Supplementary Materials - Environmental Research Letters

Arkema *et al* Assessing habitat risk from human activities to inform coastal and marine spatial planning: A demonstration in Belize

1. Supplementary Text

1.1 Method and tool for risk assessment

Conceptually, the risk dimensions we define are the same or very similar to the *Productivity-Susceptibility* axes developed recently for ecosystem-based fisheries management (Patrick *et al* 2010, Hobday *et al* 2011), *Exposure-Vulnerability* in the cumulative impact literature (Halpern *et al* 2008, Teck *et al* 2010) and *Exposure-Sensitivity* in a generalized risk assessment for ecosystem indicators (Samhouri and Levin 2012).

Our approach to estimating "consequence" and "exposure" is flexible and can accommodate additional or replacement criteria as needed to be applicable to diverse contexts. However, sensitivity testing of other risk frameworks suggests that fewer than four risk evaluation criteria will lead to skewed results (Patrick *et al* 2010). To quantify spatial overlap, the Habitat Risk Assessment (HRA) model estimates the percent of each habitat type that overlaps each activity using spatial data layers, and ranks this criterion based on cutoff values for low, medium and high overlap from previous studies (Samhouri and Levin 2012), as described in Table 1. Temporal overlap, intensity and management effectiveness scores are based on seasonal variation in habitat abundance, as documented in the peer-reviewed literature, and variation in the intensity, timing and management of human activities (typically available in government and industry reports).

The flexibility in our approach allows for alternative approaches to combining scores across the exposure and consequence criteria. For example, depending on the habitat type, activities of concern, and risk aversion of decision makers, users could instead opt to use the highest (rather than average) value, or identify specific risk triggers within a criterion that automatically assign it to a high category.

1.2 Scenario descriptions

We assess risk to habitats under three management scenarios representing three alternative configurations (location and extent) of human activities. The scenarios are based on maps of the current distribution of ocean and coastal activities, existing and pending government plans, and stakeholders' values and preferences (see Clarke *et al* 2013 for a description of the stakeholder-engagement process). Conceptually, the scenarios reflect three visions for the future of Belize in 2025. The ***Conservation*** scenario represents a vision of long-term ecosystem health through sustainable use and investment in conservation. The ***Development*** scenario prioritizes immediate development needs over long-term sustainable use and future benefits from nature. The ***Informed Management*** scenario blends strong conservation goals with current and future needs for coastal development and marine uses. Operationally, we reflect stakeholder preferences for particular activities in certain regions and information from existing legislation in changes to the current extents and locations of eight human activities. We use the HRA model to understand differences in the cumulative risk to coral reefs, mangrove forests and seagrass beds under the three alternative future scenarios. Results from the HRA are incorporated into the draft Integrated Coastal Zone Management (ICZM) Plan for Belize and serve as a metric for evaluating the preferred coastal and marine spatial plan.

1.3 Spatial data on habitats for the spatial overlap criterion

1.3.1 Coral reefs. Our map for corals is a dataset of shallow water reef locations (generally fewer than 30 m deep) for the Mesoamerican region from multiple sources (Belize Coastal Zone Management Authority and Institute (CZMAI) unpublished data, Mumby 1999). The GIS layer is based on 30-m Landsat data that were classified and converted to a shapefile. The dataset includes dense patch reefs, fore reef, reef crest, and additional reefs in and around Glover's Reef that we added after a stakeholder workshop in Belize City in October 2011. Note that some versions of this dataset also include additions made by World Resources Institute (WRI) and Wildlife Conservation Society (WCS) (Belize Coastal Data CD, WRI 2005) for shallow gorgonian beds, spur and grooves and less dense patch reefs, but we did not include these as we were primarily focused on reef structures related to coastal protection, tourism and the lobster fishery (Clarke *et al* 2013).

1.3.2 Mangrove forests. Our GIS layer for mangrove and littoral forests is from a 2010 study conducted by the Water Center for the Humid Tropics of Latin America and the Caribbean (CATHALAC) for World Wildlife Fund's Mesoamerican Reef Program (Cherrington *et al* 2010). These data are based on Landsat satellite imagery available through the Regional Visualization and Monitoring System. They cover the entire coast of mainland Belize and include the offshore atolls and islands at a scale of 1:100,000. Based on feedback from stakeholders, our final map also includes patches of mangroves for Lighthouse Reef Atoll that were not in the 2010 CATHALAC study (Fig. S2).

1.3.3 Seagrass beds. Our spatial data layer for seagrass is a combination of CZMAI surveys in 1997 (unpublished data) that were developed through joint efforts of CZMAI, the University of Exeter, the University of Newcastle and Coral Caye Conservation to delineate various types of marine habitats located offshore of Belize and a separate 2007 study undertaken by the University of British Columbia (UBC). UBC conducted regional-scale seagrass habitat mapping in the Wider Caribbean Region using Landsat sensors (Wabnitz *et al* 2008). Our map combines the large expanse of seagrass along the coast in Wabnitz *et al* 2008 with the CZMAI 1997 map, which did not document nearshore seagrass. Local experts confirm dense seagrass cover along the shoreline with the exception of some regions around communities where seagrass cover may be patchier due to development and associated dredging (Tino Chi *personal communication*).

1.4 Rankings for resilience criteria

1.4.1 Coral reefs. The potential consequence of human activities on habitats depends in part on four life history characteristics that can influence resilience to stressors: natural mortality, connectivity, recruitment and recovery time. To assess the resilience of coral reefs in Belize, we use peer-reviewed literature to rank each of these criteria based on the guidance in Table 1. Some studies are from Belize, but in other cases, we assume the ecology and life history of Belizean corals is similar to that of reefs elsewhere for which published data exist. We list the ranks in Table S3 and explain our ranking in the following paragraph.

The literature suggests that life history characteristics vary tremendously among different species of corals, which makes it difficult to score these as a single habitat type. The most common species of corals (McField 1999) in Belize has shifted with the dramatic changes in coral reefs in the Caribbean. Reefs were once dominated by larger, longer-lived species with lower recruitment, such as *Acropora servicornes* and *Montastrea annularis*. Now "weedy" species, such as *Porites asteroids* and *P. porites* dominate many reefs (Knowlton 2001, Green *et al* 2008). Their life history is characterized by shorter longevity, faster recovery from disturbance and more frequent recruitment (Knowlton 2001, Green *et al* 2008).

We rank the resilience criteria for coral reefs considering that reefs encompass species with these different life history strategies. We score natural mortality among all corals as medium, since communities include both longer-lived and shorter lived-groups (Knowlton 2001, Green *et al* 2008; Table 1, Supplementary Table S3). We give recruitment a low risk rank since the larger, long-lived species of corals spawn annually. They can experience recruitment failure in a particular year, but the weedier species reproduce and recruit throughout the year (Richmond and Hunter 1990, Soong 1991, Van Veghel 1994; Table S3). We give corals a medium rank for connectivity (10-100 km; Table 1), which is likely on the lower end for the annual spawners (Jones *et al* 2009) and higher end among populations of brooders (Richmond and Hunter 1990, Soong 1991). We rank recovery as medium (1 to 10 years; Table 1, Table S3). While *Montastrea* and *Acropora*, are on the longer end of this category and in many cases will take more than ten years to recover (Stoddart 1974, Hughes and Tanner 2000), Belize reefs are now dominated by "weedy" species (Robbart *et al* 2004, Green *et al* 2008) that recover faster (Knowlton 2001).

1.4.2 Mangrove forests. To assess the resilience of mangrove forests to exposure to human activities, we use peer-reviewed papers on mangrove natural mortality, connectivity, recruitment and recovery time. The scores assigned to each of the criteria are presented in Table 1, we list the modeled ranks in Table S3 and explain our thinking in the following paragraph. In some cases we reference studies on the most common coastal mangrove species in Belize, red (*Rhizophora mangle*) and black (*Avicennia germinans*) mangroves. In other cases we assume the ecology of Belizean mangroves is similar to forests elsewhere for which there are published data. We score risk for the natural mortality criterion as high (i.e., <20%). According to a review of studies on natural mortality of mangrove stands in Florida, Panama and Puerto Rico, 16-19% of forests are generally dead trees (Jimenez *et al* 1985). A separate study in Florida finds mortality rates of 9.6%, 4.6% and 3.2% for red, black and white mangroves, respectively (Rey 1994). We score the recruitment criterion as low risk (i.e., annual or sub-annual recruitment) because the literature suggests that mangroves in the Caribbean exhibit annual periods of dispersal and recruitment (Rabinowitz 1978, McKee 1995; Table S3). We score connectivity as medium because mangroves exhibit both long range and local dispersal (Clarke 1993, Sun *et al* 1998). We also score recovery time as medium based on a study that found mangrove seedlings 17 months after a severe disturbance in Nicaragua (Roth 1992; Table S3).

