Working Group 2 Report:

ECOLOGY AND BIODIVERSITY

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INTRODUCTION

Puerto Rico is home to some of the world's most impressive natural and manmade wonders, such as 799 miles of coastline with over 300 beaches, vibrant coral reefs, pre-Columbian ceremonial parks and artifacts, the coquí (tiny tree frog endemic to the islands), historic sugar cane and coffee plantations, El Yunque rainforest (the only subtropical rainforest in the U.S. National Forest System), Mona Island that is home to more than 100 endangered species, and 51 natural protected areas that conserve wetlands, dry forests, rainforests, caves and caverns, groundwater, cays and islets, and critical habitats for many species of flora and fauna. All these features are why Puerto Rico has been affectionately given the name *la Isla del Encanto*, the Island of Enchantment. Without healthy ecological systems Puerto Rico would not be able to boast such wonders.

Biodiversity and ecosystems are already more stressed than at any comparable period of human history (Staudinger et al. 2012). Human activities globally and in Puerto Rico have caused and may continue to cause a loss in biodiversity and natural resources through unsustainable land-use; chemical, heavy metal and nutrient pollution of our soil and water; diversion of water; habitat fragmentation and degradation; selective exploitation of species; and the introduction of non-native species, disease and pests. The current rate of biodiversity loss is greater than the natural background rate of extinction. Furthermore, these losses directly impact societies around the world and locally as we are also losing the services the ecosystems and species provide, such as fresh air, clean water, reduction in pollution and contaminant concentrations, drinking water protection, urban heat reduction, natural protection against storm surges and hurricanes, prevention of landslides, recreation and tourism opportunities, cultural and historical preservation, and even mental and spiritual well-being. The U.S. Global Change Research Program found that as species ranges shifts new community assemblages can also substantially alter ecosystem structure and function and the distribution of ecosystem services. Changes in precipitation regimes and extreme events can causes ecosystem transitions, increase transport of nutrients and pollutants to downstream ecosystems, and overwhelm the ability of natural systems to mitigate harm to people from extreme events (Staudinger et al. 2012).

Climate change is one of the most critical issues facing biodiversity and natural resource management in the world today. Land and ocean surface temperatures have warmed, the spatial and temporal patterns of precipitation have changed, sea level has risen, and we are experiencing more intense storms. These changes, particularly warmer regional temperatures, have affected the timing of reproduction in animals and plants and/or migration of animals, the length of the growing seasons, species distributions and population sizes, and the frequency of pest and disease outbreaks. Climate change is projected to affect all aspects of biodiversity; however, the projected changes have to take into account the effects from past, present, and future human activities. The effects of climate change, in terms of rising sea levels, increasing mean atmospheric and sea surface temperatures and changes in rainfall and weather patterns, are likely to be particularly severe for the ecological systems of the Caribbean islands and small island states. Small islands are variable in their marine, coastal and terrestrial biodiversity. Some are very rich. For example, coral reefs have the highest biodiversity of any marine ecosystem, with some 91,000 described species of reef taxa. And while terrestrial biodiversity is not as high as coral reefs of the Caribbean, endemism among terrestrial flora and fauna is particularly high in Puerto Rico. The International Union for the Conservation of Nature and Natural Resources (IUCN) ranks the countries with the highest percentage of endemic amphibians. Puerto Rico ranks 13th with 78.9%, above Chile, Papua New Guinea, the United States, Mexico, Brazil, India and China (IUCN 2012). Areas of high endemism should pay extra special attention to climate changes as already globally there is increasing evidence of population declines and localized extinctions that can be directly attributed to climate change (Staudinger et al. 2012).

Working Group 2: Ecology and Biodiversity of the Puerto Rico Climate Change Council (WG2 PRCCC) was tasked with assessing how Puerto Rico's ecosystems and species may be affected by climate change. The goal of this report is not to detail every potential outcome of climate change, but rather this project was a first attempt at identifying Puerto Rico's most knowledgable researchers and practitioners in the field, collecting the best available scientific knowledge, coming to agreement on key drivers of ecosystem changes and ways Puerto Rico's ecology and biodiversity may be altered, and identifying research gaps and information needs. The underlying vulnerabilities of ecosystems determine the adaptive capacity of Puerto Rico's flora and fauna, and therefore the current status and threats to each of the systems assessed were considered a priority throughout the work of the PRCCC.

REPORT ORGANIZATION

The members of Working Group 2: Ecology and Biodiversity are all highly respected in their fields of biology, ecology, oceanography, natural resource management, fisheries management, hydrology, geochemistry, forestry, and geology locally and abroad. The reviewers of this report were local experts from Puerto Rico, from the continental United States, and from overseas such as other Caribbean islands and the United Kingdom. Through a number of small sub-working group meetings, large working groups meetings, full PRCCC meetings, and a conference in San Juan, *Climate Change in the Caribbean 2011: Puerto Rico and the U.S. Virgin Islands*, WG2 members meet in-person to discuss the report compilation, data analyses, key concepts; peer-reviewed published studies, and gray literature. Subgroups of the PRCCC WG2 also meet to discuss targeted ecosystems such as coral reefs and forests. However, the majority of report writing and discussions occurred via email and PRCCC Research Library correspondences (a shared electronic research library). Additional correspondences occurred via the PRCCC email listserv (the PR-CC-L). We also consulted with and established the basis for discussions with other climate scientists and ecologists from other Caribbean islands the United States.

The Working Group 2 report is organized by the ecosystems and species the working group members determined were important for this first attempt at a climate change vulnerability assessment for Puerto Rico's natural resources (Table 1). In each section of the report, the current status and threats or non-climate change stressors (termed "underlying vulnerabilities" in this report) are discussed as well as the potential ways certain climate parameters (air and sea surface temperature, precipitation, extreme events, storms and hurricanes, sea level rise, and ocean acidification) may affect that particularly ecosystem or species assessed. (For more detailed information on the observed and projected changes for each of the climate parameters read the PRCCC Working Group 1 chapter.) The information compiled here is from literature review using a variety of databases as well as through the networks of our PRCCC members and from efforts to reach out to other Caribbean scientists and experts. As such some relevant studies may not have been received. The Working Group members and the PRCCC as a whole recognize that the information contained in this report is the best available knowledge as of July 2012 and will need to be continuously edited and expanded over the years as new and improved information becomes available. Working Group 2 also identified research and information needs to guide future studies and initiatives for natural resource management, conservation and ecosystem-based adaptation.

Table 1 Sectors to be assessed by WG 2 in report

Working Group 2: Ecology and Biodiversity Sectors Assessed
• Beach Ecosystems • Forests • Wetlands • Coral Reefs • Submerged Aquatic Vegetation (Seagrasses) • Coastal Lagoons • Bioluminescent Bays • Cays and Islets • Amphibians and Reptiles • Sea Turtles • Marine Mammals • Sea birds and Shore Birds • Coastal Fish Species • Pelagic Fish Species

BEACH ECOSYSTEMS

INTRODUCTION TO BEACH ECOSYSTEMS

A beach is a zone of loose material extending from the low water mark to a point landward where either the topography abruptly changes or transitional (from coastal to terrestrial) vegetation first appears (Cambers 1998). A wider definition of a *beach* includes the nearshore zone extending to a water depth of about 12 m where the waves are no longer able to move sediment on the bottom ((CANARI) 2008). A beach may consist of sediment ranging in size from clay to boulders. Beaches are very dynamic systems changing size, shape and even material composition from one day to another. A *dune* is an accumulation of windblown sand forming a mound landward of the beach and usually parallel to the shoreline. Coastal sand dunes provide extensive protection to many of the world's shorelines as they are important habitat for plants and animals, including rare and endangered species, as well as serve as natural barriers between coastal winds and waves and the human built environment of homes, roads, and buildings. The interaction of dunes with the adjacent beach and nearshore provides the essential basis of a stable shoreline, through the regular exchange of nutrients and minerals (Carter 1991). A *cliff* or *bluff* is a high, steep bank at the water's edge composed primarily of rock. Beach and dune systems should be seen as distinct but at the same time interacting through flows and impacts of organic and inorganic materials, such as, seeds and animals, sediment, salt spray and ground water flow.

STATUS OF BEACH ECOSYSTEMS IN PUERTO RICO

Globally it has been shown that 70% of the world's sandy beaches are eroding (Bird 1985, 1987). S*horeline retreat* or beach migration are the more accurate scientific terms, but *beach erosion* is the common term used by beachfront property owners. In the United States, approximately 86% of U.S. East Coast barrier beaches (excluding evolving spit areas) have experienced erosion during the past 100 years (Galgano et al. 2004). Widespread erosion is also well documented in California (Moore et al. 1999) and in the Gulf of Mexico (Morton and McKenna 1999). This trend is repeated in the Caribbean. Based on regular monitoring at 200 sites in nine eastern Caribbean territories over the period 1985-1995, 70% of the measured beaches were eroding and 30% were stable or accreting (Cambers 1997). Average erosion rates varied between 0.27 and 1.06 meters per year (0.9 to 3.5 feet per year), with islands impacted by hurricanes showing the highest rates. Specific beaches retreated inland by as much as 18 meters (59.1 feet)

> *"Beaches are nature's shock absorbers – they protect the coastline from the relentless power of the surf, and they protect themselves from destruction during storms. The countless grains of sand on the visible beach – and its offshore extension – are a mobile belt of cushioning. When waves strike the coast, the billions of sand grains on the full width of the beach move and roll past each other. Sometimes huge volumes of beach sand are moved either seaward or landward. A beach may change its shape drastically during a storm in the process of absorbing and dissipating the tremendenous energy of the waves".*
>
> *from Living with the Puerto Rico Shore*

during Hurricane Luis in 1995. Beach erosion is not a dominant process if there is a balance between the sediment input and the sediment output in the system under study. Seasonal changes over a period of several years followed by significant erosion during a particular storm event and slow but steady beach accretion after the storm, characterize systems with a balanced sediment input and output (ie. sediment budget). The erosion in the insular Caribbean is attributed to anthropogenic factors, e.g. sand mining, and poorly planned coastal development and man-made structures (*i.e.*, sea defenses); and to natural causes, such as winter swells and hurricanes. Anthropogenic or natural factors induce beach migration to reach a new equilibrium state sometimes resulting in net erosion, where the output of sediment is greater than the input, or total system disruption or loss. Tropical storms and hurricanes appear to be the dominant factor influencing the erosion, with beaches failing to return to their pre-hurricane levels. Indeed between 1995 and 1999, a period of severe hurricane activity for the islands of the northeastern Caribbean, it appeared that these numerous high-energy events introduced a certain vulnerability to the beach systems making recovery slower and less sustained (Cambers 2005). This loss of physical habitat has serious implications for the dependent flora and fauna.

The beaches of Puerto Rico are coastal resources of great importance. Including Vieques, Culebra, and other islets and cays, there are approximately 799 miles (1,285.86 km) of coasts, of which 129.2 miles (208 km) are beaches. In fact, Puerto Rico has over 300 beaches. The extension of the coast line is divided into 231 beaches in 42 of the 44 coastal municipalities. Of these, 97 have been classified as accessible or swimmable beaches by DNER (CPN 2002). There are three dominant groups of minerals, characteristic of Puerto Rico's beaches: quartz sands with feldspar; volcanic rock; serpentine and mineral group; and sands from calcium carbonate. Beaches in Puerto Rico are assets of public domain, or public trust lands, with minor exceptions. This is a legacy from Roman and medieval law which was transferred to Puerto Rico by the Spanish government as part of its port system and has remained despite the changes in sovereignty. Extraction of sand from the beaches for construction purposes once caused severe damage for a long period of time. The problem of extraction without proper authorization still persists, but the problem is more focused on river mouths rather than beaches, although some illegal dune extraction has occurred in recent years. Erosion has drastically reduced the width of some beaches and furthermore, water contamination interferes with the recreational use of some beaches.

Dune retreat and disappearance has also been widely documented throughout the Caribbean islands, including Puerto Rico. Extensive black sand dunes, 6 meter (18 feet) high, in St. Vincent and the Grenadines, were mined at Diamond Bay in the 1980s, leaving a flat coastal lowland devoid of vegetation and vulnerable to flooding (Cambers 1998, 2005). Cliff retreat and changes in rocky shores are less well documented, although associated with the widespread beach erosion there appears to be an increase in the exposure of beachrock ledges (Cambers 1998). Dunes provide an important barrier against the onslaught of waves during weather events along Puerto Rico's North Coast (Puerto Rico Coastal Zone Management Program 2009). In Puerto Rico, dunes (or in some cases, dune remnants) can be found in the municipalities of Isabela, Quebradillas, Camuy, Arecibo, Barceloneta, Loíza, and Carolina. Of these places, the largest concentration of sand can be found in Isabela, followed by Carolina and Loíza. However, the height and size of these dunes may be insufficient for protection of life and property in the long term. Due to the massive extraction of sand, very few dunes remain. The extraction of sand during the 1960s and 1970s for the construction industry has eliminated large expanses of protective dunes. The practice of beach extraction has been banned. At *Tres Palmitas* beach in Loíza and the coast along Isabela beach erosion has resulted in saltwater intrusion. It is estimated that approximately 2.5 million cubic meters (m^3) of sand have been extracted from dunes to the west of *Punta Jacinto* in Isabela, equivalent to the size of the Great Pyramid of Giza in Egypt. With the destruction of the dunes, the potential for shoreline damage due to storm surges has increased, although the magnitude of that potential is still unknown. The present-day conditions of some of the dunes are inadequate since their height, width, and stability offer neither security nor protection to life or property located in the coastal zone (Martínez et al. 1983, as cited by Valeiras 2007). Furthermore, the destruction of the dunes has contributed to coastal erosion as they form a reservoir of sand that replenishes the when beach levels are high and releases to supply sand to eroded beaches during storms (Coates et al. 2000). Additionally, the use of all-terrain vehicles along the beach has had negative consequences on the sand dunes and other coastal resources, such as wetlands and salt flats. These vehicles destroy dunes and rip out vegetation affecting dune equilibrium (Puerto Rico Coastal Zone Management Program 2009).

Bush et al. (1995) reported that hard rock shorelines off Guajataca on the North Coast and in parts of Old San Juan have retreat rates close to zero, while the retreat can be several meters in a single storm on sandy beaches such as El Tuque, the bluffs near Arroyo or Playita Machete on the south coast, or the beaches in the tourist areas of Condado and Isla Verde (Bush et al. 1995). More recently, in 2010 Dr. Maritza Barreto conducted a geomorphic assessment at selected beaches in Puerto Rico sponsored by the Puerto Rico Coastal Zone Management Program (Barreto 2010). The sites included: Playa Puerto Nuevo at Vega Baja, Playa Piñones at Loiza, Playa Fortuna at Luquillo, Playa Humacao at Humacao, Playa Salinas at Salinas and Playa Maní at Mayaguez (figure 1). This study integrated an evaluation of historical subaerial beach, beach profile and width changes studies. Historical subaerial beach changes were evaluated using vertical aerial photos from 1936 to 2010 (long-temporal scale study). Beach profile and width were measured using field techniques during August, September, October, November, and December 2010 (short temporal scale study). Data collections were done during normal conditions and after occurrences of storm systems (both tropical and extratropical). Historical beach changes showed loss of sand in all study sites during the 2007 period. Major beach changes were observed at Playa Fortuna, Luquillo and Playa Humacao. Major loss of sand was measured in Playa Fortuna (figure 2). Minor beach changes were identified in Playa Piñones, Playa Salinas, and Playa El Maní. The short temporal scale beach assessment indicated elevation and width suffered changes in all beaches included in the study. The final results of the study showed that geomorphic indicators (*i.e.*, eolianites, beach rock, coral reef, sand deposits, rocky shorelines) found in beach systems act as barriers that protect the beach plain from erosion processes. Furthermore, man-made structures and activities such as docks, housing, and deforestation affect sediment sources and transport that are important sediment inputs feeding beach systems and preventing erosion. Construction of breakwaters or jetties for shore protection, navigational purposes or marinas has had significant impact in Puerto Rico. A widely recognized example of a man-made structure creating severe erosion/accretion problems in Puerto Rico is the breakwater in Aguadilla. Additionally, as a result of past and current shoreline management practices, coastal development, erosion, and the already observed sea level rise (see PRCCC WG1 report) coastline around portions of Puerto Rico is retreating about one meter per year (1.0 m/yr), according to a USGS report. The authors considered sequences of past aerial photos to assess erosion and accretion (Thieler et al. 2007). Other publications have come to the same conclusion as the USGS report that most of Puerto Rico's shoreline is eroding (Bush et al. 2009; Morelock 1978, 1984; Thieler and Danforth 1993, 1994b; Bush et al. 1995; Thieler et al. 1995; Morelock and Barreto 2003). However, not all sections of all beaches in Puerto Rico are eroding. Some are stable or are accreting as they transition to new configurations and profile equilibriums as a result of beach migration; for example, Luquillo Beach, the beach between Punta Vacia Talega (the headland west of the city of Loíza) and the Loíza River mouth (just west of the city of Loíza) (Bush et al. 2009). In *America's Most Vulnerable Coastal Communities* the Geological Society of America's authors list seven areas of anthropogenic actions that contribute to Puerto Rico's overall shoreline hazard vulnerability: (1)

failure to take shoreline erosion into account; (2) constructing seawalls and revetments at the back of beaches; (3) insufficient construction setback from the shore; (4) other shoreline engineering structures like gabions (wire mesh baskets filled with cobbles or crushed rock); (5) sand mining; (6) inconsistent or uncontrolled shoreline stabilization; and (7) unrealistic cost-benefit considerations (Bush et al. 2009). These underlying vulnerabilities to Puerto Rico’s beach systems are expected to worsen due to sea level rise, increasing temperature, increasing carbon dioxide, and increasing intensity of tropical storms.

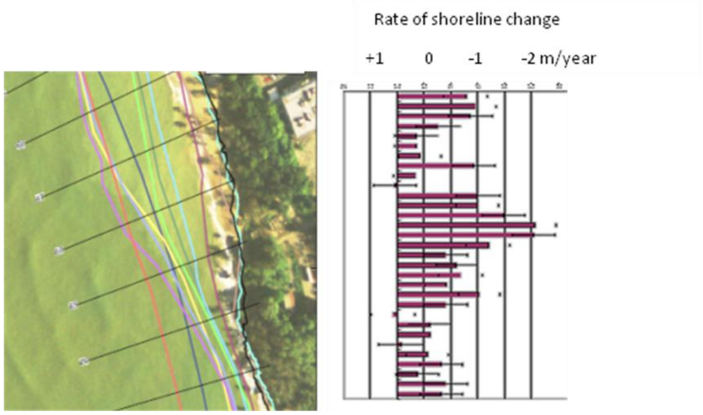


Figure 3 Coast lines every ~10 year near Rincon, PR, with matching coastal recession graph.

Figure 1 Beach study sites (source: Barreto 2010

Figure 2 Extreme subaerial beach loss at Playa Fortuna (December 2010) (source: Barreto 2010)

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE TO PUERTO RICO’S BEACHES

SEA LEVEL RISE

Sea level rise is one causative factor for dominant beach erosion (*i.e.*, inland migration) and was recognized as occurring long before the global attention from climate change discussions (Bruun 1962). The Bruun Rule (Bruun 1962), a descriptive rule of thumb, predicts that as sea level rises, sand is eroded from the upper beach and deposited on the offshore bottom so as to maintain a stable beach profile. This results in beach retreat so that for every 1 cm (0.4 in) of sea level rise, the beach retreats inland 1 meter (3.3 ft), figure 4, (note: this is a very rough approximation that applies to wide beaches with a slope of 1:100 where sediment transport is not altered by bedrock, buildings, seawalls or other obstructions, and does not include the presence of dunes as a sediment source).

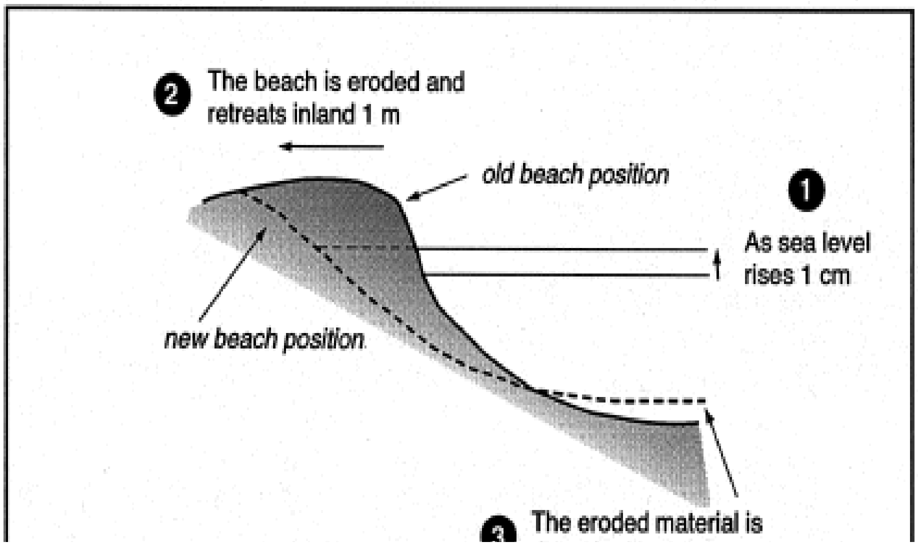


Figure 4 Beach retreat due to sea level rise (Cambers 1997)

However, such erosion may not take place regularly, but may come sporadically during storms (Williams et al. 1999). Thus, as the rate of sea level increases, the rate of beach erosion will increase. Where beaches cannot retreat inland because of other infrastructure or geological features, the rate of beach disappearance will increase. This will have implications for related systems such as dunes, coastal forests and emergent coastal wetlands. And due to the inter-connectedness of these systems, impacts to the beaches, impact other coastal ecosystems, which in turn impacts the beaches again. The concept that dunes are eroded during storms to replenish beach levels has been known since the mid-nineteenth century (Carter 1998). Sometimes the beach-dune exchange system is so well-balanced that storm losses equal interstorm gains, and the net shoreline movement is zero (Psuty 1990). With sea level rise and the diminishment of coastal sand dunes this beach replenishment after storms ecosystem service will be lost and beaches will experience greater shoreline retreat. (For other possible impacts to sand dunes from sea level rise see Christiansen et al. 1985; Christiansen and Bowman 1986; Boorman et al. 1989; Meulen 1990; Carter 1991).

There are many ecological and societal consequences of sea level rise and increased shoreline retreat. In addition to the devastating effects on the local and foreign tourism industry in Puerto Rico and on the ability of the coasts to protect life and property (see WG3 report), decreased beach area will reduce the availability of habitat for beach fauna and flora. A study by Fish *et al.* (2005) showed that a 0.5 m (1.6 ft) rise in sea level in the Caribbean would cause a decrease in turtle nesting habitat by up to 35% (Fish et al. 2005) (see sea turtle sections of this report for more information). The impacts go beyond marine fauna. For instance, certain species of land crabs, such as, *Gecarecinus lateralis* (black land crab) and *Cardisoma guanhumi* (blue land crab), found in Dominica and Puerto Rico, depend on being able to reach the sea to wash their eggs from their legs. This ability may be limited to decreased beach area, an increase in protective sea walls (as already causing problems for these crabs in Dominica), and rising sea levels. The estuarine areas these crabs inhabit will attempt to migrate inland as the seas rise and their required space may conflict with the coastal development that currently exists around Puerto Rico's coasts. This "coastal squeezing" may negatively affect the survival of land crabs. Additionally, ghost crabs or *Ocypode quadrata* inhabit Puerto Rico's beach systems and will also be affected by sea level rise. Some have stated that the decline in undeveloped beachfront habitat and alterations to the upper intertidal zone by coastal development may represent the greatest threat to the ghost crab (first to stranded oil on beaches from oil tanker traffic which may cause a reduction in the breeding rate and increased mortality at molting) (Gao and Xu 2002, SCDNR and NOAA 2000).

INCREASING TEMPERATURE

The affects of increasing temperatures on beach ecosystems are mostly related to flora and fauna and less to do with beach stability. Perhaps the most well-known affect from climate change on beach fauna are how increasing sand temperature of beaches influences the sex ratio of turtle hatchlings. To learn more about the outcomes and consequences of this see the sea turtle section of this report. Less well-known are the affects on smaller fauna of beaches. A simulation model study (Svensoon et al. 2006) showed that increased sea surface temperature caused faunal community shift and heightened the possibility of invasive species among species of barnacles. Mollusks, particularly the earlier life stages, are particularly vulnerable to changes in UV radiation, pH, and water temperature (Przeslawski et al. 2005).

> *"Coastal sand systems appear to be all but lifeless, but, in fact, they are vital, constantly changing, and quite interesting, ecologically. They are much more than just a barrier to the sea. On open wet sandy shorelines, most of the inhabitants live under the sandy surface in tubes and burrows or in spaces between sand grains. They surface when waves pass over their habitats...there are a variety of worms, crabs, sand protozoans, fungi, bacteria, and plankton (phytoplankton and zooplankton), among others present".*
>
> *from Guide to the Ecological Systems of Puerto Rico (Miller 2009: 177)*

INCREASING CARBON DIOXIDE

The great diversity of beach types around Puerto Rico reflects the fact that there are numerous sources of sand. Sand grains may be fragments of weathered rock or the skeletal remains of marine organisms. Basically, sand comes from one of four sources: riverbeds; erosion of the nearby cliffs, bluffs, and dunes; transport onto the beach via longshore currents; or landward transport of material from offshore (Bush et al. 1995). Many Puerto Rico beaches are composed of coralline sand derived from coral reefs and other marine organisms. As the oceans become more acidic, calcium carbonate exposed to sea water may dissolve, thereby reducing the supply of sand to the beaches. Similarly, the beachrock ledges that form protective barriers near the low water mark on many of the region's beaches consist of calcium carbonate cementing sand grains together – a process also likely to be impacted by ocean acidification.

The doubling of atmospheric carbon dioxide will also have a direct effect on the growth rate of plants, including dune plants and other beach vegetation, through the enhanced accumulation of carbohydrates during photosynthesis. In an enriched carbon dioxide environment, photosynthesis rates may increase up to 100% (Pearch and Bjorkman 1983). However, there are important metabolic differences between species and therefore some plants respond more readily to rises in carbon dioxide than others. In the case of dune vegetation, there are several things to consider for carbon dioxide-enhancement: 1) the metabolic response of the plants; 2) the community response, including competitive plant strategies; 3) the reaction of species to changing physical conditions including increased sand supply to the foredunes and, perhaps, changes in precipitation and groundwater (Carter 1991). The major dune forming plants in Puerto Rico and the Caribbean are C3 plants, such as *Sporobolus virginicus, Spartina patens, Ipomoea pes-caprae, Canavalia maritima* (these last two are creepers (DRNA P016) and it is expected that, if other factors remain unchanged, that dune grasses will grow more rapidly as global climate change proceeds. That is of course if the dune vegetation can adapt to other climate change-induced ecological adjustments like burial from sediment accumulation, progressive destabilization of dune systems from sea level rise, increasing drought due to enhanced levels of evapotranspiration and more competition for limited nutrients, etc (Carter 1991). It is unknown how well the vegetation will respond to such changes in Puerto Rico, and predictions are difficult to perform. There is a need for scientific evaluation and assessment of Puerto Rico's coastal vegetation and their responses to climate change.

TROPICAL STORMS AND HURRICANES

Puerto Rico was largely spared by major hurricanes during its decades of industrialization after World War II. In 1995 Bush et al. wrote: “Wind, waves, storm surge, and overwash are all most severe during hurricanes, but the majority of today’s coastal residents and property owners have never experienced the full force of a hurricane. The relatively hurricane-free period from the 1960s until the 1989 brush with Hugo led to an apathetic disregard of the hurricane menace and increased development in high-hazard zones” (Bush et al. 1995: 28). However, in the last two decades the island faced damaging storms. Hurricane Hugo (a Category 3 storm) passed over the northeast corner of PR and caused an estimated $1 billion in damages; Hurricane Georges (Category 2) crossed the island in 1998, leaving behind 12 dead and total damages amounting to $2.3 billion (CIER 2007).

The climate projections for Puerto Rico are that these occurrences will increase in intensity, but not necessarily frequency (see Working Group 1’s findings). As these events become more intense, the rate of beach erosion is likely to increase (Cambers 1996). Whether beaches recover after storms by replenishing their sand supply depends on the conditions of each beach system: (1) the existence and stability of dune systems; (2) the presence and proximity of coastal development and other man-made structures like sea walls and other hard shoreline structures; (3) the presence and health of naturally

Plants typical of sandy beach habitats in Puerto Rico are:

- *Sporobolus virginicus* (L.) Kunth. (beach dropseed or salt grass)
- *Paspalum vaginatum* Swartz. (sea paspalum)
- *Spartina patens* (Ait.) Muhl. (sea grass)
- *Cakile lanceolata* (Willd.) O.E. Schulz (sea rocket)
- *Canavalia maritima* (Aubl.) Thouars (sea bean)
- *Ipomoea pes-caprae* (L.) R. Br. (beach morning glory)
- *I. stolonifera* (Cyrillo) Poiret (fiddle-leaf morning glory)
- *Sesuvium portulacastrum* L. (sea purselane)
- *Cenchrus echinatus* L. (sandspur)

protective features such as coral reefs, seagrasses, mangroves, eolianites, and beach rock.

RESEARCH AND INFORMATION NEEDS FOR BEACH ECOSYSTEMS

1. *Inundation Mapping.* Accurate mapping of sea level rise inundation as well as storm surge combined with sea level rise inundation maps are greatly needed for Puerto Rico. At the time of this report Puerto Rico is lacking an official vertical datum (VD) for the islands. This means that accurate elevation data is simply not possible to obtain. LIDAR data, a high resolution data set, exists, however, without an accurate VD the results of an inundation analysis will not be suitable for coastal management. Without an official VD coastal areas cannot be adequately assessed for management purposes. Three studies exist for Puerto Rico to assess sea level rise on coastal areas using the North American Vertical Datum (*i.e.*, CDK 2010; Weiss et al. 2011; Capella 2010). They are an excellent first step for visualizations but more accuracy is required for scientific modelling and management decisions for beach ecosystems.

2. *Coastal Vegetation Responses.* As mentioned, more studies are needed on possible responses of coastal vegetation to climate change stressors. In particular, how will they respond to increased carbon dioxide in the atmosphere, sea level rise and associated saltwater intrusion, ecosystem migration, etc.

3. *Coastal dune geomorphology and response to sea level rise:* The morphology of the dune systems will be modified as the beach erodes, the source of sediment decreases and the energy of the waves reaches farther inland. The ability of the dune systems to migrate inland will mainly depend on two factors: the availability of inland space for the dune to migrate, which is greatly affected by the presence of permanent structures landward of the dune, and the availability of beach sediment to be transported by the wind and deposited landward of the beach. Each site will be different based on these factors and management alternatives need to be developed based on site characteristics. An assessment of current dune morphology should be conducted to evaluate the present condition of the dune systems in Puerto Rico and the factors that may affect the availability of sediment and dune migration process.

4. *Loss of Ecological Functions.* Studies on the consequences of losing ecological functions from birds and other wildlife as they migrate away for Puerto Rico's beaches. For example, impacts of seed dispersal of coastal vegetation.

5. *Beach and Dune Restoration.* Studies on best practices for beach and dune restoration for ecological functions as well as the protection of life and property.

6. *New Public perception studies*: New studies to assess the perception of the people that live and/or use the coastal spaces of Puerto Rico should be conducted. Knowing the way people think and perceive the coast can facilitate the development of management alternatives that consider the priorities and knowledge of local communities, and the creation of education programs intended to inform the public about the importance of coastal processes. Having an educated community may facilitate the implementation and maintenance of management practices.

SUMMARY OF CLIMATE CHANGE IMPACTS ON PUERTO RICO'S BEACH SYSTEMS

Globally it has been shown that 70% of the world's sandy beaches are eroding, and dune retreat and disappearance is widespread throughout the Caribbean islands, including Puerto Rico. The beaches of Puerto Rico are coastal resources of great importance. Including Vieques, Culebra, and other islets and cays, there are approximately 799 miles (1,285.86 km) of coasts, of which 129.2 miles (208 km) are beaches. In fact, Puerto Rico has over 300 beaches. Beach erosion is not a dominant process if there is a balance between the sediment input and the sediment output in the system under study. Seasonal changes over a period of several years followed by significant erosion during a particular storm event and slow but steady beach accretion after the storm, characterize systems with a balanced sediment input and output (ie. sediment budget). The erosion in the insular Caribbean is attributed to anthropogenic factors, e.g. sand mining, and poorly planned coastal development and man-made structures (*i.e.,* sea defenses); and to natural causes, such as winter swells and hurricanes. Anthropogenic or natural factors induce beach migration to reach a new equilibrium state sometimes resulting in net erosion, where the output of sediment is greater than the input, or total system disruption or loss. Tropical storms and hurricanes appear to be the dominant factor influencing the erosion, with beaches failing to return to their pre-hurricane levels.

Sea level rise is another causative factor for dominant beach erosion (i.e., inland migration) and was recognized as occurring long before the global attention from climate change discussions. For many beach systems, as the rate of sea level increases, the rate of beach erosion will increase. Where beaches cannot retreat inland because of other infrastructure or geological features, the rate of beach disappearance will increase. This will have implications for related systems such as dunes, coastal forests and emergent coastal wetlands. And due to the inter-connectedness of these systems, impacts to the beaches, impact other coastal ecosystems, which in turn impacts the beaches again. Also, as storm events become more intense, the rate of beach erosion is likely to increase. Whether beaches recover after storms by replenishing their sand supply depends on the conditions of each beach system.

There are many ecological and societal consequences of sea level rise, increased intensity of storm events, and increased shoreline retreat. In addition to the devastating effects on the local and foreign tourism industry in Puerto Rico Rico and on the ability of the coasts to protect life and property (see WG3 report), decreased beach area will reduce the availability of habitat for beach fauna and flora, such as sea turtles and crabs. Additionally, changes in temperature and carbon dioxide fertilization will affect beach and dune vegetation affecting flora and fauna as well as potentially disrupting the sand equilibrium of beaches. It is unknown how well the vegetation will respond to such changes in Puerto Rico, and predictions are difficult to perform. There is a need for scientific evaluation and assessment of Puerto Rico’s coastal vegetation and their responses to climate change, as well as accurate inundation mapping and studies on the consequences of losing ecological functions from birds and other wildlife as they migrate away for Puerto Rico’s beaches.

WETLANDS

INTRODUCTION TO WETLANDS AND THEIR ECOSYSTEM SERVICES

Since 1970, the U.S. Army Corps of Engineers (Corps) and the U.S. Environmental Protection Agency (EPA) have defined Wetlands for regulatory purposes (Section 404 of the Clean Water Act) as “…areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas." The Ramsar Convention on Wetlands and Waterfowl (1971, as amended) defines wetlands as areas that "…include a wide variety of habitats such as marshes, peatlands, floodplains, rivers and lakes, and coastal areas such as saltmarshes, mangroves, and seagrass beds, but also coral reefs and other marine areas no deeper than six metres at low tide, as well as human-made wetlands such as waste-water treatment ponds and reservoirs." Among the most widely accepted definitions is that of Cowardin et al. (1979), adopted by the U.S. Fish and Wildlife Service, "Land where an excess of water is the dominent factor determining the nature of soil development and the types of animals and plant communities living at the soil surface. It spans a continuum of environments where terrestrial and aquatic systems intergrade." These definitions comprise three main aspects - water, soil, and organisms - which are accepted by wetland scientists as the basis for recognizing and describing wetland environments. The three basic components of wetlands are summarized below (Schot 1999, Charman 2002):

1. **Ground water** -- Ground water (water table or zone of saturation) is at the surface or within the soil root zone during all or part of the growing season.
2. **Hydric soils** -- Soils are characterized by frequent, prolonged saturation and low oxygen content, which lead to anaerobic chemical environments where reduced iron is present.
3. **Specialized vegetation** -- Plants adapted for growing in standing water or saturated soils.

Water quality is not specified--salinity varies from fresh, to brackish, to marine, to hypersaline. Acidity may span the entire range of naturally occuring pH values. Depth of standing water in pools and hollows is usually too deep to walk through but too shallow to swim in. However, tidal flats are flooded daily, and many wetlands experience deeper floods from time to time. Emergent vegetation ranges from heavily forested swamps to nearly bare playas and mudflats.

In Puerto Rico, the most widely used classification system is Cowardin 1979. This system was developed by the U.S. Fish and Wildlife Service and establishes two basic types: coastal (tidal or estuarine wetlands) and inland (also known as non-tidal, freshwater, or palustrine wetlands). Working Group 2 members of the PRCCC recommended evaluating potential affects of climate variability and change for coastal or tidally influenced wetlands as well as those riverine, palustrine wetlands (non-tidal) which due to their geographic location may be impacted by air or water temperature increase, increase in salinity, increase in sedimentation due to runoff, as well as increase in salinity due to sea level rise and increased coastal inundations. This section is the first attempt to do so.

Eight basic wetland types are recognized in Puerto Rico based on studies completed by the U.S. Army Corps of Engineers (U.S. Army Engineer Waterways Experiment Station 1978) but only seven are relatively common:

1) **Saltwater aquatic** – Plants dominate in this type of wetland and may be floating or anchored to the bottom by roots. This type of wetland is permanently flooded by salt or brackish water. Seagrass beds are an example of this type of wetland (but, due to their importance, seagrasses are discussed in a separate section). Other examples are submerged flats covered by macroalgae or mudflats.

2) **Saltwater flats** – Wetlands with a vegetative cover of 25% or less. These wetlands are flooded by saltwater during high tide. Salt flats that are found in many coastal areas, often associated with mangrove forests, are an example of this type of wetland.

3) **Saltwater marsh** – Wetlands with 25% or more vegetative cover of which less than 40% of the plants are trees or shrubs. These wetlands are regularly or occasionally flooded with salt or brackish water. This type of wetland is very common in the U.S.

but not in Puerto Rico. Wetlands of this type that are found in Puerto Rico are dominated by ferns or succulent plants and are located behind mangrove forests.

4) **Saltwater swamps** – Wetlands with more than 40% cover of trees and shrubs. These wetlands are flooded occasionally or regularly by salt or brackish water. The most well-known example in Puerto Rico is the mangrove forest.

