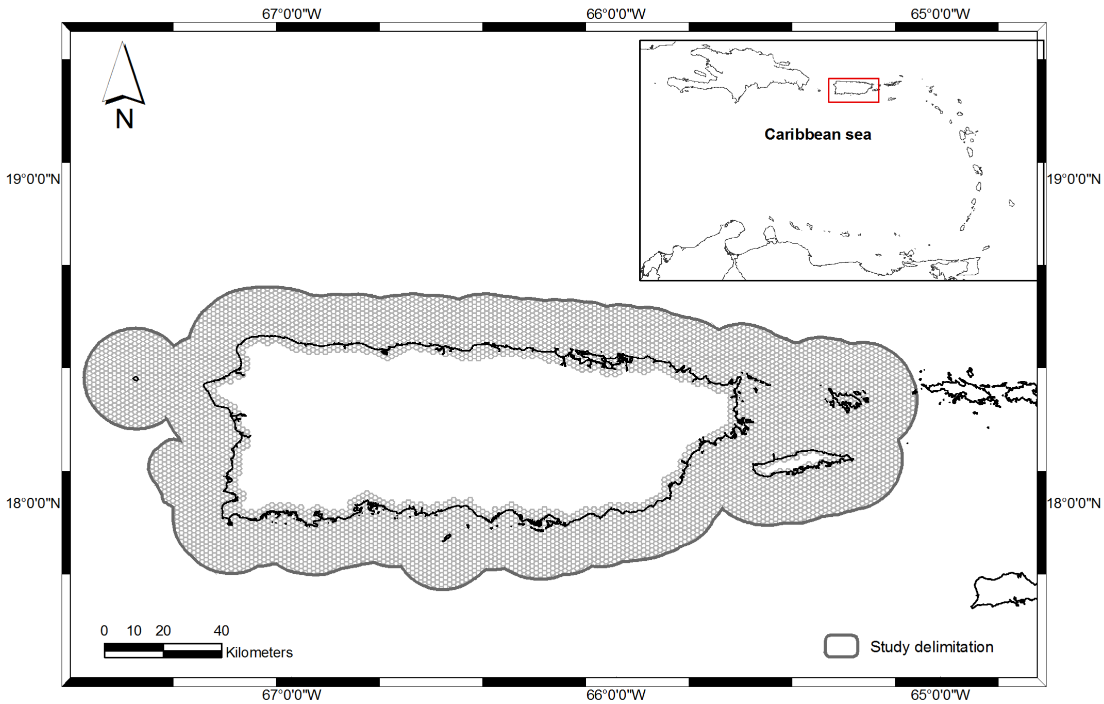


Figure 3.1 Map of the study area showing the coastal and marine habitat, which are divided into four geographic provinces for purposes of analyses: north, east, south and west provinces. Note; that the grey polygon delimits the Puerto Rico jurisdictional marine areas.

3.2 Marine working unit

To represent threats within a spatial context, the whole study area was divided into a series of non-overlapping hexagons or working units Hexagons were used because their shape approximates a circle, which has a low edge-to-area ratio (Miller et al., 2003; Warman et al. 2004), so it is the most efficient spatial packing form (Pressey and Logan, 1998). Hexagon size was selected based on the available benthic habitat map (Kendall et al., 2001), where the mean area of the polygons in the benthic habitat map is 0.46 km^2, with a standard deviation of 2 km^2. The latter value was used as the basis for a hexagon size of 2 km^2 (898.6 m on a side), which resulted in 10,828 units for the region. This size of working unit is small enough for the efficient representation of all benthic habitats, yet large enough to allow for multiple habitats to occur within a single hexagon, and it provided output that had sufficient resolution for the purposes of this study. Each hexagon was assigned a subthreat ‘score’ based on the weighted mean (by area) of the values of that factor (e.g., fisheries catch) occurring within the hexagon (Fig. 3.2). Then a resulting threat score (e.g., Overfishing) was calculated for each hexagon by calculating the weighted mean of all subthreat scores (e.g., fishery landings, number of traps, etc.). Over all threats, each hexagon was then assigned an overall threat score corresponding to the sum of the individual threat scores within that hexagon.

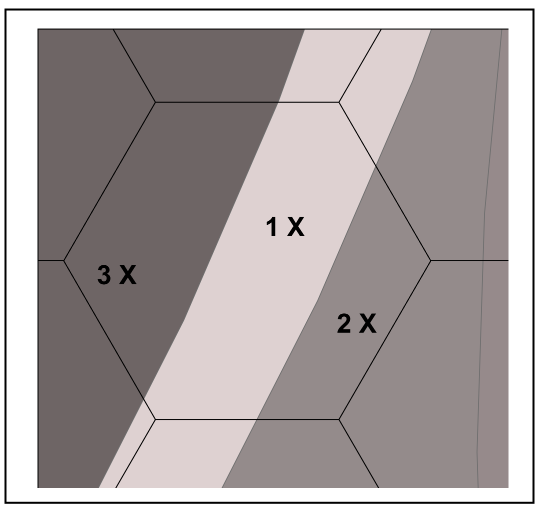


Figure 3.2 Each color inside the hexagon represents different values from the same threat layer. To weigh the stressors to a particular threats inside a working unit, all values were added within the hexagon.

3.3 Analysis of threats: general approach

All geospatial data layers were converted to the same projection (State Plane) and datum (NAD 83) within ArcView 9.1v. Numerical values within each individual data were log[X+1] transformed and rescaled between 0-1 to put them on a single scale as follows:

$$\frac{\log_{10}[X+1]}{\log_{10}[X_{max}+1]}$$

where M_{max} is the highest value for each stressor data set, which would be rescaled to a value of 1. The transformation of data allows for direct comparison among data with dramatically different native scales and units of impact. This also facilitates combination with other data sets to construct specific threat indices.

With two exceptions (traps/area and fishing grounds), the data used to construct threat indices are terrestrial in coverage. To extend the impacts of these data sets into the marine environment, the threats were treated as either a point source (e.g., river or sewage outflow, marinas, ramps, fishery landings) or as emanating from a section of coast (e.g., population density, coastal land use). For each of these data sets, a series of uniformly spaced buffers were created from the point/coast out 10 km into the marine environment, and the value of the threat was linearly decreased by 10 % for each buffer layer from the source until outside the farthest buffer, where the threat value was 10% of the value at the source. This approach was chosen to account for the expectation that the impact of each threat diminishes with distance from the source (Thattai et al., 2002).

All geospatial data sets were then converted to raster form with a uniform 250x250-m cell size within ArcView. When combining data layers into individual stress indicators, each layer was evaluated relative to its importance in comparison to the other data layers toward the modeled threat. Thus, the individual layers were weighted. Details for the weighting schemes used for each of the stress indicators are given in

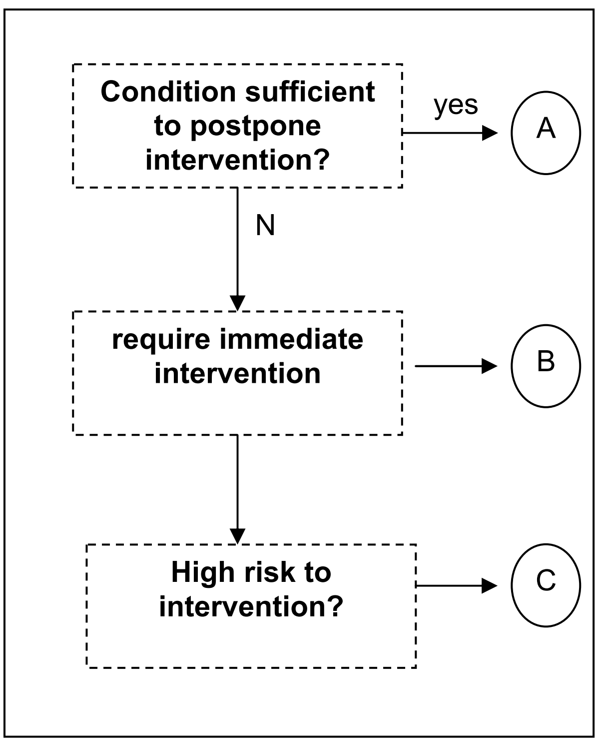
Section 3.5. To construct each of the four threat indices, their different component rasters were added (weighted as necessary) using the raster calculator tool in ArcView. After all raster data sets for a specific threat (e.g., eutrophication) were added, the values were classified on a scale of 1 to 5 ranging from very low impact (1) to very high impact (5). Classification was conducted within ArcView using the natural breaks function.

After the four individual threat indices were developed, these were simply added (using the raster calculator tool) to give an overall threat index. For this, the separate threats were considered of equal importance, so no weighting was used.

3.4 Management approach

A separate assessment of the status of Puerto Rico marine protected areas (MPAs) was made using the threat indices as developed following the procedures of Section 3.3. The boundary of each MPA was used to clip the threat index raster and individual average threat scores were calculated for each protected area. These threat scores were then reclassified within ArcView using the natural breaks function in three categories for management priorities. The three categories were based on the medical concept of triage: i) Not needing immediate intervention, ii) Requiring immediate intervention and iii) Immediate short-term intervention will not help. The purpose of triage as applied to local action management strategies is to prioritize protected areas and to identify those with threat loads requiring immediate management attention (Table 3.1). Depending on the triage priority, this may indicate either that the protected area is a first priority for management (more likely to be successful), or that it will not receive attention either because the area has not degraded sufficiently to merit immediate attention or because the cost of such management action will be too high relative to the conservation benefit (Appeldoorn, 2001). The triage approach performs a brief, focused assessment and assigns the protected area a triage acuity level, which is a proxy measure of how long marine resources can safely wait for environmental regulatory examination and management.

Table 3.1 A conceptual overview of the management triage approach and major decision points. The answers to the questions guide the manager to the correct triage level.



Level of the triage approach

TAP Leve	**Level of management**
A	Marine reserve will not receive human intervention except as needed to retain a particular species.
B	Management treatment would be successful; the resources may improve and this improvement may allow the resources to be categorized to a lower priority in the short period.
C	Resources are seriously impaired; marine reserve should not receive advanced treatment because they are unlikely to be successful.