1.4.3 Seagrass beds. The literature on natural mortality, connectivity, recruitment and recovery time of seagrasses was reviewed to assess ranks for the risk criteria listed in Table 1. We include these ranks in Table S3 and explain the scores in the following paragraph. Ranks for some criteria are based on findings for seagrass in Belize, but for others we assume the life history of Belize populations corresponds to seagrasses in places for which published data exist. We focus on information for *Thalassia testudinum*, as it is the most common of the five species of seagrass in Belize (Tussenbroek 1995).

We score natural mortality as medium because several studies from the tropics and subtropics suggest that winter populations of *Thalassia* are 30-50% the size of summer populations (Zeiman 1975, Tussenbroek1995, Kaldy and Duncan 2000). We score recruitment and recovery time risk scores as low (i.e.,< 1 yr) because seagrass beds often die off in the late summer, exhibit low biomass in the fall and winter and regenerate the spring of the following year (Zieman *et al* 1999). Further, a study conducted on *Thalassia* in the Caribbean coast of Mexico indicates turnover rates for shoots and rhizomes are 0.6 yr^{-1} (Gallegos *et al* 1993). In contrast to the other two habitats, the connectivity risk score is high, as seagrass generally disperse short distances (Orth *et al* 1994, Harwell and Orth 2002). Although *Thalassia* do reproduce sexually, they rely largely on clonal/vegetative reproduction for population growth and recovery from habitat disturbance (Gallegos *et al* 1992). Additionally, a study in the lower Florida Keys found that gene exchange in *Thalassia* resembles a "stepping stone" model of dispersal, indicating that individuals are more likely to disperse to nearby adjacent habitats within 4 km (Schlueter and Guttman 1998).

1.5 Human activity data for estimating the spatial overlap criteria and literature review for estimating ranks for remaining exposure and consequence criteria

1.5.1 Agricultural run-off. The effects of runoff from approximately 57,000 hectares of cultivated land in Belize may degrade coastal habitats. Potential effects include siltation, eutrophication, and both point and non-point pollution with a growing number of pesticides, hydrocarbons, heavy metals and faecal material from livestock, as well as cumulative water extraction and diversion to feed large-scale crop production such as banana farms. With the year-round growing season in Belize, run-off is constant, and especially high during the early rainy season starting in June (Clarke *et al* 2013).

Increased nutrient runoff accelerates phytoplankton production to the point that it increases turbidity and decreases light penetration; this reduces light availability for photosynthesis and increases sediment load on coral colonies that can have sub-lethal and lethal effects on coastal and marine habitats. Excessive sedimentation can affect the complex food web on the reef by killing not only corals, but also sponges and other organisms that serve as food for commercially important finfish and shellfish (Rogers 1990). In particular, pesticides used in agricultural practices cause an increase in the respiration rate and a decrease in the photosynthesis of coral symbionts (Kühlmann 1988, Jones 2005, Negri *et al* 2005, Cantin *et al* 2007) and a decrease in the photosynthesis and effective yield of seagrass (Walsh *et al* 1982, Flores *et al* 2013). Though these adverse effects are found even at low pesticide concentrations, increased pesticide concentration or duration increases the severity of the impact. Unfortunately there is little field evidence of the impacts of pesticides on coral and seagrass area and structure (sensitivity criteria, see Table 1), as most of the previous studies are in controlled lab environments (but see Flores *et al* 2013). However, satellite data from Belize suggest land-based runoff may have a wide range of effects on marine ecosystems in this region (Andréfouët *et al* 2002). In particular, *in situ* studies reveal a decline in water clarity over the last 15 years on the Barrier Reef by Carrie Bow Caye (Koltes and Opishinsky 2009). Based on this information from the literature, we rank the non-spatial criteria listed in Table 1; the results are presented in Tables S1-S3.

To assess the potential risk from agricultural run-off to coral reefs, mangrove forests and seagrass beds, we use the map of agricultural runoff from the WRI Threat Atlas for Belize (WRI 2005; see Fig. S3).

1.5.2 Aquaculture. Aquaculture development in Belize dates back thirty years, commencing with the commercial farming of the Pacific white shrimp *Litopenaeus vanammei* and now including Tilapia and Cobia. However, declines in the world market and losses from hurricane damage have depressed this industry. Today, there are a total of nine operational shrimp farms with a total production area of 2,673 acres within the three kilometer coastal zone (Clarke *et al* 2013).

The negative effects of aquaculture are both ecosystem-wide and specific to mangroves. Adverse effects include the loss of habitat and niche space, altered food webs, release of eutrophicating substances and toxic chemicals, the transfer of diseases and parasites to other organisms and the introduction of exotic species and genes into the environment (Rönnbäck *et al* 2002, Lui *et al* 2011, Chen *et al* 2013), all of which could affect corals, seagrass and mangroves. Direct habitat conversion for shrimp production has been substantial in the Caribbean and other parts of the world (Ellison 2008, Lui *et al* 2011). From 1985 to 2013, 11.9% of mangrove forests in Honduras were lost, primarily from pond construction and associated changes in hydrology, sedimentation, and water pollution (Chen *et al* 2013). Mangrove loss has slowed in Belize (Cherrington *et al* 2010) and most of the earlier losses were due to land development for housing and commercial uses. Thus, risk from aquaculture to mangroves is not likely to stem from overlap between this activity and the forests. However, future construction of aquaculture farms in mangrove forests would necessitate habitat conversion in those localities. Using this information from the literature we rank the non-spatial criteria listed in Table 1 and present our results in Tables S1-S3.

Aquaculture facility locations were identified from coordinates collected by the Belize Fisheries Department (2011). The footprint of each facility was digitized using satellite imagery (Fig. S3).

1.5.3 Coastal development. The coastal zone is a major focus of the country's economic development, especially to support the tourism industry, which in recent years has accounted directly for 13-15% of the Gross Domestic Product (World Travel and Tourism Council 2012). Human settlement, infrastructure, and economic activity are all included in the zone. Of the ten major residential centers in Belize, six are located within 3 km of the coast (Clarke *et al* 2013).

The ecological impacts of coastal development on seagrass, coral, and mangroves comes primarily from the indirect consequences of sedimentation and pollution and from the direct impact of habitat conversion. Sediment and pollution are the results of inadequate sewage treatment, erosion from agriculture, urban expansion, poor waste management, eutrophication and inappropriate pesticide use (Morrison *et al* 2013). Sediment reduces light intensity, which results in declines in seagrass area (Freeman *et al* 2008) and, in coral, diminished nutrition, growth and reproduction (Pastorak and Billyard 1985, Richmond 1993). Sediment physically affects coral by coating feeding surfaces and interfering with the solid substratum required by larvae for settlement and metamorphosis (Richmond 1993). Nutrient pollution and toxic substances can trigger transitions in seagrass-dominant species and, in some cases, cause transitions from seagrass to macroalgae (Peters *et al* 2009, Houk and Chamacho 2010). For example, 15 years of development was associated with a 50% decrease in seagrass cover in the Mediterranean (Frashetti *et al* 2011). In coral communities, pollution alters competitive interactions and increases the likelihood algal blooms and mortality from bacterial infection

(Pastorak and Billyard 1985, Richmond 1993, Bozec *et al* 2008, Peters *et al* 2009). Mangroves are not only adversely affected by sediment and pollution, which can cause stands to dieback, but also by the direct effect of habitat conversion for coastal development (Jusoff 2013) or dredge and fill operations. Next to littoral forests, mangroves are globally the most affected ecosystem as a result of coastal development; over 35% of mangroves have been lost in the last two decades (Valiela *et al* 2001, Polidoro *et al* 2010). Using this information from the literature we rank the non-spatial criteria listed in Table 1 and present our results in Tables S1-3.