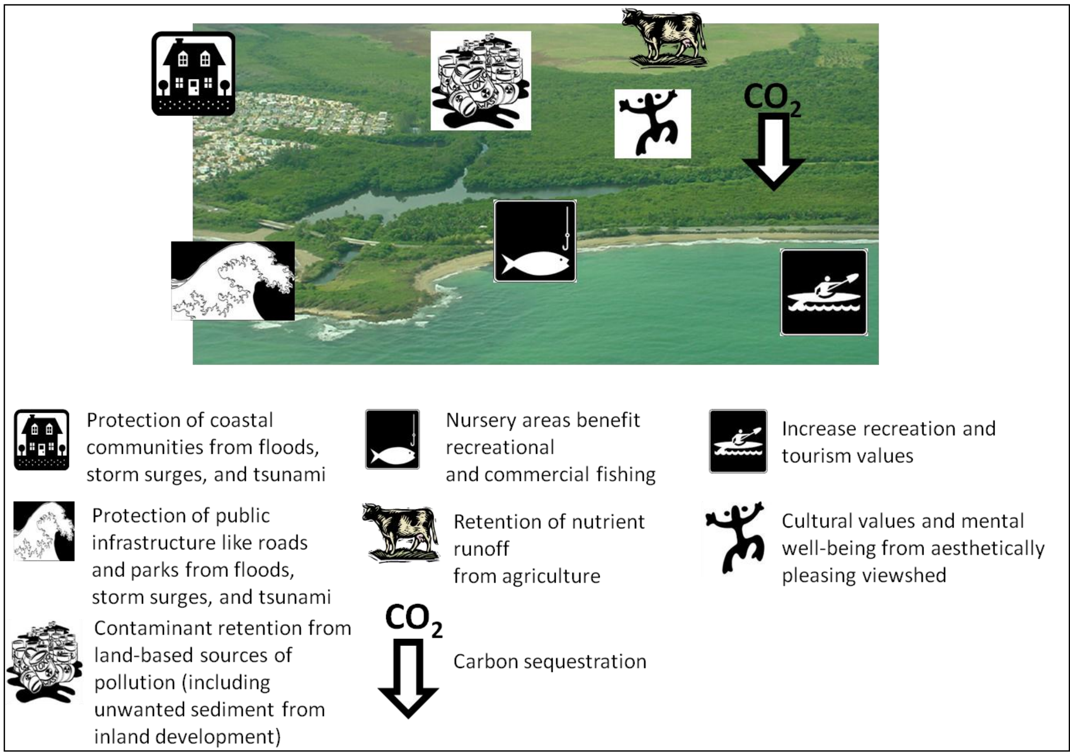
5) **Freshwater aquatic** – The dominant vegetation in this type of wetland is floating or anchored to the bottom by roots. These areas are permanently or almost always flooded. These wetlands are associated with creeks and rivers, lakes and ponds, and channels and reservoirs.

6) **Freshwater marsh** – Plants cover more than 25% of the area within this type of wetland and trees and shrubs cover less than 40% of the area. These wetlands are flooded occasionally or regularly. An example of a shallow freshwater marsh is a marsh dominated by species of grass adapted to wet conditions like juncos. Another example is a deep marsh where the most common species is cattails and water lilies or other floating species are found in deeper areas of the marsh.

7) Freshwater swamp – Trees or shrubs cover more than 40% of the area within these wetlands. These wetlands are flooded by freshwater occasionally or regularly. In addition to the trees that grow in these areas, many epiphytes (plants that grow attached to other plants for stability) and vines grow in freshwater swamps. It is likely that this type of wetland was more common in the coastal zone of Puerto Rico in the past but the conversion of lands for agricultural use, the cutting of trees for wood, and fires set by people to clear land have caused a drastic decrease in the size of these wetlands.

Emergent coastal wetlands are wetlands that emerge above sea level and are influenced by astronomical tides. They include tidal freshwater marshes, salt marshes, mangrove swamps, and tidal flats. Tidal freshwater marshes are found upstream of estuaries where the tide still influences water level, but the water is predominantly fresh. Brackish and salt marshes are found closer to the coast and are subjected to periodic flooding by the sea. Mangroves represent a large variety of plant families that have genetic adaptations which enable them to colonize saline coastal environments (Field 1995). Tidal flats are depositional formations found near estuaries and in front of mangroves.

Emergent coastal wetlands, specifically mangroves, provide many important ecosystem services, such as sequestering carbon and mitigating against possible disasters. Coastal wetlands and marine ecosystems hold vast stores of carbon.



Occupying only 2% of seabed area, vegetated wetlands represent 50% of carbon transfer from oceans to sediments. This carbon may remain stored in buried sediments for millennia. Loss of coastal wetlands and marine ecosystems sch as peatlands, forested tidal wetlands, tidal freshwater wetlands, salt marshes, mangroves, and seagrass beds lead to decreased carbon sequestration and can also lead to emissions of large amounts of CO_2 directly to the atmosphere. Mangroves are important carbon sinks, and sequester approximately 25.5 million tons of carbon every year. They also provide more than 10% of essential dissolved organic carbon that is supplied to the global ocean from land (Spalding 1997).

Carbon sequestration can be defined as the capture and secure storage of carbon that would otherwise by emitted to or remain in the atmosphere, or to prevent, carbon emissions produced by human activities from reaching the atmosphere, by capturing and diverting them to secure storage. It is through this process that agriculture and forestry practices remove CO_2 from the atmosphere (Melkania et al. 2007). Wetlands are a major sink of carbon dioxide. Wetlands' low decomposition rates associated to anaerobic soils "slow down" the increasing rates of carbon dioxide emissions in Earth's atmosphere. Increased carbon dioxide levels globally also results in higher primary productivity in most, if not all, wetlands. This "CO_2-fertilization" effect could enhance the standing stock of carbon in the ecosystem (Mitra et al. 2005). Carbon sequestration also provides associated ecosystem co-benefits such as increased soil water holding capacity, better soil structure, improved soil quality and nutrient cycling, and reduced soil erosion (Derner and Schuman 2007). Figure 5 provides a list of other ecosystem services provided in Puerto Rico by wetland ecosystems, such as contaminant retention from land-based sources of pollution, servicing as nursery areas for recreationally and commercially-important fish species, tourism value, and cultural and mental well-being from aesthetically pleasing viewsheds.

Figure 5 Ecosystem Services of Wetlands in Puerto Rico

Mangroves are wetland systems that have particularly tremendous social and ecological value. The annual economic value of mangroves, estimated by the cost of the products and services they provide, has been estimated to be $200,000 - $900,000 per hectare (Wells et al. 2006). The majority of the human population in the world is concentrated in coastal areas which are more vulnerable to natural disasters such as floods, wind generated waves, tsunami, and storm surges (Ramesh and Ramachandran 1999). Although mangrove ecosystems provide various services, the most important service provided by mangroves is the protection against coastal disasters and tsunamis (Rabindra Osti et al. 2009).

Generally, it is thought that mangroves prevent coastal erosion, and act as a barrier against typhoons, cyclones, hurricanes, and tsunamis, helping to minimize damage done to life and property (Upadhyay 2002; Dahdouh-Guebas 2006; Pearce 1996; Mazda, Y et al. 1997), although the use of coastal vegetation as a "bioshield" against extreme events like large storm surges and tsunami is still under investigation in the scientific literature (Feagin et al. 2010). Mangrove tree species that inhabit lower tidal zones can block or buffer wave action with their stems, which can measure 30 meters high and several meters in circumference (Dahdough-Guebas 2006). Mangroves defend the land from wind and trap sediment in their roots, maintaining a shallow slope on the seabed that absorbs the energy of tidal surges (Pearce 1999).

Mangroves protect the coast against waves, currents and storms and from coastal erosion. They are like live sea walls, and more efficient than concrete wall structures (Kathiresan, 2000). Some of the mangrove species, such as Rhizophora (or the red mangrove) act as a physical barrier against tidal and ocean influences and shields the coast by means of their large above-ground aerial root systems and standing crop (Dabdouh-Guebas 2005). It is also found that these mangroves species seem to act as a protective force towards this natural calamity (McCoy et al. 1996).

In India and the Phillipines, villagers tell how they have been protected from tsunamis, cyclones and other natural disasters in locations where mangroves are intact, but suffer where mangroves have been converted to shrimp farms or were lost due to human activities (Dahdouh-Guebas et al. 2005; Walters 2004). In Viet Nam mangroves have been observed to limit damage from tsunamis and cyclone waves and have led to large savings on the costs of maintaining sea dykes (Asian Wetland Symposium 2006). Remains of rows of mangroves planted by Maoris can still be seen in New Zealand to stabilize the coast indicating that mangroves helped in coastal protection (Vannucci, 1997).

Wave energy of tsunamis may be reduced by 75% if the waves pass through 200 meters of mangrove (Massal et al. 1999). It has also been found that 1.5 km belt of mangrove may be able to reduce entirely a wave one meter high (Mazda et al. 1997). It has been reported in literature that mangroves even protected villages and reduced death toll during floods and cyclones (Das and Vincent 2008). The benefits of mangroves for shoreline protection and storm damage control have been estimated to run into tens of thousands of dollars per km^2 in Sri Lanka and Malaysia. Studies carried out in Viet Nam show that the value over time of mangroves in protecting against extreme weather events lies around US$55,000 per km^2 (IUCN 2006).

It has been reported in the literature that mangroves reduce cyclone impact by dissipating wave energy and decreasing the impact caused due to cyclone (Badola and Hussain 2005; Fosberg 1971). Das and Bellamy (2007) also concluded that mangroves played an effective role in providing protection against cyclones. In one study by Narayan et al (2010) it was concluded that the mangroves have a definite positive effect on the port in terms of wave attenuation. From studies done world wide it has been concluded that cyclone impacts have been greatly lessened and losses in life and property damage would have been avoided if healthy mangrove forests had been conserved along coastlines (ASEAN 2009).

Other ecological and socioeconomic values of mangrove ecosystems are well known and increasingly recognized. Well-protected mangroves provide valuable life support services such as fisheries nurseries and habitat, coastal protection, or water quality services yielded. Mangroves also play a vital role in the interconnected nature of coral reefs, mangroves, forests, and sea grasses, which provide joint benefits to human populations. In turn, these economic benefits accrue to local, national and global populations. The economic costs of mangrove ecosystem loss and degradation is expected to be felt the hardest by the economically disadvantaged coastal communities, specificially due to their limited options for alternative livelihoods and coping capacity against hazard events.

STATUS OF WETLANDS WORLDWIDE, IN THE CARIBBEAN, AND IN PUERTO RICO

WORLDWIDE

Between 1980 and 2005, 35,000 km^2 of mangroves were cleared and drained; about 20% of the total area of mangroves was lost (Spalding et al. 2010). Coastal wetlands are under direct and increasing threat from land use changes pressures (Coleman et al. 2008), from indirect impacts of upstream disruption to sediment supply, and from development pressures and rising sea level at the coast. Altered sediment supply exacerbate sea level rise, with local rates commonly twice, and in some locations as much as 10 times global rates (Syvitski et al. 2009). Large areas of coastal wetlands worldwide have been drained and converted to other uses. Seagrass beds have declined by 29% since the 19th century, and the rate of loss is estimated to have increased by an order of magnitude in the past 40 years (Waycott et al. 2009). Salt marshes and freshwater tidal marshes have lost more than 50% of their historical global coverage, with the current rate of loss estimated at 1-2% per year (Secretariat of the Convention on Biological Diversity 2010).

CARIBBEAN WETLANDS

Emergent coastal wetlands, particularly mangrove forests are widespread and an important resource in the insular Caribbean (Spalding et al. 1997). Despite the attempts to protect them by implementing coastal management and planning programs and declaring them Wetlands of International Importance or Ramsar sites, there is still a net loss of mangroves and salt marshes in the insular Caribbean (Bacon 2000). Of the 195 wetland sites investigated by Bacon in 1991, some 47% showed evidence of serious resource degradation resulting from human impact and all sites showed some damage (Bacon 1991; 1995). A range of impacts were identified with the most important being:

- Landfill and solid waste dumping;
- Vegetation clearing, particularly unregulated cutting for timber or charcoal production;
- Reclamation for agriculture, including some fish pond construction;
- Hydrological alteration, particularly by roadways or flood diversion schemes;
- Pollution by factory and domestic effluent.

PUERTO RICO'S WETLANDS

Lugo and Cintron (1975) classified the Puerto Rican mangroves into two groups based on climate and other criteria. The northern-coast mangroves are subject to high wave energy, high precipitation, and river runoff. The southern-coast mangroves are subject to low wave energy regime, low precipitation and low river runoff if any. Basin and riverine mangroves predominate on the northern-coast type mangroves, while fringe mangroves predominate on the southern-coast type mangroves. All types are subtropical (sensu Holdridge), and subject to the same temperature regime. According to the 1989 National Inventory of Mangroves for Puerto Rico (Torres Rodriguez 1994), there are 97 “mangrove sites”, or places where mangrove ecosystems occur around the island; these sites contain typically various fragments of mangrove forest. Mangroves cover about 1,900 ha in northeastern Puerto Rico and account for 22% of all of the island’s mangrove forests. The largest remaining areas are found east of the Rio Espíritu Santo, in the Las Cabezas Reserve, in the Ceiba Forest, and at Roosevelt Roads. These mangroves are typically characterized as: *Rhizophora mangle* dominated stands on coastal and estuarine fringes; *Avicennia germinans* and *Laguncularia racemosa* in basins as pure or mixed stands; or *Conocarpus erectus* and the shrub *Suriana maritima* in mixed stands. Species richness is very low. Mangrove forest structure is strongly affected by salinity, nutrient availability, and storm disturbance. Typical mixed mangrove *Laguncularia racemosa-Avicennia germinans* associations growing at the Las Cabezas Reserve and Roosevelt Roads areas had canopy heights of 12 m, stem densities (stems ≥ 10 cm) of 344 stems ha-1, and relatively low basal area (15 m_2 ha_{-1}) and aboveground biomass (57 t ha_{-1}) (Gould et al. 2006).

Mangrove forests have gone through a period of decline and partial recovery related to human activities such as altered hydrology for agricultural practices, urbanization, and the implementation of conservation activities (Martinuzzi et al. 2009). Martinuzzi et al. (2008) used historical information and long-term landscape analyses to relate land use changes over the last 200 years. The population density of Puerto Rico has undergone dramatic increases in the last 200 years from about 50 people/km^2 to more than 400 people/km^2. Four distinct eras of mangrove change were visible over this time period. During the agricultural era (1800-1940) the area of mangroves declined 45%. As the economy changed to industrial in the late 1940s the area of mangrove increased due to reduced land use pressure on the wetlands. However, with urban expansion between 1960s and 1970s another decline was evident. Public concern for mangrove conservation resulted in the legal protection of all the mangroves in 1972, and since then their area has expanded. Between 1977 and 2002 the mangrove coverage in Puerto Rico increased by 12% from 7,443 hectares to 8,323 hectares. This study also found that past human activity altered the original proportion of mangrove species. The area of mangroves increased in rural and urban/rural sites, but not in urban ones. In urban sites the mangroves decreased by 2%. The number and size of mangrove-forest fragments were impacted by land use, and urban areas had fewer and smaller fragments than vegetated areas. Fifty five percent (55%) of today's mangroves are located in rural sites, thirty eight percent (38%) in urban/rural sites, and seven percent (7%) in urban sites. In addition, fifty nine percent (59%) are located in protected sites, or within reserves, twenty six percent (26%) are in partially protected sites, or sites that are partially included within reserves and fiftenn percent (15%) are in non-protected sites, or outside of reserves.

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON WETLAND ECOSYSTEMS

Climate modeling studies generally estimate that that global temperature will rise a few degrees in the next century (see Working Group 1 report). Such warming is likely to raise sea level by expanding ocean water, and melting glaciers and portions of the Greenland Ice Sheet (Titus and Narayanan 1995). Warmer polar ocean temperatures could also melt portions of the Ross and other Antarctic ice shelves, which might increase the rate at which Antarctic ice streams convey ice into the oceans (Titus and Narayanan 1995). The dynamics and evolution of wetlands along the coasts is strongly influenced by the gradual rise of sea level that has occurred during the last half of the Holocene period and continues to the present. Rising sea level drives wetland evolution by changing the hydrology, hydrodynamics, and sediment dynamics of the coastal zone (Nuttle *et al*. 1997). When wetland elevation is low relative to sea level, frequent inundation by tides enhances the supply of suspended sediment and nutrients to the wetland. This stimulates accretion by sedimentation. As well, the enhanced nutrient supply supports abundant growth of vegetation, and this contributes to accretion through the production of roots and rhizomes and by trapping suspended sediment for incorporation into new wetland sediment. If the elevation of the wetland increases relative to sea level, tidal inundation becomes less frequent; accretion slows; and the rate at which wetland elevation increases is reduced (Nuttle et al. 1997).

Our current understanding of how wetlands will respond to sea level rise is the prediction that they will either migrate landward or be lost, according to the Brinson *et al.* (1995) model of coastal habitat migration. According to Titus and Narayanan (1995), there is a 10 percent chance that climate change will contribute 30 centimeters (cm) by the year 2050, 65 cm by the year 2100, and two (2) meters (m) by the year 2200.

Global climate change is expected to exacerbate the loss and degradation of mangrove forests and the loss or decline of their species, and to harm the human populations dependent on their services (Millennium Ecosystem Assessment, 2005). Coastal wetlands in Small Island Developing States are especially vulnerable to impacts from relative sea level rise since they have a limited capacity to adapt, including limited space to accommodate landward migration of mangroves and other coastal ecosystems. After surveying over 200 coastal wetland sites in the insular Caribbean, Bacon (1994) suggests that responses to sea level rise would be quite variable since there is a wide range of wetland types and geomorphic settings in the region.

SEA LEVEL RISE

Salt water and tides combine to create an environment in which most plants, except salt-tolerant species (halophytes), cannot survive. Mangrove swamps, dominated by halophytic shrubs or trees, are common in warm climates such as southern Florida and Puerto Rico. Tidal freshwater wetlands form in upstream coastal wetlands where the influence of salt water ends.Under natural conditions, coastal wetlands adjust to rising seas and changes in local storm patterns, but climate changes and human activities that alter natural conditions disrupt wetland hydrology, biogeochemical cycling, and other processes that sustain wetlands (Morris et al., 2002). The potential effects from rising sea levels on mangroves are:

- Probable loss of total mangrove areas due to erosion of the seaward margin of the mangroves and loss of protective lagoon bars and sea barriers;
- Relocation and migration of mangroves inland, rather than overall loss. This landward migration can be obstructed if the landward margin of the mangrove area is steep or if there are seawalls and other developments (coastal squeeze), thereby reducing the areas of coastal ecosystems;
- Change in mangrove forest structure. Landward replacement of black mangrove (*Avicennia*) by red mangrove (*Rhizophora*) and possible increased growth and productivity of the mangrove areas from carbon dioxide fertilization;
- Increase in mangrove area and changes to associated wetland community types and distribution. Saline intrusion into inland freshwater wetlands and rejuvenation of salinas and scrub mangrove sites.

Mangrove forests in the Insular Caribbean are of four main functional types (Lugo & Snedaker, 1974) based on edaphic and hydrologic conditions: riverine, fringe, basin and scrub. Bacon (1994) stressed the importance of site-specific analysis and recommended that more attention be paid to site physiography, hydrology and ecology in predicting responses of tropical coastal wetlands to sea level rise. For instance, among the most at-risk wetland ecosystems are mangroves, but the good news is that not all coastlines with mangrove forests are projected to experience negative impacts to sea level rise. Mangrove ecosystems on low relief islands and those deprived of sediment are especially vulnerable. In contrast, mangroves with ample sediment supplies and/or room to move inland are likely to survive projected rates of sea-level rise (McLeod and Salm 2006:6). If the sedimentation rate keeps pace with local rising sea level, mangrove forests could remain largely unaffected (Snedaker, 1993; Ellison, 1996). Mangroves have demonstrated different tolerances to changes in sea level, salinity, and storms (McLeod and Salm 2006:6). However, their ability to migrate landward or seaward is also determined by local conditions, such as infrastructure (e.g., roads, agricultural fields, dikes, urbanization, seawalls, and shipping channels) and topography (e.g., steep slopes). If inland migration or growth cannot occur fast enough to account for the rise in sea level, then mangroves will become progressively smaller with each successive generation and may perish (UNEP 1994).

CHANGES IN SALINITY

Increases in salinity can be due to sea level rise, groundwater depletion owing to reduced freshwater flux, ground water extraction or reduced rainfall. This can result in reduced seedling survival and growth, and decreased photosynthetic capacity (Ball & Farquhar, 1984). Loss of freshwater wetlands with saline intrusion is documented in Florida (Ross *et al.*, 1994).

INCREASED TEMPERATURE

Temperature affects the growth, survival, reproduction, and distribution of plants and animals. Through its effects on basic metabolic processes such as respiration, photosynthesis, budburst, egg laying, and food availability, an increase in temperature could, in theory, alter biological diversity at every level in the food web. As air and water temperature increase, species ranges will likely expand toward environments that are presently cooler (IPCC 2007a; Parmesan and Yohe 2003). If dispersal capabilities are limited or suitable habitat is not available, local extirpations and extinctions are likely to occur (Thomas et al. 2004). Increased temperatures in terrestrial habitats will reduce streamflow and alter water quality into deltas, estuaries and in coastal regions, causing addition change and/or degradation of coastal ecosystems. However, mangroves are not expected to be adversely impacted by the projected increases in sea surface temperature (Field 1995 as cited in McLeod and Salm 2006). Most mangroves produce maximal shoot density with mean air temperature rises to 25^{O} C and stop producing leaves when the mean air temperature drops below 15^{o} C (Hutchings and Saenger 1987). At temperatures above 25^{o}C, some species show a declining leaf formation rate (Saenger and Moverly 1985). Temperatures above 35^{o} C have led to thermal stress affecting mangrove root structures and establishment of mangrove seedlings (UNESCO 1992; Banus, 1983). At leaf temperatures of 38.0 - 40.0^{o} C, almost no photosynthesis occurs (Clough et al. 1982; Andrews et al. 1984 as cited in McLeod and Salm 2006). Combined with higher atmospheric carbon dioxide levels, increased temperature is expected to increase mangrove productivity, by increasing growth and litter production and expansion of the geographical range of some species (Ellison, 1996 as cited in McLeod and Salm 2006). Some scientists have suggested that mangroves will move poleward with increasing air temperatures (UNEP 1994; Field 1995; Ellison 2005). Although it is possible that some species of mangrove will migrate to higher latitudes where such range extension is limited by temperature, Woodroffe and Grindrod (1991) and Snedaker (1995) suggest that extreme cold events are more likely to limit mangrove expansion into higher latitudes.

INCREASED CARBON DIOXIDE

Plants respond to higher carbon dioxide concentrations (CO_2) with increases in net photosynthesis and water use efficiency, leading to greater productivity. This “CO_2-fertilization” effect could affect the standing stock of carbon in the ecosystem. Some plants are more responsive to increases in CO_2 —these are called C3 species. C4 species are less responsive to CO_2 because they are already quite efficient at capturing CO_2. Because of species differences in sensitivity to CO_2, the composition of a wetland is one factor that will determine whether increases in CO_2 will have an effect or not. One simple prediction is that C3 species will increase in abundance relative to C4 species in a mixed community as CO_2 increases. Mangrove stands would remain under this scenario due to lack of competition.

Increased levels of CO_2 are expected to enhance photosynthesis and mangrove growth rates (UNEP 1994). For example, increased levels of CO_2 significantly increased photosynthesis and the average growth rates in two Australian mangrove species, *Rhizophora stylosa* and *Rhizophora apiculata*, but only when grown at lower salinity levels (Ball et al. 1997). Additionally, it is expected that there will be an increase in productivity in mangroves and more efficient water use due to reduced stomatal conductance (Warrick *et al.*, 1987).

PRECIPITATION CHANGES

Changes in precipitation and run-off patterns appear likely as climate change intensifies, but the uncertainties are large. Changes in freshwater runoff patterns can affect coastal and estuarine biota through several pathways. If freshwater flows to the coast decrease or the timing of freshwater runoff to estuaries changes, the salinity of coastal wetlands and estuaries is likely to increase. The distribution of coastal biota is closely linked with salinity of water and soils. Few studies have documented the interactions between runoff, salinity, and species distribution. Earlier and faster snowmelt in temperate and polar regions due to increasing global temperatures signify changes in freshwater and nutrient delivery to the coast from meltwater-dominated watersheds. Changes in the timing of freshwater runoff to estuaries could affect the productivity of many estuarine and marine fishery species (Nicholls et al. 2007). Changes in runoff can also affect sediment delivery, which has important implications for coral reef ecosystems. Freshwater inflows into estuaries influence water residence time, vertical stratification, salinity, control of phytoplankton growth rates, and the flushing of contaminants in estuaries. In estuaries with very short water residence times, phytoplankton are generally flushed from the system as fast as they can grow, reducing the estuary's susceptibility to eutrophication and harmful algal blooms. Wetland response to climate change depends on local interactions between sediment and organic matter accumulation, hydrology, subsurface processes, and storm events (Reed, 1995; Cahoon et al. 1995). Over the past several decades, water management systems (*i.e.*, Valle de Lajas- Laguna de Guánica and Caño Tiburones), urban sprawl and the increased frequency and intensity of storms have altered the timing and amount of sediment delivered to wetlands.

Changes in rainfall patterns caused by climate change may have a profound effect on both the growth of mangroves and their areal extent (Field 1995; Snedaker 1995). It is known that during long-term droughts some mangrove forests contract while salt flats expand (Cintron et al. 1978). Decreased precipitation results in a decrease in mangrove productivity, growth, and seedling survival, and may change species composition favoring more salt tolerant species (Ellison 2000, 2004). Decreased precipitation and increased evaporation is also likely to result in a decrease in mangrove area, decrease in diversity, and projected loss of the landward zone to unvegetated hypersaline flats and a decline in growth rates (Ellison 1996; Snedaker 1995).

On the other hand, increasing precipitation may allow for a greater accumulation of sediment and organic matter from runoff resulting in an increased elevation of mangroves (which in turn might help the mangroves adapt to sea level rise). Greater amounts of rainfall could increase mangrove area, diversity of mangrove zones, and mangrove growth rates in some species (Field 1995). Increased precipitation may also allow mangroves to migrate and outcompete saltmarsh vegetation (Harty 2004). Increased rainfall should result in reduced salinity and exposure to sulphates and an increase in the delivery of terrigenous nutrients. The extent of mangrove areas can be expected to increase with increased rainfall, with colonization of previously unvegetated areas at the landward fringe. The diversity of mangrove zones and growth rates should increase (Ellison 1996).

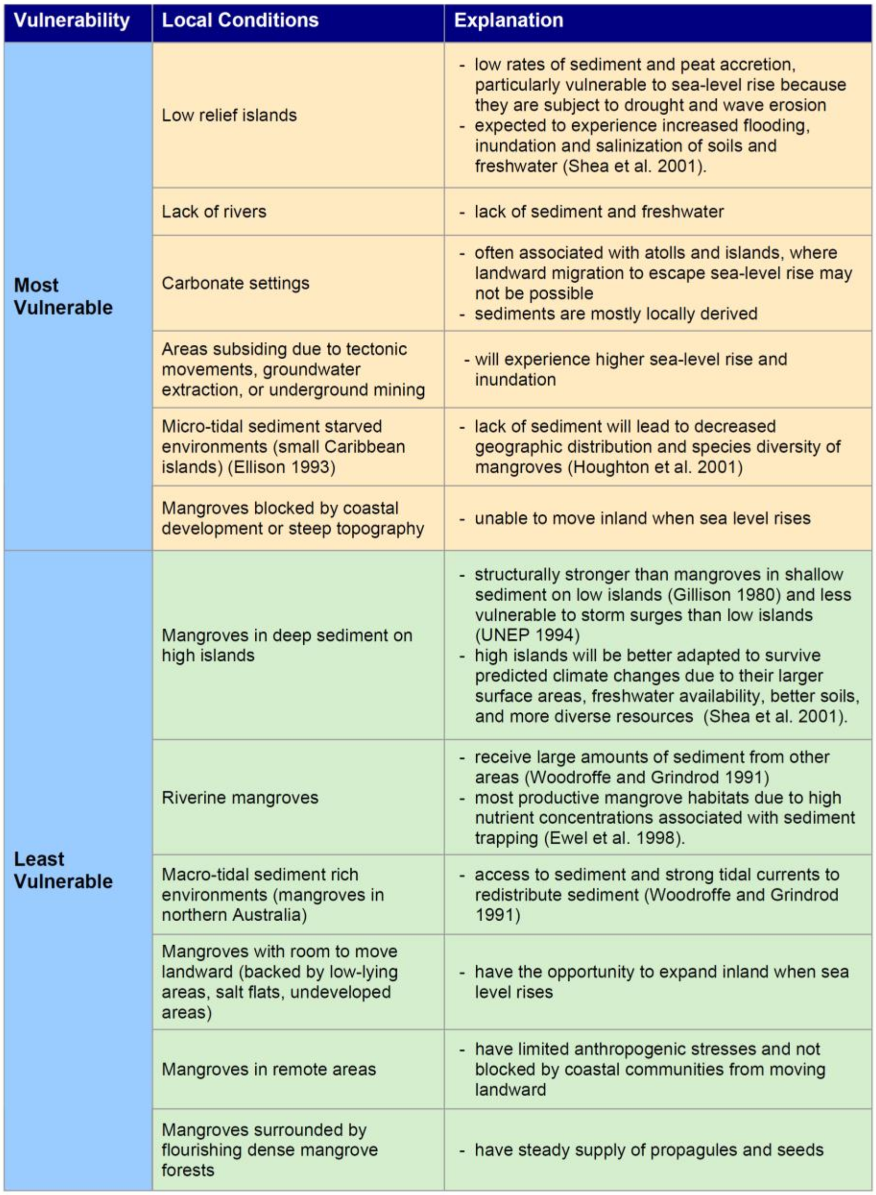
TROPICAL STORMS AND HURRICANES

Large storm impacts have resulted in mass mortality in 10 Caribbean mangrove forests in the last 50 years (Jimenez et al. 1985; Armentano et al. 1995). Studies in Anguilla before and after Hurricane Luis in 1995 showed that the mortality rate of the mangroves varied between 68 and 99% as a result of the category 4 hurricane (Bythell et al. 1996). Cahoon et al. (2003) demonstrated that mass mangrove mortality in Honduras caused by a hurricane led to peat collapse which slowed recovery rates following the disturbance. Model projections of South Florida mangroves suggest that an increase in hurricane intensity over the next century is likely to result in a decrease in the average height of mangroves (Ning et al. 2003). Major storms can also lead to a change in community structure based on a differential response to damage from the storm. Roth (1997) suggests that species proportions may shift because they have different rates of regeneration. Projected increases in the frequency of high water events (Church et al. 2001, 2004) could affect mangrove health and composition due to changes in salinity, recruitment, inundation, and changes in the wetland sediment budget (Gilman et al. 2006). Storm surges can also flood mangroves and, when combined with sea-level rise, lead to mangrove damage or mortality. As previously mentioned however, depending on the speed of change mangrove systems may be able to cope with increasing sea levels and storm surges by migrating inland. Flooding, caused by increased precipitation, storms, or relative sea-level rise may result in decreased productivity, photosynthesis, and survival (Ellison 2000). Inundation of lenticels in the aerial roots can cause the oxygen concentrations in the mangrove to decrease, resulting in death (Ellison 2004). Inundation is also projected to decrease the ability of mangrove leaves to conduct water and to photosynthesize (Naidoo 1983).

ADAPTIVE CAPACITY OF WETLANDS

The ability of wetlands to migrate inland to areas of decreasing tidal inundation along undeveloped shores is one way coastal wetlands can persist in spite of rising seas (Ross et al. 2000). However, in many areas coastal development just above the extreme high tide line has limited or eliminated opportunities for wetland migration, a phenomenon that has been labeled "coastal squeeze" (Twilley 1997). The maximum rate that wetlands can migrate into available inland areas is unknown relative to projected changes in sea level over the next century. Nonetheless, the vulnerability of coastal resources and infrastructure to sea-level rise can be expected to increase as both human development and climate change progress (Twilley et al. 2001). Assessing mangrove vulnerability to climate change, and in particular sea level rise, needs to be done at the local level since vulnerability is site-specific. To build resilience into mangrove conservation plans, managers need to identify and protect mangroves that more likely to survive sea level rise. Table 2 provides an assessment of mangrove vulnerability to sea level rise based on environmental conditions. To assess the adaptive capacity of Puerto Rico's mangroves site assessments would need to be conducted using the environmental conditions presented in the table, such as whether there is sufficient sediment and freshwater sources, and whether the mangroves are being "squeezed" between coastal development and sea level and therefore unable to migrate inland.

Table 2 an assessment of mangrove vulnerability to sea level rise based on environmental conditions. Courtesy of McLeod, Elizabeth and Salm, Rodney V. (2006). *Managing Mangroves for Resilience to Climate Change,* IUCN, Gland, Switzerland.



Vulnerability	Local Conditions	Explanation
Most Vulnerable	Low relief islands	- low rates of sediment and peat accretion, particularly vulnerable to sea-level rise because they are subject to drought and wave erosion - expected to experience increased flooding, inundation and salinization of soils and freshwater (Shea et al. 2001).
	Lack of rivers	- lack of sediment and freshwater
	Carbonate settings	- often associated with atolls and islands, where landward migration to escape sea-level rise may not be possible - sediments are mostly locally derived
	Areas subsiding due to tectonic movements, groundwater extraction, or underground mining	- will experience higher sea-level rise and inundation
	Micro-tidal sediment starved environments (small Caribbean islands) (Ellison 1993)	- lack of sediment will lead to decreased geographic distribution and species diversity of mangroves (Houghton et al. 2001)
	Mangroves blocked by coastal development or steep topography	- unable to move inland when sea level rises
Least Vulnerable	Mangroves in deep sediment on high islands	- structurally stronger than mangroves in shallow sediment on low islands (Gillison 1980) and less vulnerable to storm surges than low islands (UNEP 1994) - high islands will be better adapted to survive predicted climate changes due to their larger surface areas, freshwater availability, better soils, and more diverse resources (Shea et al. 2001).
	Riverine mangroves	- receive large amounts of sediment from other areas (Woodroffe and Grindrod 1991) - most productive mangrove habitats due to high nutrient concentrations associated with sediment trapping (Ewel et al. 1998).
	Macro-tidal sediment rich environments (mangroves in northern Australia)	- access to sediment and strong tidal currents to redistribute sediment (Woodroffe and Grindrod 1991)
	Mangroves with room to move landward (backed by low-lying areas, salt flats, undeveloped areas)	- have the opportunity to expand inland when sea level rises
	Mangroves in remote areas	- have limited anthropogenic stresses and not blocked by coastal communities from moving landward
	Mangroves surrounded by flourishing dense mangrove forests	- have steady supply of propagules and seeds

RESEARCH AND INFORMATION NEEDS

1. Accurate Sea Level Rise mapping
2. Annual monitoring of wetlands condition:
    a. Monitoring changes in salinity and hydrology. Changes in salinity and hydrology are expected due to climate change and to anthropogenic impacts such as coastal development, groundwater extraction, and dredging. Salinity and hydrology changes are important to measure because they can affect the structure and function of wetland ecosystems. To determine the salinity and hydrology in mangrove systems, a network of piezometer clusters can be installed at the site for continuous and manual measurements of salinity and water level (Drexler and Ewel 2001).
    b. Measure and monitoring changes in elevation. Annual measurements of the soil elevation deficit (elevation change minus sea-level rise) will help determine mangrove ecosystem vulnerability to sea-level rise (Cahoon and Lynch 1997). Current rates of sedimentation can be measured using artificial soil marker horizon plots. Marker horizons measure vertical accretion which incorporates both sediment deposition and sediment erosion.

Marker horizons can use sand, feldspar, brick dust and glitter, although Cahoon and Lynch (2003) recommend white feldspar as it is easily distinguishable from surrounding sediments. Marker horizons are often used with Surface Elevation Tables (SETs). SETs are used to monitor mangrove vertical accretion and subsidence and provide highly accurate and precise measurements (+/- 1.4 mm total error) of sediment elevation relative to sea-level rise (Cahoon et al. 2002a). To determine how changes in sea-level rise will affect a particular area, it is important to understand the factors influencing surface elevation, such as sedimentation rate, groundwater flow, and biological productivity (Rogers 2004; Whelan et al. 2005). Elevation changes are influenced by both surface and subsurface processes within the soil profile. Surface processes include sediment deposition and sediment erosion, and subsurface processes include root growth, decomposition, porewater flux, and compaction (Cahoon et al. 2002c). Most methods that measure surface elevation changes give only an absolute change in soil elevation and do not distinguish between elevation gain from accretion and elevation loss from subsidence (Whelan et al. 2005). However, SETs can be used with marker horizons to separate out surface accretion and subsurface expansion or compaction (Cahoon et al. 2003). According to Cahoon et al. (2003), if surface processes are controlling eleva tion change in mangroves, then elevation is controlled by erosion or sediment deposition, and if subsurface processes are controlling elevation, then it is more complicated to determine which process is dominant. New versions of the SET are able to determine where in the soil profile the influence is occurring (e.g., root zone, below root zone) (Cahoon et al. 2002b).

3. Determine mangrove and other wetland vegetation response to historical sea-level rise: Pollen and radiocarbon analyses have been used to document environmental and mangrove dynamics during the Holocene (Wooller et al. 2004; Yulianto et al. 2004; and Versteegh et al. 2004). In a study in Belize, scientists extracted a 10 meter long mangrove peat core. They extracted and radiocarbon dated fragments of mangroves leaves preserved in the core (Wooller et al. 2004). The core provided an 8600 14C year record of mangrove ecosystem changes. Pollen data from the core was used to determine changes in the floral composition of mangroves forests through the Holocene and indicated significant environmental changes such as disturbance from hurricanes or fluctuations in sea level. Changes in stand structure, which is related to changes in salinity, nutrient status, and sea level, were determined by analyzing variations in the stable carbon and nitrogen isotopes in the fossilized mangrove leaves. Historic sediment accumulation rates can also be calculated from 210Pb profiles in vibracores (Walsh and Nittrouer 2004). 210Pb is a naturally occurring radioactive isotope of lead that is used to date sediments.
4. Establishment of sentinel sites and a long-term wetland monitoring network (NOAA has a wetlands/CCSLR monitoring program)
5. Assessments of non-mangrove wetland systems in the near future are needed to better understand how climate changes may affect all wetland systems.

SUMMARY OF CLIMATE CHANGE IMPACTS ON PUERTO RICO’S WETLAND ECOSYSTEMS

Seven basic wetland types are relatively common in Puerto Rico: (1) saltwater aquatic; (2) saltwater flats; (3) saltwater marsh; (4) saltwater swamps (the most well-known example in Puerto Rico is the mangrove forest); (5)freshwater aquatic; (6) freshwater marsh; (7) freshwater swamp. Coastal wetlands provide many important ecosystem services, such as sequestering carbon, mitigating against possible disasters, assisting with prevention of coastal erosion, contaminant retention from land-based sources of pollution, servicing as nursery areas for recreationally and commercially-important fish species, local and foreign tourism, and cultural and mental well-being from aesthetically pleasing viewsheds.

Worldwide, coastal wetlands are under direct and increasing threat from land use changes pressures, from indirect impacts of upstream disruption to sediment supply, and from development pressures and rising sea level at the coast. Large areas of coastal wetlands worldwide have been drained and converted to other uses. Salt marshes and freshwater tidal marshes have lost more than 50% of their historical global coverage, with the current rate of loss estimated at 1-2% per year.

Mangroves are wetland systems that have particularly tremendous social and ecological value. In Puerto Rico, mangroves cover about 1,900 ha in northeastern Puerto Rico and account for 22% of all of the island's mangrove forests. The largest remaining areas are found east of the Rio Espíritu Santo, in the Las Cabezas Reserve, in the Ceiba Forest, and at Roosevelt Roads. These mangroves are typically characterized as: *Rhizophora mangle* dominated stands on coastal and estuarine fringes; *Avicennia germinans* and *Laguncularia racemosa* in basins as pure or mixed stands; or *Conocarpus erectus* and the shrub *Suriana maritima* in mixed stands. Species richness is very low. Mangrove forest structure is strongly affected by salinity, nutrient availability, and storm disturbance. Mangrove forests have gone through a period of decline and partial recovery related to human activities such as altered hydrology for agricultural practices, urbanization, and the implementation of conservation activities.