3.5 Weighting matrix

3.5.1 Turbidity

The modeling of the turbidity threat was based on: i) sediment outflows and ii) sewage plant discharge through marine outfalls. Estimates of sediment outflow were obtained from the US Geological Survey (USGS), which are based on 30m resolution topography using the National Elevation Dataset (NED). The sediment deliveries rate values calculated represent the area of maximum flow accumulation within each hydrological basin, where the total number of basins for Puerto Rico was 132. This produces a dataset of all grid cells within each watershed discharging to a single outflow point (Appendix A). Similarly, sewage outflows were linked to their point of discharge. Sewage treatment outflow locations were gathered from the PR-Autoridad de Acueducto y Alcantarillados (PRASA). Sewage flow was weighted based on level of treatment, where 3= primary, 2= secondary, active sludge removal and use of biolfilter, 1= tertiary, and 1.5 = sewage plants pending construction (since the latter are under construction this weighting acknowledges the future potential for turbidity stress). This weighting system thus considers primary sewage discharge to have a 50% greater impact towards turbidity than secondary sewage discharge. Sewage flow rates were then multiplied by their respective weighting factors before the data were log transformed and standardized to a scale of 0-1.

The point sources of sewage discharge and sedimentary outflow were buffered and decreasing values were assigned as specified in Section 3.3. To create the turbidity threat index, the sewage and outflow raster data sets were added. For this, the sedimentary outflow values were multiplied by a weighting factor of 2, indicating that the impacts of this source toward turbidity were thought to be twice those of sewage discharge. As a final step, the combined values were ranked on a scale of 1 (low impact) to 5 (very high impact) as specified in Section 3.3.

3.5.2 Eutrophication

In this study marine eutrophication is defined as the input of nutrients derived from land use practices and port/ship activity. For Puerto Rico no data on island wide nutrient concentrations currently exist, therefore stressors were modeled for the marine ecosystem using a combination of two layers: i) land use consisting of coastal crop land and coastal development and ii) commercial ports.

Coastal cropland was defined as areas within 1 km of the coast whose USGS land use codes indicated agricultural crop production (not including pasture land or grassland). Crop land was considered a primary source of nutrients due to the application of fertilizers, the natural nutrient content of soils, and the higher potential rate of erosion due to disturbances related to plowing, planting and harvesting. Data were based on the 2000 land use map for Puerto Rico (NOAA, 2005). Similarly, zones of coastal development were derived from those areas within 1 km of the coast whose land use codes indicate the presence of residential or commercial infrastructure. Areas of coastal development were included to account for nutrient inputs resulting from septic tanks and the use of detergents and lawn and garden fertilizers. The impact of coastal development and crop land on the incidence of nutrients was modeled as a function of the relative amount of erosion, considering that sediment laden runoff contributes between 40 and 60% of the total nitrogen stock (Arhonditsis et al., 2000). Nutrient concentrations (P, NH_4^+ and NO_3^-) were assumed to decrease as distance from land increases (Lapointe and Clark, 1992). Data for relative erosion were obtained from the National Oceanic and Atmospheric Administration (NOAA) online; http://ccma.nos.noaa.gov/ecosystems/coralreef/summit.../summit_sea2.aspx. Calculations used a simplified version of the Revised Universal Soil Loss Equation (RUSLE) model (Berner and Berner, 1987). Values of erosion from specific polygons of either developed or crop land were then divided by the area of polygon to standardized the erosion values to the length of coast (width of the polygon is a fixed 1 Km). The sources of erosion rates (contiguous lengths of coast defined as either developed or crop land) were buffered and decreasing values were assigned as specified in Section 3.3 (no log-transformation was required). The resulting vector files were rasterized and were added to form a single raster file, and the resulting cell values were transformed and reclassified on a scale of 1-5 as specified in Section 3.3.

The impact of port activity (by fertilizer, illegal sanitary and bilge water discharges) was modeled as a diffusive plume but with a maximum distance of 3 km spread from the coast (Lapointe and Clark, 1992), where nutrients concentrations decreased and were ranked as follows: 1.5 (high) = 0-2 Km; 1 (medium) = 2.1-3 Km, 0.5 (low) = areas > 3 Km distance. This model assumes that the short-term spread of nutrients (P, NH_4^+ and NO_3^-) is likely to span shorter distances due to cyanobacterial and phytoplankton consumption, in contrast to the long-term spread of suspended sediments. That values do not go to zero is consistent with the expectation that nutrients incorporated into planktonic biomass can still be available at greater distances due to rapid recycling rates (Officer et al., 1982). Ports were then buffered at the above distances and assigned the corresponding values. The resulting vector file was converted to a raster file.

The nutrients delivery through erosion was thought to result in twice the impact as the contribution from ports. Thus, to create the complete eutrophication threat index, the resulting values from these two raster layers (erosion and ports) were added, with the values in the erosion raster multiplied by a weighting factor of 2. As a final step, the combined values were again standardized on a scale of 1 (low impact) to 5 (very high impact, Section 3.3).

3.5.3 Pollution

An endless number anthropogenic substances are responsible for coastal and marine pollution. In this study the focus was on three sources of stressors: i) ports, ii) marinas and "casetas" – houses on stilts built over the water, often with not or inadequate sewer connections , and iii) coastal human population density.

Impact of ports was modeled based on oil leached as a consequence of commercial ship grounding events. First, the area of the port and associated bathymetry were delimited by hand from aerial photographs based on coastal geomorphology. Second, the benthic zone was stratified into four depth strata and these strata were scored on a scale from 1 (low) to 5 (high) based on the probability of grounding impacts by depth, based on Mignucci-Giannoni (1999) and DNER reports of grounding sites for Puerto Rico (N = 12), where: 5 = 1-6 m depth; 4.5 = 7-12 m depth; 2.5 = 13-18 m depth; 1 = >19 m depth, the potential for grounding and oil release will decrease with depth. This weighting system thus considers grounding events in the shallowest stratum to have a 50% higher probability than at depths between 13-18 m and more 75% greater probability of impact toward depth >19m. Third, to limit the spatial scale of impact, a buffer zone of 2 km per each depth strata was developed. The 2 km radius was chosen as a maximum distance that heavy oil might disperse along the bottom from where the grounding occurred. This was based on the case of the Morris J. Berman (1994) grounding event, where the energy of the surf zone did not move oil significant distances from the source (Beegle-Krause et al., 2006). To simplify the calculation, buffering was constructed around the maximum depth contour of each depth zone (i.e., 6 m, 11m, etc.) Scores, 1-5 as above, were applied to each depth-contour buffer. The respective vector polygons were converted to raster files and then added to form a single raster data set. Cell values were then were log transformed, reclassified and standardized to a scale of 0-1, as in Section 3.3.

Pollution from marinas and "casetas" largely comes from the use of Irgarol (an ingredient in antifouling paint) and discharged sewage (pathogens associated with fecal matter) because: i) "casetas" are not connected to the sewage lines and 2) the lines are below the water table. These are used as the main source of anthropogenic stressors into marine ecosystems. For modeling proposes there is a rapid decline of antifouling paint exposure from the marinas within 1.7-2.0 Km of the most concentrated source (Gardinali et al., 2004), Similarly, the concentration of pathogens is highest inshore (Milliken and Lee, 1990). The spatial extent of "casetas" along the coast (as a source of pathogens) were hand digitized from 2007 areal photographs. To account for dispersal and declining concentration of both toxic substances and pathogens with distance for their respective sources, the same buffering and weighting system used for ports (above) was applied to marinas and “casetas”, respectively. In terms of impact, the two sources were considered of equal weight.

Human population density was incorporated in the pollution index to account for the diffuse array of potential impacts ranging from the runoff from car-based hydrocarbons emanating from land to small oil and gas spills from recreational boating activities across the shelf. The goal was to define a human footprint for sections along the shelf. Human population density was calculated as the total number of inhabitants within the boundary of each coastal municipality, divided by the total coral reef cover area on the insular shelf directly offshore. The area of reef was taken as the total area of consolidated substrata as delineated in the NOAA map for Puerto Rico and the US Virgin Islands. This approach accounted both for the habitat most likely to contain corals and for variations in the width of the insular shelf off each municipality. Human population data were obtained from the Gridded Population of the World (GPW), v3 (Balk and Yetman, 2004) Online: http://sedac.ciesin.columbia.edu/downloads/docs/gpw-v3/gisn-24_web_gpwannex.pdf. Grid cells of population density (squared kilometer) were pooled by municipal boundaries and averaged over each municipality. These values were then divided by the reef areas across the immediately adjacent shelf using lines running along the four cardinal directions (depending on coast), emanated from each municipal boundary and extending to the edge of the insular shelf (50 m depth contour).

The results then were log transformed and standardized to a scale of 0-1 (Section 3.3.) and converted to raster. To create the pollution threat index, all raster data sets (marinas, “casetas”, population density) were added with equal weight. As a final step, the resulting combined values were reclassified and ranked on a scale of 1 (low impact) to 5 (very high impact, Section 3.3).

3.5.4 Overfishing

Overfishing can occur through a combination of human and natural disturbances that result in a reduction in the rate of productivity, either on the population or ecosystem scale (Hughes, 1994). These disturbances vary in their intensity of impact and in their spatial distribution across the seascape (Halpern et al., 2008). Seascape-level threats from overfishing were evaluated on the basis of i) total commercial fishery landings, ii) commercial fishing effort, iii) number of traps per fishing zones, and iv) recreational fishing, using the geographic location of marinas and boat ramp densities per squared kilometer (Appendix B).

We evaluated total fishery landings per municipality as indicator of artisanal fishing stress on seascape ecosystems with a two-step process. First, 2011 commercial fishery landings data from Matos-Caraballo (2012) were scored on a range from 0 to 1, based on the Work Environment Index formula (Heintz et al., 2005):

$$I_i = \frac{X_i - \min\{X\}}{\max\{X\} - \min\{X\}} \times S$$

Where I_i is the indicator value for the municipality *i,* X_i is the 2011 landings value for that municipality *i*, $\min\{X\}$ is the minimum value for landings across all municipalities, $\max\{X\}$ is the maximum value for landings across all municipalities, and *S* is the maximum value of the index (in this case, 1). Values of I_i close to 0 indicate less dependency in terms of the overall landings, with a value of 1 indicating the highest fishing dependency score (Appendix C). Then, the I*i* values were standardized by divided them by the length of coast for each respective municipality. Second, to limit the spatial scale of seascape impact, a buffer was applied, where the landings impact was reduced with distance from shore. For this, two buffer zones were developed, one of 10 nautical miles for the north coast, which has a uniformly narrow shelf, and one of 20 nautical miles for the rest of the island. The buffers were chosen as the average distance fishermen would travel to reach outer fishing banks and deep-water snappers, width 20 nautical miles being the maximum distance most fisherman would travel (Matos-Caraballo, 2012).