The GIS coastal development layer highlights cleared land with or without structures. The initial version of the layer comes from the Biodiversity and Environmental Resource Data System (BERDS) digital survey on Belize settlements (BERDS 2011). This layer was updated using satellite imagery from multiple free satellite image explorers (Google and Bing; Fig. S3).

1.5.4 Dredging. Dredging is considered one of the main stressors to marine ecosystems associated with development in the coastal areas. The activity usually involves the extraction of bottom sediments from one area for disposal in another area in order to open navigational channels, extract minerals, fill wetlands and mangroves for coastal development, and/or rebuild eroded beaches. Once sediment is removed, it can take on average 10 – 15 years for the area to recover and regain its biological complexity (Newell *et al* 1998). Where dredging occurs in seagrass beds, all vegetation is usually lost, causing sand to be more easily carried away by waves and currents, increasing turbidity and deepening the area (Zuidema *et al* 2011). Recovery time of seagrass communities where dredging occurs depends on the composition of the substrate and stability of the habitat (NRC 2002). Using this information from the literature we rank the non-spatial criteria listed in Table 1 and list our results in Tables S1-S3.

To date there is no policy implemented to regulate marine dredging in Belize. As it stands, authorization for dredging is issued through the Mining Unit of the Ministry of Natural Resources and Agriculture. The current map of dredging locations was created based on information in dredging permits issued by the mining department from 2005 to 2011 (Fig. S3).

1.5.5 Fishing. The Fisheries sector in Belize is comprised of three main industries: capture fisheries, aquaculture and inland subsistence fishing. The industry supports over 2,500 registered fishers and their families and provides a source of protein for local consumption and exportation to foreign markets. For the "Fishing" human activity in the ICZM plan we focus on marine capture fisheries, which in 2010 generated $23.2 million BZD with the exportation of the three main marine products lobster, conch and finfish (Belize Fisheries Department 2011).

The ecological effects of fishing on coral and seagrass, both direct and indirect, are dependent on the fishing technique used (Jennings and Palunin 1996). Lobster traps and hand-harvesting cause the greatest damage to corals, whereas the greatest impact to seagrass is from fish traps. Trawling, bottom-set gillnets, and traps impact corals through direct contact and entangling, both of which can cause coral breakage (NOAA 2001; note that as of 2010 trawling is illegal in Belize). Indirectly, overfishing, especially of top predators, can cause decreases in primary production and carbon flux, and lead to changes in abundances of lower-trophic species (Jennings and Polunin 1996, Hawkins and Roberts 2004, Knowlton and Jackson 2008, Salomon *et al* 2008). In an example from the Caribbean, coral cover was higher in infrequently fished areas (Hawkins and Roberts 2004). Using this information from the literature we rank the non-spatial criteria listed in Table 1 and present our results in Tables S1-S3.

The GIS fishing layer combines all known areas for capture-fisheries, including commercial, recreational, artisanal and sportfishing, for all relevant species. The layer comes from the Corozal Bay Wildlife Sanctuary Management Plan done by the Sarteneja Alliance for Conservation and Development (SACD Socio-economic survey 2008) and information from the Belize Fisheries Department.

1.5.6 Marine recreation. The most popular forms of marine recreation in Belize include diving, snorkeling, jet skiing, kayaking, swimming and wind and kite surfing. World-renowned snorkeling and diving draw over 800,000 tourists to the region annually, which drives the local economy through direct tourism contact and construction of new development (Belize Tourism Board (BTB) 2008, Clarke *et al* 2013). The most frequented sites by snorkelers and divers are Hol Chan Marine Reserve, Blue Hole Natural Monument, and Goff's Caye Marine Managed Area (BTB 2008).

Ecological damages associated with marine recreation result from the physical impacts of boat anchors, snorklers and divers. Corals are broken and fragmented by the anchor dropping to the substratum and by dragging chains, but can also be overturned during the retrieval process (Dinsdale and Harriot 2004). Anchoring affects seagrass by physically reducing vegetation and increasing edge density. In the short term, anchors cause localized scars and reduce seagrass shoot and leaf density (Creed and Filho 1999, Francour *et al* 1999, Ceccherelli *et al* 2007). Lasting impacts to seagrass from anchoring include reduced macrophyte standing stock (Creed and Filho 1999) and increased exposed edge—as demonstrated by a west Australian study where exposed edge increased 230% in 11 years (Hastings *et al* 1995). Divers and snorkelers cause coral breakage through direct contact. Though some studies have found no snorkeler impact on fish or macroalgae assemblages (Claudet *et al* 2010), damage to corals by snorkelers and divers has been observed by several studies (Plathong *et al* 2000, Poonian *et al* 2010, Worachananant 2008). In one observational study, structural damage to corals occurred at an average rate of 19 coral breaks per hour of diving (Worachananant 2008). Using this information from the literature, we rank the non-spatial criteria listed in Table 1 and present our results in Tables S1-S3.

The GIS layer for marine recreation is based on visitation data collected by park managers and tour operators through 2011 and locations of diving, snorkeling, swimming and kayaking sites identified on tour operator websites that we digitized and grouped into zones for recreation (Fig. S3).

1.5.7 Marine transportation. The marine transportation zone includes major shipping routes (including crude oil export *see below*), water taxi routes and ports. Industrial shipping and cruise ship traffic occur year-round and have grown in recent years because of Belize's prime location along the Caribbean coast of Central America, reputation as a tourism destination and the initiation of oil exploration in 2005. Water taxis and other smaller vessels serve leisure boating and local travel to the more than 300 cayes within territorial waters throughout the year. The regulation of marine transportation routes is by the Belize Port Authority. However, there is no policy in place that outlines specific routes for each type of activity or zones the marine area according to activity (Clarke *et al* 2013).

Industrial shipping and in particular, cruise ships, can have a number of ecological effects (Davenport and Davenport 2006, Abdulla and Linden 2008). Illegal discharge of substances, such as oil or other hydrocarbons, is common. Dredging of channels for the larger vessels increases turbidity, which thereby decreases light availability for seagrass and corals and smothers corals. Propulsion-related turbulence can also stir sediments and smother underwater communities (Davenport and Davenport 2006). Finally, cruise ships produce substantial amounts of waste water and sewage that are often discharged untreated into seagrass or coral habitats. A typical cruise ship discharges around 1 million liters of 'black water' during a 1 week voyage (United States Environmental Protection Agency 2000). Anchoring may also impact marine habitats, especially as a result of small vessel traffic, as described in *Marine Recreation*. Using this information from the literature, we rank the non-spatial criteria listed in Table 1 and present our results in Tables S1-S3.

The map for the marine transportation zone consists of buffered polylines and points depicting the location of major shipping lanes, water taxi routes and port facilities as of 2011 from the Belize Port Authority.

1.5.8 Oil exploration and drilling. The oil exploration and drilling zone represents the region where searching for and extracting the deposits of crude oil and natural gas, as well as other activities required to determine quantity and economic value, are permitted. The petroleum industry is relatively new in Belize with the first successful drilling operations commencing in June 2005 by Belize Natural Energy Ltd. Since then, both terrestrial and marine areas have been parceled out to 18 companies under exploratory license agreements with the Government of Belize. Companies are given an eight-year exploratory license, and if oil is found during that period, a twenty-five-year period for commercial exploitation. However, if no oil is discovered during the exploratory period the contract self-terminates and operations must cease.

Potential impacts from oil exploration and drilling on mangrove forests, coral reefs and seagrass beds include risk from increased marine traffic (see marine transportation), pollution from daily operations and shipping and pipeline accidents. In mangrove forests, petroleum is associated with tree defoliation, stand death, seedling mutation and the loss of associated sessile and mobile animal species (Ellison and Farnsworth 1996). Oil spills in Panama in 1968 and 1986 caused the direct death of 118 hectares of mangrove, but also reduced leaves density, leaf longevity and leaf biomass in an additional 307 hectares (Keller and Jackson 1993, Duke *et al* 1997). Oil and the dispersants used during clean-up processes reduce coral growth, tissue maintenance and reproduction (larval and nubbin mortality; Shafir *et al* 2007, Peters *et al* 2009, Yender and Michel 2010, Goodbody-Gringley *et al* 2013). Following the Bahía Las Minas 1986 oil spill in Panama, coral cover decreased by 76% and 56% at depths of 0.5-3.0 m and 3.0-6.0m, respectively (Keller and Jackson 1993). After the Deep Horizon Oil Spill, 86% of coral colonies exhibited some sign of impact—bare skeleton at the basal region, loose tissue or coverage with brown flocculent material (White *et al* 2012). Oil and dispersants cause seagrass dieback through smothering, asphyxiation, chemical toxicity, as well as reduced light availability and food sources; oil-spills have caused complete and unrecovered seagrass dieoff (Keller and Jackson 1993, Peters *et al* 2009). Using this information from the literature, we rank the non-spatial criteria listed in Table 1 and present our results in Tables S1-S3.