The negative outcomes and consequences of climate change on wetland system, specifically mangroves, are expected to be primarily due to sea level rise. When wetland elevation is low relative to sea level, frequent inundation by tides enhances the supply of suspended sediment and nutrients to the wetland. This stimulates accretion by sedimentation. As well, the enhanced nutrient supply supports abundant growth of vegetation, and this contributes to accretion through the production of roots and rhizomes and by trapping suspended sediment for incorporation into new wetland sediment. If the elevation of the wetland increases relative to sea level, tidal inundation becomes less frequent; accretion slows; and the rate at which wetland elevation increases is reduced. Our current understanding of how wetlands will respond to sea level rise is the prediction that they will either migrate landward or be lost. Globally, climate change is expected to exacerbate the loss and degradation of mangrove forests and the loss or decline of their species, and to affect the human populations dependent on their services. Coastal wetlands in islands are especially vulnerable to impacts from relative sea level rise since they have a limited capacity to adapt, including limited space to accommodate landward migration of mangroves and other coastal ecosystems. The potential effects from rising sea levels on mangroves are:

- Probable loss of total mangrove areas due to erosion of the seaward margin of the mangroves and loss of protective lagoon bars and sea barriers;
- Relocation and migration of mangroves inland, rather than overall loss. This landward migration can be obstructed if the landward margin of the mangrove area is steep or if there are seawalls and other developments (coastal squeeze), thereby reducing the areas of coastal ecosystems;
- Change in mangrove forest structure. Landward replacement of black mangrove (*Avicennia*) by red mangrove (*Rhizophora*) and possible increased growth and productivity of the mangrove areas from carbon dioxide fertilization;
- Increase in mangrove area and changes to associated wetland community types and distribution. Saline intrusion into inland freshwater wetlands and rejuvenation of saline and scrub mangrove sites.

For wetlands in general, increases in salinity due to changes in sea level, precipitation, and groundwater supplies can result in reduced seedling survival and growth, and decreased photosynthetic capacity. Temperature affects the growth, survival, reproduction, and distribution of plants and animals. As air and water temperature increase, species ranges will likely expand toward environments that are presently cooler. If dispersal capabilities are limited or suitable habitat is not available, local extirpations and extinctions are likely to occur. Increased temperatures in terrestrial habitats will reduce stream flow and alter water quality into deltas, estuaries and in coastal regions, causing addition change and/or degradation of coastal ecosystems. However, mangroves are not expected to be adversely impacted by the projected increases in sea surface temperature, but rather extreme cold events in higher latitudes are more likely to limit mangrove expansion. Assessing mangrove vulnerability to climate change, and in particular sea level rise, needs to be done at the local level since vulnerability is site-specific.

Changes in rainfall patterns caused by climate change may have a profound effect on both the growth of mangroves and their areal extent. It is known that during long-term droughts some mangrove forests contract while salt flats expand. Decreased precipitation results in a decrease in mangrove productivity, growth, and seedling survival, and may change species composition favoring more salt tolerant species. On the other hand, increasing precipitation may allow for a greater accumulation of sediment and organic matter from runoff resulting in an increased elevation of mangroves (which in turn might help the mangroves adapt to sea level rise). Greater amounts of rainfall could increase mangrove area, diversity of mangrove zones, and mangrove growth rates in some species. Increased precipitation may also allow mangroves to migrate and outcompete saltmarsh vegetation. More studies are required to better understand how precipitation may change in Puerto Rico and the corresponding changes in wetland systems.

From this brief literature review, it is found that wetlands provide critical ecosystem services to Puerto Rico, such as water quality, coastal protection from waves and storms, fisheries production and tourism. These ecosystem services are at risk from climate change. Puerto Rico agencies and organizations should continue monitoring at local level, research, conservation and restoration programs should be undertaken and whenever possible damages to wetland habitats should be avoided. Every effort should be taken by the Government and private sector to conserve and restore wetland ecosystems.

COASTAL LAGOONS

INTRODUCTION TO COASTAL LAGOONS

Coastal lagoons are shallow water bodies separated from the sea by sand bars or other dry land. These berms may break during storms allowing exchange between the lagoons and the coast, or exchange may occur through channels from the lagoon to the sea. Some coastal lagoons receive freshwater input directly from rivers and streams while others only receive freshwater from stormwater flows during rain events. Coastal lagoons are often associated with wetland systems in the coastal zone ranging from fresh to brackish wetlands. The difference in the amount of fresh and saltwater influx leads to differences in the salinities of coastal lagoons. This also affects the amount and type of flora and fauna present in coastal lagoons. Mangroves are the dominant vegetation around coastal lagoons. Lagoons with a direct connection to the sea provide habitat for seabirds and fish, including species of commercial and recreational importance. In the Virgin Islands, research has shown that coastal lagoons gradually formed in sheltered bays or inlets that were originally open to the sea (Thomas and Devine 2005). The areas were gradually separated from the sea as coral reefs grew across the mouth of the bay forming a berm that was then colonized by mangroves (Thomas and Devine 2005). A similar process probably formed many of the coastal lagoons in Puerto Rico. Researchers have found remnants of coral reefs in the Joyuda Lagoon in Cabo Rojo (E. Otero, pers. comm.), which supports the theory that the area was once a coastal embayment with coral reefs.

In Puerto Rico, there are marine lagoons, which have a free interchange with the sea and a substantial part of their volume flows with the tides; hypersaline lagoons, which are found in areas with little rainfall and high evaporation and have little freshwater input and little connection to the sea; and brackish lagoons, which are semi-closed and are connected to drainages where seawater is diluted by the freshwater input (Vivaldi and Paniagua 1986). Marine lagoons typically have sandy bottoms and clear waters and are colonized by seagrass and macroalgae and contain populations of coral reef species. Hypersaline lagoons are often relatively devoid of vegetation and fauna due to the elevated salinities. Brackish lagoons are often colonized by estuarine species that can tolerate changes in dissolved oxygen and temperature (Vivaldi and Paniagua 1986). Perhaps the best known coastal lagoons in Puerto Rico are within the San Juan Bay Estuary System, which contains the Condado Lagoon, San José Lagoon, Torrecillas Lagoon, and the Piñones Lagoon. The lagoon system in this area is also the largest mangrove forest system on the island and provides habitat for a variety of wildlife including birds, fish, reptiles, mollusks, and crustaceans (Ventosa-Febles et al. 2005). Both marine and brackish lagoons in Puerto Rico, often provide nursery habitat for species that are important in the coral reef ecosystem, as well as species that spend at least a portion of their time in estuarine environments, such as native shrimp species.

STATUS AND THREATS OF COASTAL LAGOONS

The greatest threat to coastal lagoons is development. The influx of stormwater and sediment-laden runoff from upland development into coastal lagoons results in contamination of the waters in the lagoons, as well as changes in patterns of salinity and sedimentation of the lagoons, which can result in the eventual filling of these areas. In the U.S. Virgin Islands, core samples collected from salt ponds that have become dry land have shown that terrestrial sediments from uplands led to the filling of these areas over time (Thomas and Devine 2005). Development along the shorelines of lagoons alters water quality, as well as eliminates habitat for birds and other organisms that are important in the food web. The construction of marine facilities in lagoons also affects water quality through the introduction of marine debris into the water column, as well as accidental spills of petroleum products and other contaminants (such as cleaning agents) and the resuspension of sediments or accidental groundings due to the shallow nature of coastal lagoons. Dredging of lagoons to support marine activities also affects water quality and can alter flow patterns, which affects the organisms living in the lagoons that are adapted to natural conditions within the lagoons.

Other threats to coastal lagoons include discharges of untreated stormwater and wastewater. Untreated stormwater from roads and other developed areas often contain petroleum products and other contaminants. In addition, the use of coastal lagoons as stormwater ponds alters their natural function and patterns of fresh and saltwater exchange, thus affecting the flora and fauna of the lagoons (Vivaldi and Paniagua Valverde 1986). The discharge of wastewater can lead to significant decreases in dissolved oxygen in the water column due to the increase in production by bacteria, algae, and other organisms associated with the influx of nutrients. If the dissolved oxygen levels in the water column become too reduced, fish kills can result.

OUTCOME AND CONSEQUENCES OF CLIMATE CHANGE TO PUERTO RICO’S COASTAL LAGOONS

Potential threats to coastal lagoons from climate change may come from rising sea level, changing tidal regime, changes in freshwater influx and associated changes in water quality (including salinity, nutrients, and sediment), and changes in the distribution and intensity of extreme event).

SEA LEVEL RISE

Rising sea level will have a greater affect on low-lying, shallow-gradient coastal lagoons and other ecosystems because these typically respond to sea level rise by migrating landward along undeveloped shorelines with gentle slopes (Anthony et al. 2009). However, if sea level rise accelerates, landward migration might not be fast enough to prevent inundation and developed coastlines will make migration impossible, thereby increasing the vulnerability of coastal structures to inundation and storm damage (Anthony et al. 2009). Landward migration of lagoon barriers will also lead to steep and narrower barriers and increased exchange with the sea, which will also increase the lagoon berm's vulnerability to breaching and increase flushing rates (Zimmerman 1981, Anthony et al. 2009). If breaching of the barrier increases, then the salinity in the lagoon will change and the species' composition in the lagoon may be altered.

Additionally, coastal lagoons may be more susceptible to point and non-point sources of pollution as sea levels rise. Specific to sewage treatment plants, if the plants are not able to discharge properly via gravity with higher water levels, depending on the location of nearby treatment plants, seawater may enter combined sewer systems or overwhelm the system allowing for direct discharge of sewage and other pollutants into the lagoons.

PRECIPITATION CHANGES AND FRESHWATER INFLUX

Alterations in the frequency, magnitude, and timing of rain events could result in changes in salinity and dissolved oxygen patterns in coastal lagoons due to changes in freshwater influx (Anthony et al. 2009). In areas where precipitation is expected to increase or where storms become more intense, increased runoff will lead to increased transport of nutrients and sediment to coastal lagoons and decreases in salinity (Day et al. 2008, Anthony et al. 2009). Increases in nutrient levels in coastal lagoons could result in increases in eutrophication. Studies in the Mar Menor Lagoon in Southeast Spain have shown that the main primary producer, which is a macroalgal species, has such a high biomass that it has increased the resistance of the lagoon to eutrophication due to its ability to uptake high nutrient levels from the water column that are then retained in bottom sediments (Lloret and Marín-Guirao 2008). However, if climate change projections prove true, the productivity of the algae will eventually decline and eutrophication of the lagoon will occur (Lloret et al. 2008).

Additionally, many management plans for coastal lagoons and estuaries have a high priority of controlling nonpoint sources of pollution. One such management plan is Jobos Bay National Estuarine Research Reserve's (JBNERR) on the south coast. Investigators of the USDA Agricultural Research Service detected a pesticide spike in water samples after a storm event, confirming a direct link from nonpoint source runoff from the watershed and water quality in the Reserve (JBNERR 2010). Other types of nonpoint sources of pollution that management plans are concerned about are nutrient runoff that leads to eutrophication, chemical contamination that endangers wildlife and human health, and sediment influxes that smother coral reef and seagrass communities. Changes in precipitation patterns, specifically increases in heavy downpour events, will make this management priority more challenging as more watershed-based contaminants enter Puerto Rico's coastal waterbodies.

INCREASED TEMPERATURE

Changes in air temperatures influence the water temperature of slow-moving, shallow waters in coastal lagoons (Turner 2003). In addition, due to ocean temperature increases lagoons with tidal connections to the sea will also be affected. Air temperatures increase more rapidly over land than the ocean as evidenced by a study in Narragansett Bay, Rhode Island that showed the rate of increase in the annual average surface water temperature over the past 5 years was approximately 4 times greater than that of the ocean (Anthony et al. 2009). Water temperatures affect levels of dissolved oxygen (levels decrease as temperatures increase), as well as species' physiology (Anthony et al. 2009). Therefore, increases in water temperature in coastal lagoons and associated decreases in dissolved oxygen levels will lead to changes in benthic communities, as well as highly mobile species such as fish. Changes in community composition of coastal lagoons could promote invasive flora and fauna species. Shifts in the availability of food due to changes in water temperature and impacts on species growth and cyclical migrations will also affect animals such as migrating birds.

Warmer temperatures may increase toxicity of pollutants that already exist in Puerto Rico's coastal lagoons or result in higher solubility leading to higher concentrations of pollutants that newly enter the lagoons.

Additionally, warmer temperatures may increase toxicity of pollutants that already exist in Puerto Rico's coastal lagoons or result in higher solubility leading to higher concentrations of pollutants that newly enter the lagoons. Other water quality implications of increasing temperatures on coastal lagoons could be increased stratification, greater algal growth, and a greater quantity of parasites and bacteria.

TROPICAL STORMS AND HURRICANES

Coupled with rising sea level, even slight changes in the frequency, intensity, timing, and distribution of tropical storms and hurricanes are expected to result in substantial impacts to coastal wetlands (Michener et al. 1997), including coastal lagoons. Long-term changes in storm frequency, intensity, timing, and distribution could affect biotic functions such as community structure and productivity and physical processes such as nutrient cycling (Michener et al. 1997).

RESEARCH GAPS AND INFORMATION NEEDS

Coastal lagoons in Puerto Rico have not been well-studied with the exception of some of the areas within the San Juan Bay Estuary System. Some studies of the Joyuda Lagoon were conducted in the 1980's by the Department of Marine Science of the University of Puerto Rico (J. López, pers. comm.) and DNER (Ventosa-Febles et al. 2005); however, research is also needed to determine changes in the lagoon due to the high level of development occurring in the watershed that has resulted in the alteration of the creek that is one of the main freshwater sources to the lagoon and the alteration of wetlands and road construction that has altered the saltwater connection between the lagoon and the sea. Because characterizations of the flora and fauna of many of the lagoons around the island have not been done, it will be difficult to determine the extent of climate change impacts on these systems. Therefore, research to characterize the physical and biological aspects of Puerto Rico's coastal lagoons should be completed as soon as possible. Permanent monitoring stations should be established in a representative sample of lagoons in order to track changes in the physical and biological characteristics and determine whether these are associated with climate change or human impacts. Better management of coastal lagoons is also needed in order to reduce or eliminate human impacts so that the systems can adapt to changing conditions associated with climate change.

Predictions of climate change impacts on coastal wetlands require a better understanding of the linkages between terrestrial, wetland, atmospheric, oceanographic, and human components of the ecosystem. Thus, a watershed scale approach that incorporates atmospheric and oceanographic data in modeling of scenarios to determine potential climate change impacts on coastal lagoons and potential shifts in lagoon composition and characteristics is necessary. Michener et al. (1997) presented two conceptual models of broad-scale comparative research for assessing ecological responses to climate change. One used a space-for-time substitution coupled with long-term studies to assess impacts of rising sea level and disturbance on coastal wetlands and the other used a moisture-continuum model to assess the effect of climate change and associated shifts in moisture regimes on wetland ecosystems (Michener et al. 1997). Similarly, (Bortone 2006) proposes the establishment of a sampling network with established sampling protocols to determine the important variables to monitor and a comprehensive plan to assess the environmental effects of hurricanes on coastal ecosystems. As part of this effort, Burtone (2006) recommends the mining of historical data to assess the role of hurricanes in shaping coastal ecosystems over time in order to better determine whether changes in storms due to climate change are different from processes that already occur in natural systems in response to hurricanes.

SUMMARY OF CLIMATE CHANGE IMPACTS ON PUERTO RICO'S COASTAL LAGOONS

Coastal lagoons are shallow water bodies separated from the sea by sand bars or other dry land. Mangroves are the dominant vegetation around coastal lagoons. Lagoons with a direct connection to the sea provide habitat for seabirds and fish, including species of commercial and recreational importance. In Puerto Rico, there are marine lagoons and brackish lagoons. Perhaps the best known coastal lagoons in Puerto Rico are within the San Juan Bay Estuary System, which contains the Condado Lagoon, San José Lagoon, Torrecillas Lagoon, and the Piñones Lagoon. The lagoon system in this area is also the largest mangrove forest system on the island and provides habitat for a variety of wildlife including birds, fish, reptiles, mollusks, and crustaceans. Both marine and brackish lagoons in Puerto Rico, often provide nursery habitat for species that are important in the coral reef ecosystem, as well as species that spend at least a portion of their time in estuarine environments, such as native shrimp species.

The greatest threat to coastal lagoons is development. The influx of stormwater and sediment-laden runoff from upland development into coastal lagoons results in contamination of the waters in the lagoons, as well as changes in patterns of salinity and sedimentation of the lagoons, which can result in the eventual filling of these areas. Other threats to coastal lagoons include discharges of untreated stormwater and wastewater. Untreated stormwater from roads and other developed areas often contain petroleum products and other contaminants.

Potential threats to coastal lagoons from climate change may come from rising sea level, changing tidal regime, changes in freshwater influx and associated changes in water quality (including salinity, nutrients, and sediment), and changes in the distribution and intensity of extreme event). Rising sea level will have a greater effect on low-lying, shallow-gradient coastal lagoons and other ecosystems because these typically respond to sea level rise by migrating landward along undeveloped shorelines with gentle slopes. However, if sea level rise accelerates, landward migration might not be fast enough to prevent inundation and developed coastlines will make migration impossible, thereby increasing the vulnerability of coastal structures to inundation and storm damage. Landward migration of lagoon barriers will also lead to steep and narrower barriers and increased exchange with the sea, which will increase the lagoon berm's vulnerability to breaching and increase flushing rates. If breaching of the barrier increases, from gradual sea level rise or more acute events like storms, then the salinity in the lagoon will change and the species' composition in the lagoon may be altered.

Alterations in the frequency, magnitude, and timing of rain events could result in changes in salinity and dissolved oxygen patterns in coastal lagoons due to changes in freshwater influx. Increases in nutrient levels in coastal lagoons could result in increases in eutrophication. Changes in air temperatures influence the water temperature of slow-moving, shallow waters in coastal lagoons. In addition, due to ocean temperature increases lagoons with tidal connections to the sea will also be affected. Air temperatures increase more rapidly over land than the ocean. Water temperatures affect levels of dissolved oxygen (levels decrease as temperatures increase), as well as species' physiology. Therefore, increases in water temperature in coastal lagoons and associated decreases in dissolved oxygen levels will lead to changes in benthic communities, as well as highly mobile species such as fish. Changes in community composition of coastal lagoons could promote invasive flora and fauna species. Shifts in the availability of food due to changes in water temperature and impacts on species growth and cyclical migrations will also affect animals such as migrating birds. Additionally, warmer temperatures may increase toxicity of pollutants that already exist in Puerto Rico's coastal lagoons or result in higher solubility leading to higher concentrations of pollutants that newly enter the lagoons.

Coastal lagoons in Puerto Rico have not been well-studied with some exceptions. Because characterizations of the flora and fauna of many of the lagoons around the island have not been done, it will be difficult to determine the extent of climate change impacts on these systems. Therefore, research to characterize the physical and biological aspects of Puerto Rico's coastal lagoons should be completed as soon as possible. Permanent monitoring stations should be established in a representative sample of lagoons in order to track changes in the physical and biological characteristics and determine whether these are associated with climate change or human impacts. Better management of coastal lagoons is also needed in order to reduce or eliminate human impacts so that the systems can adapt to changing conditions associated with climate change.

SEABIRDS AND SHOREBIRDS

INTRODUCTION TO SEABIRDS AND SHOREBIRDS

Puerto Rico is the easternmost and smallest of the Greater Antilles, situated 1,600 km southeast of Florida, U.S.A., between the Caribbean Sea and the Atlantic Ocean (18° 15´ N, and 66° 30´W), 110 km east of the Dominican Republic and about 37 km west of the U.S. Virgin Islands. Puerto Rico is roughly 8, 960 km², with a maximum length from east to west (from Punta Puerca to Punta Higüero) of 180 km, and a maximum width from north to south (from Isabela to Punta Colón) of 65 km. Over 70% of Puerto Rico's terrain is mountainous, with a central mountain chain transecting the island from east to west, coastal lowlands on the north and south, and a karst region in the northwest consisting of rugged, limestone formations.

The main island of Puerto Rico is surrounded by many groups of small islands and cays, and the following ones harbor one or several species of seabirds and/or shorebirds: Northwest Cays (half a km off the north coast, between Quebradillas and Manatí); Cordillera Cays (about 2 km east of Las Cabezas de San Juan, between Fajardo and the Culebra Archipelago); Culebra Archipelago (about 17 km east of Fajardo); Vieques (about 14 km southeast of Naguabo); Cayos Fríos (about 2 km south of Ponce); Caja de Muertos (about 9.6 km south of Ponce); Cayo Guayanilla (about 5 km south of Guayanilla); Montalva Cays (20-300 m south of Guánica); Parguera Cays (3-6 km south of Lajas); Mona and Monito (about 80 km west of Cabo Rojo); Isla Ratones (about 500 m west of Cabo Rojo); and Desecheo (about 19 km west of Rincón) (figure 6).

Cliffs, mudflats, salt flats, inland artificial lakes, coastal saline ponds, and sand deposits on river mouths within the main island of Puerto Rico or close to its shore are also important roosting, foraging, staging, or breeding sites for seabirds and/or shorebirds, such as: Northwest Cliffs (between Aguadilla and Camuy); Cueva del Indio (Camuy); Isla del Frío (off Ponce); the mouth of the Añasco, Cibuco (Manatí), and Arecibo rivers; Lago La Plata (Toa Alta-Naranjito); Lago Carraízo (Trujillo Alto-Caguas); Laguna de Las Salinas (Ponce); La Jungla Salt Flats (Guánica); Cabo Rojo Salt Flats (Cabo Rojo); and Boquerón Natural Reserve (Cabo Rojo).

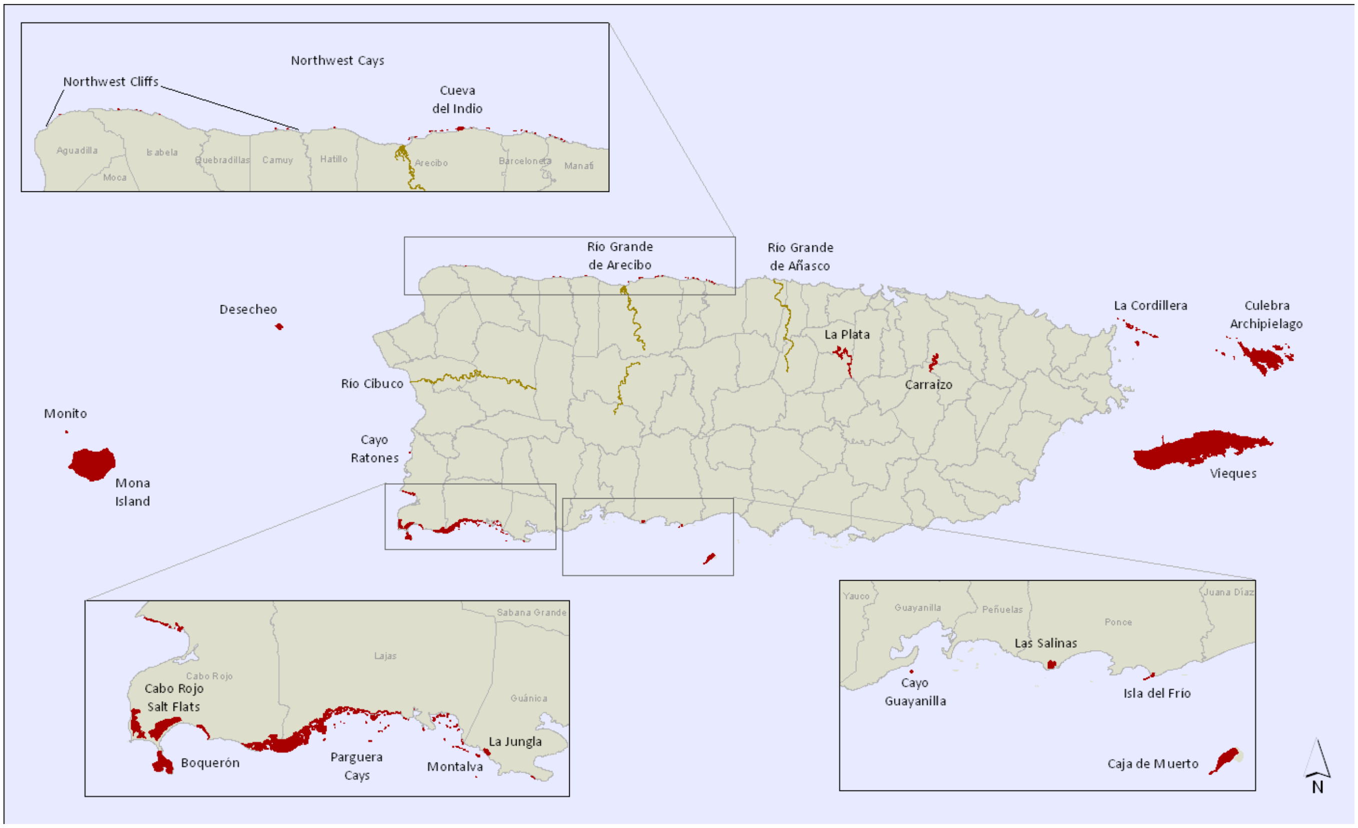


Figure 6 Cays, islets, cliffs, mudflasts, salt flats, inland artificial lakes, coastal saline ponds, and river mouth areas of Puerto Rico known to harbor one or several species of seabirds and/or shorebirds

Deep ocean waters fringe Puerto Rico: the Mona Passage to the west, between Puerto Rico and the Dominican Republic (about 1,000 m deep), the Puerto Rico Trench lying 120 km to the north (about 8.5 km deep), and to the south the sea bottom descends to the 5 km Venezuelan Basin. Pelagic seabirds travel to these areas to feed on fish attracted to the deep, rich upwelling waters.

SEABIRDS

There exists no single definition of which groups, families, and species are seabirds; and most definitions are in some way arbitrary. Schreiber and Burger (2002) state that "The one common characteristic that all seabirds share is that they feed in saltwater; but, as seems to be true with any statement in biology, some do not." For the purpose of this document, seabirds are defined as any bird species that spend most of their time in coastal waters or the open ocean. Most tropical seabirds in the West Indies, which include Puerto Rico, exist at moderate to relatively low densities;

they normally feed at sea at great distances from the breeding colonies, and typically produce just one slow-growing chick per year (Schreiber and Lee 2000, Weimerskirch 2002). The combined result is that seabirds are more vulnerable to environmental stressors (*e.g*., storms, predators, habitat modification, and man-made factors) on their breeding sites than most land birds, because of the protracted period of nest occupancy and the concentration of complete regional populations in a few sites. Furthermore, populations are slow to recover from disturbance because of their low reproductive output (Schreiber and Lee 2000).

Croxall *et al*. (2012) analyzed a broader, but less sensitive, measure of overall seabird population trends is provided by the Red List Index (RLI, from Butchart *et al*. 2004, 2007), which measures trends in extinction risk and is virtually the only trend indicator currently available for seabirds on a worldwide and/or regional basis. The RLI is based on the movement of species through IUCN Red List categories owing to genuine improvement or deterioration in status (*i.e*., re-categorizations owing to improved knowledge or revised taxonomy are excluded). It shows that, over the last 20 years, seabirds have had a substantially poorer conservation status than non-seabirds and that they have deteriorated faster over this period (Croxall *et al*. 2012). Seabirds are more threatened than a number of other similarly speciose groups (*e.g*., raptors, pigeons, gamebirds and waterbirds), and are marginally more threatened than parrots. Among seabirds, pelagic species are more threatened and have deteriorated faster than coastal species (Croxall *et al*. 2012).

SEABIRD SPECIES

Sixteen species of seabirds breed in Puerto Rico and its adjacent islands. Five species are resident throughout the year: Brown Booby (*Sula leucogaster*), Red-footed Booby (*S. sula*), Masked Booby (*S. dactylatra*), Brown Pelican (*Pelecanus occidentalis*), and Magnificent Frigatebird (*Fregata magnificens*). Eleven species are migrants absent for part of each year: Audubon's Shearwater (*Puffinus lherminieri*), White-tailed Tropicbird (*Phaethon lepturus*), Red-billed Tropicbird (*P. aethereus*), Laughing Gull (*Larus atricilla*), Royal Tern (*Thalasseus maximus*), Sandwich Tern (*T. sandvicensis*), and Cayenne Tern (*T. eurygnatha*), Roseate Tern (*Sterna dougallii*), Least Tern (*Sternula antillarum*), Bridled Tern (*Onychoprion anaethetus*), Sooty Tern (*O. fuscatus*), and Brown Noddy (*Anous stolidus*) (Saliva 2009). Although Royal and Sandwich Terns are found throughout the year, apparently most of the individuals observed outside the breeding season are transient birds. Common Terns (*Sterna hirundo*) are regularly observed early in the summer, loafing at Roseate Tern breeding colonies, but they usually depart Puerto Rico by midsummer (Saliva, pers. obs.).

Other seabird species have been reported sporadically at seabird colonies or foraging in waters off Puerto Rico such as Gull-billed Tern (*Sterna nilotica*), Arctic Tern (*Sterna paradisaea*), Black Noddy (*Anous minutus*), Black Tern (*Chlidonias niger*), Parasitic Skua (*Stercorarius parasiticus*), Herald Petrel (*Pterodroma arminjoniana*), Greater Shearwater, (*Puffinus gravis*), Double-crested Cormorant (*Phalacrocorax auritus*), Wilson's Storm Petrel (*Oceanites oceanicus*), Leach's Storm Petrel (*Oceanodroma leucorhoa*), Sooty Shearwater (*Puffinus griseus*), and Manx Shearwater (*Puffinus puffinus*) (Gochfeld *et al* 1988; Raffaele *et al* 1998; Saliva pers. obs.).

During their non-breeding season, the following migratory seabirds are regularly seen (*i.e*., at least every other year) in Puerto Rico: Ring-billed Gull (*Larus delawarensis*), Lesser Black-backed Gull (*L. fuscus*), Herring Gull (*L. argentatus*), Great Black-backed Gull (*L. marinus*), Black-headed Gull (*L. ridibundus*), Franklin's Gull (*L. pipixcan*), and Forster's Tern (*Sterna forsteri*) (Raffaele *et al* 1998; Saliva pers. obs.). Others, such as Black-legged Kittiwake (*Rissa tridactyla*), Bonaparte's Gull (*L. philadelphia*), and Black Skimmer (*Rynchops niger*) are far less common (Saliva pers. obs.).

SEABIRD NESTING HABITATS

Tens of thousands of islands across the world are used by over 250 species of seabirds for nesting and roosting, sometimes in extremely high densities (Croxall *et al.* 1984). While seabirds primarily forage at sea, their daily or seasonal returns to islands result in the transport and concentration of substantial quantities of marine nutrients on land, and can cause significant physical and chemical alterations of island habitats (Anderson and Mulder 2011). Most seabirds nest either on cliffs or on, or beneath, the slopes or flat interiors of islands (Nelson 1979), and prefer to nest in offshore islands away from large land masses; particularly ground-nesting seabirds which are vulnerable to mammalian predators. Although a variety of habitats are found within the main island of Puerto Rico; ranging from wet montane forests to dry coastal scrub, mangrove lagoons, salt ponds, limestone and volcanic cliffs, salt flats, mud flats, and sandy beaches, only five species of seabirds nest within the main island of Puerto Rico: White-tailed Tropicbird, Brown Pelican, Least Tern (Saliva 2009), Noddy Tern, and Bridled Tern (Saliva pers. obs). There are no historical records of other seabird species nesting on the main island, even in coastal areas that appear suitable for nesting. During the non-breeding season, other migratory seabirds as described above use coastal areas around the main island as staging grounds; particularly the mouths of major rivers and coastal manmade structures such as concrete pilings, buoys, and jetties.

Rocky shores are beach rock formations resulting from cementing sand, shell fragments, and precipitation of calcium carbonate coming into the sea from drainage out of the karst region (Miller and Lugo 2009). Volcanic rocks and cays, which contain jagged depressions on the rock that provide nesting habitat for some seabird species, are small rocky coastal formations frequently over washed by heavy north swells. They range from a few meters (feet) above sea level to about six meters (20 feet) high. Vegetation is only found on the larger rocky cays (> half a hectare). A few of these are connected to the main island through a thin strip of volcanic material. The rocky cays that are used by seabirds, however, are usually separated from other land and access is difficult unless sea conditions allow climbing over the jagged edge (Saliva 2009). Three species of seabirds nest on this type of habitat: Roseate Tern, Noddy Tern, and Bridled Tern (Saliva 2009).

Cliffs are vertical or near-vertical coastal features composed of sedimentary rock, pillow lava, cemented sand, or limestone rock. The great advantages of island cliffs are safety from ground predators, and the provision of wind and air space, which greatly facilitates landing and departure (Nelson 1979). A cliff face offers several distinct types of nesting site; from broad, flat ledges to caves and crevices. Some cliffs may have a dry coastal scrub community present; some of which may exhibit extreme wind/salt shear and may only be a few centimeters tall (Miller and Lugo 2009). Although many cliff features are found around the coast of Puerto Rico, only the following have been reported as seabird cliff nesting areas: Aguadilla cliffs (Aguadilla), Mona and Monito islands (Mayagüez), Cabo Rojo lighthouse (Cabo Rojo), Caño Valdivieso (Ponce), Cayo Conejo (Vieques), Caja de Muerto (Ponce), and Cayo Morrillito (Ponce). I addition, in the Culebra archipelago seabirds nest on cliffs at Punta Soldado and the islets of East Geniquí, West Geniquí, Alcarraza, Yerba, and Lobito. White-tailed Tropicbirds, Red-billed Tropicbirds, Noddy Terns, and Bridled Terns use the cavities and caves on cliff faces; whereas Noddy Terns use ledges that protrude from the cliff face, and Sooty Terns, Brown Boobies, Masked Boobies, and Laughing Gulls use flatter surfaces near the cliff edge (Saliva 2000).

Coralline cays are small, low-elevation, islands formed on the surface of coral reefs formed by the gradual deposit of sediment layers (primarily dead stony corals and sand) building up on the reef surfaces (Hopley 1981). These depositions occur on the windward or leeward areas of reef surfaces, and sometimes around an emergent outcrop of old reef. Cay sediments are largely composed of calcium carbonate produced by myriad plants (*e.g.*, *Halimeda*) and animals (*e.g.*, corals, mollusks, and foraminifera), and small amounts of silicate sediment are also contributed by sponges and other creatures (Folk and Robles 1964, Scoffin 1987, Yamano *et al*. 2000). Over time, soil and vegetation may develop on a cay surface, assisted by the deposition of seabird guano. There is much debate and concern over the future stability of cays in the face of growing human populations and pressures on reef ecosystems, and predicted climate changes and sea level rise (Kench and Cowell 2003; Hart 2003). Coralline cays in Puerto Rico are primarily found in the Reserva de La Cordillera east of Fajardo and its adjacent cays, and along the southwest coastline from Ponce to Cabo Rojo. Scattered coralline cays may also be found off Vieques Island and Fajardo-Ceiba. Royal Terns, Sandwich Terns, Roseate Terns, Least Terns, Sooty Terns, and Bridled Terns nest on the ground in coralline cays (Saliva pers. obs.).

Vegetation used by seabirds in Puerto Rico include the canopies of red mangrove (*Rhizophora mangle*), buttonwood (*Conocarpus erectus*), sea grape (*Coccoloba uvifera*), bay cedar (*Suriana maritima*), and wild banyantree (*Ficus citrifolia*); as well as shaded areas under grass clumps (*e.g*., *Cyperus* spp.), shrubs (*e.g*., *Croton*), and small/stunted trees (*e.g*., *Bursera*). Vegetation used by seabirds may be located on mangrove islets, coralline cays, at the edge of cliffs, on cliff faces, or the top of rocky cays. Most of this vegetation tends to be of short growth due to the constant wind force in coastal areas that limits their upward growth (Miller and Lugo 2009); with the exception of red mangrove. Brown Pelicans, Red-footed Boobies, Noddy Terns, and Frigatebirds construct flimsy nests on the canopy of such vegetation; whereas Sooty Terns, Bridled Terns, Roseate Terns, Laughing Gulls, Audubon’s Shearwaters, Brown Boobies, and Masked Boobies nest on the ground near or under thick vegetation (Saliva and Burger 1989; Saliva pers. obs.).

Salt flats are extensive level tracts coated with salt deposits left by evaporation of rising ground water or a temporary body of surface water. These salt flats or “salinas” are frequently positioned to the rear of coastal fringing mangrove swamps on the south coast of Puerto Rico, in areas that have access to sea water (Miller and Lugo 2009). The sea water then evaporates and produces high salt concentrations in the water and soils. The hypersaline environment cause reduced mobilization of nutrients and hyperosmotic conditions, both of which limit species richness (Copeland and Nixon 1974). Salt-tolerant plant species that can be present along the fringes of saline ponds include black mangrove (*Avicennia germinans*), glassworts (*Salicornia perennis* and *S. virginica*), saltwort (*Batis maritima*), and sea purselane (*Sesuvium portulacastrum*) (Miller and Lugo 2009). Species of blue-green and green algae tolerate hypersaline water conditions and are the major primary producers of salt flat systems (Tripp 1996). Aquatic organisms are usually limited to crustaceans and salt tolerant insects, all of which feed on algae and attain high abundances, in part, due to lack of competition and predation (Carpelan 1957, Grear 1992, Tripp 1996). Fish such as mullet (*Mugil cephalus*), mojarra (*Gerres subfasciatus*), and tilapia (*Oreochromis mossambicus*) will intrude into saline ponds during high tide or when large rain events flood the saline ponds (Saliva, pers. obs.). Brown Pelicans, Royal Terns, Sandwich Terns, Laughing Gulls, and several migratory non-breeding gull and tern species (*e.g*., Forster’s Tern, Gull-billed Terns) take advantage of these flooding events to feed on small fish. The Least Tern is the only seabird recorded nesting along the shores of saline ponds and salt flats in Puerto Rico (Saliva, pers. obs.).

SHOREBIRDS

The definition of “shorebird” is even more elusive than that of “seabird”; since it encompasses a wide group of birds, including “waders” and any bird species that is found near the edge of bodies of water at some point during their annual cycle (Hansen *et al*. 2004). However, for the purpose of this document, a shorebird is a bird species belonging to suborders Scolopaci (phalaropes and sandpipers) and Charadrii (plovers, avocets, stilts, and oystercatchers). Other waders such as egrets, herons, cranes, bitterns, ibises, and flamingoes are not considered shorebirds in this discussion (van Tuinen *et al*. 2004).

Shorebirds are among the world's greatest migrants; many species travel from the high Arctic regions to the southern limits of Australasia, Africa, and South America. Because of the huge distances travelled, virtually any of the migrant species is capable of turning up unexpectedly, even on the "wrong" continent (Hayman *et al*. 1986). Shorebirds show a high degree of site fidelity, returning to their breeding or wintering sites year after year; and they often show a remarkable return-rate to the vital "stepping-stone" wetlands on the migration route (Hayman *et al*. 1986). Many shorebirds will return to the same area of the same beach or estuary each year on migration (Saliva pers. obs.). Most shorebirds are highly gregarious and often occur in mixed species flocks; and several species remain in the region late enough to be seen in breeding plumage (Raffaele *et al.* 1998). At the Cabo Rojo Salt Flats, migratory species start to arrive in late July; with many individuals remaining through the winter, while others depart before December (Grear 1992).

SHOREBIRD SPECIES

The sandpiper group is characterized by their long necks and legs, and thin pointed bills; and most of them wade in shallow waters or on wet flats where they probe in the mud for invertebrates (Raffaele *et al*. 1998). The most common species from this group found in Puerto Rico include Semipalmated sandpiper (*Calidris pusilla*), Western Sandpiper (*Calidris mauri*), Least Sandpiper (*Calidris minutilla*), Greater Yellowlegs (*Tringa melanoleuca*), Lesser Yellowlegs (*Tringa flavipes*), Solitary Sandpiper (*Tringa solitaria*), Willet (*Catoptrophorus semipalmatus*), Spotted Sandpiper (*Actitis macularia*), Ruddy Turnstones (*Arenaria interpres*), Sanderling (*Calidris alba*), Stilt Sandpiper (*Calidris himantopus*), Short-billed Dowitcher (*Limnodromus griseus*), and Common Snipe (*Gallinago gallinago*). Small sandpipers (primarily semipalmated and western sandpipers) comprise 60-70% of the shorebirds present on salt flats during autumn migration (Wunderle *et al*. 1989, Grear 1992).