Fishing effort was modeled using the average hours used weekly by commercial fishers. The model assumes that the hours fishermen are engaged in fishing activities are a direct indicator of the potential for the removal of prey from the ecosystem. A value here represents the average hours dedicated weekly by commercial fishers to different fishing tasks categorize per municipality; data were obtained from Griffin et al., 2007. Resulting values were collaborated using PR-DNER fisheries laboratory reports (Matos-Caraballo and Agar, 2011). The same buffering system used for fishery landings (above) was applied here. The resulting values were then log transformed, reclassified and standardized to a scale of 0-1, as in Section 3.3.

A more direct geographic indicator of fishing impact was developed by modeling the distribution and density of traps and pots. Data come from two sources. Agar et al. (2005) surveyed of use of traps in the US Caribbean fishery to develop a database on fishing activity. Their survey was administered to one hundred randomly selected trap fishermen (USVI and PR), estimated to constitute 25% of the activate fisherman. These data (number of traps) were then used to divide the shelf into 69 polygons. The number of traps/area km (polygon) were then classified into three categories (Natural Break): 1 = low (0.040853-0.576740), 2 = medium (0.576741-1.504589), and 3 = high (1.504590-2.785460)trap density. The second source was Ojeda-Serrano et al. (2007), who also mapped fishing grounds for Puerto Rico. From that study the shelf area of Puerto Rico was divided into nine polygons based on a combination of geomorphology and gaps in the distribution of fishing locations around the island. Each polygon was scored based on area, where 1 = low, 2 = medium, 3 = high ground area. Both data sets were converted to raster and added. Finally cell values from the combined raster were log transformed, reclassified and standardized to a scale of 0-1, as in Section 3.3.

To account for recreational fishing impacts, the stress indicator used a combination of two layers: i) the geographic location of marinas, which focuses on larger vessels and ii) the density of ramps or piers (within 1,000 square Km), which focuses on smaller vessels (i.e., <15m length overall). Fishing activities related to marinas were stratified by depth, where threats are highest in areas shallower than 50 m and at depth exceeding 100 m (Grober-Dunsmore et al., 2008). Depth were stratified and ranked as follows: 1 (high) = 0-50 m; 0.5 (medium) = 51-100m; 1 (high) = >100 m depth. Marinas were then buffered at the above range of depths and assigned the corresponding values. The resulting cell values were then log transformed, reclassified and standardized to a scale of 0-1, as in Section 3.3.

Initial information on the locations of ramps and piers was provided by the Marine Resources Division of the PR-DNER. To generate an entire island inventory, the location of all ramp and pier structures were georeference from 2010 orthorectified aerial photos. Due to the high number of such structures, the index was expressed as the density (number/square Km) of ramps and piers combined along the coast. The impact of recreational fishing activity associated with ramps and piers was modeled to decrease with distance from shore. Recreational fishing intensity was ranked as follows: 1.5 (high) = 0-2 Km; 1 (medium) = 2.1-3 Km, 0.5 (low) = areas > 3 Km distance from the coast. Ramps and piers were then buffered at the above distances and assigned the corresponding values. These were then multiplied by the corresponding density score. Then cell values were log transformed, reclassified and standardized to a scale of 0-1, as in Section 3.3.

To create the complete overfishing threat index, the values from the above raster layers were added, with the values in the artisanal fishery landings raster multiplied by a weighting factor of 2. As a final step, the combined values were again standardized on a scale of 1 (low impact) to 5 (very high impact, Section 3.3).

Chapter 4. Results

4.1 Predicted threat impact scores

Turbidity

Turbidity (T) scores ranged from 0.09 to 2.98. These were then classified into five categories of impact, ranging from low (T = 1) to very high (T = 5). Over a half (52.03 %) of the marine area of Puerto Rico has low to medium impact scores ($\leq$2, Fig 4.1), with a small fraction (14.4 %, ~ 1924.199 Km^2) experiencing very high impact (T=5). Ecosystems with high to very high impact scores (T $\geq$ 4) include: bank-shelf/escarpment (81.97 %, ~902.782 Km^2), lagoon (9.9 %, ~109.25 Km^2), and with relatively small areas of shoreline-intertidal (5.6 %, ~61.23 Km^2), reef-crest (1 %, ~11.30 Km^2), backreef (0.8 %, ~9.74 Km^2) and forereef (0.6 %, ~7.02 Km^2).

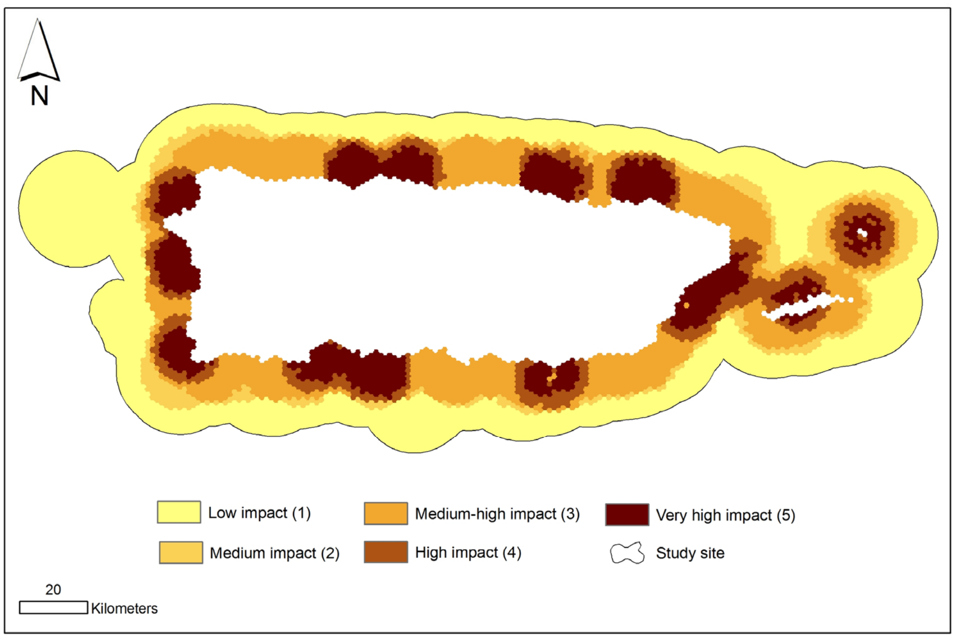


Figure 4.1 Turbidity impact across Puerto Rico coastal and marine ecosystems categorized into five classes.

Eutrophication

Eutrophication (E) scores ranged from 0.01 to 2.75, which were classified into five categories. Over an three quarters (80.01 %) of the marine extent of Puerto Rico has low to medium impact scores ($E \leq 2$, Fig 4.2), with a small fraction (7.24 %, ~ 964.89 Km^2) experiencing high to very high impact ($E \geq 4$). Ecosystems with high to very highest predicted eutrophication impact scores include: bank-shelf/escarpment (70 %, ~223.08 Km^2), lagoon (20.41 %, ~65.12 Km^2), and with relatively small areas of shoreline-intertidal (4 %, ~12.69 Km^2) reef-crest (3 %, ~9.50 Km^2), backreef (1.6 %, ~5.25 Km^2) and forereef (1 %, ~3.35 Km^2).

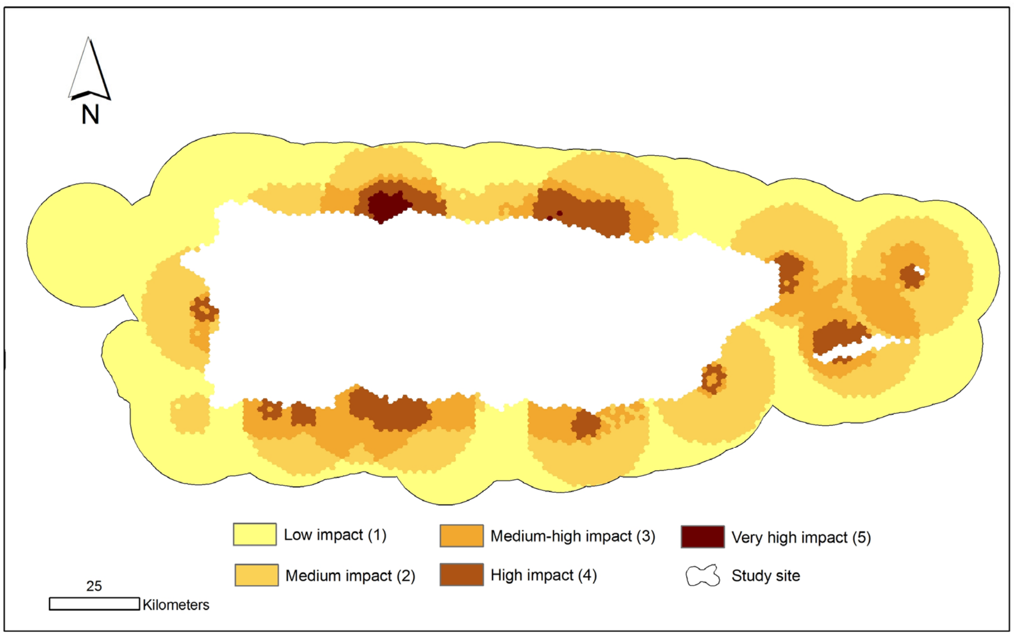


Figure 4.2 Eutrophication impact across Puerto Rico coastal and marine ecosystems categorized into five classes.