In February 2010, an oil concession map was released by the Geology and Petroleum Department in the Ministry of Energy, Science & Technology and Public Utilities that confirmed that parcels were being issued throughout Belize, including in marine reserves and national parks (Oceana 2010). To date, there is no oil spill response legislation or protocol for dealing with a spill event at sea, whether from oil exploration activities or transportation of oil products via bunkers or hydrocarbon cargoes. The Department of the Environment is in the process of developing such legislation. The map for the current oil exploration and drilling zone is based on lease boundaries and management practices available in the Proposed Petroleum Exploration Planning Framework (Geology and Petroleum unpublished report) and the 2012 Belize Petroleum contracts map (Fig. S3).

1.6 Comparison of modeled to observed data

1.6.1 Mangroves. We evaluate how well results from the HRA model align with measures of ecosystem quality for coral reefs and mangroves. For mangroves, we compared modeled cumulative risk to observed data on forest fragmentation using Receiver Operating Characteristic (ROC) curves. This is the standard approach for evaluating and comparing models, algorithms and technologies in medical diagnosis (Pepe 2004) and is becoming increasingly common in other fields such as finance, atmospheric science and machine learning (Gonen 2006). Essentially, ROC curves are a diagnostic tool for evaluating whether a logistic model discriminates between events (in this case, fragmented forest) and nonevents (non-fragmented forest patches).

In the ROC analysis, cumulative risk estimated by the HRA model is the continuous predictor variable and observed fragmentation based on changes in mangrove forests between 1980, 1989, 1994, 2000, 2004 and 2010 is the binary response variable (Cherrington *et al* 2010). Based on satellite imagery and field studies, Cherrington and colleagues (2010) classified mangrove forests along the entire coast of mainland Belize, atolls and cayes using several categorization schemes for clearing, regrowth and fragmentation. For the fragmentation analysis, the authors used ArcGIS 9.2 to identify mangrove patches whose areal extents had been altered. In some cases, the process of clearing had broken mangrove patches into multiple fragments; in other cases, the sizes of patches had merely been reduced. Areas with forest that had never been altered were classified as never fragmented. These data are depicted in Figure 10 of Cherrington *et al* 2010. During the 1980-2010 period less than 2% of the mangrove forest was further cleared beyond initial fragmentation so forest cover remained fairly constant. For the purposes of this study we categorize the forests as either fragmented or non-fragmented (Fig. S8, Fig. S9). Non-fragmented forest areas have remained unchanged in the last 30 years. Fragmented forests are areas that have been cleared and result in multiple, smaller patches. We conduct the ROC analysis at a 500 m resolution, such that each grid cell has a value for modeled cumulative risk and each is classified as either fragmented or non-fragmented.

We use ROC curves to evaluate how well the HRA model predicts mangrove fragmentation and to identify the threshold value for cumulative risk which best predicts which grid cells have fragmented forests. ROC curves have several diagnostic features that make them useful for such an analysis (Pepe 2004, Gonen 2006). Two measures are used to assess the predictive accuracy of the model: sensitivity and specificity. Sensitivity is a measure of how frequently the model produces false positives (i.e., incorrectly classifies a grid cell as fragmented) and true negatives (i.e., correctly classifies a grid cell as non-fragmented). Specificity is a function of the frequency of false negatives and true positives (Pepe 2004, Gonen 2006). A ROC analysis involves calculating sensitivity and specificity at intervals of cumulative risk (Fig. S9) and using these measures to evaluate the model and identify the optimal threshold of risk for predicting fragmentation. In the case of mangroves, cumulative risk varies from approximately 0.6 to 7.35. Using PROC LOGISTIC and the ROCPLOT macros in SAS 9.3 we calculate sensitivity and specificity at varying intervals (0.001 to 0.1; each point along the curve in Fig. S9a is a value for cumulative risk between 0.6 and 7.35), where the intervals become smaller approaching the best threshold for risk (see below). Plotting sensitivity by 1-specificity produces the ROC curve (Fig. S9a).

A statistical property of the ROC curve is that the area under the curve equals the probability of concordance between predicted and observed data. This is the number of grid cells where the model correctly predicts fragmentation divided by the total number of grid cells (Gonen 2006). The area under the mangrove ROC curve equals 0.8 which suggests the model performs well (Fig. S9a). An area of 1 indicates the model discriminates perfectly. Similarly, 79% of the predicted probabilities were associated with observed responses and the relationship between modeled and observed data was significant ($N = 11369$, $DF = 1$, Wald Chi-square = 1508, $p < 0.001$).

A second feature of the ROC analysis is that the intersection between plots of sensitivity by the continuous independent variable (i.e., cumulative risk) and specificity by the independent variable (i.e., cumulative risk) is one way to identify the optimal threshold for distinguishing between events (i.e., fragmented) and nonevents (non-fragmented areas; Fig. S9b; Pepe 2004). For the predicted and observed mangrove data, the sensitivity and specificity curves cross at a cumulative risk of ~2.9. This is quite close to the threshold (cumulative risk = 2.7) we used to distinguish between high and medium risk (top two-thirds of the distribution, see final paragraph in Section *2.2 Estimating habitat risk*) versus low risk areas (bottom third of the distribution), which suggests the regions we categorized as high and medium risk likely correspond with fragmented areas and low risk corresponds with non-fragmented areas.

1.6.2 Coral reefs. To assess the performance of the HRA model, we compare modeled cumulative risk to observed data on coral reef health 2006 at 134 sites throughout Belizean waters in 2006. The observed data include percent cover of coral and macroalgae and density of parrotfish and surgeonfish (HRI 2008). Coral and fleshy macroalgal observations were collected using the Atlantic and Gulf Rapid Reef Assessment methodology (HRI 2008). At every site coral and macroalgal cover were estimated along six transects of 10 m covering a region of about 500 m^2 per site. We average the data for the six transects to produce one observation at each site.

The modeled coral risk data set initially contained over 32,000 adjacent data points, which is computationally challenging for geospatial analyses. We therefore use only those modeled data which were in the closest proximity of the observed coral cover. Further, some locations of observed data do not have corresponding modeled data (see Fig. S10 for locations without a black star). This is because the coral reef data layer for the marine spatial planning process in Belize, which we vetted by local experts during several workshops, does not include some of the small patch reefs visited by divers during the empirical data collection.

To compare observed to modeled data, given the aforementioned challenges, we use simple kriging to construct a confidence interval for the model estimate of coral risk at each observed location. Kriging essentially interpolates the modeled data based on the scale at which it varies spatially (Chiles and Delfiner 1999, Wackernagel 2003). This enables us to make statements about the confidence of model predictions at all locations where we have observed data. Because modeled coral risk is, by construction, measured on a different scale than observed coral health, we start by normalizing and then standardizing the data. This standardization makes the observed and modeled data more comparable. Yet, an analysis on standardized data assumes that the relative increments of both measurement units are the same and that “average risk” measured by the model inherently measures the same as ”average risk” measured by coral health indicators. After normalizing and standardizing the four empirical indicators of coral health (i.e. coral cover, macroalgal cover, parrotfish and surgeonfish density), we average their scores to create an overall coral health indicator.