Sandpipers such as Upland Sandpiper (*Bartramia longicauda*), Hudsonian Godwit (*Limosa haemastica*), Marbled Godwit (*Limosa fedoa*), Whimbrel (*Numenius phaeopus*), Red Knot (*Calidris canutus*), White-rumped Sandpiper (*Calidris fuscicollis*), Pectoral Sandpiper (*Calidris melanotos*), Dunlin (*Calidris alpina*), Buff-breasted Sandpiper (*Tryngites subruficollis*), Ruff (*Philomachus pugnax*); as well as phalaropes such as Wilson's Phalarope (*Phalaropus tricolor*) and Red-necked Phalarope (*Phalaropus lobatus*), are uncommon or vagrant in Puerto Rico (Raffaele *et al*. 1998).

Plovers are chunky birds with relatively shorter necks, bills, and legs than sandpipers; and a distinctive broadening at the bill tip which sandpipers lack (Raffaele *et al*. 1998). Plover species regularly found in Puerto Rico include Black-bellied Plover (*Pluvialis squatarola*), Snowy Plover (*Charadrius alexandrinus*), Wilson's Plover (*Charadrius wilsonia*), Semipalmated Plover (*Charadrius semipalmatus*), and Killdeer (*Charadrius vociferus*). Piping Plover (*Charadrius melodus*), and American Golden Plover (*Pluvialis dominica*) are rare visitors to Puerto Rico.

Stilts and avocets are noisy, gregarious, mostly-black-and-white wading birds with very long legs; with one species, the Black-necked Stilt (*Himantopus himantopus*), commonly found in Puerto Rico. The American Avocet (*Recurvirostra americana*) is vagrant and very rare in Puerto Rico (Raffaele *et al*. 1998).

Oystercatchers are very distinctive, fairly large shorebirds with large, brightly colored bills that are unusual in that they are laterally compressed (Raffaele *et al*. 1998). One species is found in Puerto Rico; the American Oystercatcher (*Haematopus palliatus*).

SHOREBIRD NESTING HABITAT

The lower reaches of Puerto Rico's rivers are estuaries where fresh and salt water mix, and saltwater can flow upstream in these rivers as far as several kilometers inland (Miller and Lugo 2009). Shorebirds use estuarine forested shrub swamps found along the coast; where there are tides, currents, and storm surges that expose the habitats to salty conditions as exemplified by the coastal forests dominated by red and black mangroves (Miller and Lugo 2009). Some shorebird species also use coralline cays as foraging, breeding, and roosting sites (Saliva, pers. obs.). Some species would also forage along sandy beaches or frequent hypersaline areas to utilize the abundant prey resources found there (Mahoney and Jehl 1985). The Cabo Rojo salt flats, part of the Caribbean Islands National Wildlife Refuge, have been identified as an important migratory staging area for at least 22 species of shorebirds that use these salt flats annually (USFWS 1993, Collazo *et al*. 1995). Findings indicate that this hypersaline lagoon system is the single most important converging point for migrating shorebirds on the island during autumn migration and winter (Grear 1992).

Most of the shorebird species that are found in Puerto Rico only use the available habitats as stopovers during their northward or southward migrations to re-fuel. Four shorebird species breed on the ground in salt flats: Black-necked Stilt, Killdeer, Wilson's Plover, and Snowy Plover (Grear 1992); but Killdeer would also nest in manmade, recently-mowed fields (Saliva, pers. obs.). American Oystercatchers and Wilson’s Plovers regularly nest in offshore coralline cays southwest of Puerto Rico (Saliva, pers. obs.).

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON SEABIRDS AND SHOREBIRDS

Effects of weather and climate on birds can be long term, occurring over hundreds of years, or as short as a passing rain storm (Schreiber 2002). The long-term effects of weather on birds undoubtedly have helped shape their particular demography and other life-history characteristics affecting the decision to nest each year, where and when to nest, annual nest success, chick growth rate, and adult survival (Schreiber 2002). Effects can be direct (*e.g.*, difficulty flying, thermal stress, flooding of nests, loss of eggs, extirpation of the species) or indirect (*e.g.*, destruction of nesting habitat, change in food distribution, decreased visibility of prey). Weather can affect the availability of food, cost of catching food, transportation cost of food, ability to find food, timing of the breeding season, number of birds that attempt to nest in a given season, clutch size, reproductive success, chick growth, thermoregulation, and adult survival (Schreiber 2002). At both high and low latitudes, unpredictable changes in food availability induced by environmental events cause changes in the onset of breeding and increased mortality of adults and chicks (Finney *et al.* 1999). When changes in local environmental conditions cause a decrease in the available food supply, adult birds often desert nests going elsewhere to find food (Schreiber 2002).

The following are climate change stressors affecting seabirds and shorebirds in Puerto Rico:

a. Sea level rise/salt water intrusion

b. Increased precipitation

c. Increased frequency/severity of storms

d. Increased irradiation/increased mean temperatures

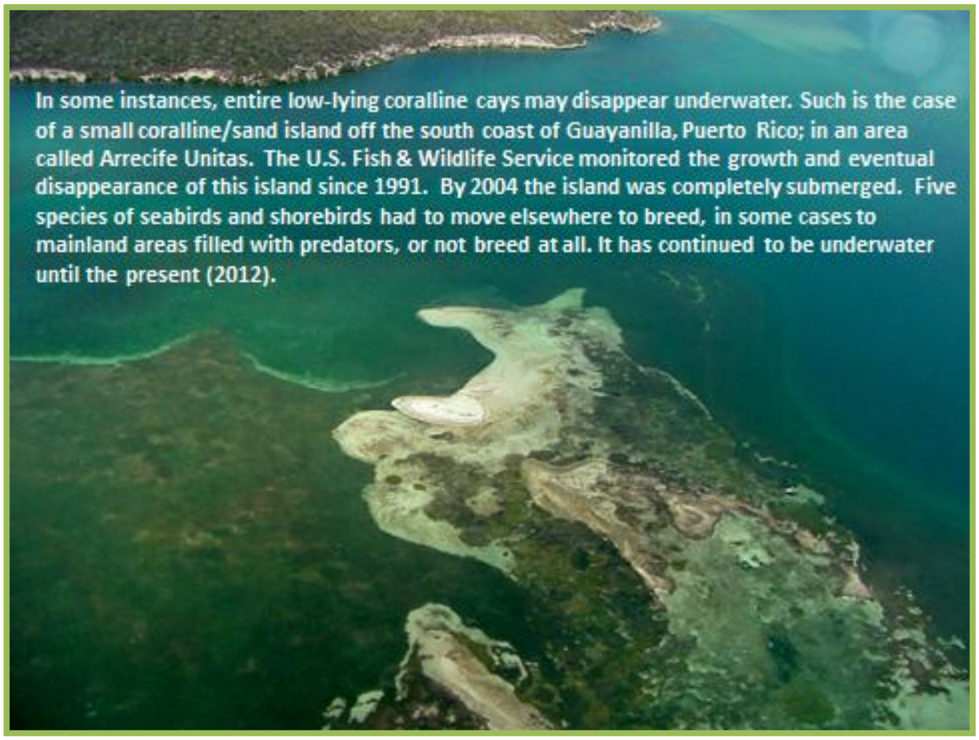
e. Ocean acidification

SEA LEVEL RISE/SALT WATER INTRUSION

Significant change in water level would impact prey availability and nesting habitat conditions for shorebirds. Grear (1992) found that the dominant abiotic factors likely to affect the macro-invertebrate community preyed upon by shorebirds at the Cabo Rojo Salt Flats are water level and salinity. Of the alternative explanations advanced to examine the distribution patterns among selected habitat units, it appears that changes in calidrid distribution were due to shifts in both densities and depth (*i.e.*, accessibility) of preferred prey in the eastern and middle lagoons versus the mud flat (alternative 3). Thus, marked changes in the distribution of calidrids among these habitats in 1990 may be explained on the basis of physical conditions and invertebrate food resources (Grear 1992). Grear (1992) documented the higher quality of deeper areas in 1991 in terms of prey abundance; where he found that prey densities were several orders of magnitude higher than in similar areas in shallower water. However, although prey abundance may be higher, deeper waters would exclude short-legged shorebirds from accessing prey; and water levels above 30 cm would exclude almost all shorebirds from using this habitat despite the abundance of prey. Indirect effects of the expected sea level rise on shorebirds include starvation during migration stopovers for re-fueling, displacement into less optimal habitat, potential increase in predation in less optimal habitat, and nest failure.

Nesting habitat would be affected by the expected sea level rise as low-lying islands, such as, the coralline offshore cays and coastal salt flats of southwestern Puerto Rico, may be the most affected by sea level rise of all coastal areas. Most of these islands are at, or less than two meters (about three feet), above sea level. Storm-driven swells and high tides frequently wash over the lowest sections of these cays, and deposit large chunks of coral that form a berm on the seaside of the cays. Species that use these cays prefer nesting in the low-lying sections, where some sand and small pieces of coral aggregate, rather than on the coral berm. Unfortunately, their preferred nesting areas are the ones most prone to coastal flooding and destruction from storm surge. Nests located in these areas are regularly abandoned after a strong storm washes over the eggs (Saliva, pers. obs.), and mortality of eggs in some years may include a large portion of a colony (Douglas 2000, Saliva pers. obs.). Roseate Terns that lose their nests to storm surge may relay their nests elsewhere within or outside of the colony area; but re-nesters late in the breeding season usually have poor reproductive success (Douglas 2000, Saliva pers. obs.). For example, a Roseate Tern colony of 207 pairs at Cayo Media Luna East in southwest Puerto Rico (presumably late nesters and re-nesters from a failed colony at Cayo Turrumote II) failed completely (Douglas 2000).

In some instances, entire low-lying coralline cays may disappear underwater. Such is the case of a small coralline/sand island off the south coast of Guayanilla, Puerto Rico; in an area called Arrecife Unitas. Arrecife Unitas is a shallow sand bank surrounded by extensive seagrass beds located at (17°57'51" N, 66°46'49" W) between Punta Verraco and Punta Gotay, Guayanilla. When the U.S. Fish and Wildlife Service began surveying tern nesting areas along the southwest coast of Puerto Rico in 1991, Arrecife Unitas was composed of submerged wetlands and an above-surface small area of less than 300 square meters of coralline sand; with an elevation of less than half a meter (Saliva pers. obs.). This small sand island became to be named “Cayo Guayanilla”, and qualitative assessments were made of its development. Dr. Jorge Saliva and his FWS team observed a group of Royal Terns loafing on the island, but no nesting activity in 1991. Their first record of seabird nesting activity at Cayo Guayanilla was in the summer of 1993, when about 110 pairs of Least Tern were observed nesting on the island; which had grown considerably to a little less than a quarter acre and had a small patch of vegetation (grass) on it. In subsequent years, more submerged areas surfaced that provided nesting habitat for Least Terns, Sandwich Terns, Roseate Terns, Wilson’s Plovers, American Oystercatchers, and a Bahama Pintail (*Anas bahamensis*).



In some instances, entire low-lying coralline cays may disappear underwater. Such is the case of a small coralline/sand island off the south coast of Guayanilla, Puerto Rico; in an area called Arrecife Unitas. The U.S. Fish & Wildlife Service monitored the growth and eventual disappearance of this island since 1991. By 2004 the island was completely submerged. Five species of seabirds and shorebirds had to move elsewhere to breed, in some cases to mainland areas filled with predators, or not breed at all. It has continued to be underwater until the present (2012).

By 1995, another section of submerged wetland surfaced to form a small sand spit about 50 meters southeast of Cayo Guayanilla, and by the summer of 1996 the two sand pits had joined to form a horseshoe shape cay with an open pond in the middle and short vegetation on both spits. The cay maintained this horseshoe shape and acquired more sand and vegetation through the summer of 1999 (surprisingly, it did not appear to lose size or shape from the passing of Hurricane Georges in September 1998), but by the summer of 2000 the island had split in half by a shallow canal near its center. In 2001, it was still split by a deeper canal and the north portion was smaller in size and with more vegetation than observed in 2000. By 2003, the southern portion of Cayo Guayanilla was submerged and the northern portion was about a third of the size observed in 2001, and in 2005 the entire island was underwater. It has continued to be underwater until the present (2012), although during aerial surveys at other times of the year (*e.g.*, December) we have seen a small accumulation of sand on the original northern portion of Cayo Guayanilla. Therefore, all shorebird and seabirds species that nested at Cayo Guayanilla were forced to look for nesting areas elsewhere or not breed at all. The disappearance of Cayo Guayanilla has resulted in the loss of the main Least Tern nesting area in Puerto Rico. We believe that, Sandwich and Roseate Terns joined the nesting populations further west, whereas Least Terns shifted to nest at Punta Gotay in the mainland. However, we did not have individually banded/radio-tagged birds that we could follow to determine where the colonies moved. Any Least Terns that moved to Punta Gotay would have been exposed to potential predation by rats, mongoose, cats, and dogs; as well as human trampling.

INCREASED PRECIPITATION

The value of the Cabo Rojo Salt Flats as a migratory stopover has been established by cumulative data collected since 1985. The continued use and value of this stopover is related to the availability and abundance of food resources (Grear 1992). Furthermore, variation in autumn rainfall, both within and between years, causes salinity changes and the seasonal flooding and desiccation of algal mats to occur unpredictably. This poses a sharp contrast to shorebird stopover areas in higher latitudes where high-amplitude tidal cycles result in more predictable patterns. Thus, invertebrate species richness at the Cabo Rojo Salt Flats is likely to have been truncated by both the magnitude and the unpredictable timing of environmental stresses (Grear 1992). The timing and predictability of invertebrate productivity is an important aspect of the traditional use of stopover areas by shorebirds. Intrinsic biological rhythms of invertebrates may be of critical importance to the seasonal predictability of these areas and their traditional use by shorebirds. Although the Cabo Rojo Salt Flats do not exhibit highly predictable bursts of invertebrate productivity, the site may still be relatively predictable in comparison to other tropical wetland sites in the Caribbean (Grear 1992).

Most seabirds and shorebirds are able to avoid some changes in precipitation by simply flying to a different area (Schreiber 2002). Species whose plumage is not water-resistant (Sooty Terns, Bridled Terns, and Frigatebirds) would stay airborne and continuously shake off excess water (Saliva, pers. obs.). Although there are few studies on the behavior of birds during extreme weather conditions at their breeding sites, probably because of human risk and difficulty in studying, some observations suggest that many adult seabirds such as Sooty Terns, Roseate Terns, Sandwich Terns, Laughing Gulls, Boobies, and Pelicans can withstand heavy rains while on the nest (Saliva, pers. obs.). They would crouch down over the eggs and remain motionless waiting for the rain to pass. Species that nest at the edge of cliffs or on top of vegetation would be minimally affected by heavy precipitation, since excess water would percolate through the nest structure and into the ground. However, heavy rain that causes flooding of the nests would invariably result in nest abandonment in ground nesters such as Wilson's and Snowy Plovers, Killdeer, American Oystercatchers, and several seabird species. Increased/heavy precipitation may be fatal to White-tailed Tropicbirds, Red-billed Tropicbirds, and Audubon's Shearwaters that nest in borrows or under large rocks; due to flooding of the nests and collapse of burrows over eggs, young, or adults.

Increased precipitation may also affect shorebird and seabird species in Puerto Rico, by modifying nesting habitat conditions. Heavy rain is typically followed by increased abundance and thickness of vegetation cover in and around nesting areas. Changes in vegetation may make a colony site unsuitable for nesting, causing a decline in the number of birds in an area (Schreiber and Schreiber1989). Bird species that require or prefer nesting in open areas such as Killdeer, Wilson's Plover, Black-necked Stilt, Snowy Plover, American Oystercatcher, Brown Booby, Masked Booby, Royal Tern, Sandwich Tern, and Least Tern may find their nesting habitat encroached by vegetation; thus, unsuitable for breeding. Other species such as Roseate Tern and Laughing Gull, which may nest either on bare ground or under/near vegetation, would probably adapt well to increased vegetation cover; moving to whichever appropriate site is available. Species that prefer some vegetation cover over or near their nests, such as the Sooty Tern, Bridled Tern, and Audubon's Shearwater may benefit from increased vegetation; if vegetation is not so dense to impede their movement in and out of the nest. However species such as the Magnificent Frigatebird, Brown Pelican, and Red-footed Booby, which nest on top of thick vegetation, would likely benefit from additional native vegetation cover.

INCREASED FREQUENCY/SEVERITY OF STORMS

Flying to a different area may not be a suitable alternative for nesting seabirds and shorebirds during severe storms; especially when hatching approaches and the parents are reluctant to leave their nests unattended. Mortality may be particularly high if adult birds do not leave the colonies soon enough before a severe storm hits the area. Additionally, if storms are frequent and severe, species may not have sufficient time to successfully relay and raise young. Not much is known about the foraging behaviour of seabirds during severe storm events, but food may not be accessible to most seabird species during severe storms primarily because of mechanical constraints imposed by their foraging techniques (plunge diving) in high, gusty winds. Shorebirds, being less aerial than seabirds, may experience heavier mortality at their breeding areas during severe storms; directly from the impact of strong winds, and indirectly from impaired food acquisition for both adult and young birds.

Philopatric species (those that return to nest at the same site every year) may be the most affected by severe storms that destroy habitat and make it unsuitable for nesting in subsequent years. If nest loss occurs early enough in the breeding season, many species will relay their nests; but when it occurs late in the season, few to no birds relay, possibly due to insufficient time to complete the cycle or because of energy constraints, or both (Schreiber 2002). Species that depend on vegetation for nesting may find their nesting habitat temporarily modified (*e.g.*, heavy salt spray that withers vegetation), severely modified to the extent that is unsuitable for the species (*e.g.*, severe defoliation), or destroyed (*e.g.*, coastal shrubs uprooted). Although some species may make relatively fast adjustments to the loss or modification of vegetation (*e.g.*, Brown Pelicans and most tern/gull species), other species may take longer or totally abandon the area (*e.g.*, Magnificent Frigatebird and Red-footed Booby). After Hurricane Hugo in September of 1989, Brown Pelicans at Dutchcap Key in St. Thomas, U.S. Virgin Islands nested on bare ground and the top of small bushes, after the canopy of tall trees such as *Bursera simaruba* and *Pisonia subcordata* was eliminated by the storm (Saliva, pers. obs.). Red-footed Boobies, on the other hand, did not nest on their usual *Ficus citrifolia* tree at the top of Cayo Geniquí in the Culebra archipelago, after the same hurricane defoliated that tree; and did not return to that cay until two years later when they nested on another, younger tree (Saliva, pers. obs.).

Severe storms are accompanied by heavy winds which could be fatal to species that nest on top of vegetation. Magnificent Frigatebirds, Brown Pelicans, and Red-footed Boobies construct feeble nests that are easily destroyed during strong winds. Eggs and young birds may fall from the nests, and perishes from ground impact, predation, or starvation (Saliva, pers. obs.). Juveniles of these three species are clumsy and gangly, and can be knocked off from their nests or perching branches by strong winds, falling through the vegetation and becoming either fatally entangled or falling to the ground and starving to death from abandonment (Saliva, pers. obs.). Although Boobies and Pelicans may nest throughout the year, their peak nesting periods are during the winter months (November-February) or early spring (April-May). Nesting of these species outside of the rainy season may be an adaptation to avoid mortality of eggs and young birds during strong winds. However, if climate change results in changes in the seasonality and duration of storms, these species may experience significant mortality of eggs and young during severe storms.

INCREASED IRRADIATION/INCREASED MEAN TEMPERATURES

Irradiation and mean habitat temperatures affect the behaviour of shorebirds and seabirds, and may have profound effects on breeding outcomes. Seabirds use various behavioural methods to thermoregulate when overheated or chilled; from changing incubation postures in gulls and terns during hot days to huddling in groups during cold spells in penguins (Bartholomew and Dawson 1979, Saliva 1995).

Although seabird species that nest under thick vegetation may not normally experience heat extremes, unusual weather events may negatively affect, directly or indirectly, the behavior of adult and young birds. At temperatures above the effective evaporative cooling temperature, Sooty Terns will leave their nests unattended and fly off to the water to dip their chest/belly feathers to cool off; leaving their nests unattended and exposed to predators and heat stress during this time (Saliva 1995). Thus, increased temperatures brought about by climate change may alter adult nest attendance and indirectly contribute to nest failure, not only in species whose nests are directly exposed to solar radiation (*e.g.*, Roseate Tern, Sandwich Tern, Royal Tern, Least Tern), but also species that nest under shelter such as the Sooty Tern, Bridled Tern, and Laughing Gull. During prolonged, extreme temperature periods, the shelter provided by vegetation may prove ineffective in maintaining sufficiently cool temperatures on their nests to allow normal incubation and chick rearing.

The extent to which adult survival is affected by environmental variability is poorly known due to the lack of sufficient studies using marked birds (Weimerskirch 2002). However, environmental conditions affecting food supply, such as increasing ocean and water temperatures, may influence survival of long-lived species such as seabirds and shorebirds. A classic example is the relationship between climate (El Niño Southern Oscillation-ENSO), schooling fish, and seabirds (Guanay Cormorant- *Leucocarbo bougainvilli*, Peruvian Booby- *Sula variegata*, and Brown Pelican- *Pelecanus occidentalis*) off the coast of Perú. Warm temperatures brought about by ENSO cause fish to change their travelling paths; avoiding the waters near seabird colonies and causing mass seabird die offs. Through techniques in modeling of survival it has been possible to relate the survival of adult Emperor Penguins (*Aptenodytes forsteri*) to oceanographic anomalies related to the Antarctic Circumpolar Wave that produces warm events (Weimerskirch 2002). During the warm events that occur every 4 to 5 years, adult survival drops to low values; some years to 0.75, whereas in other years survival is 0.92 to 0.97. However, although adult seabird survival may not be affected in some species because they can fly to follow food, young seabirds may be abandoned and starve to death when fish behavior is altered by climatic events. For some species like Roseate Tern and Least Tern, which depend on predatory fish such as species of tuna (*Thunnus* spp.), mackerel (*Scomberomorus* spp.), dolphinfish (*Coryphaena* spp.), and jacks (*Caranx* spp.) to drive schools of larval or small fish up near the water surface; absence of predatory fish may result in unavailability of prey fish, even though the prey fish may be present.

OCEAN ACIDIFICATION

Ocean acidification reduces the ability of marine algae and free-swimming zooplankton to maintain protective shells, and thus the survival of larval marine species, including commercial fish and shellfish, is also reduced (NOAA 2008). This could have indirect profound consequences on seabirds that depend on larval fish for feeding young chicks. If larval fish are not available to seabirds because plankton stocks have dwindled due to ocean acidification, massive mortality of young seabirds may occur; reducing the ability of those seabird species to replenish the populations. Seabird species that forage on larger fish or that can switch prey items to larger fish may have a slight, albeit temporary, advantage while stocks of larger fish are available.

Long term, indirect impacts of ocean acidification include interference with the formation of coralline cays where several shorebird and seabird species nest, which may be hindered by the disappearance of reef-building calcareous corals and algae (Wootton *et al*. 2008). Therefore, seabird and shorebird species that depend on coralline cays would be forced to select less suitable nesting areas, or not reproduce at all.

RESEARCH AND INFORMATION NEEDS FOR SEABIRDS AND SHOREBIRDS **(NOT IN ORDER OF IMPORTANCE)**

1. Study the effects of water level and salinity fluctuation on *Dasyhelea* (midge flies), *Trichocorixa* (waterboatmen), and algae through controlled experiments. For example, the existing seasonal cycles of desiccation and flooding, as opposed to permanent flooding, may be critical to the maintenance of algal mats and insect productivity for shorebirds.

2. Establish a careful banding program for the long-term monitoring of populations and colony-site shifting, and conduct studies on the demography and dispersal rates of selected shorebird and seabird species using band-recapture techniques. Main target species should be Least Tern, Roseate Tern, Sandwich Tern, Royal Tern, and American Oystercatcher.

3. Establish feral mammal eradication programs on islands currently or potentially used by shorebirds and seabirds.

4. Conduct Least Tern habitat assessment and studies of breeding success at different colony

sites. Least Terns nest in areas prone to flooding, sea level rise effects, and mammalian predators. Their numbers are very small and little is known about the status of their populations and breeding success.

5. Habitat restoration of seabird nesting areas at historical nesting locations in Cayo Matojo (Royal and Sandwich terns), Cayo Lobito (Royal and Sandwich terns), Cayo Geniquí West (Brown Booby), Peninsula Flamenco (Sooty Tern), and Cayo Yerba (Sooty Tern)-Culebra archipelago; through habitat manipulation (removal of invasive plants and encroaching dead vegetation) and the use of decoys to attract terns to the manipulated habitat.

6. Attracting Red-footed Boobies and "seeding" of their historical nesting areas in Desecheo Island and Cayo Geniquí; through vocalization playbacks, the use of decoys, and translocation of immature Red-footed Boobies from Mona or Monito Islands.

7. Colony assessment and habitat enhancement for Least Terns nesting in salt flat areas of Balneario de Boquerón and Refugio de Vida Silvestre de DRNA in Cabo Rojo; mud flats of La Jungla, Guánica; and Punta Gotay, Tallaboa, through the construction of elevated platforms, use of decoys, and vocalization playbacks.

8. Enhancement of Roseate Tern nesting habitat on coralline cays off Lajas and Guánica, PR; through the use of artificial, wooden nest structures (boxes) and Roseate/Sandwich Tern decoys.

9. Re-assessment of rat eradication at Cayo Don Luis (southwestern Puerto Rico) and determination of Brown Pelican re-occupancy and nest success after eradication. Brown Pelican breeding success has been very low on this island, possibly due primarily to rat predation. Rats were eradicated from this island in 2010, but we have not been able to follow up on potential rat re-invasion and Pelican nest success after eradication. This would serve as part of post-delisting monitoring that the Service has to conduct as part of the Brown Pelican delisting process.

10. Assessment of rat invasion at Sooty Tern, Noddy Tern, Brown Booby and Masked Booby nesting sites at several low-lying cays in the Reserva de La Cordillera, and manipulation (removal) of invasive plant species to enhance nesting habitat for these species.

11. Conduct habitat vulnerability assessments for low-lying cays that currently harbor shorebird and seabird species, and other such cays not currently occupied.

12. Conduct status surveys, habitat characterization, and breeding success studies of Snowy Plover and American Oystercatcher in Puerto Rico.

SUMMARY OF CLIMATE CHANGE IMPACTS ON SEABIRDS AND SHOREBIRDS

Over the last twenty years, seabirds have had a substantially poorer conservation status than non-seabirds, and they have deteriorated faster over this period. Seabirds are more vulnerable to environmental stressors (*e.g.*, storms, predators, habitat modification, and man-made factors) on their breeding sites than most land birds, because of the protracted period of nest occupancy and the concentration of complete regional populations in a few sites. Furthermore, populations are slow to recover from disturbance because of their low reproductive output.

Cliffs, mudflats, salt flats, inland artificial lakes, coastal saline ponds, and sand deposits on river mouths within the main island of Puerto Rico or close to its shore are also important roosting, foraging, staging, or breeding sites for seabirds and/or shorebirds, such as: Northwest Cliffs (between Aguadilla and Camuy); Cueva del Indio

(Camuy); Isla del Frío (off Ponce); the mouth of the Añasco, Cibuco (Manatí), and Arecibo rivers; Lago La Plata (Toa Alta-Naranjito); Lago Carraízo (Trujillo Alto-Caguas); Laguna de Las Salinas (Ponce); La Jungla Salt Flats (Guánica); Cabo Rojo Salt Flats (Cabo Rojo); and Boquerón Natural Reserve (Cabo Rojo).

The following climate stressors affect seabirds and shorebirds in Puerto Rico: (1) sea level rise and saltwater intrusion; (2) increasing precipitation; (3) increased severity of storms; (4) increased irradiation and mean temperatures; and (5) ocean acidification (figure 7). The expected sea level rise (see Working Group 1 report) have the indirect effects on seabirds and shorebirds of starvation during migration stopovers for re-fueling, displacement into less optimal habitat, potential increases in predation in less optimal habitat, and nest abandonment and mortality of eggs and chicks. Such is the case of a small coralline/sane island off the south coast of Guayanilla; in an area called Arrecife Unitas. The U.S. Fish and Wildlife Service monitored the growth and eventual disappearance of this island in 1991. By 2004 the island was completely submerged. Five species of seabirds and shorebirds had to move elsewhere to breed, in some cases to mainland areas filled with predators.

Increased precipitation may modify nesting habitat conditions, availability and abundance of food resources, cause nest abandonment, or even death of young and adults. Increasingly more intense storms may destroy habitat and result in unsuitable nesting areas in subsequent years, and impaired food acquisition. Increasing irradiation and mean temperatures may alter adult nest attendance and prey fish behavior and indirectly contribute to nest failure. And lastly, ocean acidification could interfere with critical processes such as reef building, carbon sequestration via phytoplankton sedimentation, and consumer-resource interactions.

More research is needed to determine how and to what extent Puerto Rico's seabirds and shorebirds are vulnerable to climate changes. Colony assessments and habitat enhancement for Least Terns nesting in salt flat areas of Balneario de Boquerón and Refugio de Vida Silvestre de DRNA in Cabo Rojo; mud flats of La Jungla, Guánica; and Punta Gotay, Tallaboa, through the construction of elevated platforms, use of decoys, and vocalization playbacks should be conducted. Status surveys, habitat characterization, and breeding success studies of Snowy Plover and American Oystercatcher in Puerto Rico should also be a research priority. And habitat vulnerability assessments for low-lying cays that currently harbor shorebird and seabird species, and other such cays not currently occupied.

1. **Increased precipitation** may modify nesting habitat conditions, availability and abundance of food resources, cause nest abandonment, or even death of young and adults.

2. **Increased severity of storms** may destruct habitat and result in unsuitable nesting areas in subsequent years, and impaired food acquisition.

3. **Increased irradiation and mean temperatures** may alter adult nest attendance and prey fish behavior; indirectly contributing to nest failure.

4. **Ocean acidification** (declining pH) could interfere with critical processes such as reef building, carbon sequestration via phytoplankton sedimentation, and consumer-resource interactions.

5. **Sea Level Rise** Indirect effects of the expected sea level rise on seabirds and shorebirds include starvation during migration stopovers for re-fueling, displacement into less optimal habitat, potential increase in predation in less optimal habitat, and nest abandonment and mortality of eggs and chicks.

Figure 7 Figure 1 Summary of how five climate stressors could potentially affect Puerto Rico's Seabirds and Shorebirds

FORESTS

INTRODUCTION TO PUERTO RICO'S FORESTS

Forest composition and structure are strongly tied to climate – at large scales as climate governs seasonality, temperatures, and precipitation – and at local scales where climate interacts with topography, elevation, and soils to affect site characteristics. In this section we first provide an overview of the type and extent of the dominant forests in Puerto Rico, and identify the major stressors related to climate change that are likely to affect those forested systems. We then treat each forest type in more detail and discuss the potential outcomes and consequences of specific climate stressors on the biological communities of forested ecosystems. The resulting effects on human society regarding the provision of important ecosystem services are considered briefly as well. We conclude the section with a summary of the multiple stressors and forest types together, and highlight important research and information gaps that remain.

More than half of the Puerto Rican archipelago is covered by forests, woodlands and shrublands (Gould et al. 2008a). We have categorized this forest cover in four broad groupings (figure 8) in order to assess outcomes of changing climate stressors on: (1) Coastal Lowland Forests – including lowland moist forests and woodlands and freshwater Pterocarpus swamps (mangrove forests are not discussed in detail here having already been

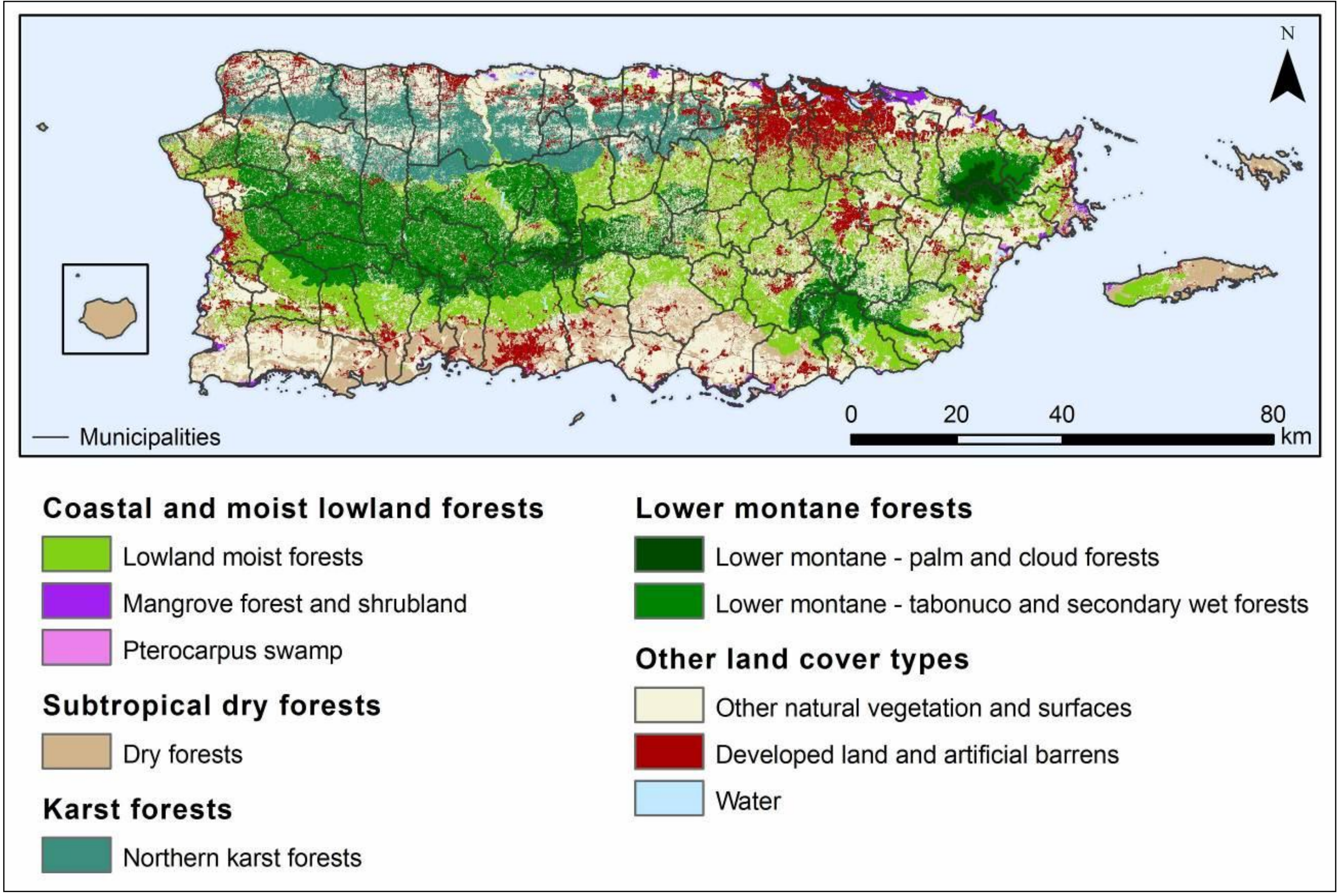


Figure 8 The dominant forest classes of Puerto Rico as considered in this climate change vulnerability report

addressed in the previous section on Wetlands); (2) Dry Forests – including forests, woodlands and shrublands in the subtropical dry Holdridge lifezone (Ewel and Whitmore 1973); (3) Karst Forests – including the forests of the northern limestone region; and (4) Montane Forests of the Central and Luquillo Mountains.

Non-limestone lowland moist forests are found on the coastal plain and lower mountain slopes below about 150 m in northeastern Puerto Rico, and ranging to 400 m or more in the Central Mountains. These forests and woodlands occupy 165,000 hectares or about 16 percent of the islands, and are typically secondary forests regenerating on former pasture or other agricultural lands with *Spathodaea campanulata* being a common nonnative tree. In the coastal wetlands, Pterocarpus swamps occupy just 261 hectares, and mangroves comprise some 8700 hectares, for a combined total of 1 percent of the islands. Other forests in coastal wetlands are typically secondary forests regenerating on occasionally or seasonally flooded former pastures.

The dry forests are found near the coast in the northeast on volcanic substrates and somewhat more inland on the south and southwest coasts on limestone substrates. They also occur on the ultramafic substrates of the Susúa State Forest in the southwestern on part of Puerto Rico. Together dry forest communities occupy about 70,000 hectares, about 8 percent of the islands. Much of this area is woodland and shrubland and only about 4 percent of the islands are closed dry forest.

The northern Karst forests on limestone substrates occupy over 72,000 hectares, or just over 8 percent of the islands. Forest composition and structure reflect past human uses as well as steep topographic gradients associated with the haystack hills, or "mogotes" typical of karst topography. The majority of the region can be characterized as having steep slopes, with a lesser portion as valley bottoms, followed by summits (Martinuzzi et al. 2007). The summits tend to have drier, less well-developed soils, with the greatest soil moisture and development in the valley bottoms. Summits and slopes tend to have calcareous soils, with deeper, noncalcareous alluvial soils in the valley bottoms. As a whole the forests are very diverse, with a large number of endemic species. Variation in composition and structure tends to correlate with topographic position, and canopy heights and basal area increase from summit to slope to valley bottom (Lugo 2005b).

The lower montane forests are to be found, depending on location, at elevations above 150 to over 400 m in the four wetter Holdridge life zones that occur in Puerto Rico (Ewel and Whitmore 1973): subtropical wet forest, subtropical rain forest, subtropical lower montane wet forest, and subtropical lower montane rain forest. These forests collectively occupy about 158,000 hectares, or nearly 18 percent of the islands. They include the well described tabonuco, palo colorado, sierra palm and elfin woodland forests of the El Yunque National Forest in the Luquillo Mountains, several forested parks in the Central and Carite Mountains, and forest growing on abandoned pasture and coffee plantations throughout the Central Mountains.

We have identified four major climate change stressors that are likely to affect Puerto Rico's forests:

- Sea level rise
- Increased severity of tropical storms
- Decreased precipitation/increased drought/increased seasonality
- Increased irradiation/increased mean temperature and extremes

The Working Group 1 report addresses in detail the driving mechanisms behind each of these stressors, as well as observed trends and future projections. In the following paragraphs we will discuss these stressors in the context of each of the four dominant forest categories presented.

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON PUERTO RICO'S FORESTS

COASTAL AND MOIST LOWLAND FORESTS – INCLUDING FRESHWATER PTEROCARPUS SWAMPS, AND OTHER LOWLAND MOIST FORESTS AND WOODLANDS

Elevational and hydrologic gradients control the occurrence of natural vegetation and also favor certain human activities in the coastal zone. Among them are topographic and associated salinity gradients at or very near sea level that influence the distribution of mangrove species (Lugo and Snedaker 1974); the growth of secondary littoral woodlands under the influence of wind and salt spray along shorelines; the regeneration of secondary semi-evergreen seasonal forest on abandoned pastures and hills up to 100 m; and the development of coconut plantations on relatively level terrain. The northeastern sea coasts were originally occupied by littoral woodland and mangroves (Wadsworth 1950), freshwater swamp forests, and different types of subtropical dry forests. Pterocarpus formed extensive stands in freshwater swamps and extended to higher elevations along riparian corridors. Species associated with drier conditions include Bursera simaruba and Guaiacum officinale, which were likely prevalent on less protected coastal slopes and hilltops, and Bucida buceras which also grows in coastal basins and along streams (Gill 1931**,** Little and Wadsworth 1964). In protected locations, species associated with moist forests such as Calophyllum calaba, Manilkara bidentata, Tabebuia heterophylla, and Mastichodendron foetidissimum were likely common.