Pollution

Pollution (P) scores ranged from 0.001 to 2.48. After categorizing these into five classes, over a half (70.14 %) of the marine area Puerto Rico has low to medium impact scores (P<2, Fig 4.3), with a small fraction (2 %, ~ 249.93 Km^2) experiencing very high impact (P> 5). Ecosystems with high to very highest pollution impact scores include: bank-shelf/escarpment (80 %, ~810.75 Km^2), lagoon (11.7 %, ~119.03 Km^2), with relatively small areas of shoreline-intertidal (5 %, ~49.94 Km^2), , reef-crest (1 %, ~14.53 Km^2), and backreef (1 %, ~13.29 Km^2) and forereef (0.88 %, ~9.01 Km^2).

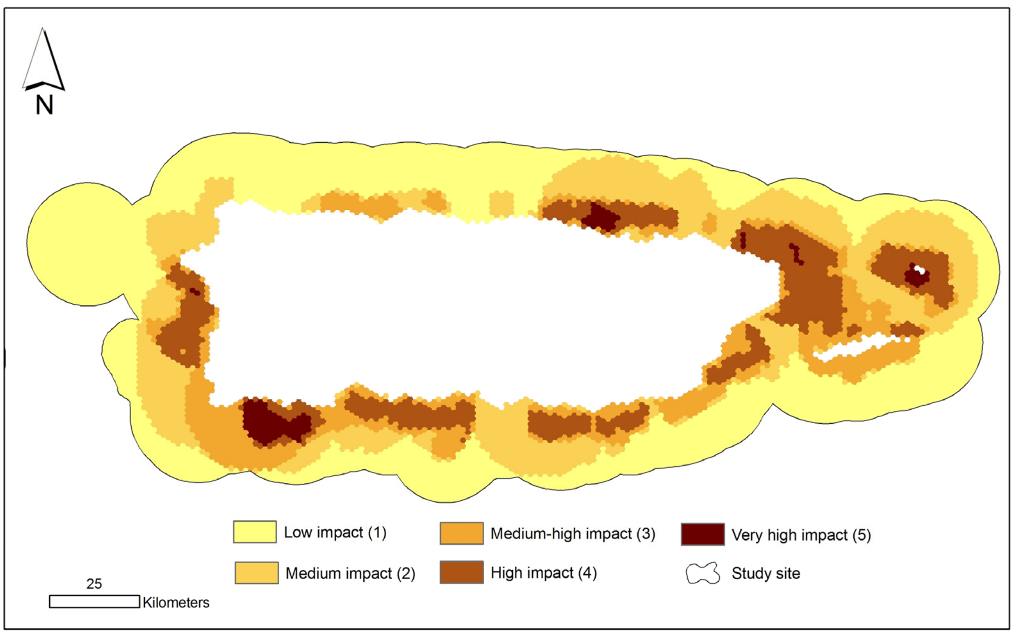


Figure 4.3 Pollution impact across Puerto Rico coastal and marine ecosystems categorized into five classes.

Overfishing

Overfishing (F) scores ranging from 0.52 to 5.35. After categorizing them into five classes 39.52 % within the Puerto Rico marine waters experienced high to very highest impact scores (F $\geq$ 4, ~5264.224 Km^2, Fig 4.4), with a small fraction (19.69 %, ~ 2622.852 Km^2) experiencing low impact (F = 1). Ecosystems with high to very highest overfishing impact scores include: pelagic (~1887.86 Km^2), bank-shelf/escarpment (88.57 %, ~1651 Km^2), lagoon (6.17 %, ~115.16 Km^2), with relatively small areas of shoreline intertidal (3 %, ~54.87 Km^2), backreef (1 %, ~18.63 Km^2). forereef (0.7 %, ~13.94 Km^2), reef-crest (0.50 %, ~9.39 Km^2).

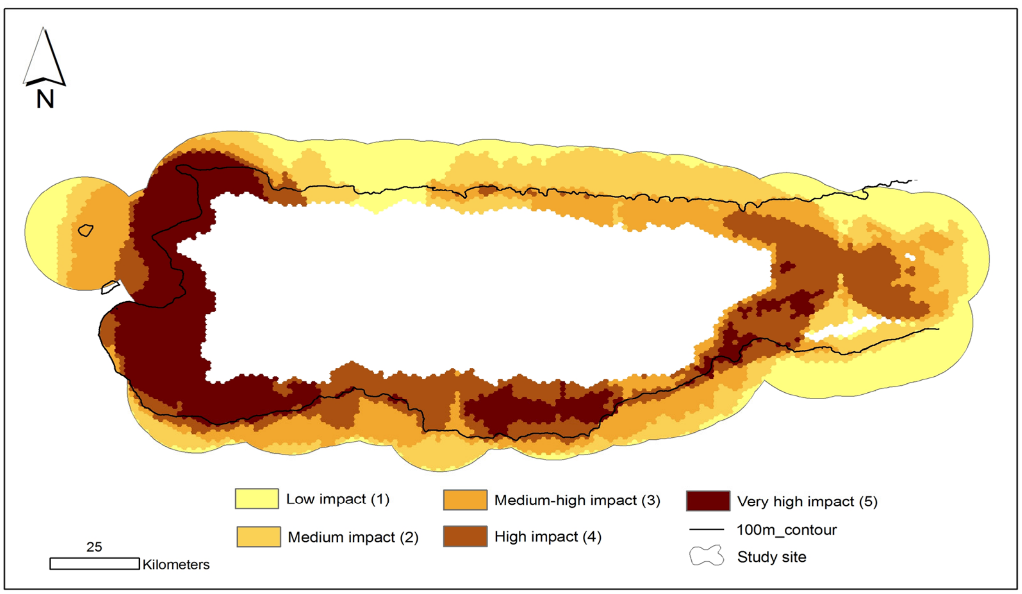


Figure 4.4 Overfishing impact across Puerto Rico coastal and marine ecosystems categorized into five classes. The solid line delineated the pelagic zones (100 m contour).

4.2 Overalls threat cumulative impact

The pooling impact score from these four threats demonstrated that the entire study area (~13322.11 Km^2) is being impacted by one or more threats (Table 4.1). While of the four threats modeled in this study, the most pervasive direct socioeconomic threat to coastal and marine habitats is overfishing. Overall 40.68 % of the marine planning unit experienced high to very highest impact scores ($\geq$ 4, ~5419.59 Km^2, Fig 4.5), were 17.8 %, ~ 2375.70 Km^2 experiencing low impact (F = 1). Within the Puerto Rico shelf the benthic ecosystems most threaten was bank-shelf/escarpment ($\geq$ 5, 84.91 %, ~1413.24 Km^2) followed by lagoon (8.30 %, ~138.21 Km^2), with relatively small areas of shoreline intertidal (4.2 %, ~70.15 Km^2), reef-crest (1%, ~15.89 Km^2), backreef (1 %, ~15.03 Km^2) and forereef (0.7 %, ~10.88 Km^2).

Comparing the mean accumulative scores among the four geographic provinces, the western province obtain the highest collectively scored with 4.7 $\pm$ 0.019 (S.E). It was followed by the southern province, with 4.55 $\pm$ 0.016. The eastern province obtained a 4.43 $\pm$ 0.015, while the northern province obtained a 4.35 $\pm$ 0.019 respectively. A variance on ranks analysis demonstrated that at the regional level there were significant differences in the median values among the four geographical provinces (Kruskal-Wallis H Test; p<0.001, d.f.=3, 218.94). To isolate the geographic provinces the Dunn's method for multiple comparison procedure (p=0.05) was used, which showed that eastern provinces did not differ significantly from the northern province (p>0.05, Table 4.2), which could be significant in terms of management. However, the western province had significantly higher cumulative threat scores versus the east, south and north provinces.

In terms of management triage approach, 15 protected areas manage by DNER have a triage level = 3 (C; high risk to intervention), with four having level = 2 (B; require immediate intervention) and one ranking = 1(A; not needing immediate intervention). The remaining protected areas scored in two or more triage levels (Table 4.3, Fig. 4.6).

Table 4.1 Classification of area by percent for individual and cumulative stressor scores

	Study area (Km^2) 13,322	Threat Scores				
		Low	Medium	Medium -High	High	Very High
Overall scores		17.8	23.3	18.2	22.8	17.8
Turbidity		44.4	7.6	23.6	9.9	14.4
Eutrophication		48.6	31.2	12.7	6.7	1.0
Pollution		45.7	24.5	14.3	13.6	1.9
Overfishing		19.7	20.4	20.3	19.7	19.8

Table 4.2 All geographic provinces multiple comparison procedures, Dunn's Method. Significance is indicated by (*) p-values < 0.05

Provinces	Differences on ranks	Q
South vs East	894.437	13.912*
South vs North	770.239	11.390*
South vs West	563.336	8.057*
West vs East	331.101	4.987*
West vs North	206.903	2.972*
North vs East	124.198	1.942

Table 4.3 Overview of management triage levels for marine protected areas managed by the PR Department of Natural and Environmental Resources. : A) Not needing immediate intervention B) Requires immediate intervention and C) High risk to intervention.