We then compute the percent of observed values that lie in the 95% confidence interval of the model data. To do this, we first use simple kriging to obtain model values of estimated coral risk at the locations where we have values of observed coral health (Chiles and Delfiner 1999, Wackernagel 2003). The prime geostatistical tool for creating such kriging maps through spatial interpolation is the variogram; a mathematical function that defines the spatial dependence between stochastic processes in different locations of a spatial field (Wackernagel 2003). As such, the variogram helps us define the extent to which we can interpolate observed values (Wackernagel 2003). Of the several possible variograms that we fitted to the modeled risk and observed health datasets, the best combined an exponential fit with a nugget-effect. The nugget effect refers to a jump at the origin of the variogram and, in our case, it represents a form of micro-scale variability related to the fact that the observed data is measured at a relatively coarse spatial scale (500 m^2). We also verified whether the data is anisotropic—in other words, whether the spatial correlation depends on the directional axis. We find that the modeled risk results are indeed anisotropic; in particular, they vary more along the north-south axis than along the east-west axis. We thus applied the appropriate conversion rates to obtain the final kriging maps, and kriged values, of modeled data. Finally, we compared the kriging values of modeled data to the observed data and find that 74.4% of the observed data lie within the 95% confidence interval of kriged model data.

While the 74% agreement between the observed and kriged model data suggests good model performance, we can use supplementary Fig. S10 to explore where the observed points do not fall into the 95% confidence intervals. We see that the model appears to perform well over much of the barrier reef, Turneffe Atoll and Lighthouse Caye, but that the observed data suggest significantly higher risk at Glover's Reef than the model prediction. Several areas along this reef were designated as no fishing zones in 2002, and the model uses this layer of information to attribute a relatively low risk to these places. However, the observed data suggest that by 2006 the coral ecosystems had not yet fully recovered.

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3. Supplementary Tables

Table S1. Exposure criteria for eight human activities and three habitats given the current zoning scheme (Cur), and three future management scenarios (Con=Conservation, IM=Informed Management, Dev=Development). Red = high risk (rank 3), yellow = medium risk (rank 2) and blue = low risk (rank 1).

Exposure

	Coral reefs				Mangrove forests				Seagrass meadows			
	Cur	Con	IM	Dev	Cur	Con	IM	Dev	Cur	Con	IM	Dev
Agriculture												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	0	0	0	0	0	0	0	0	0	0	0	0
Aquaculture												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	225	225	225	225	225	225	225	225	225	225	225	225
Coastal Development												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	1500	1500	1500	1500	1500	1500	1500	1500	1500	1500	1500	1500
Dredging												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000
Marine Fishing												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	100	100	100	100	100	100	100	100	100	100	100	100
Oil Exploration												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000
Recreation												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	100	100	100	100	100	100	100	100	100	100	100	100
Transportation												
Spatial overlap	*Calculated for each grid cell and region by the HRA model*											
Temporal overlap												
Intensity												
Management effectiveness												
Zone of influence	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000

Table S2. Consequence ranks for sensitivity criteria for the eight human activities, three habitats, one current and four future scenarios. Red = high risk (rank 3), yellow = medium risk (rank 2) and blue = low risk (rank 1).

Consequence - Sensitivity

	Coral reefs				Mangrove forests				Seagrass meadows			
	Cur	Con	IM	Dev	Cur	Con	IM	Dev	Cur	Con	IM	Dev
Agriculture												
Change in Area	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue
Change in Structure	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue
Frequency of Natural Disturbance	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
Aquaculture												
Change in Area	Blue	Blue	Blue	Blue	Yellow	Yellow	Yellow	Red	Blue	Blue	Blue	Blue
Change in Structure	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Red	Blue	Blue	Blue	Blue
Frequency of Natural Disturbance	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
Coastal Development												
Change in Area	Blue	Blue	Blue	Blue	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
Change in Structure	Blue	Blue	Blue	Blue	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
Frequency of Natural Disturbance	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
Dredging												
Change in Area	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Red	Red	Red	Red
Change in Structure	Yellow	Yellow	Yellow	Yellow	Blue	Blue	Blue	Blue	Red	Red	Red	Red
Frequency of Natural Disturbance	Blue	Blue	Blue	Blue	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow
Marine Fishing												
Change in Area	Blue	Blue	Yellow	Yellow	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Yellow
Change in Structure	Blue	Blue	Yellow	Yellow	Blue	Blue	Blue	Blue	Blue	Blue	Yellow	Yellow
Frequency of Natural Disturbance	Blue	Yellow	Yellow	Yellow	Blue	Blue	Blue	Blue	Yellow	Yellow	Yellow	Yellow
Oil Exploration												
Change in Area	Yellow	Yellow	Yellow	Red	Yellow	Yellow	Yellow	Red	Yellow	Yellow	Yellow	Red
Change in Structure	Yellow	Yellow	Yellow	Red	Yellow	Yellow	Yellow	Red	Yellow	Yellow	Yellow	Red
Frequency of Natural Disturbance	Red	Red	Red	Red	Red	Yellow	Yellow	Red	Red	Red	Red	Red
Recreation												
Change in Area	Yellow	Blue	Yellow	Yellow	Blue	Blue	Blue	Blue	Yellow	Blue	Blue	Yellow
Change in Structure	Blue	Blue	Blue	Yellow	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Yellow
Frequency of Natural Disturbance	Blue	Blue	Blue	Yellow	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Yellow
Transportation												
Change in Area	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Yellow	Yellow	Yellow	Yellow
Change in Structure	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Yellow	Yellow	Yellow	Yellow
Frequency of Natural Disturbance	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue	Blue

Table S3. Consequence ranks for resilience criteria for the three habitats (see table 1 for explanation of criteria). These ranks are the same for the current and three future management scenarios. Red = high risk (rank 3), yellow = medium risk (rank 2) and blue = low risk (rank 1). Note that these colors refer to the rank for the criteria in terms of how it influences risk and do not necessarily mean that the life history characteristic is high, medium or low for a particular habitat (see Table 1 and supplementary text for further explanation).

	Consequence - Resilience		
	Coral reefs	Mangrove forests	Seagrass meadows
Natural mortality	Yellow	Red	Yellow
Recruitment	Blue	Blue	Blue
Connectivity	Yellow	Yellow	Red
Recovery time	Yellow	Yellow	Blue

4. Supplementary Figures

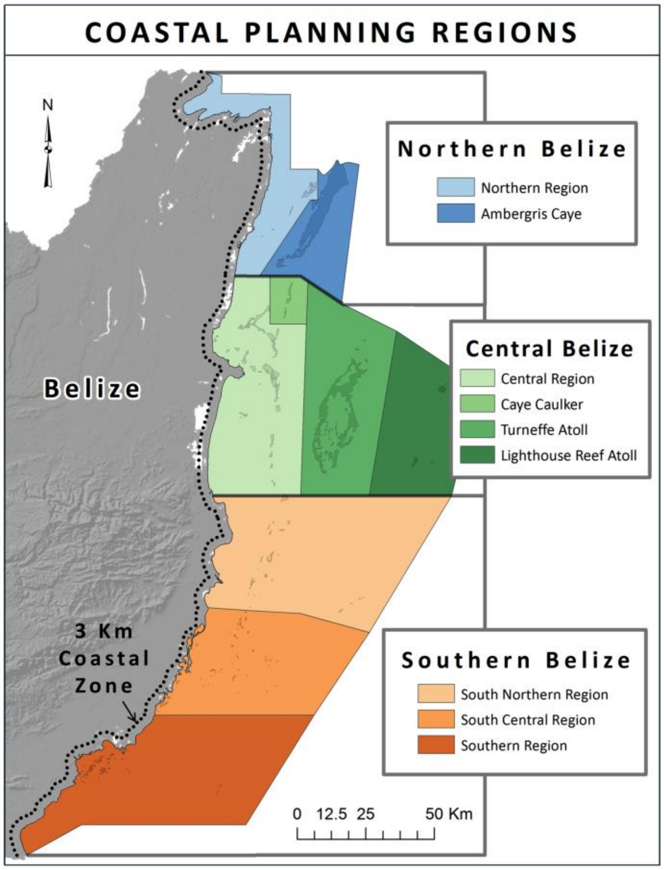


Figure S1. Nine coastal planning regions in Belize.