PTEROCARPUS FORESTS

In the Caribbean, swamps formed by *Pterocarpus officinalis* trees (hereby Pterocarpus forests) occur in topographical depressions influenced by freshwater inputs, particularly in low-lying coastal areas (Bacon 1990). Adapted to flooded ecosystems, Pterocarpus forests occur in river floodplains, coastal basins and subtropical rain forests in Puerto Rico (Alvarez-Lopez 1990). Early assessments of the Puerto Rico flora described *Pterocarpus officinalis* as an abundant species covering most of the mountainous region of the island (Bates 1929, Alvarez-Lopez 1990). However, during the early 1900's, Puerto Rico lost most of its Pterocarpus cover to deforestation driven by agricultural practices. By 1980, the total area of Pterocarpus forests in Puerto Rico was estimated to cover only 238 ha in 15 locations (Cintrón 1983), while the most recent estimate calculates their extent to 261 hectares based on 2000 Landsat imagery analysis and classification (Gould 2008a). This increase in wetland cover can be attributed to the abandonment of agricultural practices in coastal plains and the associated reestablishment of natural hydrological cycles (Martinuzzi et al. 2009). Nevertheless, Pterocarpus forests now occur in genetically isolated populations typically restricted to the coast, abutting mangrove ecosystems (Eusse and Aide 1999, Rivera-Ocasio et al. 2007), an important factor when considering the amount of biodiversity implied in a particular forest. The remaining stands of Pterocarpus forests in Puerto Rico are highly diverse, including basin and riverine forests in coastal and mountainous areas. In the Humedal Punta Viento Natural Ecological Reserve, located in the town of Patillas, a groundwater-fed spring provides a small Pterocarpus stand enough freshwater to survive in spite of being encroached by mangroves in a coastal basin site that receives less rainfall than the Guánica Dry Forest (Alvarez-Lopez 1990, Van Bloem et al. 2007). Over the next decades, these remaining stands of Pterocarpus forests face a new set of threats, mainly because of environmental changes driven by sea level rise.

STRESSORS AND CONSEQUENCES TO PTEROCARPUS FORESTS

The importance of hydrology in wetland ecosystems is well known (Mitsch and Gosselink 2007). The pulsating nature of the wetland's flooding and drying periods transfers energy and materials through the system driving many aspects of the ecosystem structure and function in the process (Odum et al. 1995). It is precisely this close relation between hydrology and the ability to maintain a functional ecosystem what makes freshwater forested wetlands in coastal plains particularly vulnerable to climate change. For Pterocarpus forests, stressors related to increased temperatures – such as sea level rise and increased variability in precipitation patterns – could have a profound effect in the hydrological processes of the wetland.

SALT-WATER TOLERANCE AND SEA LEVEL RISE

The majority of the remaining stands of Pterocarpus forests in Puerto Rico are located in the coast, restricted to the limits of their physiological distribution in terms of salinity tolerance (Eusse and Aide 1999). This proximity to the ocean represents yet another threat for these remaining stands, due to the increased risk of saltwater intrusion. Research conducted on Pterocarpus forests has shown that increased soil and water salinity affects litter, flower and fruit production and that salinities above 14‰ can kill populations of these trees (Eusse and Aide 1999, Rivera-Ocasio et al. 2007). In addition, salinity affects recruitment and growth, and even small changes (~1‰) reduced nodulation in the roots of seedlings (Saur et al. 1998, Fougnies et al. 2007, Rivera-Ocasio et al. 2007, Dulormne et al. 2009). Still, this species has several adaptations that allow the tree to tolerate and avoid lower level salinities (Eusse and Aide 1999). For instance, their buttress roots create mounds of sediment that modify the micro-topography of the forest by providing areas of higher elevation around trees (Koponen et al. 2004). In addition, they accumulate excess of sodium in the rachis of the leaf, away from photosynthetic tissue, and have a shallow root system that is likely driven by a strategy to avoid the denser saltwater in the lower depths of the water table (Medina et al. 2008). Nevertheless, the increasing presence of salinity in the environment creates stressful conditions that disrupt the normal ecological processes of Pterocarpus forests.

Over the course of this century, sea level rise will bring more salt water into Pterocarpus dominated ecosystems in Puerto Rico. Our current understanding of how these important wetlands will respond is the prediction that they will either migrate landward or be lost, according to the Brinson et al. (1995) model of coastal habitat migration. During long term events of salt water intrusion into coastal wetlands, freshwater species can be substituted by salt tolerant species, such as mangroves, and the zonation of existing mangroves forests can also change according to the migration of species according to their salt tolerance (Lugo 2006). In small islands like Puerto Rico, landward migration of coastal ecosystems could be limited by the concentration of human populations in the coast, poor land use practices and the further development of areas adjacent to the coast (Helmer 2004).

Coastal wetlands like mangroves forests can cope with the inundation threat of sea level rise by increasing their relative elevation to the sea through vertical accretion of peat, in some cases by up to 4 mm year^{-1} (McKee et al. 2007, McKee 2011). Other studies suggests that coastal swamps will be unable to cope with the projected rates of sea level rise (3-5 mm year^{-1}, IPCC 2007) although there is still some debate on the different methodologies used to measure accretion rates and draw substantial conclusions (McKee 2011). The accretion of sediments in Pterocarpus forests as a coping strategy against sea level rise has received limited research attention and, therefore, needs further study. However, in the case of Pterocarpus forests, the threat of increased inundation periods could be secondary to saltwater intrusion, which can continue to affect the ecosystem even after the retreat of an inundating tide. Furthermore, the effect of salinity in the production of organic matter in Pterocarpus forests (Eusse and Aide 1999) could hinder the ability of these ecosystems to establish accretion rates capable of coping with long term sea level rise.

DROUGHTS AND RAINFALL EVENTS

Regardless of the uncertainty or disagreements in climate models that seek to predict and quantify changes in precipitation patterns related to climate change, most models agree on the expected long term trends for the Caribbean region (Neelin et al. 2006, Allan and Soden 2007). The general agreement states that regions that lie outside the convection zones of atmospheric circulation – particularly the region of Central America and the Caribbean – are expected to go through a significant drying trend over the next 100 years (Campbell et al. 2011). IPCC (2007) data show that the maximum number of consecutive dry days in the Caribbean is decreasing. This result is consistent with the decreasing trends of precipitation over the last 50 years in Puerto Rico, where higher air and sea surface temperatures affect the hydrological cycle at a smaller scale.

Coupled with this long-term drying trend, warmer sea surface temperatures are likely to increase the frequency and intensity of pulsated rainfall events like hurricanes and tropical storms. For instance, the most active hurricane season ever recorded in the Atlantic Ocean – which included devastating storms like Hurricane Stan in Guatemala and Hurricane Katrina in New Orleans – occurred during the record breaking sea surface temperatures of 2005 (Trenberth and Shea 2006). The 2011 hurricane season was the third most active on record, tied with the 2010 season (Strachan 2012), and brought unusual amounts of rain to Puerto Rico during the late summer months.

Extreme variability in the quantity and temporal distribution of precipitation in the Caribbean (as directly related to storm events) could affect ecological processes like recruitment, root development and seedling survival in forested wetlands (Lopez and Kursar 2007). Regarding Pterocarpus forests, the predicted conditions can interact to exacerbate salt water intrusion events, particularly among riverine forests sites. For instance, if annual rainfall is concentrated in pulsated rain events that will become more frequent and intense during the two peaks of rainy seasons observed in the Caribbean (Comarazamy and González 2011), and these events are followed by increasingly dry periods, tidal influence can be intensified on riverine wetlands by the opening of the river mouth during the pulsated rain event and the subsequent lack of freshwater inputs during the extended dry periods. Hydroperiods affected under altered precipitation patterns can alter important life history events like seed distribution and establishment (Lopez and Kursar 2007). Pterocarpus trees produce floating seeds that are dispersed through the landscape via hydrochory (dispersal of seeds by water) near the end of the flooding period. This adaptation allows seeds to reach areas of higher elevation in the complex microtopography of the forest and establish during the following dry season (Alvarez-Lopez 1990, Lopez 2001). However, prolonged flooding periods can affect germination of seeds by reducing the oxygen content of the soil and depleting the seeds' reserves, reducing the overall success of hydrochory as an adaptation strategy. In contrast, if heavy rainfall becomes more common during the dry season, establishment of Pterocarpus seedlings will likely be facilitated by this adaptation {Lopez, 2007 #41}.

In the context of sea level rise, freshwater inputs delivered by precipitation and its associated runoff have not yet been discussed as a potential mediator of saltwater intrusion in Pterocarpus forests, though they have for coastal dune plants (Greaver and Sternberg 2007, 2010) and hardwood hammock tree species (Sternberg and Swart 1987, Ish-Shalom et al. 1992). The benefits of freshwater inputs to the resilience of Pterocarpus forests to saltwater intrusion needs further research attention due to the possibility of catchment-scale hydrological restorations as possible strategies to manage these forests in the light of climate change-related events.

Each of the discussed climate change-related stressors has the potential to affect Pterocarpus forests in distinct ways, with the added complexity brought by the interactions between them (Lugo 2006). Between 2004-2009, over 630,000 acres of forested wetlands were lost in the tropical and subtropical regions of the United States (U.S. Fish and Wildlife Service 2009). As for mangroves forests, there is a lack of data of the ecosystem's capacity for carbon storage (Donato et al. 2011), making Caribbean wide studies on the amount of carbon stored in freshwater forested wetlands important to better understand the consequences of this loss. Studies of the hydrology, biodiversity and ecosystem processes across varying scales in the Caribbean could help us have a clearer picture of the future of these wetlands and develop useful management strategies for long term disturbances related to the changing climate of the earth.

LOWLAND MOIST FORESTS

Forest occurring on noncalcareous substrates within the subtropical moist lowland Holdridge life zone (Ewel and Whitmore 1973) occupy 165,000 hectares or about 16 percent of Puerto Rico (Gould et al. 2008a). These are typically very young to mature secondary forests regenerating on former pasture or other agricultural lands on the coastal plain, hills, and lower slopes of the eastern and north central part of the island. They are a mix of native and non native species. Of the top four species in terms of importance values recorded by the US Forest Service Forest Inventory and Analysis in a 2003 survey, the nonnative Spathodaea campanulata had the highest value, followed by the native species Guarea guidonia, Andira inermis, and Cecropia schreberiana (Brandeis et al. 2007).

STRESSORS AND CONSEQUENCES TO LOWLAND MOIST FORESTS

Sea level rise will likely affect only a small portion of these forests, but this includes littoral forests on beach strands, coconut plantations, and floodplain forests that may experience increased salinity, or altered flooding regimes. A potentially more serious stressor to these forests is increased seasonality, lower levels of precipitation or extensive drought. Effects may be to reduce growth rates, alter species composition and increase the likelihood of fire. The most abundant species in moist lowland forest are not well adapted to extensive dry periods and increased mortality, opening of forest canopies would likely lead to an increase in fine fuels (grasses) and increased occurrence of fire. Increasing storm frequency and severity could also impact these forests. Although they are well adapted to storms and recover quickly, the combination of storm damage (loss of canopy, downed limbs, mortality) and drought would enhance fire danger (Gould et al. 2008b).

These moist lowland forests serve as habitat for a number of common native bird, amphibian, and reptile species, as well as a number of migratory bird species (Gould et al. 2008a). Loss in forest area or shifts in composition, fruiting and phenology would likely affect a number of species. Additional research is needed in this area, as the particular impacts of climate change would have to be assessed on a species by species basis.

SUBTROPICAL DRY FORESTS

About 40% of the tropics and subtropics are covered by open or closed forest. Of this 42% is dry forest, 33% is moist forest and about 25% is wet and rain forest. The dry forests account for about 22% in South America and 50% in Central America. The subtropical dry forest life zone (Ewel and Whitmore 1973) covers approximately 17% of Puerto Rico and its adjacent islands and cays. It includes the outlying islands of Mona, Desecheo, Culebra, Caja de Muertos and the majority of Vieques. On the main island of Puerto Rico it covers southwestern Puerto Rico from Santa Isabel to Cabo Rojo and a small area near Ceiba and Fajardo. It is the driest of the life zones, with a mean rainfall of about 600 to 1,000 mm per year. Dry forests vary in species composition and structure depending on soil types, which vary across the life zone (Ewel and Whitmore 1973).

Generally, dry forests are smaller and less complex floristically and structurally than wet forests. The vegetation tends to form a complete ground cover and, on most soil types, is deciduous. Leaves are often small and succulent or coriaceous, and species with thorns or spines are common. Tree heights do not usually exceed 15 meters and the crowns are broad, spreading and flat. Plants are usually low in moisture content and the wood of most species is hard and durable. In these forests the seasonality of rainfall is a dominant ecological factor and patterns of growth and reproduction are synchronized with the availability of water. In most tropical and subtropical dry forest, two dry periods are characteristic, a minor one in the summer months and a major one in the winter (Ewel and Whitmore 1973).

Sutropical dry forest varies in structure and species composition depending on substrate types, which vary across the life zone. Substrates in the life zone include limestone, volcanic, alluvial and ultramafic igneous (Gould et al. 2008a). The subtropical dry forest has been converted and fragmented in large part to agriculture and grazing, urban and rural development, and tourism. Remaining protected areas in this life zone include the Guánica Commonwealth Forest, the Cabo Rojo, Laguna Cartagena, Culebra, Desecheo and Vieques National Wildlife Refuges, the Mona, Monito, and Caja de Muertos Natural Reserves, among others. The integrity of these natural areas is threatened by encroaching adjacent development, invasive species, and fire.

STRESSORS AND CONSEQUENCES TO SUBTROPICAL DRY FORESTS

The principal climate related stressors or threats to the dry forests in Puerto Rico include those associated with precipitation, increased temperature, and increased storm severity. While it is not necessarily clear what the changes in precipitation will be, there is consistency in the predictions concerning increases in temperature. The areas containing seasonally dry tropical forests are expected to receive increases in temperature from between 2 to 4 C by 2100 (IPCC 2007). Even in the absence of changes in precipitation, this warming will result in a reduction in soil moisture availability, or in other words, increasing drought (Meir and Pennington 2011). Increased severity of tropical storms is predicted to result from warming sea surface temperatures (Emanuel 2005). Sea level rise may affect dry forest communities in coastal areas.

INCREASED TEMPERATURE/DECREASED PRECIPITATION/INCREASED DROUGHT

Dry forests in Puerto Rico typically are subject to a primary dry season, from December to April. The severity and extension of this dry season is of particular concern in that it may affect phenology (flowering and fruiting patterns) and the successful establishment of seedlings. While it varies among species, there is general agreement that in dry forests the availability of moisture plays a major role in seedling recruitment and survival. Seeds from most dry forest tropical species mature in the dry season and are dispersed at the beginning of the wetter season when moisture is available for germination and growth (Ewel 1980; Singh and Singh 1992). Vegetative regeneration is an important mechanism in disturbed dry forest sites, and would become even more important when regeneration by seed is affected by changes in the seasonality of precipitation (Murphy and Lugo 1986). This sensitivity to changes in moisture will make dry forests particularly vulnerable to climate change, potentially eliminating the more drought sensitive species and resulting in shifts in species composition in these forests (Condit et al. 1996). The increased dependency on vegetative regeneration or coppicing will likely cause shifts in species composition to species with a greater ability to resprout. This could have consequences for the breeding success of faunal species that depend on fruiting and flowering resources, such as frugivorous birds (Faaborg 1982; Faaborg and Arendt 1992; Dugger et al. 2000).

Species composition may also be affected by differing growth rates as a result of decreasing moisture availability and increasing concentrations of CO^2. One study has indicated that growth of a dry forest tree may be reduced by 12% under a medium emission scenario and as much as 21% under a high emission scenario. This is particularly important due to the aerial extent of dry forests in the tropics. If tree species exhibit differing rates of growth reduction, then species composition in those forests may be affected (Brienen et al. 2009). Tree mortality may be increased by increasing length and intensity of drought. Trees may respond in differing ways to moisture stress, either through stomatal control to maintain leaf water potential or through maintaining a water supply from the soil. The former reduce the risk of hydraulic failure but increase the risk of mortality by "carbon starvation." Some studies indicate that the major cause of mortality in seasonally dry forests is hydraulic failure (Meir and Pennington 2011). Shifts in species composition may occur as a result of this increased mortality.

Increases in temperature and increased drought periods have been shown to result in the increased potential for more frequent and intense wildfires. While in Puerto Rico most wildfires are human-related, an extended and more severe drought can create conditions conducive to more intense and larger fires. High intensity fires may lead to significant loss of woody vegetation, and if seed sources are not available, regrowth of native woody vegetation may be restricted (Meir and Pennington 2011). This often permits the dominance of invasive species, and in dry forests of Puerto Rico such as Guánica this includes African forage grasses. With successive burning these grasses expand into intact forest. Germination and survival of native forest seedlings and saplings is low due to high water deficit, high temperatures and high light levels (Wolfe and Van Bloem 2012).

Restoration of degraded dry forests may become more difficult. Dry forests that have been cleared or degraded often develop into exotic grassland communities that are maintained by fire and are difficult to reforest. Areas that are replanted are subject to high mortality from fire, but also from increased moisture stress and temperature (Wolfe and Van Bloem 2012). Studies have shown that successful restoration projects can be high maintenance and therefore costly, requiring mulching, irrigation or both. The application of such treatments can increase survival significantly (Martinez and Van Bloem 2009). Increased temperatures and more intense and prolonged droughts can therefore make restoration a more costly endeavor.

SEA LEVEL RISE

It is anticipated that rising seas will affect coastal morphology, causing mangroves to move inland and encroach on dry forest, with consequences for the species that depend on that habitat (Kerr 2009, Blunden et al. 2011). Resident and migratory birds that rely on dry forest for breeding and over-wintering, respectively, could be particularly vulnerable if dry forest does not expand into new areas at the same rate it is converted to estuarine habitats. For example, Rodriguez Colón (2012) investigated North American migrant songbirds that depend simultaneously on interconnected mangrove and dry forest habitat in the south coastal region of Jobos Bay National Estuarine Research Reserve, and concluded that these populations will be imperiled by sea level rise if dry forest habitat is reduced, degraded, or eliminated. Avian populations in other coastal dry forest areas in Puerto Rico may face a similar threat.

INCREASED SEVERITY OF STORMS

Most models indicate that while storm frequencies may not increase, they do agree that storm intensity will increase. The physical impacts of this increased intensity include increased damage from high winds and increased landslides and flooding. This may result in changes in species composition and structure of the forest. The structure of Caribbean dry forests is shorter and contains a larger proportion of multi-stemmed trees than other neotropical dry forests. Following disturbance such as hurricanes, dry forests resprout near the base. Studies in the Guánica dry forest have shown that hurricane-induced tree sprouts have the longevity and growth characteristics to contribute to the structure of the forest (Van Bloem et al. 2005). More intense hurricanes may result in a structure that is comprised of a greater proportion of multi-stemmed species and, in addition, the species composition may shift towards a greater proportion of species that have the ability to sprout. More intense storms may also result in mudslides and landslide that open the area for invasion of non-native species.

KARST FORESTS

The northern Karst forests on limestone substrates occupy over 72,000 hectares, or just over 8 percent of Puerto Rico. Aside from the protected Luquillo Mountains these represent the largest area of contiguous forest on the island. The steep topography of the haystack hills region and the underground drainage of the limestone affect both the hydrology and the human use of the landscape. In spite of the steep terrain the land was heavily used for agriculture in the last century and nearly all forests in this region are secondary forests on abandoned agricultural land (Chinea 1980, Rivera and Aide 1998, Aukema et al. 2005). In spite of the past human deforestation much of the reforested land is characterized by native species, and reflects natural gradients in soil characteristics and site moisture associated with topographic variation. Hill tops are well-drained and relatively dry, hill slopes are steep, well drained, but with more soil moisture, and valley bottoms typically have deep humid soils. There is a north-south gradient in rainfall in the Karst region, with more rainfall in the higher elevations of the southern portion of the region and less rainfall in the north (Lugo and Castro 2001).

STRESSORS AND CONSEQUENCES TO KARST FORESTS

Given that strong site moisture gradients are already present within the karst region, these forests may to some extent be well-buffered against changes in temperature and precipitation. Small increases in dry season length, occasional droughts, or reductions in annual rainfall may not affect dry-adapted species on hill tops – and moist adapted species may remain sheltered in valley bottoms that continue to receive adequate runoff. Extensive drought or reductions in annual rainfall that are significant enough to alter the subterranean hydrologic system and dry out valley bottoms could have significant effects by decreasing growth rates and increasing canopy openness and mortality – leading to increased likelihood of fire and shifts in species composition, structure, and ecological function. Increased storm severity and accompanying damage could further exacerbate these effects (Peng and Wang 2012).

LOWER MONTANE FORESTS

Montane forests cover a total habitat area of about 158,000 hectares, or nearly 18 percent of Puerto Rico, and include elevations above ~400 m in the Central and Carite Mountains, as well as above ~150 m in El Yunque National Forest (ENYF). Within this broad elevation gradient, specific forest types are distinguished by edaphic conditions, combined with rainfall and topographic relief, as defined by Holdridge life zone associations (Ewel and Whitmore 1973). At lower elevations subtropical wet forests dominate, where average annual rainfall can vary between 2000-4000 mm, and the weathered soils are derived from volcaniclastic sediments and serpentinite substrates (Ewel and Whitmore 1973, Miller and Lugo 2009). The subtropical wet forest assemblage is characterized by stratified, closed canopies reaching heights greater than 30 m and with a rich diversity of species (Little et al. 1974). The upper elevations are characterized primarily by lower montane wet, subtropical rain, and lower montane rain forest. Abundant annual rainfall (as much as 5,000 mm in EYNF) produces swampy and nutrient impoverished soils resulting in evergreen forest assemblages with trees of 18 m average height. At the stable mountain peaks the forest displays the gnarled and heavily epiphyte-laden characteristics of the elfin forest. Average canopy heights of only 6 m and extremely wet and depleted soils distinguish this cloud forest habitat. More fragile, steeply-sloped mountaintop areas are dominated by palm-brake forests (Ewel and Whitmore 1973, Lugo 2005a, Miller and Lugo 2009).

In the many centuries since European colonization of Puerto Rico, the lower montane forests of the Central and Carite Mountains (excluding EYNF) have gone through a cycle of clearing, conversion to agriculture, and subsequent forest regeneration with the advent of industrial development and agricultural abandonment (Franco et al. 1997). As of the most recent forest inventory in 2004, 57% of mainland Puerto Rico was covered in forest, woodland, or shrubland, with considerable increases in forest cover in subtropical moist, wet, and rain forests, much of which is still in the early stages of development (Brandeis et al. 2007).

Lower montane forests harbor most of the Puerto Rican endemic flora and fauna including birds, amphibians, bats, reptiles, invertebrates, trees, and plant species (Miller and Lugo 2009). In addition, watersheds in these habitats supply water to all major reservoirs in Puerto Rico. Therefore, the resulting effects of climate stressors will be relevant to several important ecological functions and related ecosystem services.

STRESSORS AND CONSEQUENCES TO MONTANE FORESTS

The dominant climate change-induced stressors that are likely to have a direct impact on montane forest communities in Puerto Rico include increased severity of tropical storms, decreased precipitation and an associated increase in drought, and increased irradiation and temperature. Indirectly, sea level rise may have an impact as well. Global circulation models currently project an increase of sea surface temperatures in tropical regions of 1-2 °C during the next century (IPCC 2007). This may result in greater cyclonic storm intensity (Emanuel 2005, Shepard & Knutson 2007), and may also cause a decline in precipitation by as much as 50% in some regions, including the Caribbean (Hulm & Viner 1998, Neelin et al. 2006, Gamble & Curtis 2008).

INCREASED SEVERITY OF TROPICAL STORMS

A large body of research allows us to make general predictions about how Puerto Rico's montane forests might respond to changes in the intensity of tropical storms. Lugo (2000) summarized the research regarding the effects of hurricanes on Caribbean forests, identifying 12 dominant bio-geophysical outcomes: sudden and massive tree mortality; delayed patterns of tree mortality; alternative methods of forest regeneration in gap vs. landslide areas; changes in plant successional direction due to novel dispersal opportunities and recruitment sites; high species turnover and opportunities for species change in forests; diversity of age classes; faster biomass and nutrient turnover; species substitutions and changes in turnover time of biomass and nutrients; lower aboveground biomass in mature vegetation; creation of carbon sinks; selective pressure on organismal reproduction; and convergence of community structure and organization.

Should the intensity of hurricanes increase, Lugo (2000) predicts that they will cause more damage (*i.e.*, larger gaps and landslides) and reset the recovery trajectory further each time. This will lead to a greater average abundance of shade-intolerant secondary species in forests over time, as mature forests yield to the augmented pressures of cyclonic disturbance. Consequently, the quantity and average height of aboveground biomass in these secondary forests will decrease due to the recurrent interruptions of vegetation growth. And species that better reproduce under disturbance conditions will thrive at the expense of species that require long periods of stability in order to mature. Long-term data collected from the Luquillo Forest Dynamices Plot, a 16-hectare grid of tabonuco forest located in EYNF, corroborate some of these predictions. Since 1990, repeated monitoring of trees, seedlings, and reproductive flowering/fruiting have demonstrated that forest community dynamics are clearly affected by the close connection between hurricane damage and light availability (Zimmerman et al. 1994, Uriarte et al. 2004, Uriarte et al. 2009). The animal populations associated with these plant communities are also vulnerable. Wunderle (1995), for example, reported a homogenizing of understory and gap assemblages of birds in the EYNF, when comparing the results of post-Hurricane Hugo (1989) monitoring surveys with prior results. This pattern persisted even after new gaps and the forest understory regained their unique foliage profiles. More frequent and intense cyclonic storms could therefore result in permanently altered faunal assemblages and food webs.

The impacts of cyclonic storms on tropical forest communities must, however, be placed in the context of the legacy effects of human land use (Chazdon 2003, 2008). The recovering forests of Puerto Rico exhibit legacies of land use in the form of altered species composition that persist for decades (e.g., Thompson 2002, Garcia-Montiel 2002). Although human disturbance promotes secondary species that are able to colonize more rapidly after human disturbances (Grau et al. 2003), Uriarte et al. (2009) found that the interaction of the legacy of past human use and the contemporary natural disturbance caused by hurricanes will probably result in a forest community that includes both primary and secondary species in a composition not previously witnessed. Thus, continuing interactions of humans and hurricanes are expected to remix native and introduced plant and animal species into novel communities adapted to anthropogenic environmental conditions (Lugo and Helmer 2004, Lugo 2009).

DECREASED PRECIPITATION/INCREASED DROUGHT

Drought might seem an unlikely event in Puerto Rico's moist and wet lower montane forest communities. Yet several studies in the Luquillo Mountains have highlighted the impacts of droughts on both aquatic and terrestrial systems (Covich et al. 2000, Beard et al. 2005, Heartsill-Scalley et al. 2007). As mentioned earlier, climate models based upon field and satellite data predict a decrease in rainfall of 5-50% over the next 100 years for the Caribbean and Central America (Neelin et al. 2006). If precipitation in the region declines as anticipated then the frequency of short-term droughts is also likely to change, with an increase in the number of rainless or low-rain days per observation period.

Seed production and seedling recruitment in tropical forests are closely linked to inter-annual variation in climate, indicating the potential sensitivity of these processes to global and regional anthropogenic influences (Wright 2005, Wright et al. 2005, Wright and Calderon 2006). Changes in climate could directly affect relative competitive abilities of species specialized to different environmental niches, or indirectly affect recruitment via effects on pathogens and herbivores. In contrast to dry forests, decreased precipitation in Puerto Rico may ultimately boost greater seed production in lower montane forests, because a reduction in average seasonal evapotranspiration and cloudiness would lead to increased overall irradiation and likely an increase in flowering/fruiting fecundity in lower montane forests (Wright 2005, Zimmerman et al. 2007). Seedling establishment and growth are also likely to be affected by periods of lower or no precipitation, depending upon the degree to which shade tolerance and drought sensitivity are correlated. Any increase in drought frequency could cause drought-insensitive secondary species to become even more common in the community (as has been shown in many temperate species, see Sack 2004, Niinemets & Valladares 2006). On the other hand, if shade and drought tolerance are not related, then we can expect a pronounced change in drought frequency to encourage different, novel assemblages of species and distribution patterns. Either way, the probable result of a drier climate combined with the effects of land use history and hurricane disturbance is a change in the composition, abundance, and distribution of species in Puerto Rico's lower montane forests.

SEA LEVEL RISE

As for sea level, the impacts of this stressor are most obvious for coastal areas (Nicholls and Cazenave 2010): in recent decades there has been a massive shift in Puerto Rico towards an industrial and service-dominated economy, which has resulted in rapid urban expansion in low-lying zones (Lopez Marrero et al. 2001, Lopez Marrero 2003). It is worth considering if the potential climate-induced displacement of human communities in coastal habitats will trigger increased competition for resources on mid-upper elevation lands and forested environments. The intensity of development for housing, recreational, infrastructure, and large, continental-style strip malls and shopping centers in Puerto Rico is already considerable in the lower cordillera forests, fragmenting important landscapes with human infrastructure (Helmer 2004). Should these development pressures migrate upslope in response to sea level rise, the conversion of forest to built environment will further reduce and fragment lower montane forest habitat.

INCREASED IRRADIATION AND TEMPERATURE

Changes in local and regional climate can lead to increased vulnerability of cloud forests (Mulligan 2010). Although development is not likely to directly encroach on the higher elevations, there is evidence to support a coupling of land use and air temperatures in lowland regions with orographic cloud formation in tropical montane forests (Scatena 1998, Still et al. 1999, Lawton et al. 2001, Ray et al. 2006). Modeling of Costa Rican cloud forests demonstrated that the orographic cloud bank intersected the mountains at the lowest elevations, covered the largest land surface area, and remained longest on the surface in the montane regions when the lowland and premontane regions were completely forested. In contrast, deforestation raised the surface sensible heat fluxes, warming the air temperature and lowering the dew point of air masses, ultimately reducing the cloud forest area covered with fog in the montane regions by as much as 13% and raising the orographic cloud bases by 75 m (Ray et al. 2006).

In Puerto Rico the importance of lowland land cover on the height of the cloud base has also been documented. Scatena and Larsen (1991) reported that immediately following the severe deforestation of the Luquillo Mountains by Hurricane Hugo in 1989 the cloud condensation level was temporarily lifted by several hundred meters. According to Lugo (2005a), the daytime heat produced by the ring of urban infrastructure outside the National Forest affects the cloud condensation level of the Luquillo Mountains, and is potentially modifying the mountains' climate as well. Increases in temperature and solar irradiation may ultimately enhance the heat-island effects that accompany deforestation and urban expansion in coastal areas, with consequences for ecosystems and species at the highest elevations (Pounds et al. 1999; Colwell et al. 2008). In a pantropical study of hydrological processes in montane cloud forests, Bruijnzeel, Mulligan, and Scatena (2010) concluded that epiphytes and bryophytes in the upper parts of the forest canopy are particularly susceptible to climatic warming and drying, given their exposed position and dependence on precipitation inputs for their moisture and nutrient needs. If simultaneously subjected to increasingly intense tropical storm disturbance as well, it is probable that cloud forest vulnerability will be compounded even further.

AREAS OF GREATEST VULNERABILITY

Biological communities that are restricted to environments with specific temperature and precipitation ranges, independent of forest type, are most vulnerable in the short-term to the projected changes associated with a changing climate. This includes habitats located at the "extremes" – coastal lowlands, high elevation peaks, and the dry forests. While some species may potentially migrate upslope (e.g., Colorado trees, Cyrilla racemiflora, see Scatena 1998), species already bumping up against the upper limits of their range may not be so fortunate, and could be diminished or lost from Puerto Rico altogether due to the lack of suitable environmental conditions, or the biological environment (e.g., pathogens, competition from congeners). Fluctuations in fruiting/flowering phenology and upward shifts in vegetation could in turn create niche vacancies for previously suppressed or opportunistic exotic species to subsequently occupy (Platt and Connell 2003).

Directional changes in forest dynamics will in turn be determinant of how well upper elevation communities harbor animal populations. Two avian examples from the lower montane forests are the endemic Elfin-woods Warbler and the Puerto Rican Sharp-Shinned Hawk. As with high elevation plants, it follows that climate change might further restrict the birds' range as a result of habitat loss due to hurricane damage, or an altered prey base (Delannoy 1997, Anadón-Irizarry 2006). Dry forest avifauna that could potentially be affected by climate change include endangered species such as the Yellow-shouldered Blackbird and the Puerto Rican Nightjar, species whose distribution is restricted primarily to the arid habitats along Puerto Rico's southern coast (Atlas 2009; Vilella and Gonzalez 2009). Similarly, (Pounds et al. 1999) implicated dramatic declines in anuran populations in Costa Rica with recent warming of global sea surface temperatures and a corresponding movement upward of the orographic cloud base.

Certain lowland species are at risk not necessarily because of environmental barriers associated with temperature and precipitation gradients, but rather because of the lack of available habitat due to the presence of humans and their associated infrastructure. As discussed with regards to Pterocarpus forests, which are already severely limited in geographic scope, burgeoning urban populations in adjacent coastal areas could effectively halt the landward migration of the Pterocarpus in the face of rising sea level (Helmer 2004). There is an obvious need for more research to support the link between climatic change and the consequent impacts on biodiversity. Even so, a warming climate will almost certainly create challenges for high elevation, coastal, and dry forest species situated in insular settings like Puerto Rico.

SOCIETAL CONSEQUENCES

Up to this point we have discussed the biophysical and ecological outcomes and consequences that would likely occur in Puerto Rico's forests as the direct results of various climate change stressors. There may also be important consequences for human communities. In addition to the high toll of emotional and economic damage caused by destruction of built infrastructure, human life, and property (Pielke et al. 2003, Pielke et al. 2008), an increase in hurricane intensity and the accompanying deterioration or loss of habitat would potentially reduce the ability of Puerto Rico's forests to provide essential ecological services. These services include absorbing storm impacts of wind and rain, conserving soil resources and processing organic matter, storing carbon, supplying millions of gallons of water, filtering urban air pollution and regulating air temperatures, affording opportunities for recreation and tourism, and functioning as essential habitat in the life cycle of estuarine organisms that contribute to the coastal food chain (Lugo 1994, 2005a, Scatena et al. 2002).

Taking water yield as an example, Bruijnzeel, Mulligan, and Scatena (2010) reported that the impact of climate change on the water balance of cloud-affected forests is dominated by the anticipated changes in rainfall. Therefore, if Puerto Rico experiences an overall drying trend as is currently projected, the supply and flow of water from upland to lowland areas will likely be diminished. With public demands for fresh water increasing – water withdrawl from streams draining the El Yunque National Forest increased by 70 percent from 1994 to 2004 (Cook et al. 2007) – any reduction in the forests' capacity to supply abundant water could have unfavourable consequences for the island's human population.

Harder to anticipate are the outcomes that will ensue as a result of changes in species composition, structure, and growth rates, due to increased temperatures, drought, solar irradiation, and fire frequency. It is reasonable to assume that novel forest communities in Puerto Rico will continue to protect soils, cycle nutrients, support wildlife, store carbon, maintain watershed functions, and mitigate species extinctions, as put forward by Lugo (2009). Yet there will clearly be changes in the types of and the extent to which these services will be provided, the details of which are not yet well understood.

Forested ecosystems also have cultural, historic, aesthetic, and economic values that will change in response to climate stressors. In El Yunque National Forest and southwestern Puerto Rico, for example, activities such as bird watching and wildlife photography bring tourists, both local and foreign, and contribute to the local economy. Many people come searching for a particular species or experience that is afforded by these unique habitats. Climate stressors that result in the decreased abundance of and, therefore, visibility of high profile forest species may in turn affect the island's tourism industry.

Society's vulnerability to changes in the natural environment is determined in large part by numerous socio-economic and political factors acting at diverse temporal and spatial scales (Turner et al. 2003, Uriarte et al. 2011). Accordingly, climate-induced changes to ecosystem services must be examined in the context of contemporary land use practices as well, for loss in habitat and service provision will be amplified by other threats to natural resources, including urban and rural development.

RESEARCH GAPS AND INFORMATION NEEDS FOR PUERTO RICO'S FORESTS

There are currently a broad range of institutions and agencies engaged in multi- and interdisciplinary research endeavors in Puerto Rico. These efforts are aimed at investigating the long-term changes in disturbance regime and climate (including natural and human pertubations), and their effects on floral and faunal assemblages, biogeochemical dynamics, and ecosystem services. Yet several large knowledge gaps remain, particularly with regards to the specific details of habitat loss, shifts in fruiting/flowering patterns, and altered species composition. The linkages between climate change, ecological function, and the provision of ecosystem services need considerable illumination as well.

Additional environmental monitoring of all forest types will help fill these gaps. It is especially important in lowland and coastal areas where the combined stressors of human development pressures and climate change are likely to have the most significant impacts on forested landscapes. Karst areas, too, are lacking in data compared to lower montane and dry forests. More data from more sites will improve our ability to interpret the scales and rates at which climate-induced stressors affect forests, the degree of resilience to these pressures, and the thresholds beyond which irreversible change occurs. Directed studies focused on individual stressors are not sufficient; understanding of the combined, synergistic effects of multiple stressors on different forest types will be crucial to developing a nuanced model of the system dynamics that drive ecological transitions, and thereby improve society's ability to predict the consequences. Finally, clarification of the relative impacts of similar climate change stressors on diverse forest ecosystems of different spatial scales will help better identify areas and species of highest priority. In an era of rapid climatic change, this knowledge can then be translated into policy mechanisms aimed at adaptively managing forested ecosystems into the future.

SUMMARY OF PUERTO RICO'S FORESTS AND CLIMATE CHANGE

Puerto Rico's dominant forest types (coastal and lowland and Pterocarpus forests; dry forests; karst forests; and lower montane forests) will all be affected by one or more of four major stressors associated with a changing climate: sea level rise, increased severity of tropical storms, decreased precipitation/increased drought, and increased irradiation/temperatures. Complex interactions among these stressors make it difficult to tease out simple cause and effect relationships among climate drivers and the resulting impacts on forested ecosystems. Nevertheless, we can still draw some general conclusions about the anticipated outcomes and consequences (Table 3).

All forest types are likely to see alternations in flowering and fruiting phenology. The establishment and survival of seedlings will be affected as well, which will result in changes in species composition, distribution, and abundance. Over the long term, increased seasonality coupled with cyclonic storm impacts will signal directional changes in forest regeneration pathways. This will ultimately lead to novel communities that harbor unique assemblages of plants and animals, and exhibit altered forest structure and ecological functions. Biological communities located at the extremes of moisture, temperature, and elevation gradients – mountain peaks, dry forests, and coastal lowlands – are most at risk. Increased solar irradiation and temperatures and the effects on cloud condensation will impact forest communities at the highest elevations. Increases in the intensity and length of droughts and increased wildfires are a serious concern in dry forest communities. In coastal areas salt water intrusion will be a factor, as sea level rises and inundates lowlying forests. Development pressures could further complicate the ability of lowland forests to adapt. Forested ecosystems also have cultural, historic, aesthetic, and economic values that will change in response to climate stressors. The delivery and flow of important forest ecosystem services may be compromised by climate-induced alterations, with potentially detrimental effects for the human communities that depend on them.