Province	Natural Area Protected	Triage Level
East	NR-Canal Luis Peña	**C**
North east	NR-Corredor Ecologico Noreste (Fajardo)	**C**
North	Piñones State Forest	**C**
North	NR-Cienaga las Cucharillas	**C**
North	MR- Arrecife de Isla Verde	**C**
North	NR-Canal Martín Peña	**C**
West	MR-Tres Palmas	**C**
West	Boquerón Wildlife Refuge	**C**
West	NR-Laguna de Joyuda	**C**
South	Guanica State Forest (Marine Extention)	**C**
South	NR-Punta Cucharas	**C**
South	NR-JOBOS Bay	**C**
South	NR-Arrecifes de Guayama	**C**
East	NR-Humacao	**C**
East	Roosevelt Road	**C**
West	MR-Desecheo	**A**
South	NR-Punta Petrona	**B**
South	NR-Isla Caja de Muerto	**B**
Southeast	NR-Punta Vientos	**B**
North	NR-Laguna Tortuguero	**B**
	Others	**Multiple levels**

Acronym: NR: Natural Reserves; MR:Marine Reserves

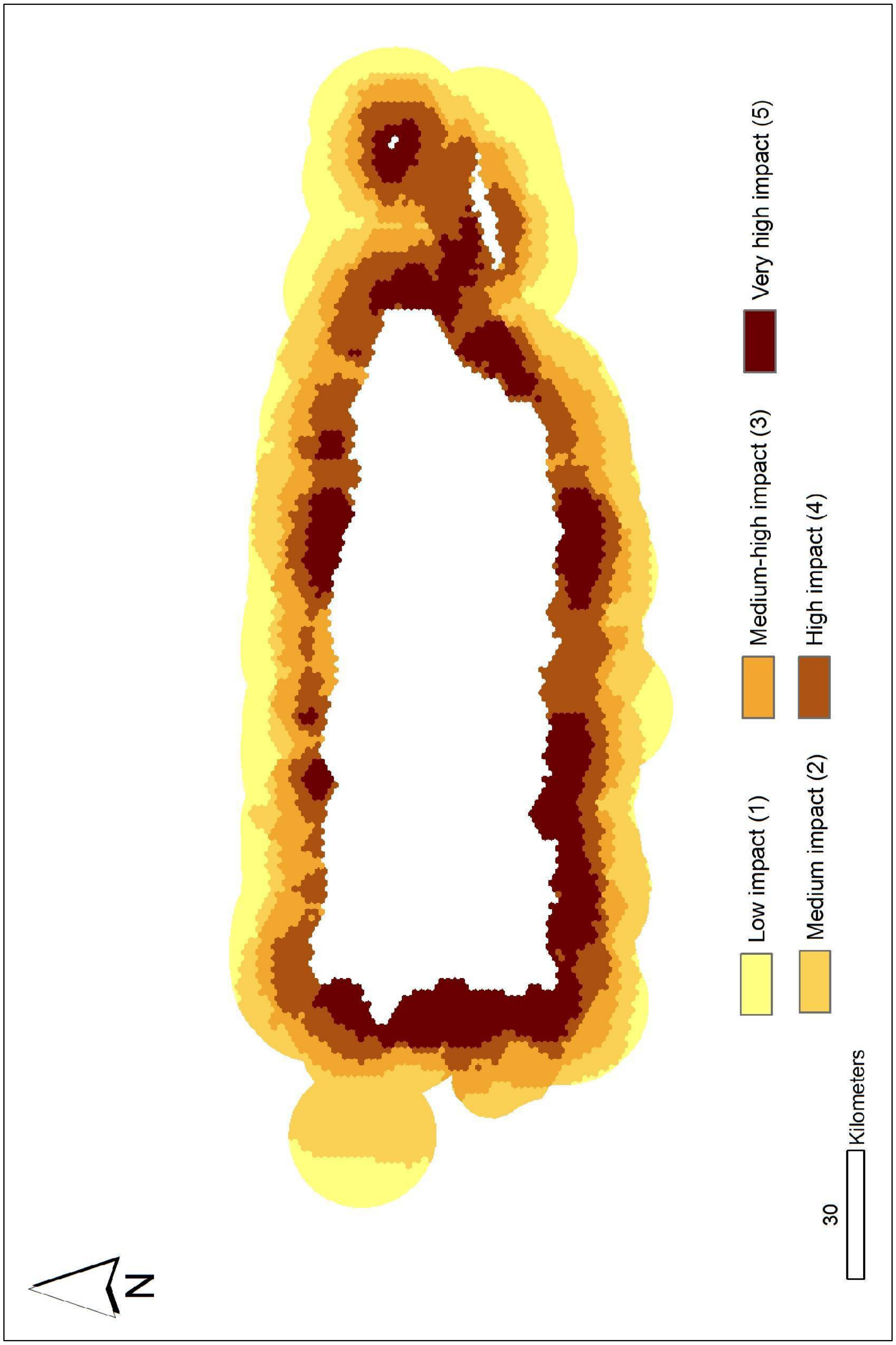


Figure 4.5 Overall pooled threats for Puerto Rico coastal and marine benthic ecosystems, using natural breaks classes. Western provinces have the higher scores for cumulative threat index.

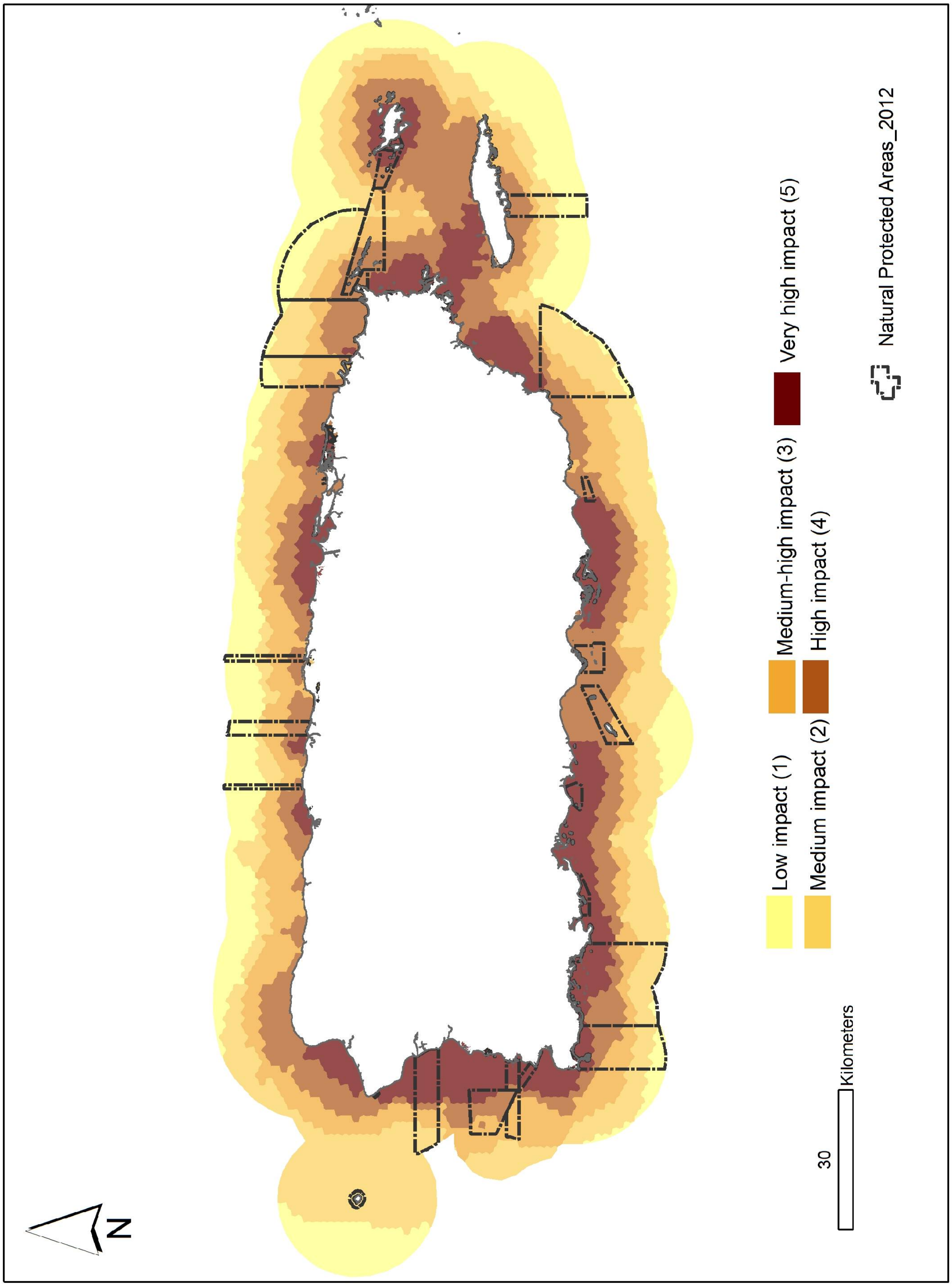


Figure 4.6 Geographic locations of PR Marinel Protected Areas manage by the Department of Natural and Environmental Resources. Federal and PR Conservation Trust MPAs are not shown.

Chapter 5. Discussion

There is no direct way to quantify how anthropogenic drivers threaten coastal and marine ecosystems in a synoptic manner that would facilitate management decisions. While remote sensing techniques may provide such information in the future, at present only indirect methods are available. Here we use available data to develop proxy indicators that may reflect these threats and their cumulative impact in order to provide a more accurate and robust framework for management assessment and action. Overall, our study highlights the cumulative impact of multiple anthropogenic activities on coastal and marine ecosystems. The ability to model cumulative impacts, both with and across stressors in a manner that can be represented geographically is of great importance because they have direct application to management through the mapping of valid coastal and marine vulnerability indices (McLaughlin et al., 2002; Cooper and McLaughlin, 1998). Nevertheless, estimating cumulative impacts across widely different types of stressors entails a number of assumptions on their relative importance and their respective geographic scales, so the results of any attempt to reduce this complexity to a few, or even a single, index will be open to interpretation. Thus, the validity of any such indices should be assessed before their general application can be recommended.

To assess the validity of the predicted stress levels, model results were compared to the actual state of ecosystem condition in selected areas where such determinations were made (Table 5.1). On the large scale, turbidity threats were found to have relatively high spatial extents and predicted cumulative impacts (Fig. 4.1). The spatial distribution of turbidity impacts is positively spatially correlated to the major rivers in Puerto Rico. Therefore, the north coasts from Arecibo throughout Loiza are the most significantly threatened area. This is expected since the north shore is going to be influenced by (1) the more abundant and frequent precipitation in northern Puerto Rico, resulting in more persistent and higher flow rates of streams that discharged into the Atlantic Ocean (Warne et al., 2005), and that (2) these rivers will support a substantial sediment loading (Morelock, 1984). In contrast, turbidity within the southern geographic province is significant lower due to reduced river influence and sediment loadings. Additionally, the south coast generally has low wave energy, and this would keep the sediments near the coast and more localized (Acevedo et al. 1989; Warne et al. 2005). These trends are reflected in the result in lower turbidity scores within the southern province. The western province turbidity values are more spread where allowing the sediments move offshore.

Table 5.1 Summary of coastal and marine degraded areas in Puerto Rico.