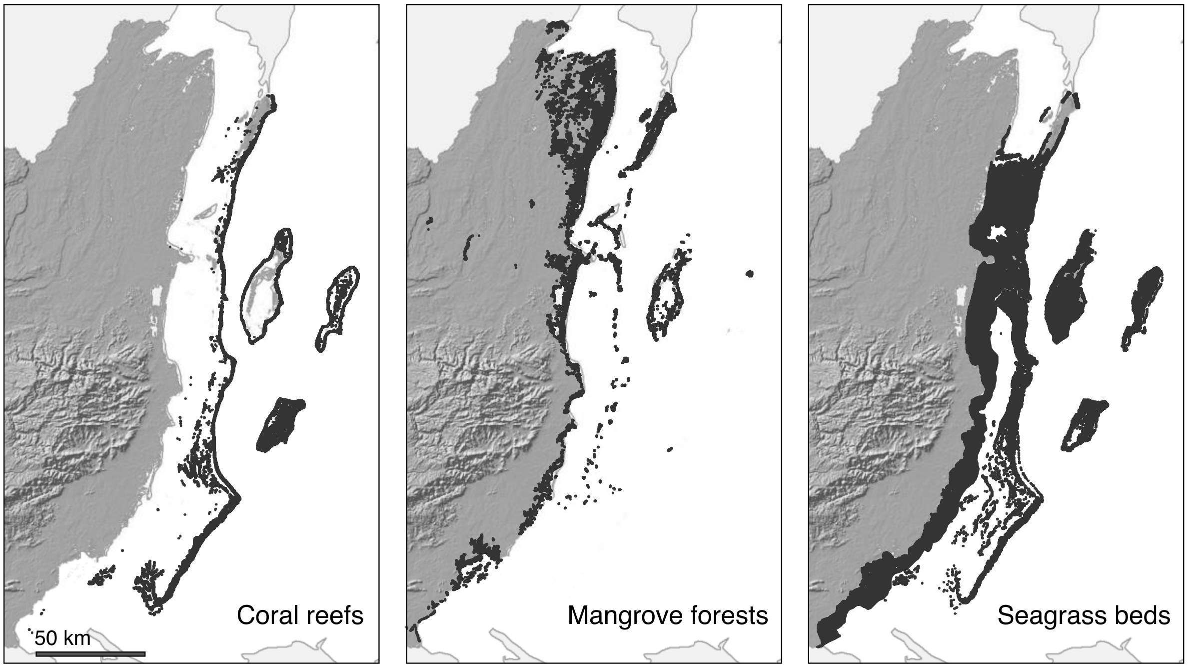


Figure S2. Three coastal habitats, coral reefs, mangrove forests and seagrass beds, for which we assessed the risk of human activities in Belize.

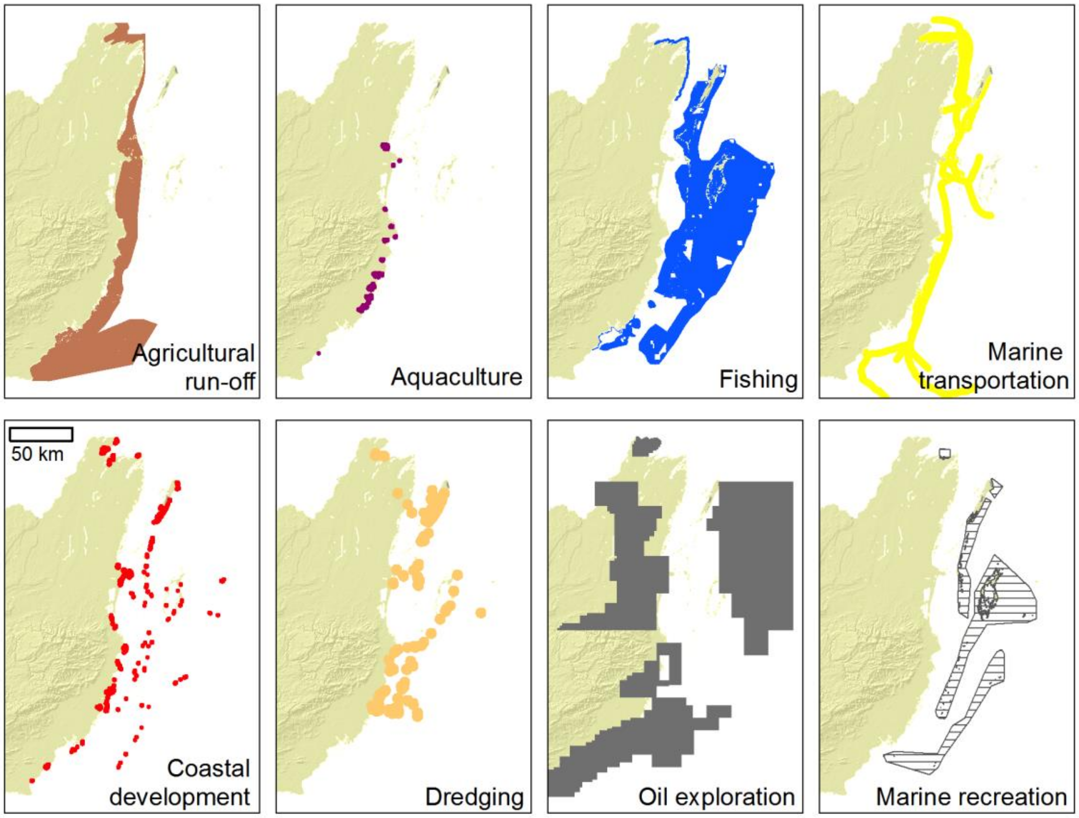


Figure S3. Current distribution of eight zones of human activity.

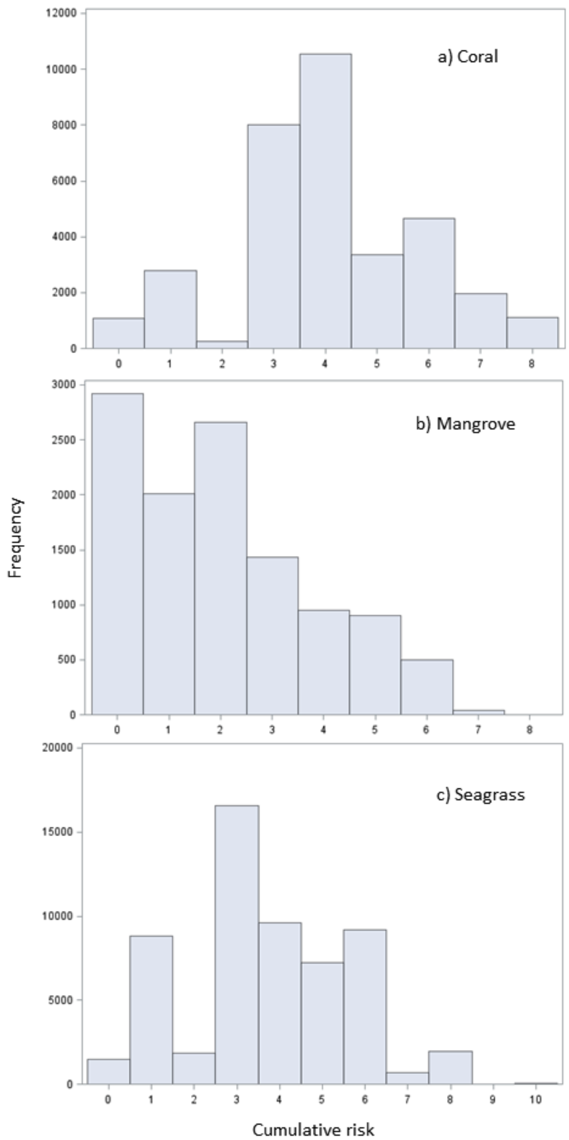


Figure S4. Frequency distribution of cumulative risk for (a) coral reefs, (b) mangrove forests and (c) seagrass beds for Belize where each count is a 500 m grid cell of habitat. Note the difference in x-axis for seagrass.