Table 3 Summary of outcomes/consequences of climate change stressors on Puerto Rico's dominant forest types

FOREST TYPE	**STRESSORS**				**BIOPHYSICAL OUTCOMES**						**SOCIETAL CONSEQUENCESES**
	Sea level rise	Increased storm severity	Decreased precipitation/ increased drought	Increased solar irradiation/ temperature	Salt water intrusion	Increased seasonality/ changes in flowering & fruiting phenology, seedling establishment	Increased wildfires	Elevated cloud condensation level	Changes in species composition, distribution, and abundance	Novel plant and animal assemblages with unique structure and function	Altered quantity and quality of forest ecosystem services (e.g., provision of clean water, carbon storage, economic and recreation opportunities)
Pterocarpus forests	√	√	√		√	√			√	√	√
Lowland moist forest & woodlands	√		√		√	√			√	√	√
Dry forests	√	√	√	√	√	√	√		√	√	√
Karst forests		√	√	√		√	√		√	√	√
Lower montane forests	√	√	√	√		√		√	√	√	√

AMPHIBIANS AND REPTILES

INTRODUCTION TO AMPHIBIANS AND REPTILES

AMPHIBIANS

The extant amphibians (from the Greek “amphi” or both ways and “bios” or life; Class Tetrapoda, Sub-Class Lissamphibia) are tetrapods that include frogs, salamanders, and caecilians. Classification of living amphibians includes three Orders (in decreasing order of species richness): Anura (from the Greek “an” or without and “oura” or tail), with the familiar frog and toad forms; Caudata (from the Latin “cauda” or tail), with a long body plan, tail, and four (some with only two) limbs; and Gymnophiona (from the Greek “gymnos” or naked and “ophis” or snake because it was believed that these amphibians were most closely related to snakes). Species in Gymnophiona are frequently called caecilians from the Latin “caecus” or blind, which refers to the small eyes or complete absence of eyes in some species. Caecilians are the least known amphibians with fewest species, and characterized by an elongated body with skin folds that facilitates borrowing (similar the external body plan of an earthworm), the body lacks limbs, and has a short tail.

Amphibians are ectotherms–as opposed to endotherms like humans–and regulate metabolism mostly by external environmental regimes of temperature: in simple terms, as ambient temperature raises or lowers amphibians can raise or lower their body temperature as needed. They also produce eggs with several permeable jelly membranes (similar to fish eggs) that make amphibians highly dependent of water and humidity for reproduction and normal development of embryos. External fertilization and a free-swimming larvae are predominant among amphibians. Amphibians are characterized also by a humid permeable skin that lacks hair, scutes, scales, and any structure for protection from external ambient variables and factors (Duellman & Trueb, 1994). Their skin is full of glands for various purposes (from hydration to defense), but that must be humid for normal physiology and gas exchange, most like breathing through skin. This makes an amphibian virtually like a fish out of the water. Humid conditions are most prevalent during nighttime hours, which may explain why amphibians are mostly nocturnal in habits. The diet of adult amphibians mostly includes animal prey, while the diet of aquatic larvae mostly includes vegetable matter. Amphibians are distributed worldwide except in the coldest Polar Regions, highest mountaintops that are mostly frozen, and in strictly marine environments (Duellman & Trueb, 1994; Wells, 2007).

Notable exceptions to these descriptions do exist and some species of amphibians live in deserts (e.g., the Australian frog *Neobatrachus centralis*), have internal fertilization (e.g., the Puerto Rican Common Coquí *Eleutherodactylus coqui*), are ovoviviparous (*i.e.* females retain the eggs in oviducts and “give birth” to juvenile replicas of adults like the Puerto Rican frog *Eleutherodactylus jasperi*), are diurnal in habits (e.g., the Central American frog *Oophaga pumilio*), inhabit salty mud flats (e.g., the Southeast Asian frog *Fejervarya cancrivora*), and are fully aquatic (e.g., the Bolivian frog *Telmatobius culeus* from Lake Titicaca) (Duellman & Trueb, 1994; Wells, 2007). In the Caribbean, only members of the Order Anura (frogs and toads) are represented.

REPTILES

The extant reptiles (from the Latin “reptile”, neutral of “reptilis”, or creping, crawling; Class Tetrapoda) are tetrapods that include lizards and snakes, turtles, crocodiles, and tuataras. Specifically, the extant reptiles include four Orders (in decreasing order of species richness): Squamata (from the Latin “squama” or scale for their skin covered by scales), which includes lizards, snakes, and amphisbaenians; Testudinata (from the Latin “testa” or shell, an ossified carapace), which includes turtles, tortoises, and terrapins; Crocodylia (from the Greek “kroke” or pebbles and “drilos” or worm, which referred originally to the Nile crocodile presumably for its basking behavior), which includes crocodiles, alligators, caimans, and garials; and Rhynchocephalia (from the Greek “rhynchos” or beak, snout and “kephalē” or head), which includes the tuataras. The Order Squamata is subdivided further into two Suborders: Lacertilia (from the Latin “lacerta” or lizard) and Serpentes (from the Latin “serpens” or snake). Although snakes and some lizards lack limbs, all reptiles evolved from a four limb (tetrapod) ancestor.

Reptiles, like amphibians, are also ectotherms and regulate metabolism mostly by external environmental regimes of temperature. The protection brought in by their impermeable skin, however, makes reptiles somewhat more independent of humid conditions, compared with amphibians, for reproduction and normal physiology: reptiles do not have an aquatic larval stage and their dry skin is covered with scutes, bony external plates similar to scales, for protection against dehydration. In general, reptiles are most active during daytime hours, but nocturnal species (*e.g.,* snakes) are common. The diet of reptiles generally includes animal material, vegetable material, or both. Internal fertilization is the norm among species and eggs are usually deposited in burrows or under debris in the ground (Vitt & Caldwell, 2009). Temperature-Sex Determination (TSD), instead of chromosomal-sex determination, occurs in reptiles in which males or females are produced more frequently than expected along portions of a vertical thermal gradient within the underground nest (Janzen, 1994). High prevalence of TSD in reptiles occurs most frequently in turtles and crocodilians; a low incidence of TSD is prevalent in the species-rich lizard Families Iguanidae and Polychrotidae; and a lack of TDS is notable in snakes (Valenzuela 2004).

The egg produced by reptiles, unlike amphibians, is an amniote egg (a membrane protected egg that is fertilized internally in the female). Different membranes of this egg are specialized for specific functions and provides a structure that is rather independent of external environmental conditions: gas exchange and protection from external environmental conditions (egg shell); internal protection (chorion); excretion of embryonic waste (alantoids); provide nutrients to the developing embryo (yolk); and keeps an adequate aqueous environment (amniotic liquid) for normal development. Consequently, this amniote egg represents a major evolutionary adaptation–only shared by birds and mammals–to life in dry land, making reptiles independent of water for reproduction as almost all resources for normal development are enclosed in the egg. Some species of reptiles live mostly in aquatic environments, including the open oceans, but none is fully aquatic. Consequently, reptiles need to always return to dryland to lay eggs and for normal development of embryos. Reptiles are distributed worldwide except in the coldest Polar Regions and highest mountaintops that are mostly frozen (Vitt & Caldwell, 2009).

Exceptions to these descriptions do exist and many species of reptiles are ovoviviparous (e.g., *Boa* spp.), some are obligate parthenogenic (e.g., populations consist of only one gender; the brahmini blind snake, *Ramphotyphlops braminus*, for example) and prefer humid and cooler environments over drier and warmer environments (*i.e.,* forest interior lizard like the Puerto Rican Yellow-Chinned anole, *Anolis gundlachi*, versus open habitat species like the Puerto Rican grass anole, *Anolis pulchellus*). In the Caribbean, members of the Order Testudinata, Crocodylia, and Squamata are the ones represented.

THE HERPETOFAUNAL PROBLEM

Despite the facts that amphibians and reptiles are ectotherms and that they are frequently considered collectively, differences between the two groups are substantial (see above) and their responses to climate change deserve separate analyses, which is beyond this review. Some generalities are shared, however, as their physiology and phenology are most dependent on external environmental variables related with climate change like ambient temperature and humidity, rainfall, hydroperiod, and processes affecting these variables (e.g., sea-level rise, disturbances from storms, pathogenic infections, anthropogenic activities related with land use, storms and hurricanes, among others). How precisely each of the two groups will respond to climate change challenges scientists for various reasons. For example, in Puerto Rico, seasonal variation in temperature and rainfall is small (compared to continental and temperate areas), and shows a similar annual pattern: mean monthly temperature and monthly total rainfall is somewhat higher between May and November and lower during late January and early April. In coastal ecosystems, the relative contribution of climate variables to activity patterns in amphibians and reptiles, and between amphibians and reptiles, is difficult assess as to make formal generalizations because of the similarity in the seasonal variation of climatic variables and the similarity of the activity patterns despite physiological, behavioral, and ecological differences among species (N. Ríos-López, unpubl. data). Some general principles may exist, however, based on the differences in physiological requirements between amphibians and reptiles: in general, the abundance and distribution of amphibians is mostly influenced by humidity while the abundance and distribution of reptiles is mostly influenced by ambient temperature (Pianka, 1986; Duellman, 1999; also see Ríos-López and Aide, 2007 for a discussion from Puerto Rico).

Unfortunately, a straight forward analysis of the direct effect of these climate change variables on amphibians and reptiles may be misleading because a myriad of interacting factors exist that may counteract each other (e.g., Saenz et al., 2006), exacerbate responses to climate change (e.g., Pounds et al., 2006), and because of the uncertainty of how long the effects of climate change will remain as currently projected (note, however, that IPCC [2007] documents warming and sea level rise effects, due to anthropogenic activities, to last for more than a millennium because this is the timescale required for removal of carbon dioxide from the atmosphere). The next two examples, using the most referenced climate change variable, temperature, illustrate the complexity for predicting responses to climate change by amphibians and reptiles.

In amphibians like the Puerto Rican Common Coqui (*Eleutherodactylus coqui*) Stewart & Woolbright (1996) documented a gradual decrease in species abundance, from >24,000 coquis ha^{-1} in the lowlands to < 2,500 coquis ha^{-1} in high elevation areas along a 1000-m elevation range. This difference between a high temperature and a low temperature area is presumably due to a decrease in prey availability with elevation (Stewart & Woolbright, 1996). Low temperature, however, increases developmental time of embryos in egg clutches (Townsend & Stewart, 1985), and decreases metabolism and growth rate, increases time to maturity, and increase age at first reproduction in *E. coqui* (Ríos-López, 1999). As individuals from high temperature areas (*i.e.*, lowlands) grow faster, reach sexual maturity twice as fast as those from low temperature areas (*i.e.*, highlands), and breed more frequently, population size may be higher with increased temperature (Ríos-López, 1999). Most likely, however, a cascading effect due to climate warming may include increased decomposition rates and nutrient cycling in humid areas (*i.e.*, tropical montane forests), increased carrying capacity (*i.e.*, availability of herbivorous insects), and a synergism between increased metabolism and population growth along the altitudinal gradient. If increased biomass of coquies is interpreted as increased productivity, the former scenario could be interpreted as a non-negative effect of warming to ecosystem processes like nutrient, carbon, and energy flows, as increased productivity is frequently associated with increased diversity. Obviously, this is not trivial and climate change brings up diverse consequences to ecological systems and herpetofaunal species.

In reptiles with TSD, increases in ambient temperature will affect sex ratios in nests, nest survival, and normal development and growth of hatchlings (Janzen, 1994; McCallum et al., 2009). In the American Alligator (*Alligator mississippiensis*) temperatures of 30°C and below in the nest produce females, 32°C produces males and females, 33.5°C produces only males (Ferguson & Joanen, 1982, 1983), but temperatures 35°C and above produce 90% females which are usually not viable (Murray, 2002). In contrast, cooler incubation temperatures produce only males while warmer incubation temperatures produce only females in *Trachemys scripta elegans* (Cadi et al., 2004; Tuker et al., 2008), a close relative of the Puerto Rican Jicotea, *T. stejnegeri*. Increased ambient temperature due to climate warming will influence sex ratios, and unless selection in female nest-site choice (*i.e.*, open versus shade areas) to keep pace with warming, altered sex ratios in nests will have profound consequences for population demography and dynamics (Gibbons et al., 2000). In Puerto Rico, species such as the Hawksbill Turtle (*Eretmochelys imbricata*), which nest under the vegetation in tropical beaches, may not be as affected as other species such as Leatherback (*Dermochelys coriacea*), Green Turtle (*Chelonia mydas*), and Loggerhead Turtle (*Caretta caretta*), which lay their eggs in nests in open, sandy areas in the beach. The effect of warming can also be different between species (e.g., terrestrial versus aquatic turtle species; McCallum et al., 2009), which requires further study.

THE CARIBBEAN HERPETOFAUNA AS A STUDY MODEL

The herpetofauna is nonetheless an important component for many ecosystem processes, especially in tropical islands. On Caribbean islands, which lack the large mammals that characterize continents, amphibians and reptiles, along with birds, constitute the top predators and are the most abundant and conspicuous land vertebrates (Reagan 1996a; Duellman 1999). In a Puerto Rican rain forest, for example, the combined population density of three species of lizards (Polychrotidae: *Anolis*) and one species of tree frog (Eleutherodactylidae: *Eleutherodactylus*) reached 49,800 individuals ha^{-1} (Reagan 1996b; Stewart and Woolbright 1996). One of these species, *Anolis stratulus* (Cope), has the second highest density documented for any lizard worldwide with > 2 individuals/m^2 (Reagan 1996a). Collectively, these species regulate ecosystem processes indirectly through predator-prey interactions as they consume > 550,000 prey items ha^{-1} d^{-1}, mainly insects (Reagan 1996b). This makes the Caribbean herpetofauna a major component in the local ecosystem energy, nutrient, and carbon flows (Roughgarden 1995; Beard et al. 2003; Hillman et al. 2009). As a result, given that amphibians and reptiles are particularly at risk due to climate change, responses to projected climate change will have profound consequences for herpetofaunal species diversity and ecosystem processes.

Examining responses to climate change by amphibians and reptiles for adequate conservation, management and policy making efforts requires a first exploration of projected changes in climate, both at global and regional scales. Herein, Dr. Neftali Rios-López for the PRCCC reviewed the pertinent literature and examined the relationship between climatic factors and population dynamics, species distribution, species abundance and extinction and on changes in diversity that most likely will occur with predicted climate change. Climate projections from contributions of Working Group 1 to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (2007) are briefly discussed when appropriate and responses to temperature, humidity/rainfall, sea level rise, pathogenic disease, and storms by Puerto Rican amphibians and reptiles, whenever possible, are highlighted. Areas where information is needed are also highlighted with suggestions for research agendas for the benefit of herpetofaunal conservation efforts in light of projected climate change.

CLIMATE CHANGE SCENARIO: CURRENT STATUS AND EXISTING VULNERABILITIES IN THE CARIBBEAN AND PUERTO RICO

TEMPERATURE

In marine turtles, their capacity for long-distance dispersal is one wonder of the natural world, with some species traveling thousands of kilometers across the ocean (Meylan, 1995). Sea surface temperature and chlorophyll concentrations strongly influence migration routes of marine turtles (Thomas & Dabo, 2005), which suggest that projected increase in sea surface temperature may affect migratory patterns of marine turtles. In addition, increased water temperature and ocean acidification (with increased CO_2 input from anthropogenic sources) may contribute to the death of corals, sponges, and sea grasses, which are highly important food sources for marine turtles. How sea turtles will respond to climatic change remain mostly a matter of speculation, and according to the State of the World’s Sea Turtles (SWoT) Report (2006), comprehensive long-term data sets are urgently needed to examine responses to projected changes in climate by marine turtles.

A low capacity for long-distance dispersal characterize virtually all herpetofaunal species (exceptions include marine turtles), which makes them highly dependent on local environmental conditions for reproduction and survival (Duellman & Trueb, 1994; Gibbons et al., 2000; Root et al., 2003; Vitt & Caldwell, 2009). For example, lowland species with highly patchy, disrupted distribution (e.g., in Puerto Rico, the Dry Forest Anole, *Anolis cooki* [Rodríguez-Robles et al., 2008], *A. poncensis*, and several *Sphaerodactylus* species) and those living in montane areas the ones who will suffer the most with warming (and precipitation) (Pounds et al., 2006; McCain & Colwell, 2011). Species with limited geographic distribution living in montane areas (*i.e.*, in Puerto Rico: the Dwarft Coqui, *Eleutherodactylus unicolor*, and the Yellow-Shinned Anole, *Anolis gundlachi*) will be forced to migrate upland to keep up the pace with increased ambient temperature. Lowland herpetofaunal species better suited for increased temperature (e.g., in Puerto Rico, the Antillean Coqui, *Eleutherodactylus antillensis*, the Common Grass Anole, *Anolis pulchellus*, or the Crested Anole, *A. cristatellus*) will expand their geographic distribution upland, which suggests a complete replacement of many of the montane, specialized species by lower elevation, generalist species with projected estimates of warming. This could also lead to extinction of high elevation residing species like *E. unicolor*, the Cricket Coqui, *E. gryllus*, and the forest-dwelling *Sphaerodactylus klauberi*, to name a few, if their adaptive capacity is compromised (e.g., Root et al., 2003) and because of altitudinal shortage of space beyond the highest peaks. Responses to warming by amphibians and reptiles will not be so linear, however, and other factors acting in synergism will influence these.

Species with limited geographic distribution living in montane areas (i.e., in Puerto Rico: the Dwarft Coqui and the Yellow-Shinned Anole) will be forced to migrate upland to keep up the pace with increased ambient temperature.

Many studies have documented population declines and extinction of herpetofaunal species worldwide (Wake, 1991; Blaustein et al, 2010; Gibbons et al., 2000; Reading et al., 2010). Originally, many cases of declining populations of these species have been enigmatic, but recent evidence points at climate change as a main driver for cascading effects on amphibians and reptiles at different temporal and spatial scales (Cary & Alexander, 2003; McCallum et al., 2009; Blaustein et al., 2010; Reading et al., 2010). In amphibians, for example, Pounds et al. (2006) conclude with ‘very high confidence (>99%, following Intergovernmental Panel on Climate Change, IPCC) that large-scale warming has been a key factor for amphibian population declines in tropical montane cloud forests in Central America. Pounds et al. (2006) concluded that warming encourages outbreaks of the pathogenic chytrid fungus, *Batrachochytrium dendrobatidis* (*Bd*), by converging nocturnal and diurnal ambient temperature towards growth optimum conditions of this amphibian-demising fungus, particularly during warm years associated with El Niño events. In extreme warm years, after an El Niño event, the physiology of amphibians may became compromised, the fungus growth at an optimum, resulting in observed drastic population declines and species extinctions in Central America (Pounds et al., 2006).

The phenomenon described by Pounds et al. (2006) may not be exclusive to Central America and it is highly possible that a similar situation occurs with Puerto Rican amphibians (*e.g.*, Burrowes et al. 2004): here, population declines and species extinctions of several amphibians correlate strongly with the the pathogen is first detected, last time of species detection from the wild, years with strong El Niño events, years with largest difference between mean winter temperature and mean summer temperature, and years when the difference between minimum daily temperature (*i.e.*, nocturnal temperature) and maximum daily temperature (*i.e.*, diurnal temperature) converges towards the thermal optimum growth range of the pathogen Bd (N. Ríos-López, unpubl. data). Increased pathogenic outbreaks with increased sea surface temperature have also been documented for marine turtles: Haines and Kleese (1976) argued that increased outbreaks of diseases in marine turtles peaked during El Niño events, suggesting a synergism between increased sea surface temperature and pathogenic outbreaks.

In reptiles, Readings et al. (2010) documented an alarming and steady trend of declining populations of many species of snakes (which are top predators) from distant countries like the United Kingdom, France, Italy, Nigeria, and Australia. Readings et al. (2010) found that 75% of declining populations are from species characterized by small home range, sedentary habits, sit-and-wait foragers, and most came from well-protected areas. Readings et al. (2010) reasoned that synchronized patters of declines across regions are indicative of a change in habitat quality (instead of habitat loss) suggesting a common stochastic environmental factor (e.g., climate change). In contrast, Shine (1991) documented a dramatic decline in the Australian common black snakes (*Pseudechis porphyriacus*) that specializes in frogs. Consequently, a plausible hypothesis for most snake population declines in Readings et al. (2010) may include a reduction in the availability of their prey due to a cascading effect from global warming, pathogenic outbreaks among potential prey, and a reduction in the carrying capacity among all habitat areas: the chytrid fungus has been documented in amphibians from each of the countries examined by Richards et al. (2010). Synergism between temperature (and humidity) and pathogenic infections in reptiles may also lead to drastic population declines with projected climate change. For example, the widespread upper respiratory tract disease–a contagious respiratory ailment caused by the bacterium *Mycoplasma agassizii*–is a potential cause of population declines in desert in Southwest U.S.A and in gopher tortoises in Southeast U.S.A (Jacobson, 1993; Smith et al., 1998). An unidentified bacteria that can cause a similar respiratory ailment in snakes have been found in the Puerto Rican Boa (*Epicrates inornatus*), and increased physiological stress due to climate change on the Puerto Rican Boa may exacerbate its response to pathogenic infection with profound consequences for population persistence and species survival (A. Puente-Rolón, per. comm.).

An unidentified bacteria that can cause a similar respiratory ailment in snakes have been found in the Puerto Rican Boa (Epicrates inornatus), and increased physiological stress due to climate change on the Puerto Rican Boa may exacerbate its response to pathogenic infection with profound consequences for population persistence and species survival.

Species of snakes that need high body temperature to regulate their metabolism under hot, sun-lit environments, like most dry forests in Puerto Rico, are a greater risk of overheating because dispersal requires contact with the hot substratum (e.g., Lilliwhite, 1987), thus affecting dispersal behavior. In some species like the grass snakes (*Natrix natrix*) the rate of gastric secretion increases with temperature in the range between 25°C and 35°C (Skoczylas, 1970), which the rise in digestive rates increases the frequency of feeding in the snake. If the availability of potential prey decreases with climate change, some snake species will die due to starvation. The projected warming will also increase rates of water exchange in snakes threatening their survival as pulmonary water losses in ventilation increases (Lillywhite, 1987) and because the majority of water loss occurs through skin (Dmiél, 1972).

In a recent study, Gunderson and Leal (2012) compared the locomotor performance of the Crested Anole, *A. cristatellus*, between populations from cooler, humid forests and from dryer, hotter forests in Puerto Rico. The authors estimated thermal performance curves from sprint speed (a proxy for locomotor performance, which is a temperature-dependent performance trait frequently used to examine the influence of climate change on reptiles) and found similar performance curves between anoles from humid, cooler habitats and hotter, dryer habitats: sprint speed performance increased steadily between ~20°C and 35°C body temperature, and declined sharply beyond 35°C body temperature. The authors found an ambient temperature of 33.4°C in the dryer, hotter forest, and an ambient temperature of 28.9°C in the humid, cooler forest, and concluded that individuals living in the dry forest may be living in their physiological thermal limit with profound consequences for survival with the projected 3°C warming estimate for the 21st century. Individuals living in the cooler, wetter forest will be little impacted (if at all) with the same projected warming as the ambient temperature may raise to 31.9°C, well within the anole operative temperature for normal locomotor capacity (Gunderson and Leal, 2012). Increased temperature in cooler, wetter forests, however, may be detrimental to anole species whose thermal niche is limited to these forests. One such species is *A. gundlachi*, (and presumably other forest-interior species), which has a poor tolerance to high temperature (Rogowitz, 1996, 2003). Therefore, while warming may have little impact on *A. cristatellus* (a species adapted to a wide range of environmental conditions) in a wet and cool forest, warming will be highly detrimental to *A. gundlachi* in the same area. Presumably, the geographic distribution (and abundance) of many other species whose habitat use may be related with their thermal physiology capacity will be compromised in similar ways by the projected level of warming.

PRECIPITATION

Projections from climate models revealed a global increase in the intensity of precipitation events and precipitation extremes (compared with the mean in tropical areas), along with longer periods between rainfall events (Meehl et al., 2007; see Working Group 1 report). This may lead to profound changes in the annual variability and distribution of rainfall among months. In the Caribbean, for example, summer rainfall (between June and August) is likely to decrease in the Greater Antilles, but changes elsewhere and in winter are uncertain (Christensen et al., 2007). In Puerto Rico, a decrease between 20% and 30% in summer rainfall (between June and August) could occur by the end of the 21st century; a decrease between 5% and 10% in winter rainfall (between December and February); and a decrease between 15% and 20% in annual rainfall (Christensen et al., 2007; see Working Group 1 report). In addition, increased temperature has raised the height for orographic formation of clouds in tropical montane forests, which has reduced the frequency and daily regime of cloud immersion and humidity (Foster 2001; Pounds et al., 2006; Van der Molen et al., 2006; Wu et al., 2006). This phenomenon has been documented in Puerto Rico by Van der Molen (2002) after hurricane Hugo defoliated most cloud forests at the Luquillo Experimental Forest (LEF) in 1989. The cloud base was uplifted beyond the highest peak at the LEF (Wu et al., 2006), which could has reduced daily mist frequency and relative humidity similar to that documented from Monteverde, Costa Rica, due to warming (Pounds et al., 2006).

The projected reduction in rainfall and in forest immersion in clouds due to warming in Puerto Rico will exacerbate responses to temperature by herpetofaunal species, particularly by amphibians. For example, Stewart (1995) monitored a deme of the Common Coqui, *E. coqui*, for 14 years (from 1979 to 1993) in the LEF. Stewart noted a decreased in population size since 1983 and a marked drop in abundance in 1984, for which the species never recovered during the studied period. Interestingly, Stewart (1995) also documented having seen the Puerto Rico Mountain Coqui, *E. portoricensis*, regularly before 1984, after it was seen only twice. At a short time scale, the decrease in numbers of *E. coqui* was correlated with an increase in the frequency of the number of periods of days with ≤3mm of rain (a threshold below which frogs are critically stressed; Pough et al., 1983; Beuchat et al., 1984; Stewart 1995). At a longer time scale, species abundance was correlated negatively with the longest dry period during the previous year, which revealed a lag-time response between reduced rainfall and changes in population size (Stewart, 1995). Stewart (1995) concluded that rather than total monthly or annual rainfall, it was the distribution of rainfall that determines population dynamics and abundance. While increased humidity is critical to amphibians for normal behavior, other variables (warming) and factors (pathogens) may have likely influenced Stewart's observations, which reflects important interactions between variables driving responses to climate change by amphibians.

The projected increase in ambient temperature means that the height of the cloud base will increase; presumably well above the highest mountain peaks, with a reduction in humidity levels, and the elevation shift of lowland ecosystems into highland areas (and presumably the substitution of highland ecosystems like tropical montane cloud forests). This is particularly relevant for pathogenic infections documented in Puerto Rican amphibians from montane forests. For example, Burrowes et al. (2008a) documented a higher prevalence and infection by *Bd* in populations of *E. coqui* from a 650 m of elevation compared with a population of *E. coqui* at 850 m of elevation at the LEF. They also found a 20% higher prevalence of this pathogen on juveniles compared to that from adults across sites. The authors reasoned that runoff from higher elevation sites is greater compared with runoff at lower elevation sites, which serves as a source and means for the spread of the aquatic zoospores of *Bd* from high elevation areas to lower elevation areas (Burrowes et al., 2008a). The projected decrease in humidity levels at the highest elevation areas at the LEF (and presumably elsewhere in Puerto Rico) would imply the increased prevalence and epidemic spread of *Bd* to species with limited distribution and adapted to humidity and temperature regimes typical to high elevation could forests (e.g., *E. unicolor*, *E. gryllus*, *E. portoricensis*, among others). In samples from the mid 1970s, *Bd* has already been found in *E. unicolor*, *E. portoricensis*, and in other species with geographic distribution limited to high elevation montane forests (Burrowes et al., 2008b), most of them inhabiting the forest understory and not the canopy (e.g., *E. gryllus*, *E. hedricki*). As Puerto Rican tropical montane could forests are centers of high diversity, endemism, and ecologically specialized species of amphibians, the projected decrease in rainfall and humidity regimes, along with warming, will have detrimental consequences for most Puerto Rican amphibians.

In Puerto Rico, studies examining responses to water availability by reptiles in the climate change context have lagged considerably, as far as the author is aware, but a reduction in rainfall and humidity must certainly affect several physiological and behavioral aspects of reptiles, particularly of those with limited geographic distribution in highland areas.

SEA-LEVEL RISE

Sea levels have risen and will continue for centuries, significantly affecting islands in the Caribbean (IPCC, 2007; see Working Group 1 report). There are seven living species of sea turtles (Family Chelonioidea and Demorchelidae) found in all the world's oceans, three of which are frequently found nesting in Puerto Rico (*i.e.*, the Green Turtle, *Chelonia mydas*, Hawksbill, *Eretmochelys imbricata*, and Leatherback, *Dermochelys coriacea*). The latter one is only found during the nesting season, while hawksbills and greens are found in several life stages near Puerto Rican coasts. In sea turtles, a direct consequence of sea-level rise will be loss of coastal area, particularly sandy beaches vital for nesting. For example, Bräutigam and Eckert (2006) and Fish et al. (2005) documented loss of nesting habitat, due to erosion caused by sea level rise, storms, and sand-mining activities, as a major factor influencing declining population of sea turtles. Sea level rise will also impact foraging grounds such as seagrass beds, coral reefs, and the open ocean according to SWoT (2006). The expected increase in tidal height, which accompany sea level rise, may also increase the risk of eggs in nest being flooded (and die) as the available nesting area encroaches between the sea level to the coast and the belt of sandy beaches that remains along urban areas inland.

Sea-level rise due to thermal expansion of warmer seawaters will also lead to salt-water intrusion in coastal areas with increased degradation and loss of larger areas of coastal wetlands worldwide (Allen et al., 1996; Howard and Mendelssohn, 1999; Williams et al., 1999; Nichols, 2004, Rivera et al., 2007). Increased salinity, for example, has been accompanied by landward migration rates of mangroves 12 to 37 times the relative sea-level rise in American Samoa (Gilman et al., 2007), with expected replacement of brackish-water and fresh-water wetland forests by mangroves (Rivera et al., 2007). In coastal Puerto Rico, species richness of amphibians and reptiles increases inland with distance from sea (figure9), and population size of amphibians decreases with increased ambient salinity (Ríos-López, 2008). Brackish-water and fresh-water *Pterocarpus officinalis* forests, however, are specially important habitat for several species of amphibians and reptiles, as tree holes in these forests provide retreat and breeding sites for amphibians, hummocks formed by buttresses provides breeding sites for amphibians and reptiles, and the architecture of the dense canopy provides important microhabitat for reptiles, including the densest Puerto Rican anole, *A. stratulus*. The loss of *Pterocarpus* forests will be highly detrimental for amphibians and reptiles in coastal areas because *Pterocarpus* forest characteristics suitable for amphibian and reptile reproduction and survival are not shared by mangroves (see forest section and box for more information on climate change and Pterocarpus forests). In addition, responses to forest loss will also be exacerbated by sea-level rise and salt-water intrusion because a slight increase in water salinity (between 6 ppt and 8ppt) result in impaired osmotic physiology and metamorphosis, morphological deformities, and low larval survival in native and introduced amphibians (Ríos-López, 2008).

Responses to sea-level rise by amphibians and reptiles will also be exacerbated considering the fact that more than 70% of the world’s human population live on or near coastlines (Mitsch & Gosselink, 2007) and the urban/industrial development inland will exacerbate the level of contraction of populations of amphibians (Ríos-López, 2008) and reptiles (Ríos-López, 2007). If adaptation to these environmental changes does not occur at the rate of current changes in climate, we expect many populations and species with limited geographic distribution (e.g., the Puerto Rican Crested Toad, *Bufo lemur*; several *Anolis* spp.; see Gunderson & Leal, 2012) be destined for extinction if conservation of coastal areas and the establishment of coastal-to-inland corridors beyond urbanized areas are not implemented.

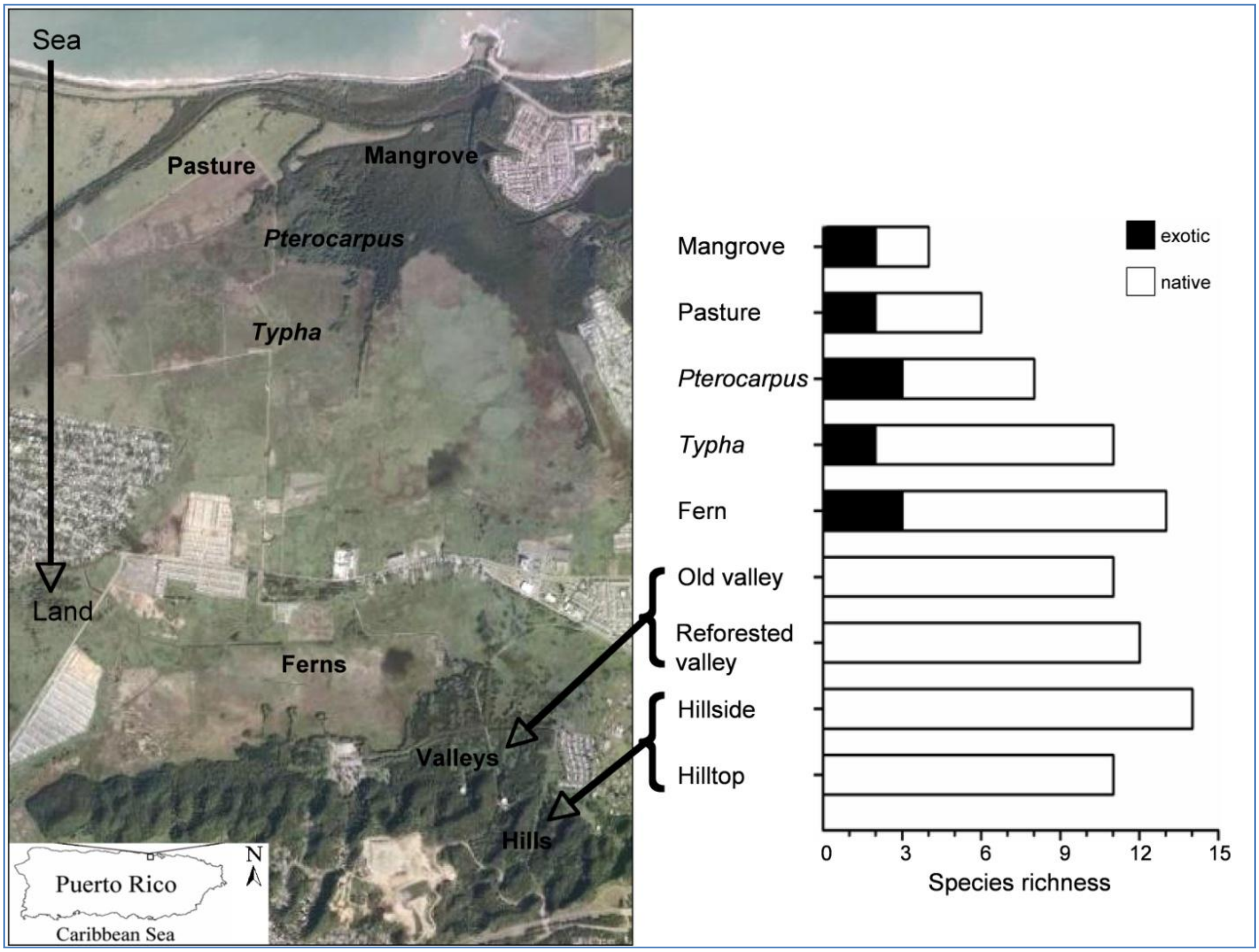


Figure 9 Herpetofaunal species richness increases with distance from the coast. Sea-level rise and salt-water intrusion will force many species to inland migration while urbanization and land-use will set an effective dispersal barrier inland. Many species will be at their physiological limit unless effective conservation measures, including natural corridors from coast-to-inland regions beyond urbanized areas, can be implemented. Otherwise, loss of species and wetland ecosystems will be lost with predicted changes in climate. Other responses include introduced species replacing native species in coastal areas, apparently facilitated by anthropogenic disturbances related with changes in land-use. Synergism between climate change and anthropogenic disturbances brings a complex suit of threats to Puerto Rican herpetofaunal diversity, which requires well planned, multidisciplinary conservation strategies and management plans (from Ríos-López, 2007).

TROPICAL STORMS AND HURRICANES

Future tropical storms are projected to be more intense, with larger peak wind speeds, and more precipitation associated with increases of tropical storms (IPCC, 2007; see Working Group 1 report). In coastal areas, frequent and more intense storms will compound the impacts of sea-level rise, increasing erosion of sandy beaches (impacting nesting behavior and nest survival of sea turtles), and increasing the magnitude of tidal storm surges and the duration of sea-water intrusion further inland (impacting amphibians and non-turtle reptiles that frequently breeds and lay eggs in the ground). To illustrate, Losos et al. (2003) addressed the question of how populations of the Cuban Anole, *Anolis sagrei* survived after Hurricane Floyd devastated many small Bahamian islands in 1999. Sea water had inundated these islands apparently for up to 6 hours, but anole populations colonized these islands shortly afterwards (~ two months): these new anole populations, however, consisted of only juveniles. Losos et al. (2003) documented viable eggs of *A. sagrei* and normal development compared with controls (no immersion) after being immersed in sea water for up to 6 hours under laboratory conditions. Similar results has been found in other reptiles (e.g., geckoes) and a combination between resistance to immersion in sea water and the force of the storm surge inland facilitated egg survival and recolonization of impacted islands by reptiles (Losos et al., 2003). However, if the force of these storm surges increases due to the expected increased in the magnitude of storms and hurricanes with climate change (along with the level of coastal exposure to the surges) we would expect less resilient populations of reptiles (and amphibians) in tropical coastal areas. Moreover, we would expect the complete wash out of adult, juveniles, and eggs of species of reptiles, particularly in smaller tropical islands and cays, as suggested by Losos et al. (2003).

In montane forests, an increase in height was documented for orographic formation of clouds after hurricane Hugo defoliated most cloud forest areas in the LEF, Puerto Rico, in 1989 (Van de Molen et al., 2010). Beyond obvious effects of increased temperature and evaporative water loss due to defoliation and increased tree and stem falls, the increased height of orographic cloud formation reduces the duration of cloud immersion and humidity across the vertical distribution of forests. Woolbright (1991) sampled population sizes of *Eleutherodactylus coqui* before and after Hurricane Hugo stroke Puerto Rico in 1989, and found a drastic population decline of juveniles. This decline was accompanied by drastic decreases in abundance of calling frogs–between 30% and >83% reduction–of species of *Eleutherodactylus* inhabiting montane forests (*E. wightmanae*, *E. portoricensis*, *E. richmondi*) (Woolbright, 1991, 1996). Presumably, reduced humidity and increase temperature after Hurricane Hugo were highly detrimental to juveniles of *E. coqui* (which will have profound consequences for population resilience with the expected increase in storm and hurricane frequency and magnitude due to climate change.

Also in tropical montane cloud forests from Puerto Rico, species of amphibians frequently associated with the canopy layer like the tree-hole breeder, *Eleutherodactylus hedricki*, and the bromeliad-dweller, *E. gryllus* will be significantly affected by stronger and frequent storms and hurricanes that damage . Forest-interior species of reptiles like the Puerto Rican Giant Anole, *Anolis cuvieri*, the Dwarft Anole, *Anolis occultus*, and ground-dweller reptiles the live bearing Puerto Rican Galliwasp, *Diploglossus pleii*, may also be impacted significantly from large changes in ambient and ground temperature, and reduced humidity after increased storm and hurricane damage due to climate change.