Location	Threats drivers to Puerto Rico marine and coastal habitats	Study Scores
Northern province	Dramatic decline in hard bottom and coral cover (Goenaga et al., 1979) due to silt-laden water and currents (Kaye, 1959) also significant amount of terrigenous sediments from river discharge (Morelok et al., 2000) makes bottoms unstable for coral reef development.	
Río Grande, Luquillo & Fajardo (Pta Iglesias, Pta San Agustin)	Significantly inshore reef degraded due to high sedimentation and water turbidity (Goenaga, 1989; Herdandez, 2000).	3-5
Río Espíritu Santo	Benthic habitat degradation by coastal development and sewage discharge that promote eutrophication (Hernandez-Delgado, 1995).	3
San Juan Bay Estuarine System	The estuary water quality index rated poor potentially caused by sewage discharge and non-point sources of pollution (Seguinot-Barbosa et al., 1999), although the area has been significantly influenced by higher concentrations of metals in sediments, fish and crab tissue (Otero et al., 2011).	5
Eastern province	Several anthropogenic threats are impacting the coastal and marine habitat resources in this area, including coastal development, recreational structures (e.g. marinas), and overfishing (Appeldoorn, 1987; García-Sais et al., 2003).	
Yabucoa Bay (Cayo Sargento)	Degradation to coral reef habitat and poor coral cover - has been attributed to a proliferation of fleshy and filamentous algae, resulting from eutrophication and sedimentation (Hernandez-Delgado and Alicea-Rodriguez, 1993).	5
Culebra / Vieques watershed	Significant amount of sediments and runoff flowing to the shoreline (Ramos-Scharrón, 2009).	5/ 3-5
Fajardo and Humacao	Anthropogenic activities associated to the operation of several marinas (Hernandez-Delgado et al., 1995; Hernandez-Delgado and Alicea-Rodriguez, 1993), inducing turbidity and eutrophication.	4-5
Arrecife de La Cordillera	Coral reef fishes overfishing threats by recreational activities (Hernandez-Delgado 1994).	4

Southern province	In general benthic communities have been chronically affected by turbidity, sedimentation input and industrial development (Goenaga and Boulon 1992; Tetra Tech., 1991).	
Ponce and Jobos Bay	Damage and threat to coral reef associated to anchoring, turbidity and sedimentation (Acevedo, 1986; Ramirez, 1992; García- Sais and Castro, 1997).	5
Peñuelas	The area has been influenced by dredging and industrial activities, where auspicious impulse of eutrophication, high turbidity and pollution (Goenaga and Cintron 1979 ; Acevedo, 1986).	5
Western province	Dramatic decline in grouper fisheries resources (Matos-Caraballo, 1997, 2000). Overall, threats are associate to non-point source of pollution (Bonkosky et al., 2009), however condition of benthic habitat and species associations improves the farther the are from the coast (Hernandez-Delgado, 2000, Loya, 1976).	
Cabo Rojo, Mayaguez	Several threats are affecting western inshore and fringing reefs (Goenaga and Boulon 1992; Morelock, 2001). The most significant drivers were land-derived activities that produce frequent non-point source pollution pulses (Hernandez-Delgado, 2000)	4-5
Desecheo and Offshore benthic habitat	Offshore reefs, such as Desecheo (75% living coral, Hernandez-Delgado, 2000), represented one of the areas with the fewest anthropogenic threats. Good results were documented for Manchas Interiores, Tourmaline, Escollo Rodriguez, Cayo Ron and Algarrobo, indicating that the farther away from the coast, the better the condition of benthic habitat (Hernandez-Delgado, 2000; Loya, 1976).	2
Rincón	South coast of Rincon (Ensenada county subdivision) has recurring pulses highly turbid coastal waters due to runoff; bethic habitat impacted by anthropogenic sources of pollution and nonpoint source fecal contamination (Norat et al, 2013).	5

Contrary to turbidity, eutrophication threats showed small spatial extents and low predicted cumulative impacts (Fig. 4.2). The eutrophication threats are relatively high near agriculture lands with the exception of the eastern province, where, predicted impact scores may be higher than anticipated because many anthropogenic stressors are not readily observable. Overall, the impacts of eutrophication on coastal and marine areas correspond to the lowest scores among the four threats modeled in this study. However the spatial distribution and concentration on nutrients will be influenced by benthic filter feeding communities as natural control where by prevalence on shallow water (Officer et al., 1982).

The spatial distribution of pollution threats estimated among the four geographic provinces shows that the eastern and southern areas are the most threatened, along with the areas adjacent to San Juan off the north coast. The predicted impact scores are relatively high in areas associated to coastal boating structures (e.g., docks, boat ramps) and tourism facilities (e.g., "casetas" La Parguera). These human recreational activities result in illegal sewages discharges and accidental oil spills (Otero and Carrubba, 2007; Jeffrey et al., 2010). The predicted threat distribution of pollution was also restricted to the insular shelf (<50m, Fig. 4.3).

Predicted threats due to overfishing had the broadest distribution across the insular shelf, suggesting that this is the most pervasive anthropogenic stress. Still, overfishing impacts are highly heterogeneous across the geographic provinces. Among the four provinces, 42 % of high predicted cumulative impact occurs in the western province (Fig. 4.4).

In general, predicted anthropogenic impacts on coastal and marine ecosystems shows spatial heterogeneity (Fig. 4.5), yet every square kilometer is affected by some stressors. Over a third (40.6%) of the study area experienced high to very high impact scores (~5,419 Km^2), while a much smaller fraction (18%) and area (~2,375 Km^2) experienced low (1-2) impact scores. Most of the highest predicted cumulative impacts occur over areas of the insular shelf, where ecologically important and diverse coral ecosystems occur. Large areas of highly predicted impact (scores 4-5) occur in the west and south provinces (Fig. 4.5). These results suggest that almost half of all coral reefs experience medium to high impact.

Specific benthic habitats within the insular shelf (<50 m depth) support different biotic assemblages of fishes (or ontogentic stages) and invertebrates. The predicted results of stress distributions showed variations among the individual impact scores indicating differential threats to unique marine assemblages. Among these habitats the most affected are coastal lagoons including estuary and intertidal zones that support many crustaceans and bivalves, which make up an important component of the fisheries (Edwards, 1978; Stoner, 1988). These areas are naturally more turbid and nutrient rich even in the absence of anthropogenic threats, so assessing the scale of such threats remains difficult using in-situ obsevations. Nevertheless, outflows and relative erosion from land use practices coupled with wind and tidal currents that mix throughout the water column will increase turbidity and disperse fine sediments over the lagoon floor, and may also transport pollutants such as fecal coliform (Steets et al., 2003) from inflowing streams and rivers. Identifying such threats is critical for management. Coastal lagoons with sediments rates faster than sea level rise will fill in and be short-lived, as observed in the case of the salt flat marshes in Guanica Bay (i.e., Río Loco discharge) and in Cabo Rojo.

The majority of low impact areas occur offshore in the western and northern provinces, especially in areas at greater than 100 m depth. While these areas are still subject to fishing impacts, regulations such as seasonal restrictions or permanent bag limits as well as more extreme weather conditions limit human access. However, the analysis did not account for illegal and unreported fishing nor unreported sewage or land clean discharging, which may be extensive in the Caribbean. Seamount and pelagic deep-water ecosystems had the lowest scores (<10% of these ecosystems), partly because of the lower vulnerability of these ecosystems to most natural and anthropogenic stressors.

One of the driving factors of this study was to produce threat maps for Puerto Rico that better represented the local situation and thus have greater management utility than those previously produced by the World Resources Institute in their Reefs-at-Risk (R@R) assessment, which was based on Caribbean-wide spatial data sets. The R@R assessment estimated sedimentation and pollution from inland sources as threatening over 60 % of the coastal and marine ecosystems. In comparison, the results of the present model show more nuanced spatial distributions for these threats both in terms of spatial resolution and spatial variability. Additionally, relative threat levels seem to be lower. Because the layers generated during this study represent these threats on a finer scale, resource managers can identify potential hot spots where these treats are particularly high and target those areas for remediation. Most of the difference between studies is due to the effects of turbidity. Overall, the level and distribution of pollution threats predicted in this study is comparable to that predicted by

R@R, with both studies accounting for low cumulative threat levels within marine and coastal ecosystems.

Both R@R and this study found overfishing to be the most pervasive direct anthropogenic threat to coastal and marine ecosystems. Yet, the R@R assessment overestimates the potential overfishing threat, particularly within the northern geographic province. In the R@R analysis 84% of the area was scored as having high impact levels, while in the present study these high impact levels only accounted for 60% of the area around Puerto Rico.

The developed cumulative stress indices can be used to assess the spatial distributions of the threats and their intensity relative to the management challenges faced by the Puerto Rico government, particularly under the management mandates of the Department of Natural and Environmental Resources (DNER). Not all sites are threatened by all stressors. One of the most obvious applications in this context is to compare predicted cumulative stresses with the distribution of the existing reserves (no-take marine reserves, natural reserves, forest reserves), which represent areas of high management priority. Overall the spatial distribution of threat impacts is heterogeneous, with the exception of the Desecheo Island Marine Reserve, which was less threatened (Fig. 4.6). Cumulative threats were generally higher along the south and west coasts in comparison to the east and north coasts, and this applies generally to reserve areas as well. This suggests that management of reserve areas must occur at a larger spatial scale, as many of the anthropogenic stressors may occur on the scale of the coast. However, there is significant patchiness to this general pattern. For example, the marine reserve at Canal Luis Peña on Culebra Island is among the areas in the east coast with the greatest exposure to multiple anthropogenic stressors, in contrast to the Vieques Natural Reserve, where turbidity is the only significant threat and overall threat scores are otherwise lower. While there are gaps among the high threat areas along the north coast, among the north coast reserves the predicted accumulative threats were rated as high, even within the limited coastal environments, such as coastal lagoons and estuarine ecosystems already exposed to naturally high levels of turbidity and nutrients, and also within much of the shelf due to its narrow extent. As a consequence, very high cumulative threats scores within the north province were found in the Piñones Forest Reserve, the Caño Martin Peña Natural Reserve and the Coral Reef Natural Reserve of Isla Verde.