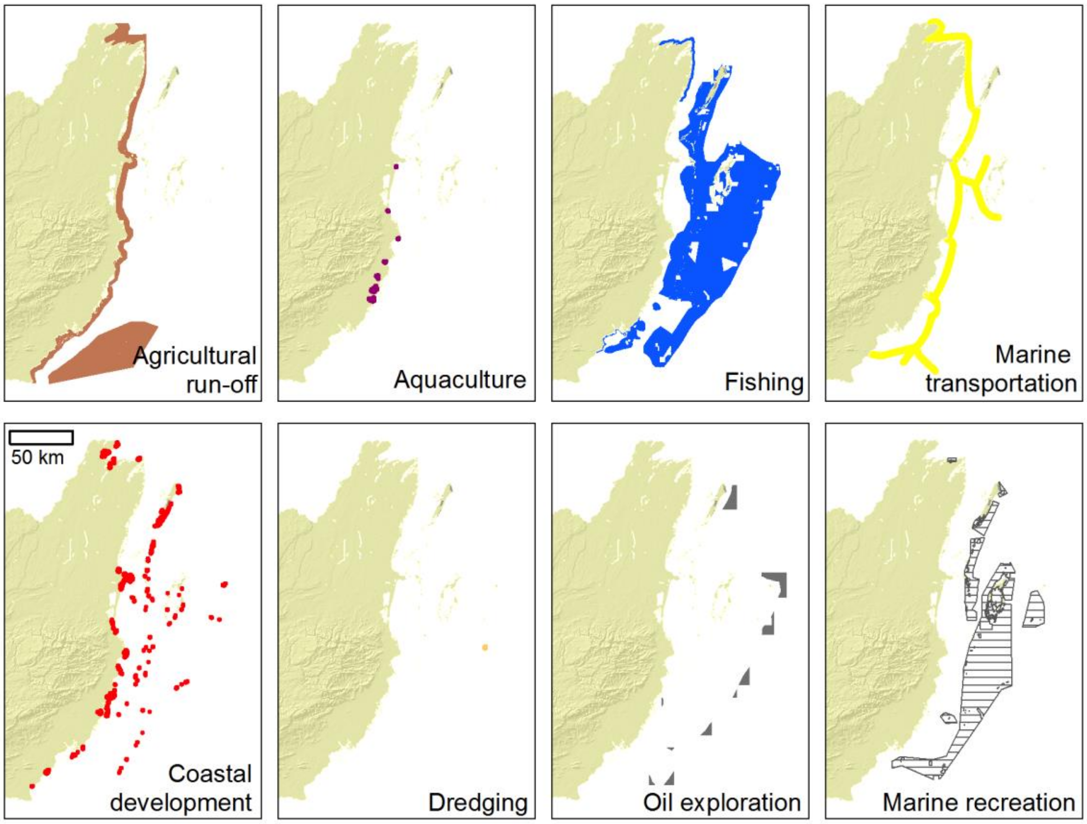


Figure S5. Distribution of eight zones of human activity for the Conservation scenario.

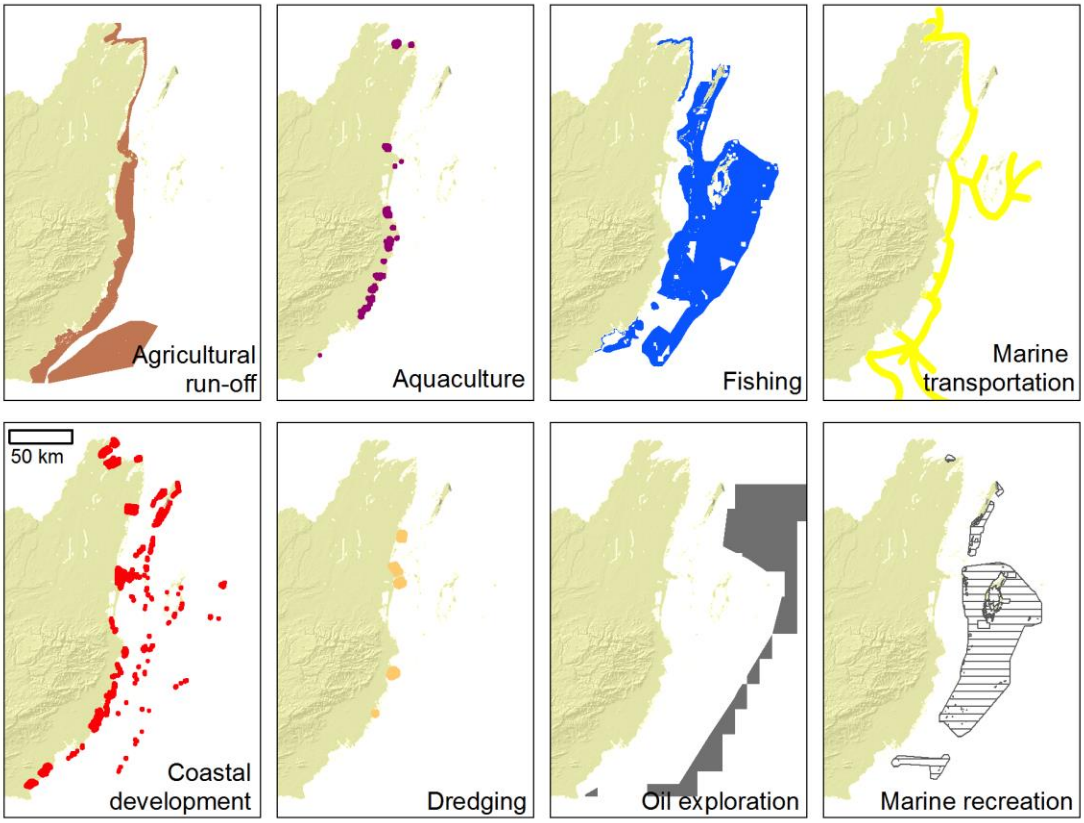


Figure S6. Distribution of eight zones of human activity for the Informed Management scenario.

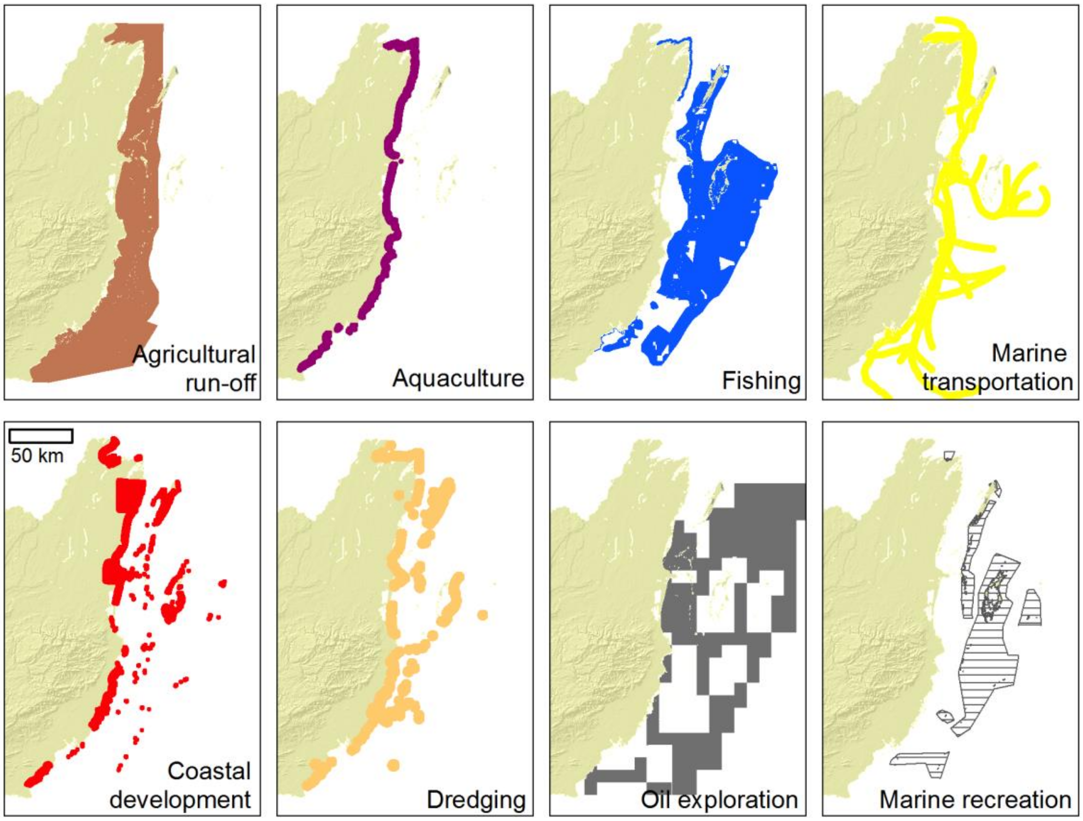


Figure S7. Distribution of eight zones of human activity for the Development scenario.