RESEARCH AND INFORMATION NEEDS

Effective conservation and management of important ecosystems for amphibians and reptiles requires research agendas that address the following knowledge gaps:

1. Optimal thermal and hydric physiology, and correlated habitat use for herpetofaunal species, particularly amphibians from montane forests and forest-interior and fossorial species of reptiles.

2. Genetic diversity data and selective data, particularly for herpetofaunal species with limited geographic distribution and patchy distribution.

3. Long-term monitoring of herpetofaunal species, particularly snakes, ground-dwelling lizards, amphibians from montane cloud forests, and marine turtles.

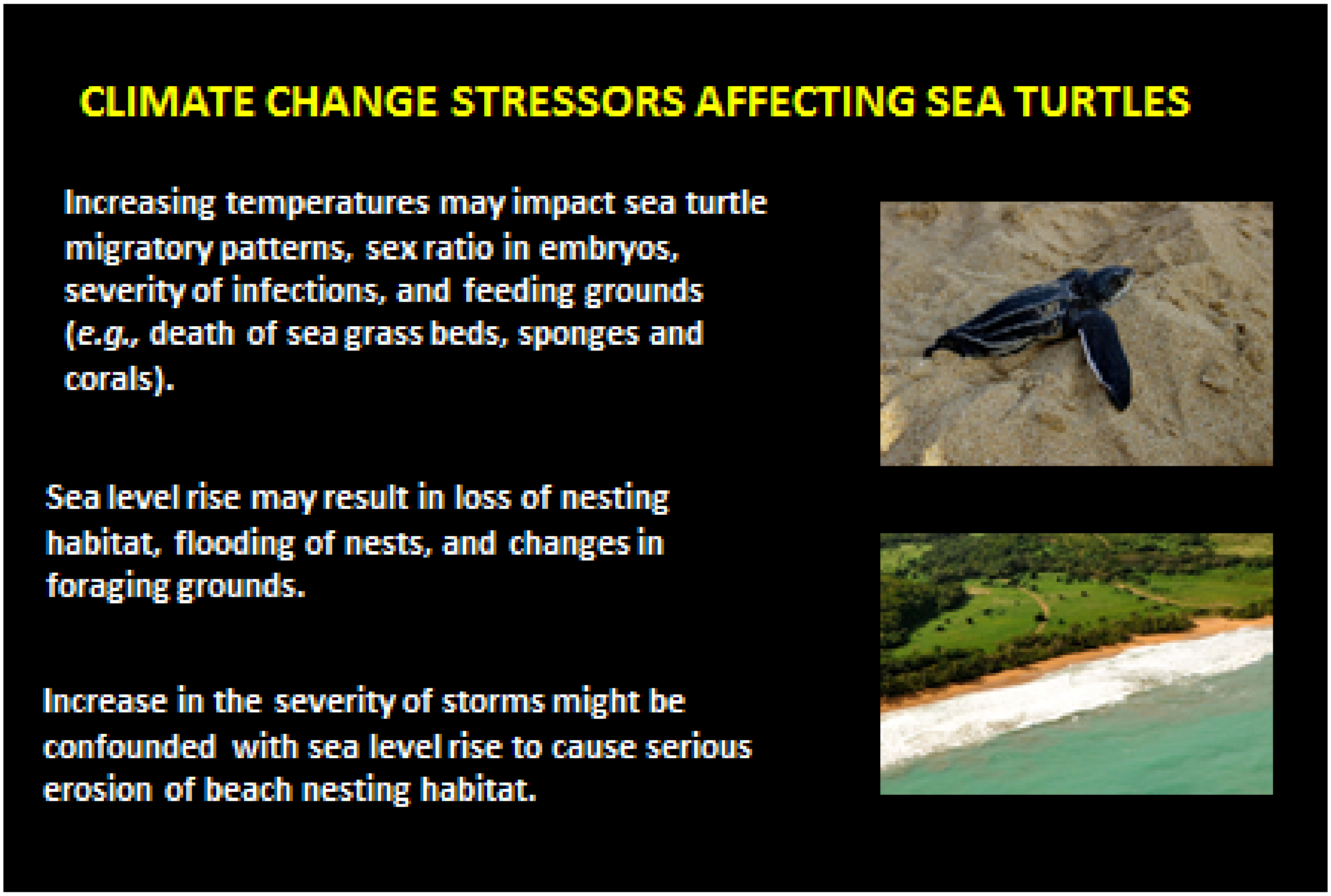
4. Predictive habitat modeling that generate detail species distribution maps based on known or inferred species physiology and habitat-use ecological requirements (e.g., bioclimate, elevation, vegetation type, among other environmental parameters) to examine responses to climate change by herpetofaunal species.

5. Studies on the population, community, and ecosystem ecology perspectives of introduced herpetofaunal species in relation to herpetofaunal assemblages in coastal and montane areas most likely to be affected by climate change.

Amphibians and Reptiles will be affected significantly by projected warming, changes in precipitation pattern, a reduction in humidity in tropical montane cloud forests, sea-level rise, tropical storms and hurricanes. Species from coastal ecosystems and from montane humid and wetter forests will be most affected due to space limitation for migration with climate change from coast-to-inland spatial shift in environmental conditions and from lowland-to-highland shift in environmental conditions up

SUMMARY OF CLIMATE CHANGE OUTCOMES/CONSEQUENCES ON PUERTO RICO’S AMPHIBIANS AND REPTILES

Responses to predicted climate changes by herpetofaunal species, with emphasis on Puerto Rican species, were examined from published literature, discussions, and unpublished data. Species of amphibians and reptiles regulate their thermal physiology and metabolism with changes in external, ambient climate variables (*i.e*, ectotherms). Consequently, they will be affected significantly by projected warming, changes in precipitation pattern, a reduction in humidity in tropical montane cloud forests, sea-level rise, tropical storms and hurricanes. Species from coastal ecosystems and from montane humid and wetter forests will be most affected due to space limitation for migration with climate change from coast-to-inland spatial shift in environmental conditions and from lowland-to-highland shift in environmental conditions up to the highest mountain peaks. Most information deals with responses to increased ambient temperature, followed by precipitation and pathogenic infections (an indirect consequence of climate change) by amphibians, with little information on reptiles.Recommendations for much needed research agenda include studies addressing the thermal and metabolic physiology for a wider range of herpetofaunal species, particularly reptiles, along with long-term monitoring of population dynamics and climate change. Conservation efforts and the development of effective management plans must be focused at coastal ecosystems, particularly fresh and brackish-water wetlands, and at tropical montane cloud forests. Planned conservation actions are urgently needed and must include natural corridors from coastal-to-inland areas beyond current urbanized areas; more difficult to develop are plans oriented at conserving species limited to tropical montane could forests as climate change may have pushed many herpetofaunal species to their physiological thermal limit. It is recognized that species losses will be inevitable and use of limited conservation resources should be prioritized, possibly at species and areas with greater probability for long-term persistence.



CLIMATE CHANGE STRESSORS AFFECTING SEA TURTLES

Increasing temperatures may impact sea turtle migratory patterns, sex ratio in embryos, severity of infections, and feeding grounds (*e.g.*, death of sea grass beds, sponges and corals).

Sea level rise may result in loss of nesting habitat, flooding of nests, and changes in foraging grounds.

Increase in the severity of storms might be confounded with sea level rise to cause serious erosion of beach nesting habitat.

INTRODUCTION TO MARINE SYSTEMS

The following sections about coral reefs, seagrasses, bioluminescent bays, coastal and pelagic fishes, and marine mammals are all affected in two major ways by climate change. First, the overall increase in global surface temperatures is evident regionally by increasing sea surface temperatures around Puerto Rico and the rest of the Caribbean. The continued uptake of heat has wide-ranging implications for marine resources as will be discussed in each of the following sections. Generally, as described by Kennedy et al. 2002, temperature changes in coastal and marine ecosystems will influence organism metabolism and alter ecological processes such as productivity and species interactions. Species are adapted to specific ranges of environmental temperature. As temperatures change, species' geographic distributions will expand or contract, creating new combinations of species that will interact in unpredictable ways. Species that are unable to migrate or compete with other species for resources may face local or global extinction (Kennedy et al. 2002). Thermal expansion of the ocean, together with melting of land ice, is resulting in rising sea levels. Climate change is also likely to alter patterns of wind and water circulation in the ocean environment. Increases in ocean temperatures are changing the strength and direction of currents, and may influence the vertical movement of ocean waters (*i.e.*, upwelling and downwelling), increasing or decreasing the availability of essential nutrients and oxygen to marine organisms. Reductions in the supply of nutrients usually limit the primary production at the base of the food chains that support marine resources (especially fisheries). Increasing temperatures can also increase incidence and virulence of disease. Changes in ocean circulation patterns can also cause substantial changes in regional ocean and land temperatures and the geographic distributions of marine species. Other climate stressors acting on marine systems are changes in the hydrological cycle (more intense downpour events and more sustained dry days) that may increase the risk of coastal flooding or drought. There is also the possibility that warmer conditions may result in more intense hurricanes (tropical cyclones) and storms, resulting in greater storm surges and greater physical disturbance of coastal environments. The second major way that climate change is expected to affect marine resources is through changes in seawater chemistry. More specifically, changes to oceanic concentrations of carbon dioxide and the resulting affect on ocean acidity. Recently, marine scientists have started to realize the substantial implications of ocean acidification (Wootton *et al*. 2008). Ocean acidification is the process whereby hydrogen ion concentrations (pH) in the ocean increase (acidification) due to absorbance of carbon dioxide from the atmosphere released during the burning of fossil fuels and other manmade activities (NOAA 2008). Changing pH levels potentially have vast consequences for marine ecosystems because of the critical role pH plays in mediating physiological reactions. Furthermore, many important groups of marine organisms have a skeleton of calcium-carbonate, which dissolves when it reacts with free hydrogen ions (Wootton *et al*. 2008). Hence, declining pH could interfere with critical processes such as reef building, carbon sequestration via phytoplankton sedimentation, and consumer-resource interactions. Marine plankton is a vital food source for many marine species and their decline could have serious consequences for the marine food web (Turley 2008).

Working Group 1's report on each of the climate stressors provides more details to the science, trends, and projections for these stressors. Figure 10 is a generalized summary of climate change effects on oceanic and coastal ecosystems.

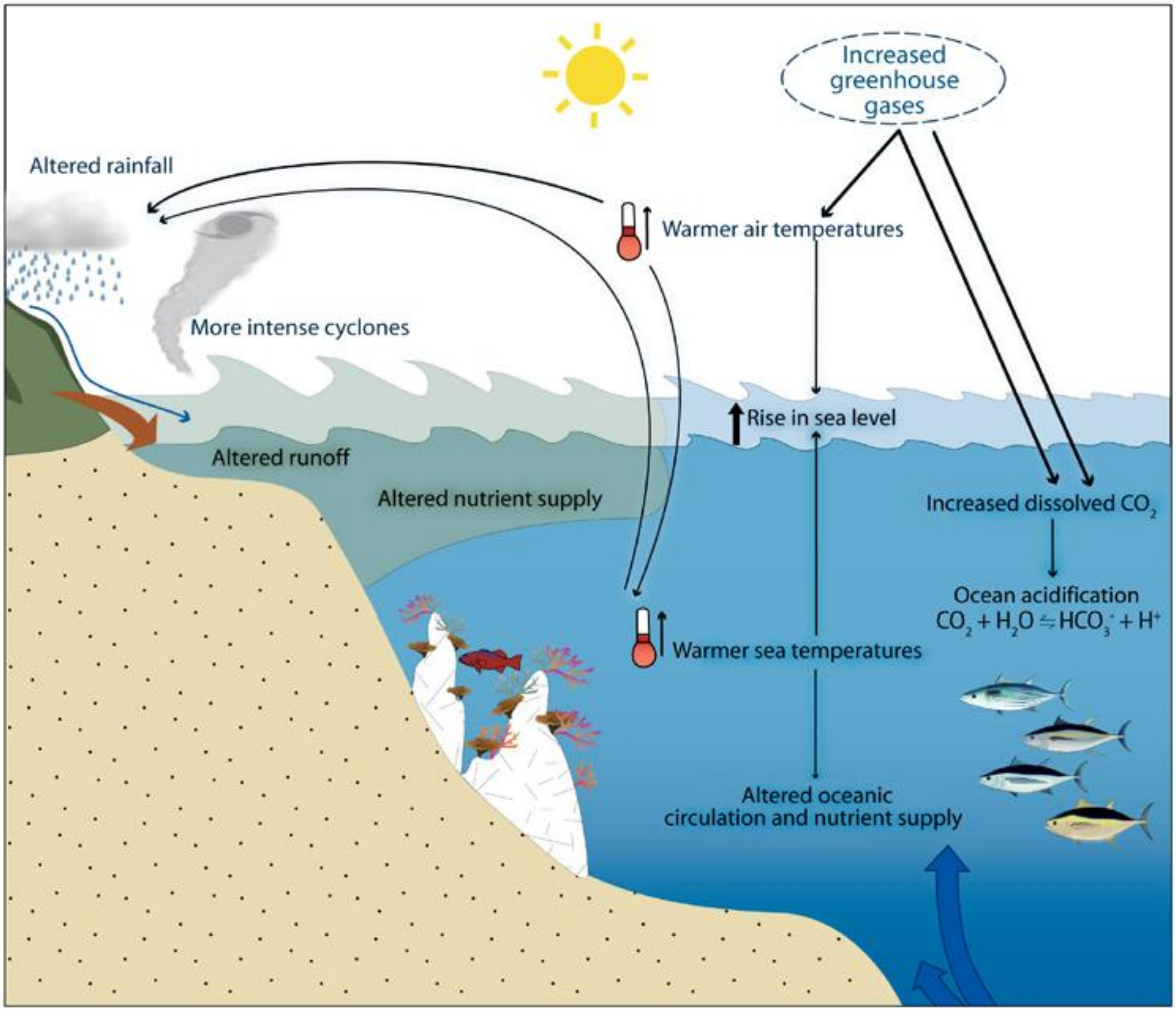
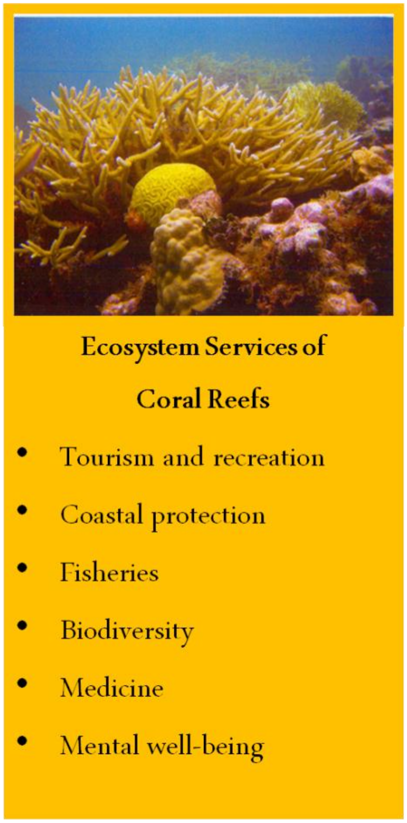


Figure 10 Generalized effects of increased greenhouse gases on oceanic and coastal ecosystems in the tropics. Adapted and printed with permissions from Bell JD, Johnson JE and Hobday AJ (eds) (2011) *Vulnerability of Tropical Pacific Fisheries and Aquaculture to Climate Change*. Secretariat of the Pacific Community, Noumea, New Caledonia.

CORAL REEFS

INTRODUCTION TO CORAL REEFS

Coral reefs, known as the rainforests of the oceans, are unique ecosystems in that they are defined by both biological ("coral" community) and geological ("reef" structure) components. Coral reefs are made of limestone (calcium carbonate) that is secreted as skeletal material by colonial animals (coral polyps) and calcareous algae. Reef building coral polyps house single-celled microalgae, called zooxanthellae, within their body tissues. This symbiotic relationship benefits both partners, in that, the coral obtains food from the plant photosynthesis and the microalgae benefit from nutrients released as waste by the coral. Furthermore, the two have complementary effects on carbon dioxide exchange that has been shown to account for the rapid rates of coral skeletal growth. Due to their high biodiversity and uniqueness, these structures are some of the most studied marine ecosystems.



Ecosystem Services of Coral Reefs

- Tourism and recreation
- Coastal protection
- Fisheries
- Biodiversity
- Medicine
- Mental well-being

Figure 11 Ecosystem Services of Coral Reefs

Coral reefs provide a number of values to humans, as well as the health of the biosphere (figure 11). Reefs support fisheries, and reef structures provide natural breakwaters that protect shorelines, other ecosystems, and human settlements from wave activity. Humans use reefs and their products extensively for food, building materials, pharmaceuticals, the aquarium trade, and other uses. In addition, due to their beauty and novelty, reefs have become a major part of the tourism product for the Caribbean region. As such, coral reefs form part of our economic foundation.

Unfortunately, these valuable ecosystems are being rapidly degraded by localized human activities such as coastal development, sedimentation, overfishing and marine pollution ((UNEP) 2004). Approximately 36% of Caribbean coral reefs lie within 2 km (1.2 mi) of the coast and this makes them highly susceptible to pressures arising from coastal population centers (Burke and Maidens 2004). The following provides a description of some main stresses affecting coral reef systems world-wide.

SEWAGE POLLUTION

It is estimated that less than 20% of sewage water generated in the Caribbean region is treated before entering the ocean (Burke and Maidens 2004). Untreated sewage is a major source of nutrients entering coastal waters which, under normal circumstances, would be devoid of nutrients. High nutrient conditions favor algal growth at the expense of the corals (Souter and Linden 2000), since coral reefs thrive in low nutrient (oligotrophic) waters. Sewage pollution has been shown to have rapid negative and often irreversible impacts on coral reef surrounding waters' environmental and microbiological quality over very large spatial scales (Bonkosky et al. 2009). This has also had paramount negative implications on coral reef benthic community composition (Hernández-Delgado et al. 2010), often on threatened Elkhorn coral (*Acropora palmata*) stands (Patterson-Sutherland et al., 2004, 2010) Hernández-Delgado et al., 2010, 2011).

LAND USE AND LAND COVER CHANGE

The conversion of land to agriculture increases soil erosion and sediment delivery to coastal waters, bringing with it pesticides and nutrients. Nearly a quarter of all the land area draining into the Caribbean is agricultural land (Burke and Maidens 2004). Increased sediment causes stress on coastal ecosystems in a variety of ways. It screens out the light needed for photosynthesis (Te 1997, Goh and Lee 2010), decreases the amount of suitable substrates for juvenile corals (Goh and Lee 2010), can have pronounced negative physiological impacts on corals (Vargas-Ãngel et al. 2007), and in extreme cases, the interaction of high sediment loads with eutrophication (Meesters et al., 1998; Weber et al. 2012) can completely smother corals (Rogers, 1990), often over very large temporal and spatial scales (Perry et al. 2008, Perry et al. 2009), which make damages permanent and irreversible over human temporal scales (Lybolt et al. 2011). Traditionally, sediments and nutrients coming from the land were filtered by mangrove forests and seagrass, however, the loss of these important areas is widespread throughout the Caribbean (Jameson et al. 1995). These impacts have largely rocketed following current coastal urban and tourism development trends (Hernández-Delgado et al. 2012), particularly with the construction of steep non-paved roads on small islands (Ramos-Scharrón 2012, Ramos-Scharrón et al. 2012), and following extreme rainfall events (Larsen and Webb 2009).

MARINE-BASED SOURCES OF POLLUTION

Marine-based sources of pollution, including oil discharge and spills, sewage, ballast and bilge discharge, and the dumping of other human garbage and waste from ships, are a cause for great concern in the Caribbean region (Burke and Maidens 2004), particularly given the escalating geographical extent of localized impacts by the cruise ship industry across the region (Hernández-Delgado et al. 2012).

FISHERIES

Coral reefs and associated habitats provide important commercial, recreational and subsistency fishery resources globally. Fishing plays a central social and cultural role in many island communities and can represent a critical source of protein. However, coral reef fisheries, though often relatively small in scale, may have disproportionately large impacts on the ecosystem if not conducted sustainably ((NOAA) 2009). And fisheries have been rarely sustainable (Pauley et al. 2002), therefore have also impacted coral reefs over very large temporal and spatial scales (Jackson, 1997, 2001). Rapid human population growth, demand for fishery resources, use of more efficient fishery technologies, and inadequate management and enforcement have led to the depletion of key reef species and habitat damage in many locations world-wide. Furthermore, fishermen typically target the largest fish on the reef since these have the highest market value. The depletion of larger fish leads to a reduction in the average size of the targeted species, and can cause fishermen to fish for lower valued species, removing even more components of the coral reef food web (McManus et al. 2000). The removal of certain species can significantly alter the reef structure (Roberts 1995, Hawkins and Roberts 2004). For example, herbivorous fish are responsible for controlling algae growth on the reef and if these fish are removed from the system, algae can flourish and reduce coral cover (Bohnsack 1993). But also, declining benthic biodiversity and environmental conditions of coral reefs (Jones et al. 2004), as well as climate change-related massive coral mortality events (Pratchett et al. 2006, Pratchett et al. 2008). The increasing temporal and spatial scale of reef ecological turnover events across the Caribbean region (Aronson et al. 2002) has caused major ecological change across coral reefs (Gardner et al. 2003) and associated nursery ground habitats which might have largely influence long-term declining trends on Caribbean-wide reef-based fisheries (Paddack et al. 2009) and on reef fish functional redundancy (Bellwood et al. 2004). In addition to scientific investigations revealing declining reef quality, socioeconomic studies have also documented fishers' perceptions through generations of observation that fish are less abundant and coral reef health has declined (e.g. Griffith et al. 2007).

STORMS AND HURRICANES

Storms and hurricanes are part of the natural variability of coral reef systems . However, hurricanes can cause extensive damage to coral reefs (Stoddart 1985, Harmelin-Vivien 1994, Salazar-Vallejo 2002), for example as described for Hurricane Gilbert in Jamaica (Bacon 1989), for Hurricanes David, Frederick and Hugo in the US Virgin Islands (Rogers et al. 1982), for Hurricane Hugo in Guadeloupe (Bouchon et al. 1991), for Hurricane Lenny in St. Lucia (Wulf 2001) for Hurricane Mitch in the Mexican Caribbean (Bahena et al. 2000), and for Hurricane Georges in Puerto Rico (1998) (Causey et al. 2002). Depending on factors such as hurricane intensity, trajectory, coastal morphology, reef depth and position in relation to its trajectory, and its interactions with different anthropogenic factors (*i.e.*, coastal pollution), hurricanes may reduce the physical complexity of coral reefs and the abundance of living corals (Goenaga, 1992)(Steneck 1994). These effects are greatest at shallow depths where wave action is greatest. However, shallow corals are adapted to wave action and hurricanes can cause considerable damage in deeper water where corals seldom experience wave action under normal conditions (Harmelin-Vivien and Laboute 1986). Hurricanes may also cause extensive impacts on tropical forests, which could in turn have significant indirect consequences on coastal coral reef ecosystems (Lugo 2000, Lugo et al. 2000). Thus, storms and hurricanes are considered threats to coral reefs because of the damages that occur when combined with additional anthropogenic stresses on the ecosystems the impacts of hurricanes linger a lot longer than they did in the past. The threat becomes great when we consider the frequency and intensity of the disturbance and the time to recover. With more and more stresses on coral reefs existing today, the adaptive capacity of reefs may be compromised.

UNSUSTAINABLE TOURISM

The tourism industry, a sector of major importance to the regional economy, also threatens the reefs in a number of ways (Hernández-Delgado et al. 2012). Dive boats can damage reef structure with their anchors, divers also cause physical damage, and resort development and operations increase pollution and sewage in coastal waters, as does the construction of tourism infrastructure and facilities (roads, marinas, airports). Also, the expanding cruise ship industry also increases environmental and socio-economic impacts that include habitat destruction, increased recreational misuses, increased pollution, increased exploitation of water resources, altered quality of life and livelihoods of local communities, and revenue leakage, among others (Hernández-Delgado et al. 2012). Under current climate change trends and under projected coastal urban and tourism development in Puerto Rico, most of this development will be carried out at areas highly vulnerable to sea level rise and to the impacts of extreme events such as storm tides, therefore under the risk of significant long-term negative economic impacts.

The cumulative impacts of natural stressors, in combination with a diverse array of localized human impacts, fishing, and climate change, have caused significant coral reef community phase shifts (Knowlton 2001, Hughes et al. 2004), as well as a significant loss of functional redundancy (Micheli and Halpern, 2005; Nyström, 2006; Hoey and Bellwood, 2009). Consequently, coral reefs are considered to be in crisis (Bellwood et al. 2004). This has been well-documented and has stimulated numerous publications on the future of coral reefs (e.g., Hoegh-Guldberg, 1999; Knowlton 2001, McClanahan et al. 2002) and their vulnerability to climate and environmental change (Bryant et al. 1998, Hughes et al. 2003). The causes of this crisis are not only related to the above mentioned stress factors, but are a complex mixture of both human-imposed and climate-related stresses, and include factors such as coral disease outbreaks, which have suspected, but unproven connections to both human activities and climate factors (Miller et al. 2006, Cróquer and Weil 2009a, b, Miller et al. 2009, Weil and Cróquer 2009, Weil et al. 2009, Hernández-Pacheco et al. 2011). Notably, by 1998, an estimated 11% of the world's reefs had been destroyed by human activity, and an additional 16% were extensively damaged in 1997–98 by coral beaching (Wilkinson 2000). Conditions of coral reefs have further deteriorated over the past decade as well (Wilkinson and Souter, 2008).

STATUS OF PUERTO RICO REEFS

The most extensive coral reef systems in Puerto Rico are found in the Southwestern and Northeastern shelf, although coral communities can be found in other areas (García-Sais et al., 2003; Hernández-Delgado, 2005). In the Northeastern coast of Puerto Rico, the insular platform is partially protected from waves by a chain of emerging fringing reefs which provide protection to the coast, thus creating calm waters and favorable conditions for the formation of sandy beaches. There are fewer coral reef formations along the rest of the Northern and Northwestern coasts because coastal waters receive substantial amounts of sediment and nutrients from the unloading of the largest and fastest flowing rivers in Puerto Rico (Goenaga and Cintrón 1979). This inhibits appropriate growth or development by reducing the photosynthesis process and increasing respiration and mucus production. Similarly, the island shelf of these coasts is narrow, exposing the coasts to strong waves. However, on Puerto Rico's North coast – in the municipality of Vega Baja – a substantial development of biogenic coral reefs has been identified which has maintained itself in excellent condition because strong wave action has allowed for its survival (Hernández-Delgado et al. 2011). For example, during the 2005 bleaching episode, which caused a massive mortality of corals through the northeastern Caribbean due to a post-bleaching mass white plague-like coral mortality event (Miller et al. 2006, Miller et al. 2009, Eakin et al. 2010), including Puerto Rico (Hernández-Pacheco et al. 2011), corals growing under strong currents exhibited higher colony survival rates, probably facilitated by higher oxygenation and planktonic food supply (Hernández-Delgado et al., 2007). The rest of the eastern, southern and western sections of the shelf support extensive reef systems representative of the northeastern Caribbean reef biodiversity (Hernández-Delgado 2005).

Puerto Rico is surrounded by approximately 500,000 ha of easy access coral reef ecosystems, where depth does not exceed 20 meters (Puerto Rico Coastal Zone Management Program 2009). In 2007, the DNER's Coral Reefs Management and Conservation Program commissioned a study to determine the economic value of coral reefs and associated environments in Eastern Puerto Rico, specifically in Fajardo, Arrecifes La Cordillera, Vieques and Culebra. According to estimates, the value of these resources is $1.853 billion, or $4.2 million/linear km of coastline. The findings showed that tourism and recreation are the activities that derive the most value ((ETI) 2007).

Despite their high value, the coral reefs of Puerto Rico have been degrading rapidly during the past decades due a combination of natural and anthropogenic factors. Their current state rated among the most critical of the Caribbean, particularly due to the unchecked urban and industrial development on the coast during the past 40 years (Causey et al. 2002). Natural factors associated with degradation have also been well documented. These include factors such as extreme rainfall events (Goenaga and Canals 1979), hurricanes (Goenaga, 1992), and mass mortality of Long-spined sea urchin (*Diadema antillarum),* which fostered major algal overgrowth (Ruiz-Ramos et al. 2011; Vicente and Goenaga, 1984).

Other natural factors which have resulted in mass coral mortality events have included outbreaks of white band disease, black band disease, white plague, yellow band disease, and massive bleaching (Bruckner and Bruckner 1997, Weil et al. 2003, Weil 2004, Hernández-Delgado 2005, Bruckner and Bruckner 2006, Bruckner and Hill 2009, Cróquer and Weil 2009a, b, Weil et al. 2009, Hernández-Pacheco et al. 2011, Weil and Rogers 2011). The extraction of corals also poses a danger to Puerto Rico's reefs. Although the removal of coral is banned in Puerto Rico, some people destroy them for ornamental or commercial purposes.
There have been several important reviews of anthropogenic impacts on coral reef ecosystems in Puerto Rico (McKenzie and Benton, 1972; Loya, 1976; Goenaga, 1988, 1991, Goenaga and Cintrón 1979, Goenaga and Boulon, 1992, Hernández-Delgado 1992, 2000, 2005, Ballantine et al. 2008, García-Sais et al. 2003, 2008, Hernández-Delgado and Sandoz-Vera 2011). These authors have consistently pointed out at declining water quality (due mostly to increased sedimentation, turbidity and eutrophication, sewage pollution) and fishing as the

primary human factors that have resulted in significant chronic reef degradation. For a more detailed review of the current status of Puerto Rico's coral reefs see García-Sais et al. (2008).

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON PUERTO RICO'S CORAL REEFS

Climate change has had significant deleterious impacts to coral reefs across the Caribbean region, including Puerto Rico (Wilkinson and Souter, 2008). Although in a very long-term climate change has the potential to yield some benefits for certain coral species in specific regions, such as the expansion of their geographic ranges to higher latitudes, most of the short-term effects of climate change are highly stressful rather than beneficial (Buddemeier et al. 2004). The following sections summarize some of the expected negative impacts of climate change on coral reef systems.

SEA LEVEL RISE

Sea level rise is and will continue to be a major cause of concern for coral reef ecosystems considering their state of degradation through the region, particularly towards the end of the 21st century. The consistent decline of the most significant reef-building species across the region suggests that overall reef accretion rates might be slower than in the past. If the vertical growth rate of corals is likely to be slower than the increase in sea level rise, then the ability of reef ecosystems to keep pace with the forecasted sea level rise trends might be deeply compromised (Veron et al. 2009). There is already evidence that current mortality trends in the Star coral, *Montastraea annularis*, species complex in Puerto Rico as a result of recurrent massive coral bleaching events and mass coral mortalities may drive this species to extinction within the next three to four decades (Hernández-Pacheco et al. 2011). Current percent cover loss of this species, as well as in *M. faveolata*, reaches values of 70% to 97% at some locations (Hernández-Delgado, in preparation). Therefore, losing the most critical reef building species across Puerto Rican reefs may result in a significant decline in overall reef accretion. Under current predicted increasing sea level rise, reduced reef accretion may result in increased vulnerability to coastal communities from erosion, storm surge, and increased SLR itself. Further, SLR will result in flooded coastal urban zones, which will include flooding of sewage and storm water sewer systems, sewage treatment facilities, and other types of sources of pollution. Coastal waters adjacent to urban centers will likely have higher turbidity, poorer microbiological water quality and potentially higher nutrient concentrations. Reefs adjacent to these areas will

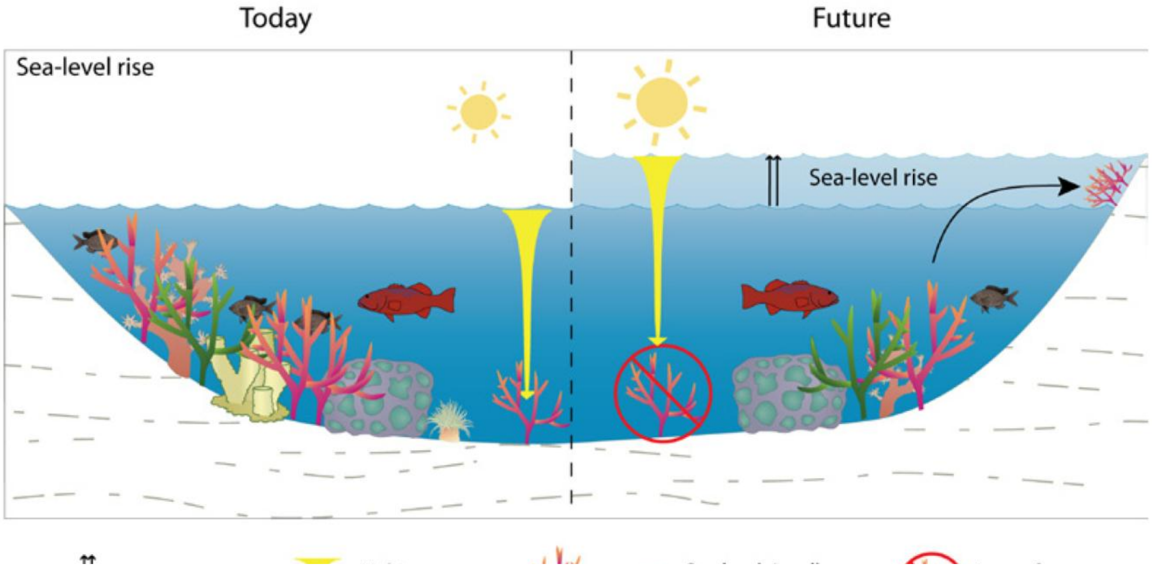


Figure 12 Effects of projected sea-level rise on coral reefs. Where corals and other reef building organisms are healthy, reefs should keep up with rising sea levels. Unfortunately, the combined effects of increased sea surface temperatures and ocean acidity are likely to mean that corals in some places will not grow fast enough to maintain their positions in the water column with the projected rapid sea-level rise. Some coral reefs may 'drown' as sea levels increase, but new areas may also be colonised where conditions for coral growth remain favourable. Printed with permissions from Bell JD, Johnson JE and Hobday AJ (eds) (2011) *Vulnerability of Tropical Pacific Fisheries and Aquaculture to Climate Change*. Secretariat of the Pacific Community, Noumea, New Caledonia.

likely have lower chances of survival and keeping pace with sea level rise than healthier remote or oceanic reefs.

While coral reefs may be fairly resilient ecosystems, the cumulative effects of the threats discussed are taking an alarming toll on coral reefs throughout the Caribbean and could thus reduce their adaptive capacity to climate change-induced sea level rise. If this were to occur we could see increased coral mortality or smaller extents of coral reefs. The ecological, physical, and societal consequences of this are great considering all the ecosystem services of coral reefs previously mentioned. A critical societal impact of this would be the reduced ability of corals to protect our shorelines as effectively and wave energy could increase in strength further eroding beaches and coastal properties. The consequences of this are locally specific. However, they will depend on seven factors (as identified by the WG 2 Coral Reef Subcommittee): (1) rate of sea level increase; (2) rate of coral reef accretion; (3) benthic community structure; (4) health of corals; (5) increase in coastal erosion; (6) sediment movement; and (7) area of island (north coast is tipping) and energy.

The vulnerability of Puerto Rico's coral reefs to sea level rise will vary. Corals with reduced growth rates due to thermal stress and ocean acidification, or corals with naturally slow growth rates in marginal environmental conditions (e.g. deeper water), may have a high vulnerability to the accelerating rise in sea level. However, in order to know which of Puerto Rico's reefs are most vulnerable, more research is necessary. There is a need to address historical changes in coral skeletal extension, density and calcification rates on different coral species through sclerochronolgical methods, comparing different depth zones, and under different environmental conditions, as a function of: (1) sea surface warming (using SST satellite data and Oxygen isotopic signals for calibration and further historical and modelling analysis), (2) address changes in pH from coral skeletons (using Boron as a proxy signal), and (3) address changes on coral growth as a function of historical changes in land use patterns. Also, there is a need to assess actual coral growth rates on different species under different conditions and as a function of net bioerosion rates to assess actual net reef accretion balance. These will provide paramount data to parameterize mathematical models of reef accretion rates as a function of SLR under different climate and environmental degradation scenarios. See figure 12 for a summary of sea level rise impacts to coral reefs.

INCREASING TEMPERATURE

One of the key characteristics of reef-building corals is their mutualistic symbiosis with populations of dinoflagellates (*Symbiodinium*; zooxanthellae). These tiny plant-like organisms occupy vacuoles within cells associated with gastrodermal tissues of corals, imparting an overall brown color to their animal host. Zooxanthellae photosynthesize while they are living within the cells of the host coral, and produce organic carbon. This carbon is then transferred to the coral, where it fuels growth, reproduction and calcification. As a result of this abundant energy, corals are able to grow and calcify rapidly, providing in return much-needed inorganic nutrients to the zooxanthellae, and calcium carbonate to the reef community as a whole. This beneficial relationship between the corals and zooxanthelle, termed symbiosis, breaks down under stressful situations. Sudden reductions in salinity, or increases in chemical toxins, sea surface temperatures or solar irradiance, will cause corals to turn brilliant white as the brown zooxanthellae are expelled from the coral tissues (Hoegh-Guldberg 1999). This phenomen is referred to as coral bleaching. Deprived of their energy source, corals become much more susceptible to competitors, such as macroalgae (seaweeds), starvation, disease and death (Harvell et al. 1999; Diaz-Pulido and McCook 2002; Diaz-Pulido et al. 2009). The most direct evidence of stress caused by climate changes on coral reefs comes in the form of massive coral bleaching often due to stressful conditions as a result of prolonged sea surface warming. A small increase of 1.0°C, particularly during periods of 8 weeks or longer, can trigger a massive bleaching event (Buddemeier et al. 2004, Eakin et al. 2010). Bleaching events lasting 8-12 weeks or longer may trigger significant coral mortality (Eakin et al. 2010). Based on current trends, climate models have shown that coral bleaching will become an annual event by the year 2020 (Hoegh-Guldberg, 1999, Hoegh-Guldberg et al. 2007, Hoegh-Guldberg and Bruno, 2010). A PRCCC WG 1 analysis by Dr. Jorge Corredor found that an increase of 1.17^{o} C (2.1^{o}F) over a 50 year period can be expected for sea surface temperatures around Puerto Rico. Additionally, SST above the threshold for coral bleaching will be exceeded over a third of the year (Hoegh-Guldberg 1999) (see PRCCC WG 1 report).

Notably, no incidents of mass coral bleaching were formally reported in the Caribbean before 1983 (Glynn, 1996). However, according to Reefbase (2012), from the early 1980s to 2012 more than 4000 observations had been reported. One of the earliest incidences was documented during the 1982-83 El Niño-Southern Oscillation (ENSO) (Glynn 1988). Further, bleaching incidents have also been recorded for 1987 and throughout 1998 (Williams et al. 1987, Williams and Bunkley-Williams 1988, Goenaga et al. 1989, Hernández-Pacheco et al. 2011), including Puerto Rico (Williams et al. 1987, Williams and Bunkley-Williams 1988, Goenaga et al. 1989, Williams and Bunkley-Williams 1990, Vicente and Goenaga, 1990, Hernández-Delgado and Alicea-Rodríguez, 1993; Hernández-Delgado, 2000; Hernández-Delgado et al., 2000). More recently, in 2005, Caribbean reefs again experienced massive bleaching. Massive decline of corals across the entire Caribbean basin, with the average hard coral cover on reefs reduced by 80%, from about 50% to 10% cover, in three decades mostly as a combined result of hurricane impacts and Caribbean-wide scale factors, probably associated to climate change (Gardner et al. 2003). But the 2005 record-breaking SST across the northeastern Caribbean and subsequent bleaching event ended up in a large-scale white plague-like condition that caused a 60 to 80% loss in % coral cover (Miller et al. 2006, Miller et al. 2009), particularly impacting significant reef-building species such as the Star corals, *Montastraea annularis*, species complex (Hernández-Pacheco et al. 2011).