In general these threat impacts imply changes in water quality, coral reef disease by fecal pollution and increases in marine debris (Loya, 1990; García-Sais et al., 2008; Bachoon et al., 2010). High overfishing scores imply that both artisanal and recreational fishermen, without distinction to geographical province, target the same benthic substratum across Puerto Rico. As a consequence 1) reef fishes are likely the fish group most threatened by anthropogenic drivers (CFMC; 1985, Ault et al. 2008, Pittman et al. 2010) and 2) there is high pressure targeting essential fish habitats. Nevertheless, despite these generalities, by analyzing threats in a high resolution spatially explicit manner, management can reduce uncertainty of threat exposure within specific areas and act to reduce specific threats within local areas.

For MPAs, practical new approaches need to be developed to improve their management and governance, and this must occur on scales larger than individual reserves. These approaches should follow from the specific threats identified in the current analysis and take into account their overlapping nature. The combination of threat driven management will result in cost and time efficiencies to improvements across the seascape within each MPA.

Second financing models needed to provide long-term success of each reserve must also consider the health condition of the resource. One approach to condition considered here is to employ the triage concept. In general over half of the Puerto Rico natural and marine reserves near the coastline are in high risk, suggesting that the cost of management intervention to reduce anthropogenic threat pressures is high and the expectation of restoration success is uncertain. In this scenario management actions need to encourage the local environment regulatory agencies and multilateral agencies (e.g. EPA, NRCS, PR-Planning Board) together to improve management governance across the seascape. The absence of active involvement and cooperation among local and multilateral agencies will be detrimental to both the natural resources and economic activities that depend on them, with consequent ripple effects throughout the economy.

Only a few reserves near coastline approach the intermediate triage level; these were the nature reserves of Tortuguero Lagoon (north), Punta Petrona (south) and Punta Viento (southeast). The reason that these areas are not at highly threatened is because they are only being affected by singular threats, unlike other areas influenced by multiple stressors. Management by the DNER or environmental agencies is thus cost effective as it need only reduce a single threat to get immediate results rather than needing to reduce multiple sources of threats with substantially greater effort. However, these areas are not necessarily biodiversity hotspots, so the choice of management intervention must also weigh the potential value of the resource under threat. Finally the healthiest reserves systems in Puerto Rico are those found offshore, such as the Desecheo marine reserve.

The south and west reserve systems represent the largest coverage of coral reef areas around the island, and thus also contain over half of the reserves areas designated by the state and federal governments. Therefore, management for anthropogenic stressors related to the deterioration of coral reefs in the southern and western provinces has to be treated in coordination with the regulating agencies responsible for urban planning and agriculture department. In some way, the regulatory agencies have to encompass jurisdiction within the watersheds in order to minimize the human activities and drivers that underlie threats to coastal and marine ecosystems.

Chapter 6. Conclusion

This study provides the current best estimates for spatially variant cumulative threats in the coastal and marine waters of Puerto Rico. Most of these activities primarily affect coastal lagoons and nearshore ecosystems rather than offshore ecosystems. This suggests that greater attention by government agencies should be toward the natural reserves systems in the western and southern part of the island, with the southern region being a particular priority (due to significantly greater coral habitat).

The estimations derived from this model are conservative, because the level and spatial distribution of threats can be influenced and enhanced by other environmental and oceanographic factors. As a consequence the results should be considered as illustrative of the power of the model as a tool and the type of management advice that this tool could provide. One of the most powerful aspects of this tool is that it potentially can fully encompass or integrate human activities into the management of biological resources, and as a consequence helps move management toward a more complete ecosystem-based management (EBM) approach. This study focused on turbidity, eutrophication, pollution and overfishing in formulating cumulative threat levels and their spatial distributions, but many others approaches or drivers can be incorporated into the model, such as occurrences of a priority species, oceanographic information (e.g., current pattern, sea level, sea surface temperature and chlorophyll distribution), and land-based activities (e.g., landfills, septic tanks, road density and coastal structures). The standardized methods developed under this study provide local regulatory agencies with a finer scale indicator of both the level and location of potential pressures on essential fish habitat (EFH) and coral reefs ecosystems. This approach also can contribute directly to the development of spatially explicit cost estimations that managers and stakeholders may subsequently use in the design of marine protected areas (MPAs) through the application of multivariate models and decision support tools such as Marxan. The resulting threat layers also provide valuable information to implement an effective marine zoning or management plan for current protected areas, as well as for coastal and marine spatial planning in general. In scenarios where threat conditions are not included in local or region-wide spatial planning, regulatory agencies can select areas of high cumulative stress as areas targeted for conservation instead of coastal and marine areas in healthier condition (Linke et al, 2012).

The variable stressors, as well as the distances at which benthic zones and threats are connected, are important factors relevant to integrating eco-system-based management for coral reef fisheries. Multivariable models, such as Marxan, that can incorporate these results within a cost function can give government agencies specific criteria and tools for selecting priority areas for spatial management that would lessen the negative impacts of these threats on these ecosystems. The scenarios generated in this study provide the basic frame of representation to 1) model multiple sources of anthropogenic stressors, 2) identify the spatial capacity for resilience (habitats to recover from disturbance), 3) provide spatial and geographical distribution of threats on habitats across the shelf (information necessary to account for the habitat requirements of all life stages of ontogenetically migrating fish species), and 4) spatially link distributions of these threats to up-land human sources and activities.

Resource managers and the scientific community should work together to develop and integrate monitoring programs to test the linkages between land-base activities and subsequent responses in marine ecosystem condition over time. These programs would then identify specific areas and activities for priority management concern. Additional research should be compiled into regional databases of empirical measurements, which could be used to further validate the efficacy of the models developed here as well as supplement regular monitoring programs. Modeling and monitoring efforts may also be expanded to include others types of conservation targets in order to explore multiple risk or cost scenarios. Resulting products would aid regulatory agencies in comparing multiple scenarios with different costs (e.g., socio-economic, species occurrences, and oceanographic drivers), in order to achieve conservation target goals (maintaining ecological function by reducing threats to coastal or marine sites).

The analytical structure used here can be extended by incorporating other types of information to identify high cumulative natural and anthropogenic potential impacts that justify conservation priority. This study is expected to enhance agency effectiveness in selecting essential marine habitats and helping identifying locations and strategies to minimize ecological impacts and maintain sustainable use for the benefit of those stakeholders dependent upon these resources. Finally, data compilation and model building in this study were created in a compatible format, and these were the most time consuming steps in designing a accurate threat scenarios. This compatibility will be significant in future analyses as it will greatly reduce the time necessary to explore alternate threat scenarios and assist stakeholders, regulatory agencies and planners to pursue a systematic approach to issues of sitting and scaling MPAs, critical habitats, coastal and marine spatial zones, etc., using available data. Finally spatially modeling of threats can facilitate government agencies determine marine management strategies.

Chapter 7. Recommendations

7.1. Sensitivity Analysis

To gain insight into which model assumptions are critical (i.e., which stressor layers or their calculation affect results) a sensitivity analysis is required. The process should engage various ways of changing input layers of the model to see their effect on the cumulative impact output value. The decision maker may investigate the best scenario assumptions before acting to reduce any particular stressor. Consultation with stakeholders will be essential during this process, as they may propose interesting changes to the model, and the impacts of such changes could be tested prior to management action. However, complex sensitivity analysis will require programming the model (e.g., using Python in ArcGIS) so it can be run repetitively. Programming would greatly facilitate scenario testing with managers and other stakeholders. This would allow governmental agencies to economically analyze costs while developing planning scenarios before management action take places.

7.2. Data Needs

This study was designed to ensure that users appreciate the uncertainties contained within the environmental stressors data and models. The series of models needed to calculate to what degree coastal and marine ecosystems are exposed to threats requires data on human activity and natural stressors (e.g., weather), both of which have to be modelled even before the results are combined. Moreover, the nature of the data available from these sources is not widely available within the necessary time and spatial scales and locations. For example, while there is a need to quantify and map parameters related to eutrophication as a function of nutrients, dissolved oxygen is also an important factor as biological oxygen demand, mediated by eutrophication and algal production, could be detrimental for plant and fish survival. Such data, however, are not available. Additionally more data are need for quantifying sediment or nutrient loads from the watershed out into the marine environment, which may be mediated by ocean waves and currents.

Nevertheless there are a number of local institutions (e.g., CariCOOS, CCRI) working to help scientists and managers contend with these data approaches. Additional data may also be available from distributed sources (e.g., Environmental Quality Board, Department Public Health and other governmental agencies). To simplify data identification and access, detailed metadata are needed that describe the available data from such distributed data sources. For example available water quality measurements from governmental agencies and academia are often available for a particular geographic zone (subject to a number of different laboratory processes and yield a number of measurements), but the metadata necessary to describe that is much larger than for the collection of stream stage measurements from a single gage.

Finally, if the model is incorporated as a management tool, a next step may be to periodically renew predictions based on changes resulting from management actions or additional anthropogenic stressors. For this it will be necessary that agencies update the available data on a periodic basis (e.g., every five years) to have representation in both time and space.

7.3. Groundtruthing

Further validation of the model can be obtained through more extensive groundtruthing. There should be a strong correlation with model results on the distribution of threats and direct measures of those threats (e.g, turbidity levels). Correlations with model results and the current health of marine resources will be helpful, but these will not account for the effects of cumulative stress over time. Correlation between the threat maps and groundtruth data will improve the results obtained and hence the quality of the maps. However is difficult to perform such groundtruthing in practice, not only because of weak spatial correlations due to cumulative processes over time, but also because marine data are expensive to collect. One particularly promising approach is the use of available remote sensing data (e.g., for chlorophyll or suspended sediments) as they have wide spatial coverage and temporally discrete. Using remote sensing techniques allows water quality to be measured and helps to understand how various parts of an ecosystem are exposed to or react from natural and anthropogenic stress.

For example, the concentration of chlorophyll has been used as indicator of eutrophication (Carlson 1977), and data for Puerto Rico are available spatially and temporally. One disadvantage, however, is that the broad wavelength spectral data available on current satellites (i.e., Landsat) do not permit discrimination of chlorophyll in waters with high suspended sediments (Ritchie *et al.*, 1994). A likely chlorophyll remote sensing sensor has been used to estimate suspended sediment concentration (SSC) where in situ monitoring is insufficient or impractical (Pavelsky and Smith, 2009).

Newer satellites (e.g., with MODIS channels) offer more refined remote sensing and thus an alternative for identifying coastal waters with significant SSC and their temporal variation (Li et. al., 2009; Pavelsky and Smith, 2009).