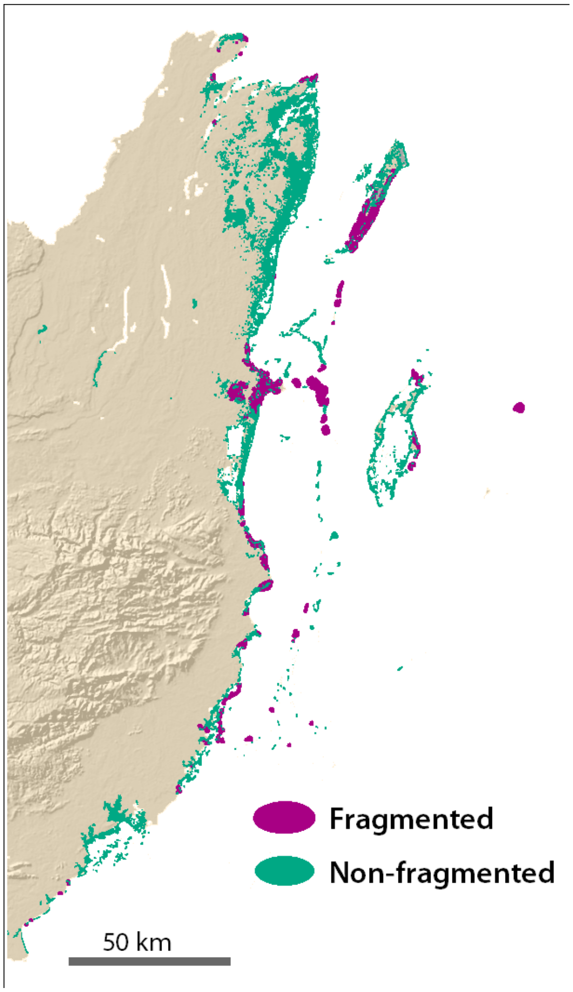
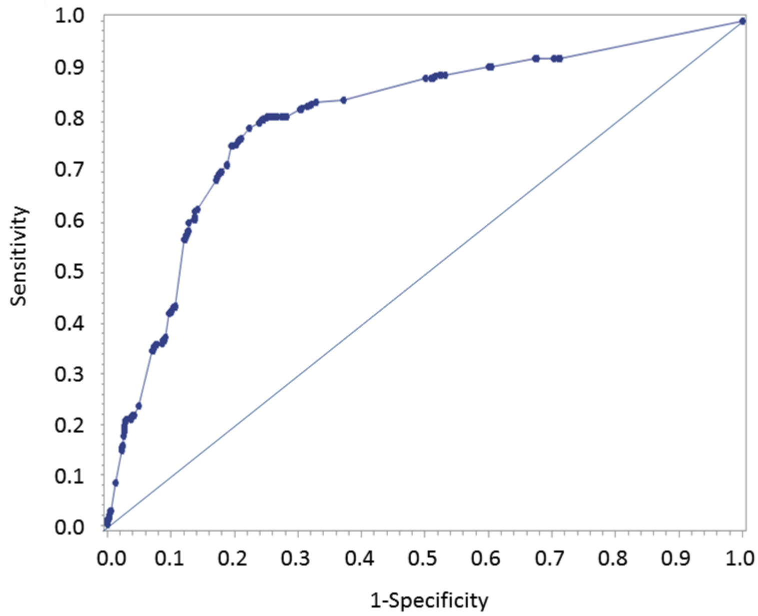


Figure S8. Distribution of fragmented (120 km^2) and non-fragmented (625 km^2) mangrove forests (data are reclassified from data reported in Cherrington *et al* 2010).

A)



B)

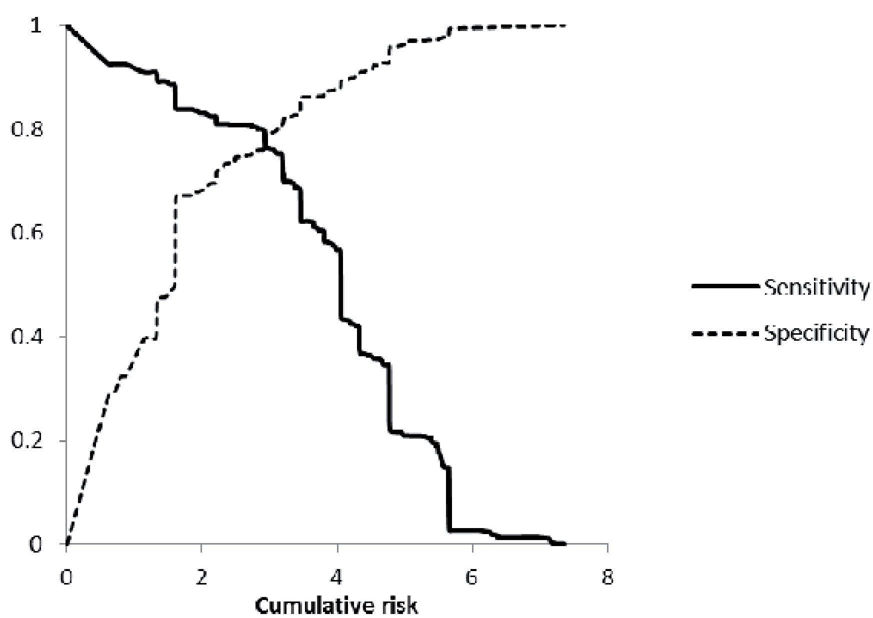


Figure S9. A) ROC curve for cumulative risk predicting mangrove fragmentation. Each point corresponds with a value of cumulative risk and possible threshold for distinguishing between fragmented and non-fragmented grid cells. The unity line represents the ROC curve for a null model for forest fragmentation and a visual lower bound for the ROC curve of the model to out-perform. B) Cumulative risk plotted against sensitivity and specificity, indicating the best threshold for distinguishing between fragmented and not fragmented areas is where the curves cross at ~2.9.

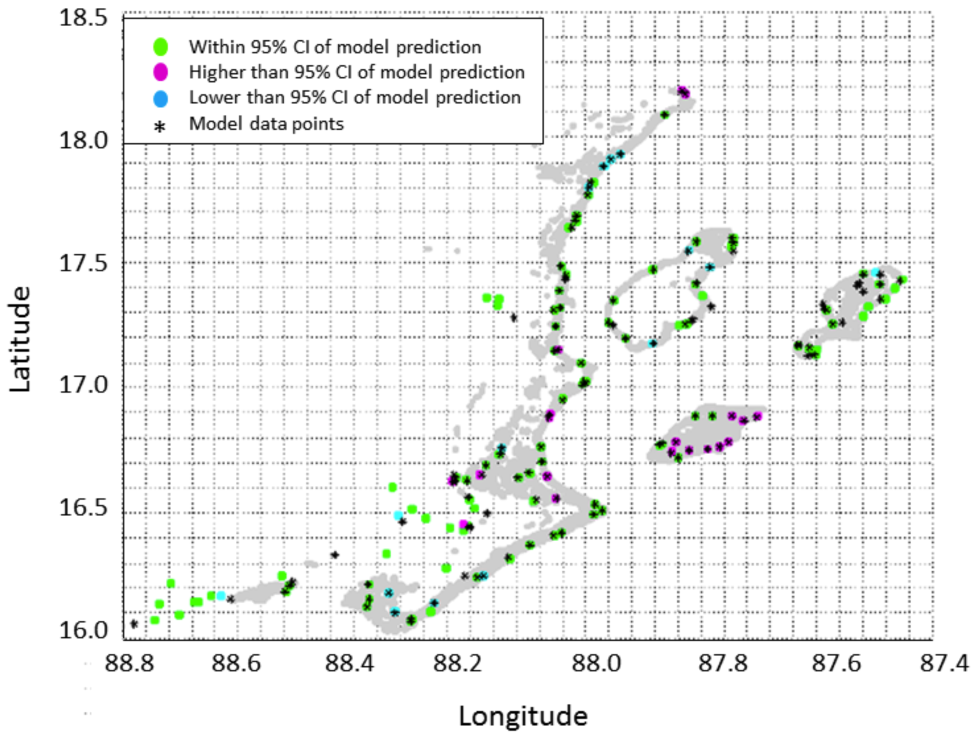


Figure S10. Performance of modeled cumulative risk for predicting observed index of coral health. Colored circles are observed data points and black stars are model data points. Through simple kriging of the model data, we produce confidence intervals for the model data at all locations of observed data. Colors indicate which points of observed data fall within or outside (above or below) the confidence intervals of the kriged model data.