7.4. Input from Stakeholders

Stakeholder participation needs to be incorporated into the model to help managers answer questions such as: How robust are the results to initial conditions or assumptions? What regulatory agencies need to act in order to minimized natural and anthropogenic threats to coastal marine ecosystems? The use of stakeholders would allow input of both additional scientific or agency data as well as traditional ecological knowledge of resource users. It would also facilitate the setting of conservation priorities and marine ecoregional assessments. That will help to find solutions that better align human uses with their most compatible places to reduce marine conservation conflicts.

As they go through each step in this process (designing), stakeholders can i) provide monitoring indicators needed to determine the effectiveness of the model, and ii) prioritize which actions are need to focus on cost-effective strategies to minimize threats. This process will also facilitate stakeholder buy-in to the management process by giving them a sense of ownership in the model that will then underlie subsequent management action.

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Appendix A. Puerto Rico, sediments deliveries per basin for the year 2000. The data were obtain from the NOAA Coastal Program. Longitude and latitude are based on the WGS84 datum. Online at:
http://ccma.nos.noaa.gov/ecosystems/coralreef/summit_sea/summit_sea2.aspx

Basin ID	Sediments deliveries-2000	Longitude	Latitude	Basin ID	Sediments deliveries-2000	Longitude	Latitude
46	53668.56	-67.1145	18.51253	26	996838.8	-67.1708	18.40117
49	115609.6	-67.0541	18.51384	122	167466.2	-65.8328	18.4218
48	149749.3	-67.1029	18.51251	114	247367.5	-65.8308	18.41993
47	8398.192	-67.1074	18.5116	118	153409	-66.0282	18.41499
43	61626.75	-67.0203	18.51023	25	237899.8	-67.2134	18.38373
41	221433.5	-66.8743	18.48865	115	41688.03	-65.8076	18.40484
42	161811.5	-66.8268	18.48691	110	491017.9	-65.8061	18.4035
45	98846.03	-67.1681	18.47916	116	211639	-65.795	18.40177
38	1361264	-66.8353	18.4862	64	0	-65.5848	18.38497
44	56354.66	-66.9521	18.48325	104	543601.5	-65.7521	18.38154
39	774935.9	-66.9518	18.48299	106	180679.4	-65.7308	18.37995
151	188250.3	-66.9016	18.48295	70	1241.64	-65.6445	18.37629
37	186253.9	-66.5565	18.48786	103	291559.2	-65.7119	18.3702
156	190489.4	-66.7643	18.48221	105	573422.6	-65.6997	18.36793
32	143608.1	-66.7549	18.47861	113	0	-65.5655	18.34866
34	72246.51	-66.537	18.47875	50	46338.93	-65.2574	18.33359
31	2743435	-66.533	18.47666	101	870487.6	-65.6326	18.3239
36	221814.1	-66.3931	18.47732	154	161.1045	-65.6311	18.31471
158	454946.9	-66.6776	18.47174	56	35700.05	-65.2297	18.31793
130	1323205	-66.3695	18.47449	167	376828.5	-65.2815	18.30864
131	118880.8	-66.3142	18.47439	57	40577.6	-65.3336	18.30178
30	335058.7	-66.7083	18.46356	24	430554	-67.1775	18.26764
27	2899375	-66.708	18.46356	23	1330583	-67.1752	18.26769
29	424131.6	-66.7078	18.46357	135	745.5698	-65.6079	18.2603
128	29343.05	-66.2548	18.47027	22	183380.6	-67.1489	18.20765
127	3418748	-66.2576	18.46941	138	323.1737	-65.6038	18.21322
126	142739.5	-66.4563	18.4602	100	714201.4	-65.6924	18.20495
35	43304.61	-66.0918	18.46623	21	1434536	-67.1682	18.16394
124	99259.61	-66.2251	18.45806	99	854875.7	-65.7334	18.18188
123	1517095	-66.1874	18.44975	97	611069	-65.7591	18.15631
60	27124.87	-65.9535	18.44596	98	270738.4	-65.7671	18.14859
121	1185406	-65.8934	18.43362	96	724164.9	-65.7869	18.12396
129	87751.14	-66.111	18.42854	169	206328.7	-65.3904	18.16075
119	4714614	-65.9743	18.43182	173	110966.5	-65.5488	18.11911
120	507268.3	-66.1056	18.42837	92	310118.4	-65.7899	18.09791
171	287416.5	-65.4686	18.09389	4	104779	-66.5797	17.97386

Appendix A Continue

20	324606.7	-67.1698	18.02618	8	320822.4	-66.3407	17.97623
93	918539.8	-65.8373	18.05035	2	634476.3	-66.6628	17.97014
9	600326.6	-67.169	18.00428	79	44893.53	-66.3404	17.97596
7	66744.96	-66.7666	18.01037	72	46455.08	-66.3149	17.97586
10	473966	-66.7661	18.01038	75	202.6761	-67.2092	17.95823
15	1041907	-66.7669	18.0101	80	4794.897	-67.0803	17.96044
91	1849739	-66.3255	17.97909	6	1068574	-67.0987	17.95952
14	1486021	-66.7338	17.99585	82	1728.661	-67.0887	17.95918
5	201457.2	-66.8025	17.99316	73	43964.82	-66.0221	17.97767
11	1057554	-66.6385	17.98386	78	548568	-65.9654	17.97367
18	1200440	-66.8098	17.98029	12	721306.4	-66.4273	17.96497
153	1509936	-66.6232	17.98279	95	1392443	-66.4271	17.96497
3	63973.65	-66.5985	17.97946	1	297571.7	-66.85	17.95406
19	3440943	-66.5979	17.9792	77	337235.2	-66.0751	17.96708
84	356702	-65.8969	17.99014	83	343560.2	-66.0674	17.96612
86	519548.3	-65.8974	17.98986	81	1021621	-66.2215	17.95739
155	0	-67.0451	17.96903	112	424.8491	-67.1866	17.93653
76	39025.91	-67.0047	17.96743	111	556.1172	-67.192	17.93614
13	1975663	-66.9162	17.9687	68	163905.3	-66.2423	17.95001
90	793290.5	-66.0143	17.98483	59	86988.27	-66.242	17.95001
85	453140.7	-66.1352	17.94229	162	109702.3	-65.9061	18.42719
71	34193.67	-66.1566	17.93788	163	323907.9	-65.9757	18.4326
150	5684.129	-66.197	17.93505	164	215733.5	-66.5072	18.47578
152	0	-66.5164	17.89298	166	121172.4	-65.2836	18.28746
157	362453.6	-67.1735	18.25287	165	238132	-65.2855	18.32973
158	454946.9	-67.1603	18.1763	168	232798.4	-65.4309	18.1578
159	29255.2	-66.6379	17.98416	170	259410.1	-65.3611	18.1318
160	292901.5	-66.4051	17.95344	172	266972.6	-65.5232	18.09046
161	0	-65.7426	18.16928	58	177786.5	-67.1844	18.40167

Appendix B. Socio-economic fishing stressors scheme for Puerto Rico Island.

Province	Municipality	Average Hrs/fishing	Num. dock & ramp	Num. Marina
East	Patillas	40.3	15	1
East	Río Grande	24.88	1	0
East	Fajado	30.52	15	9
East	Ceiba	23.87	0	3
East	Vieques	34.46	2	2
East	Culebra	21.71	41	11
East	Naguabo	30.24	0	1
East	Humacao	35.42	0	3
East	Yabucoa	26.5	1	1
North	San Juan, Toa Baja	34.8	2	3
North	Cataño	30.16	48	7
North	Carolina	24.79	0	2
North	Loíza	27.7	5	5
North	Arecibo, Hatillo, Camuy, Quebradillas, Isabela	24.5	3	1
North	Barceloneta, Manatí, Vega Baja, Vega Alta, Dorado	12.36	4	3
South	Guayama	24.55	9	4
South	Guánica, Yauco, Peñuelas	33.81	66	4
South	Guayanilla	28.95	5	1
South	Santa Isabel	30.53	17	2
South	Salinas	26.59	82	7
South	Arroyo	23.81	2	1
South	Lajas	31.31	115	16
South	Ponce	35.93	11	4
South	Juana Diaz	31.8	3	2
West	Cabo Rojo	32.36	80	13
West	Mayagüez	32.02	2	6
West	Añasco	40.65	2	2
West	Rincon	40.31	1	2
West	Aguada	36.76	2	0
West	Aguadilla	25.73	2	4

Appendix C. Evaluated fishery landings dependency per municipality as indicator of artisanal fishing stress on seascape ecosystems. Values of I*i* close to 0 indicate less dependency in terms of the overall landings, with a value of 1 indicating the highest fishing dependency score.

LOCATION	POUNDS (LB)	INDEX	LOCATION	POUNDS (LB)	INDEX
Isabela	892	0.00	Lajas	27,662	0.08
Camuy	12,153	0.03	Cabo Rojo	363,382	1.00
Hatillo	906	0.00	Mayagüez	72,860	0.20
Arecibo	21,460	0.06	Añasco	38,710	0.11
Barceloneta	1,748	0.00	Rincón	165,180	0.45
Manatí	3,514	0.01	Aguada	21,507	0.06
Vega Baja	14,145	0.04	Aguadilla	113,252	0.31
Vega Alta	3,952	0.01			
Dorado	1,059	0.00			
Toa Baja	4,161	0.01			
Cataño	3,740	0.01			
San Juan	28,325	0.08			
Carolina	190	0.00			
Loíza	5,354	0.01			
Río Grande	2,857	0.01			
Luquillo	539	0.00			
Fajardo	49,981	0.14			
Ceiba	35,871	0.10			
Naguabo	13,058	0.04			
Humacao	11,522	0.03			
Yabucoa	7,327	0.02			
Maunabo	20,324	0.06			
Culebra	337	0.00			
Vieques	17,675	0.05			
Patillas	485	0.00			
Arroyo	13,880	0.04			
Guayama	21,884	0.06			
Salinas	51,787	0.14			
Santa Isabel	16,052	0.04			
Juana Díaz	45,653	0.13			
Ponce	30,293	0.08			
Peñuelas	38,535	0.11			
Guayanilla	6,272	0.02			
Guánica	45,615	0.13			