Mass bleaching of corals in the past two decades has been clearly linked to ENSO events (Hoegh-Guldberg, 1999; Glynn, 2000). These have increased in frequency, severity, and duration since the 1970s (Stahle *et al.*, 1998; Mann *et al.*, 2000). This combination (sea surface warming and intense ENSO events) has resulted in a dramatic increase in coral bleaching (Glynn, 1993; Brown, 1997, Wilkinson 2000). A rising baseline in warm-season SST on coral reefs (Fitt et al., 2001; Lough, 2001) suggests that physiological bleaching is at least partly to blame in the three most significant massive bleaching events in the Caribbean in 1987, 1998, and 2005. Each massive bleaching event over the last three decades has caused major coral mortality on different localities. Although, Field data indicate that coral bleaching was much worse during the 1982-83 El Niño than in 1997-98. Temperature extremes during the two events were similar (Glynn et al., 2001, Podestá & Glynn, 2001). The difference in responses to these comparable events offers some support for the idea that corals or communities can adapt to higher temperatures over decades, either through adaptive bleaching (Baker, 2003) or through evolutionary selection for more heat/irradiance-tolerant corals that survive bleaching events (Glynn *et al.*, 2001). This is a debatable point, however, as adaptation occurring over such a short time period seems unlikely and no evidence of an acclimatization response has been found to-date.

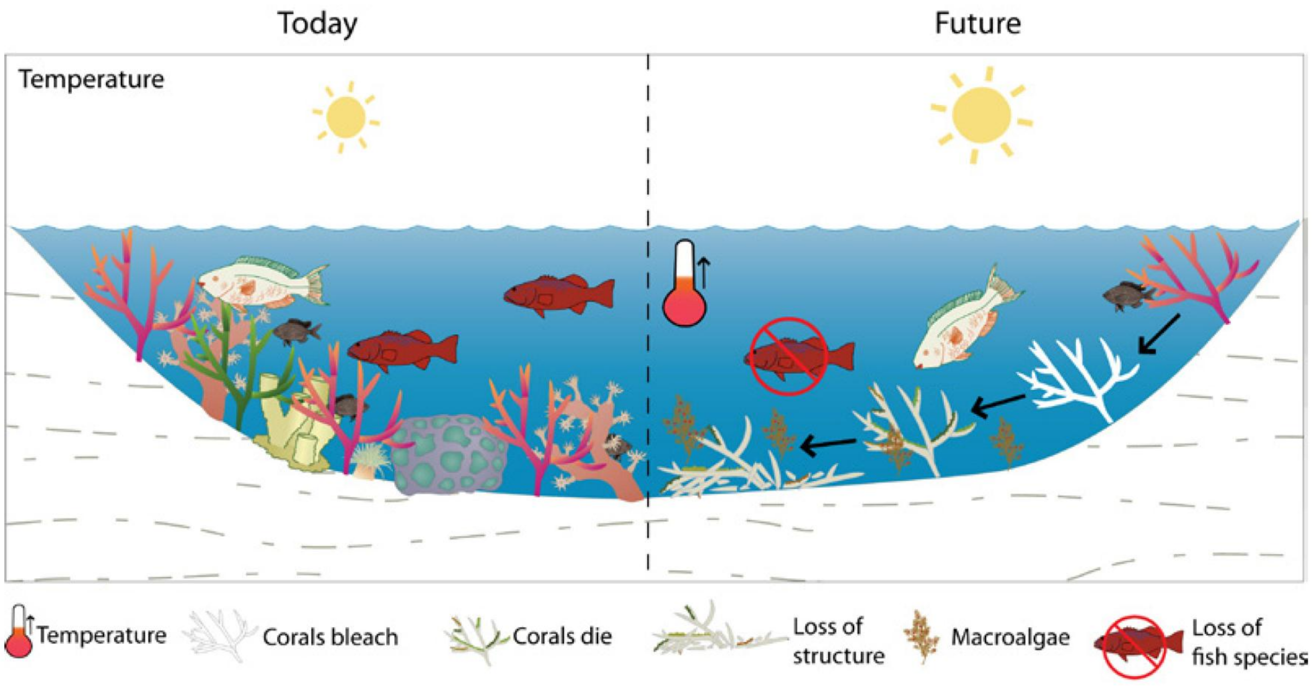


Figure 13 Effects of projected increases in sea surface temperature (SST) and light on coral reefs. Under present-day conditions, short periods of warmer-than-average maximum SST cause bleaching events. Healthy coral reefs recover from these disturbances and the abundance of corals remains high. As SST increases by 1°C or more above preindustrial levels, bleaching events are expected to become more intense and frequent, with the loss of corals, reef structure, and associated fauna, and increases in macroalgae, over time. Printed with permissions from Bell JD, Johnson JE and Hobday AJ (eds) (2011) *Vulnerability of Tropical Pacific Fisheries and Aquaculture to Climate Change*. Secretariat of the Pacific Community, Noumea, New Caledonia.

The 2005 Caribbean bleaching event, which resulted from increased sea temperatures of up to 31.8 ^{o}C (89.2^{o}F) at depths at 30 meters (98.4 feet) and about 33.1(91.6 ^{o}F) on some reef crest zones off eastern Puerto Rico (Hernández-Delgado et al., 2006). In late 2005, the coral bleaching in the south and west of Puerto Rico was devastating, being more evident on the reefs where the *Montastraea annularis* species complex was dominant in terms of substrate coverage (Hernandez-Delgado, as quoted in Garcia-Sais et al. 2008). This bleaching episode was followed by a massive outbreak of white plague, which caused deaths and a reduction of between 60 and 80% of live coral cover in Eastern Puerto Rico. This bleaching episode has alerted about the susceptibility of Puerto Rico's coral reefs to the impact of climate change as is predicted that increasing sea surface temperatures will result in more intense bleaching episodes. The impacts of the bleaching depend on the frequency, severity and spatial extent of the bleaching event. While corals can recover from these events, the frequency and severity of exposition to higher temperatures will increase coral mortality (figure 13). It should be noted that depending on the species of coral and the depth they are located; we might see an increase or change in growth rate of corals as well. Further, massive bleaching events have been shown to trigger significant declines in coral reef fish abundance and diversity (Pratchett et al. 2006, Pratchett et al. 2008). Climate-related factors, in combination with long-term fishing impacts, have been pointed out as a cause of long-term reef fish declining trends across the wider Caribbean region (Paddack et al. 2009).

Apart from the coral mortality and habitat fragmentation impacts of coral bleaching, discussed earlier, thermal expansion of the ocean from increasing sea surface temperatures and ice melt water will result in an increase in the pressure gradient which could cause changes in upwelling patterns (Bakun, 1990). In the Caribbean, upwelling areas off the Guianas-Brazil Shelf, downstream of island passages, and off Venezuela are known to influence fishery production. Changes in upwelling or other circulation patterns could significantly affect large-scale reef connectivity, therefore affecting the dispersal and transport of larvae and nutrients, and potentially having long-term impacts on the distribution of corals and associated reef species across the wider Caribbean. Connectivity could also be compromised by the increased fragmentation of reef habitat (due to the effects of coral bleaching and ocean acidification). Changes to the spatial and temporal scales of connectivity have implications for the management of coral reef ecosystems, especially the design and placement of no-take marine-protected areas (Munday et al. 2009b). The size and spacing of protected areas may need to be strategically adjusted if reserve networks are to retain their efficacy in the future.

While there is not much information available on how increased SST will affect metamorphosis, survival rate and other aspects of larvae or juvenile reef species, increased temperatures may negatively impact these (Bassim & Sammarco, 2002, 2003). Nozawa and Harrison (2007) found two different effects of elevated temperature on the early stages of recruitment process of scleractinian corals; a positive effect on larval settlement and a negative effect on post-settlement survival under prolonged exposure. (Edmunds 2004) also found increased coral recruitment rates, but increased post-recruitment mortality in St. John, USVI, with increasing SST. Studies on species such as the sand dollar (*Arachnoides placenta*) show that temperatures of or greater than 31°C (87.8°F) negatively affect larvae and juveniles (Chen and Chen 1992).

Increasing sea surface temperatures off the coast of Puerto Rico could cause corals physiological stress. This added stress could increase coral susceptibility to disease (and therefore an increase in virulence of bacterial and fungal pathogens). The most profound and widespread changes in Caribbean coral reefs in the past 30 years have been attributed to disease, however, the reasons for this sudden emergence and rapid spread are not well known (Buddemeier et al. 2004). Twenty three diseases and syndromes affecting corals have been identified in the Caribbean, and in most cases, the pathogen causing the disease is not known (Weil 2004, Weil and Rogers 2011). Disease outbreaks and consequent mortality among corals and other reef organisms have been a major cause of the recent increase in coral reef degradation (Epstein *et a*l., 1998; Harvell *et al.*, 1999; Rosenberg and Ben-Haim, 2002). Although diseases and syndromes of corals and other reef organisms remain incompletely characterized (Richardson and Aronson, 2002), they are known to be caused by both bacterial and fungal agents. These diseases are commonly lethal, but they exhibit a wide range of rates of progression. Most appear to affect some species more than others, but few, if any, are species-specific (Buddemeier et al. 2004).

Two specific outbreaks have radically altered the ecology of Caribbean coral reefs (Richardson and Aronson, 2002). One disease killed more than 97% of the black-spined sea urchin (*Diadema antillarum)* (Lessios, 1988), some populations of which are now beginning to recover, like on several reefs on the north coast of Jamaica (Aronson and Precht, 2000) , St. Croix (Miller et al. 2003), and the North Coast and island of Culebra in Puerto Rico (Ruiz-Ramos et al. 2011). Another disease, white band disease (WBD), has killed much of the elkhorn (*Acropora palmata*) and staghorn (*A. cervicornis*) coral throughout the Caribbean. These were dominant reef-building corals in the Caribbean for tens of thousands to hundreds of thousands of years (Gladfelter et al., 1980; Aronson and Precht, 1997, 2001, 2002).

Other bacterial diseases of Caribbean corals, including black-band disease, “plague,” and “white pox,” have caused significant coral mortality (Richardson, 1998; Richardson *et al.*, 1998; Patterson *et al.*, 2002). A disease caused by a fungus of terrestrial origin, *Aspergillus sydowii* (Geiser *et al.*, 1998), has killed large numbers of sea fans and sea whips (Kim & Harvell, 2001).

Prior to the 1980s, the most important reef herbivores in the Caribbean were parrotfish, surgeonfish, and the black-spined sea urchin (*Diadema antillarum),* but in many areas the fish populations had been greatly reduced (Hughes 1994). When a disease outbreak destroyed most of the *Diadema* populations throughout the Caribbean in 1983–84 (Lessios, 1988), acute episodes of coral mortality (due to hurricanes and other factors) combined with the absence of crucial herbivores to convert coral-dominated Caribbean reefs to seaweed-dominated communities (Hughes 1994, Aronson and Precht, 2000).

Warming can increase the virulence of pathogens, since optimal water temperatures for those infectious agents for which data are available are at least 1°C (2°F) higher than the optima of their coral hosts (Harvell et al. 2002). Recent increases in the frequency and virulence of disease outbreaks on coral reefs are consistent with this prediction, suggesting that the trend of increasing disease will continue and strengthen as global temperatures increase (Buddemeier et al. 2004).

The ecosystem-level impacts of mass mortality of corals from increased sea surface temperatures include an increase in algae and therefore a change in benthic community structure (Hernández-Delgado, in preparation; Hernández-Pacheco et al., in preparation). Corals can recover after short periods of stress but as the length and severity of the stress increases so does coral mortality. Changes in reef biodiversity as a result of these events depend on whether the reef has been already undergoing reef degradation. More research is needed to determine how the tropical fish that inhabit the coral reefs will respond to temperature increases.

INCREASED CARBON DIOXIDE

The impacts of anthropogenic climate change so far include decreased ocean productivity, altered food web dynamics, reduced abundance of habitat-forming species, shifting species distributions, and a greater incidence of disease (Hoegh-Guldberg and Bruno, 2010). Although there is considerable uncertainty about the spatial and temporal details, climate change is clearly and fundamentally altering ocean ecosystems. This has occurred mostly as a result of increased sea surface temperature and massive coral bleaching (Hoegh-Guldberg and Salvat 1995). Another cause of concern for coral reefs from climate change is “the other CO_2 problem” – ocean acidification.

Photosynthesis and respiration by marine organisms affect seawater carbon dioxide concentration, but the overwhelming driver of carbon dioxide concentrations in shallow seawater is the concentration of carbon dioxide in the overlying atmosphere. Changes in the carbon dioxide concentration of seawater through well-known processes of air-sea gas exchange alter the pH (an index of acidity) and the concentrations of carbonate and bicarbonate ions. Surface seawater chemistry adjusts to changes in atmospheric carbon dioxide concentrations on a time scale of about a year. Projected increases in atmospheric carbon dioxide may drive a reduction in ocean pH to levels not seen for millions of years (Caldeira & Wickett, 2003).

Many marine organisms, including corals, use calcium and carbonate ions from seawater to secrete calcium carbonate skeletons. Reducing the concentration of either ion can affect the rate of skeletal deposition, but the carbonate ion is much less abundant than calcium, and appears to play a key role in coral calcification (Langdon, 2003). The carbonate ion concentration in surface water will decrease substantially in response to future atmospheric carbon dioxide increases, reducing the calcification rates of some of the most important calcium carbonate producers, including corals.

However, calcification rates of corals also depend on other factors such as temperature (Kleypas et al. 1999). An estimated average decline of reef calcification rates is 6–14% as atmospheric carbon dioxide concentration increased from pre-industrial levels (280 ppmv) to year 2004 values (370 ppmv) (Buddemeier et al. 2004). However, studies have shown that while declining saturation state from ocean acidification may be suppressing calcification rates, increasing temperatures may be enhancing them. This is evident by calcification rates of large heads of the massive coral *Porites* increased rather than decreased over the latter half of the 20th century (Lough & Barnes, 1997, 2000; Bessat & Buigues, 2001). Temperature and calcification rates are correlated, and these corals have so far responded more to increases in water temperature (growing faster through increased metabolism and the increased photosynthetic rates of their zooxanthellae) than to decreases in carbonate ion concentration (Buddemeier et al. 2004). The temperature enhancement is thought to be a short-term effect, because the coral calcification response to temperature is not linear; rather, it reaches a maximum near the coral's optimal temperature, and then decreases at temperatures higher than that optimum. Thus, in order to boost calcification, the temperature increase must remain below the corals' upper thermal limit.

The projected changes of Puerto Rico's seawater chemistry are expected to aggressively affect coral reefs because the rate of calcification of corals and crustose coralline algae is highly sensitive to declining carbonate ion concentrations (Kleypas and Langdon 2006). Furthermore, processes such as bioerosion are likely to increase as the concentration of carbonate ions decreases (figure 14). The critical carbonate balance of coral reefs degrades when atmospheric concentrations of carbon dioxide exceed 450ppm and maintenance of carbonate coral reef systems may be too much above 350ppm (below our current concentration of 395 ppm). Experimental studies (Kleypas and Langdon 2006) and field studies (De'Ath et al. 2009; Cooper et al. 2008; Tanzil et al. 2009) indicate that substantial decreases in calcification are expect to occur in reef systems that are more fragile and degraded. In other words, health reef systems stand better against ocean acidification, although they are not buffered completely from negative effects. The reduction in calcification rates at low carbonate ion concentrations suggests that corals, and the reefs they build, are highly vulnerable to ocean acidification, and that increases in atmospheric CO_2 above 450 ppm are likely to result in erosion of some coral reefs. This outcome will have important societal consequences for the ecosystem services coral reefs provide to Puerto Rico, such as fisheries and coastal protection (see Table 4).

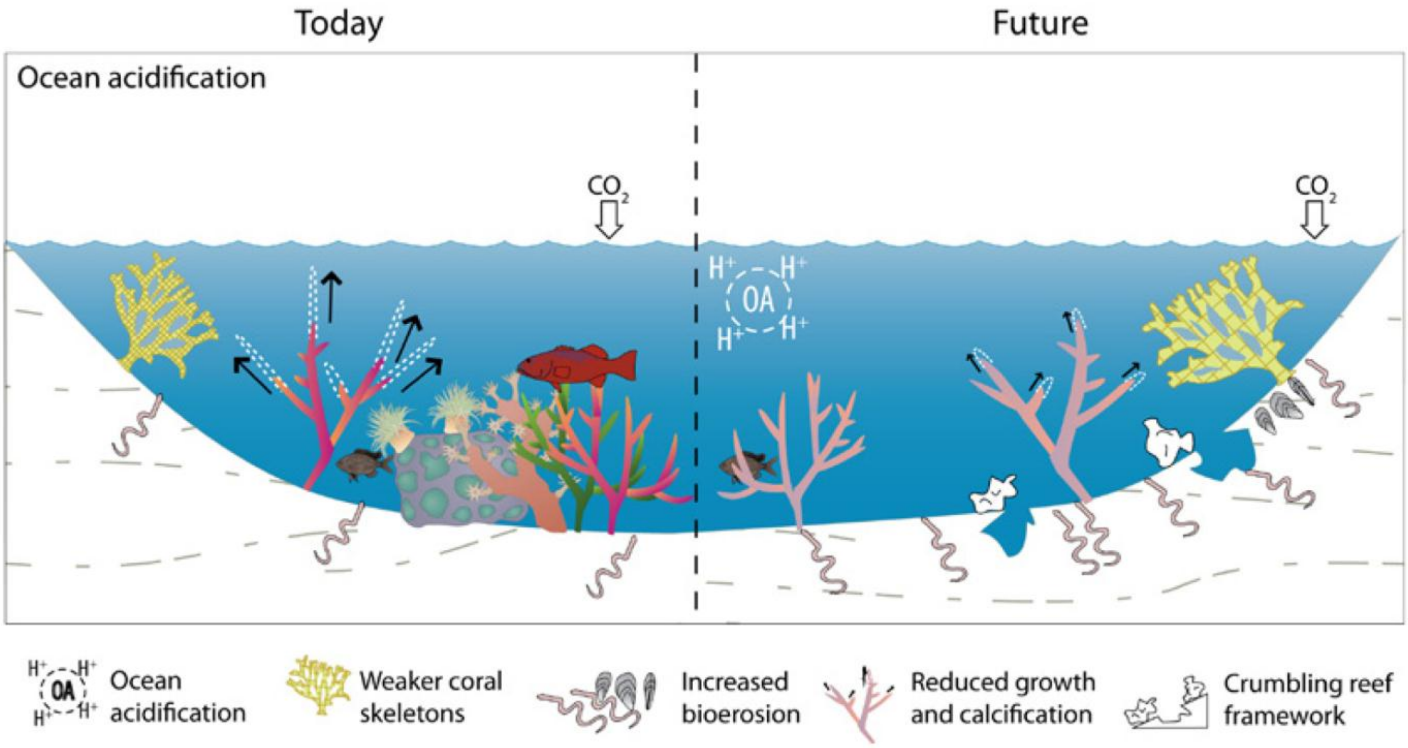


Figure 14 Effects of projected ocean acidification (OA) on coral reefs. The rising concentration of carbon dioxide (CO2) in the atmosphere from the burning of fossil fuels is acidifying the world's oceans and decreasing concentrations of carbonate ions. Reduced calcification of reef-building corals and crustose coralline algae when CO2 exceeds 450 ppm is expected to change the balance of reef processes from net construction to net erosion, leading to loss of corals and reef frameworks. Printed with permissions from Bell JD, Johnson JE and Hobday AJ (eds) (2011) *Vulnerability of Tropical Pacific Fisheries and Aquaculture to Climate Change*. Secretariat of the Pacific Community, Noumea, New Caledonia.

PRECIPITATION CHANGES

Climate change is expected to bring about changes in precipitation. According to the PRCCC's WG 1 Puerto Rico can expect to experience heavier downpour events and more consecutive dry days. Mathematical models have unequivocally shown that average rainfall is expected to significantly decline across the wider Caribbean region increasing the frequency of drought conditions (see PRCCC Working Group 1 report). Increases in heavy precipitation (high magnitude, acute events) can lower salinity and increase sediment discharge and deposition near river mouths, sometimes leading to mass mortalities on nearby coral reefs (van Woesik 1991, Coles and Jokiel 1992). Supporting this are findings from other studies which have shown that algal biomass is highest in nearshore habitats, especially in the vicinity of river mouths (Roberts 1997). Even under reduced long-term average rainfall conditions, extreme rainfall events as those seen since year 2010 can produce large amounts of rainfall and runoff in localized areas on very short temporal scales. If the frequency and/or severity of extreme precipitation increases this can lead to increased runoff events on the island and an increased sediment and nutrient input to coastal ecosystems as it has already been documented (Larsen and Webb 2009). This leads to decreased light availability in quantity and quality and resulting physiological stress can have low energy and polyp stress effects (Erftemeijer et al. 2012). Additionally, the corals may need to use more energy to expel the sediment cap that the runoff brought to the reef. This chain of events may lead to a decrease in productivity and growth of corals and a change in species composition (community structure) and/or a reduction in coral cover. Runoff can also increase the amount of pollutants that get to corals and those pollutants can decrease productivity. The heavier the downpour the greater the outflow of land based sources of pollution and the more damage to the reefs. Increased runoff could also lead to an increase in nutrient loads from land which causes algal blooms and/or eutrophication to occur (Cloern and Jassby 2010). The result is less sunlight penetration to corals, which as mentioned previously can be devastating to coral reef ecosystems. Whether sediment, nutrients, or pollutants reach the reefs from downpours depends entirely on soil conservation practices on land, urban development, riverine buffers, wetland conservation, and conservation of other natural systems. Therefore, if downpours increase as a result of climate change, coral reefs can still be protected if land based sources of pollution are controlled properly.

On the other side of changes in precipitation, if Puerto Rico experiences increased drought conditions, this could lead to a decrease in freshwater flows or a changing in the timing of freshwater runoff to coastal waters. This could increase coral growth and productivity. This potential climate change outcome needs more study as there are many areas with a freshwater lens in Puerto Rico that have healthy and/or resilient corals. On the other hand, the frequency and intensity of droughts are also expected to increase, which may cause changes in vegetation cover and land use that can lead to erosion and sediment stress when rains return (Buddemeier et al. 2004).

TROPICAL STORMS AND HURRICANES

Kjerve *et al.* (1986) point out, that our perception of the ability of coral reefs to withstand hurricane damage may depend largely on how long it has been since the previous hurricane. It is thought that Caribbean reefs require about eight years to recover from a storm (Gardner et al. 2005). However, climate change may result in increases in the intensity of storms and changes in ENSO and precipitation patterns (see PRCCC Working Group 1 Report)(Maul 1993). These will in turn affect turbidity, salinity and runoff; altering the oceans on a large scale, changing their circulation patterns, chemical composition and increasing advection (Harley et al. 2006), negatively impacting coral reefs. Increased storm activity and intensity will hinder the ability to coral reefs to recover. In fact, some studies conducted in the region already show that while some recovery of reef in deeper waters occurs after storms, there is no noticeable recovery in nearshore areas (Roberts 1997).

If Puerto Rico experiences an increased intensity of storm events and hurricanes (waves and wind the primary concerns of these events) we could see increased coral mortality and coral reef growth may not be able to keep pace with these destructive events (other climate stressors influence this as well). Coral mortality from storms is caused mostly by physical damages (Goenaga, 1992). Mortality, net loss, and recuperation are possible outcomes of the physical damages and all three lead to changes in base structure (figure 15). With recuperation, corals reestablish and grow in new areas by movement and fragementation (asexual reproduction).

If the frequency of storm events does not change, then our coral reefs may have a chance to bounce back. The big question is how much time do reefs need between storm events to return to previous conditions? There is a possibility that there has been coral acclimation of our Puerto Rico reef communities to storm events, but more research is needed in this area. With a background of chronic stress from non-climate impacts like unsustainable land use practices and overexploitation of key functional groups like herbivores, combined with climate stressors like acidification and higher sea surface temperatures, more intense tropical cyclones could also act as key agents for change in coral reef habitats. The described compounding of threats has the potential to lead to sudden change in ecological state (from coral dominated to macroalgae dominated, for instance (Hughes 1994)), where any single factor could well cause the collapse of Puerto Rico’s coral reef systems.

INCREASED DUST FROM ARID REGIONS

The Sahara is the major source on Earth of mineral dust (60-200 millions of tons per year), which are eroded mineral soils from the Sahara Desert and the transition zone (Sahel) between the Sahara Desert and tropical forests to the south. This mineral or Saharan dust is lifted by convection to high altitudes where it is transported by the trade winds across the Atlntic Ocean to the Americas and the Caribbean basin. Besides depositing nutrients

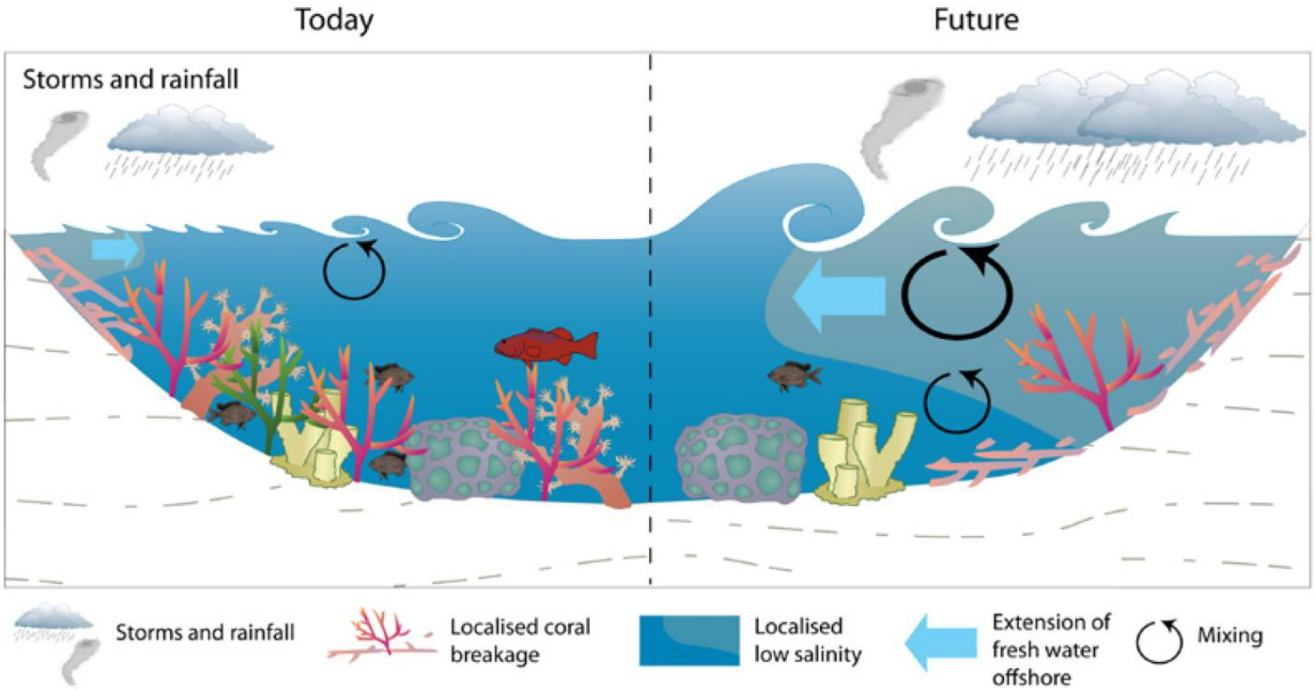


Figure 15 Effects of stronger cyclones and heavier rainfall on coral reefs, which are expected to lead to greater loss and degradation of habitat through damage by more powerful waves, reduced salinity and increased turbidity of coastal waters. The effects of stronger cyclones and increased rainfall are also expected to interact with higher sea surface temperatures, increased acidification and local factors, such as poor catchment management, to cause severe problems for coral reefs. Printed with permissions from Bell JD, Johnson JE and Hobday AJ (eds) (2011) *Vulnerability of Tropical Pacific Fisheries and Aquaculture to Climate Change*. Secretariat of the Pacific Community, Noumea, New Caledonia.

utilized by Atlantic phytoplankton and other organisms, the dust also contains harmful contaminants. Found in the dust are viable pathogenic microorganisms, metals, heavy metals, pesticides, polychlorinated biphenyls (PCBs), and polycyclic aromatic hydrocarbons. A known coral pathogen, the fungus *Aspergillus sydowii,* which produces the disease aspergillosis in sea fans, is also transported by Saharan dust.

Caribbean coral events, such as coral reef diseases, appear to coincide with African dust production as well as other factors (Shinn et al. 2000, Garrison et al. 2003, Weir-Brush et al. 2004). Based on data recorded in Barbados, the years of highest cumulative dust flux occurred in 1983-1985 and 1987 (UNEP/GPA 2006). This has also been linked to increasing aridity and desertification in Northern Africa (Prospero and Lamb 2003). Various peaks in the dust record at Barbados and elsewhere in the western Atlantic coincide with benchmark perturbations events on reefs through the Caribbean (UNEP/GPA 2006). Since climate change is a global phenomena, the Saharan Desert will also be affected by climate changes. However, how the dust emissions in Africa might change is not currently known. Some global climate models predict a wetter North Africa and some a drier. The mechanism by which dust may affect corals include direct fertilization of benthic algae by iron or other nutrients interacting with ammonia and nitrite, as well as nitrate-rich submarine ground water and as a source of pathogens by broadcasting bacterial, viral and fungal spores (Shinn et al. 2000, Garrison et al. 2003).

RESEARCH NEEDS AND INFORMATION GAPS

There is a need to expand coral reef long-term monitoring efforts at selected representative localities to address climate change-associated impacts, but also to address impacts associated to other localized human stressors, such as high water turbidity, sedimentation, eutrophication and fishing. Also, special attention should be placed on monitoring high-energy reefs to test hypotheses about thermal resistance of corals under high energy conditions. Also, there is a need to address long-term ecological changes on deeper water corals which appear to be more resilient than those from shallower zones. There is also a need to provide special attention to the long-term response of reef fish communities to habitat degradation. There is also a critical need to foster low-tech coral aquaculture and reef rehabilitation. According to Hernández-Delgado and Suleimán-Ramos (2012), low-tech coral aquaculture and reef rehabilitation approaches in PR have proved to be successful, reliable and highly cost-effective tools to conserve and restore Staghorn coral, *Acropora cervicornis*, populations with minimum intervention and maintenance.

SUMMARY OF CORAL REEFS AND CLIMATE CHANGE

Coral reefs, known as the rainforests of the oceans, are unique ecosystems that provide a number of values to humans, as well as the health of the biosphere. Reefs support fisheries, and reef structures provide natural breakwaters that protect shorelines, other ecosystems, and human settlements from wave activity. Humans use reefs and their products extensively for food, building materials, pharmaceuticals, the aquarium trade, and other uses. In addition, due to their beauty and novelty, reefs have become a major part of the local and international tourism product for Puerto Rico. As such, coral reefs form part of our economic foundation. Unfortunately, these valuable ecosystems are being rapidly degraded by localized human activities such as coastal development, sedimentation, overfishing and marine pollution. The most extensive coral reef systems in Puerto Rico are found in the Southwestern and Northeastern shelf, although coral communities can be found in other areas. Climate change has had significant harmful impacts to coral reefs across the Caribbean region, including Puerto Rico. Sea level rise is and will continue to be a major cause of concern for coral reef ecosystems. The consistent decline of the most significant reef-building species across the region suggests that overall reef accretion rates might be slower than in the past. If the vertical growth rate of corals is likely to be slower than the increase in sea level rise, then the ability of reef ecosystems to keep pace with the forecasted sea level rise trends might be deeply compromised.

Perhaps more of an immediate concern is that the beneficial relationship between the corals and its symbiotic zooxanthellae breaks down under stressful situations. Sudden reductions in salinity, or increases in chemical toxins, sea surface temperatures or solar irradiance, will cause corals to turn brilliant white as the brown zooxanthellae are expelled from the coral tissues. This phenomen is referred to as coral bleaching. Deprived of their energy source, corals become much more susceptible to competitors, such as macroalgae (seaweeds), starvation, disease and death. The most direct evidence of stress caused by climate changes on coral reefs comes in the form of massive coral bleaching often due to stressful conditions as a result of prolonged sea surface warming. Based on current trends, climate models have shown that coral bleaching will become an annual event by the year 2020 and that the temperature of the ocean will exceed the threshold for coral bleaching over a third of the year. Another concern with increasing SSTs s is that Puerto Rico's corals could suffer from physiological stress that increases coral susceptibility to disease (and therefore an increase in virulence of bacterial and fungal pathogens). The most profound and widespread changes in Caribbean coral reefs in the past 30 years have been attributed to disease, however, the reasons for this sudden emergence and rapid spread are not well known.

Another cause of concern for coral reefs from climate change is "the other CO_2 problem" – ocean acidification. The projected changes of Puerto Rico's seawater chemistry are expected to aggressively affect coral reefs because the rate of calcification of corals and crustose coralline algae is highly sensitive to declining carbonate ion concentrations. The reduction in calcification rates at low carbonate ion concentrations suggests that corals, and the reefs they build, are highly vulnerable to ocean acidification, and that increases in atmospheric CO_2 above 450 ppm are likely to result in erosion of some coral reefs.

Increases in heavy precipitation (high magnitude, acute events) can lower salinity and increase sediment discharge and deposition near river mouths, sometimes leading to mass mortalities on nearby coral reefs. The heavier the downpour the greater the outflow of land based sources of pollution and the more damage to the reefs. On the other side of changes in precipitation, if Puerto Rico experiences increased drought conditions, this could lead to a decrease in freshwater flows or a changing in the timing of freshwater runoff to coastal waters. This could increase coral growth and productivity. This potential climate change outcome needs more study as there are many areas with a freshwater lens in Puerto Rico that have healthy and/or resilient corals. On the other hand, increasing frequency and intensity of droughts may cause changes in vegetation cover and land use that can lead to erosion and sediment stress when rains return.

There is a need to expand coral reef long-term monitoring efforts at selected representative localities to address climate change-associated impacts, but also to address impacts associated to other localized human stressors, such as high water turbidity, sedimentation, eutrophication and fishing. Also, special attention should be placed on monitoring high-energy reefs to test hypotheses about thermal resistance of corals under high energy conditions.

All existing information leads to the conclusion that coral reefs have very high vulnerability to further increases in sea surface temperatures, sea level, heavy downpour events, and ocean acidity (Table 4). While coral reefs may be fairly resilient ecosystems, the cumulative effects of the threats discussed are taking an alarming toll on coral reefs throughout the Caribbean and could thus reduce their adaptive capacity to climate change.

Coral Reefs and Climate Change	**BIOPHYSICAL OUTCOMES**									**SOCIETAL CONSEQUENCES**
	Reduced reef accretion	Increased coral mortality	Altered food web dynamics	Shifting species distributions	Greater incidence of disease	More frequent and widespread annual coral bleaching	Physical damages	Increased algal growth and changes in community structure	Decreased light availability	Increased vulnerability to coastal communities from erosion, storm surges, and SLR; Decreased tourism and recreation; Loss of fisheries; Loss of potential medical discoveries
Sea Level Rise	√	√		√		√			√	√
Increased Sea Surface Temperatures	√	√	√	√	√	√				√
Increased Tropical Storm and Hurricane Intensity	√	√					√		√	√
Increased Precipitation and Increased Intensity of Downpour Events		√	√		√			√	√	√
Increased Carbon Dioxide	√	√	√	√			√			√
Increased Saharan Dust to the Caribbean from Africa					√			√	√	v

Table 4 Summary outcomes/consequences of climate change stressors on Puerto Rico's coral reefs

SUBMERGED AQUATIC VEGETATION - SEAGRASSES

INTRODUCTION TO SEAGRASSES

Seagrasses are aquatic flowering plants that grow in the soft or sandy bottoms of estuaries and along the coastal margins of tropical, temperate and sub-arctic marine waters (Duarte 2000, Green and Short 2003). They are found throughout the insular Caribbean growing in reef lagoons between beaches and coral reefs, or forming extensive meadows in more protected bays or estuaries. *Thalassia testudinum* is the most abundant species (Duffy 2006) growing in monospecific beds, or intermixed with *Syringodium filiforme, Halodule wrightii*, or *Halophila spp*. and macroalgae.

Seagrasses form extremely complex ecosystems that are highly productive, faunally rich and ecologically important (Duarte 2000). As such, seagrass beds have been recognized as productive fishery areas in the Caribbean (Muehlstein 1989, 1992). They are important breeding grounds and nurseries for finfish and shellfish population (Nagelkerken *et al.*, 2002). The plants filter suspended sediments, and nutrients from coastal waters, stabilize sediments, dissipate wave energy, and remove carbon dioxide from the ocean-atmosphere system which could play some role in the amelioration of climate change impacts (Barbier et al. 2011).

Seven seagrass species have been reported for Puerto Rico, including *Thalassia testudinum*, *Halophila decipiens*, *H. baillonis*, *H. engelmannii*, *Syringodium filiforme*, *Halodule wrightii* (now *beaudettei*), and *Ruppia maritima* (Vicente 1992, Hemminga and Duarte 2000) with the dominant species being *Thalassia* and *Syringodium* (L. Roberson, pers. obs.).

Globally, seagrasses play an important role in maintaining productivity**,** as they are responsible for approximately 1% of total primary production and 15% of total carbon storage (Hemminga and Duarte 2000). For the seagrass species found in Puerto Rico, Duarte and Chiscano (1999) reported that the average above-ground to below-ground biomass (AG/BG) was approximately 250/500 g dry wt m^{-2} for both *Thalassia* and *Syringodium* and 200/100 g dry wt m^{-2} for *Halodule*. The AG/BG average productivity (g dry wt m^{-2} d^{-1}) for *Thalassia*, *Syringodium* and *Halodule* were approximately 3.5/2, 3/2, and 7/1, respectively. These biomass and productivity measurements demonstrate that belowground biomass for turtle and manatee grass in Puerto Rico is higher than the global average, while above-ground biomass productivity is similar to the global average for these two species. It also may indicate that local seagrass populations are stressed and therefore invest more production in BG biomass as storage. For shoal grass, the measurements demonstrate its capacity for high AG productivity although its BG productivity was lower than the global average, probably due to the limited development of its rhizome system. In La Parguera, Puerto Rico, González-Liboy (1979) found that the AG productivity of turtle grass ranged from 2-7 g dry wt/m^2 per day with an average of 5, again indicating the high productivity of the local system. In addition to the productivity of the grasses, the productivity of other components of seagrass meadows may be more than 20-60% of total production (Hemminga and Duarte 2000). For instance, epiphytes in *Halodule* stands may be more productive than the grass, possibly accounting for the high AG productivity measurements in the review by Duarte and Chiscano (1999).

The importance of seagrasses in the marine food webs is greater than the extension of their beds. Material from seagrass beds is exported to areas very distant from the beds and often to deep waters as dissolved and particulate material. Leaves of manatee grass, and to a lesser extent turtle grass, have been observed in depths of thousands of meters. Seagrass leaves have been photographed on sediments in the Puerto Rico trench at depths of 7,860 meters (25,787.4 feet) (Webber and Thurman 1991). At these depths, this material is an important part of the diet of organisms such as echinoderms (*i.e.*, starfish, urchins). Furthermore, accumulation of seagrass wracks on beaches provide significant amounts of nutrients to local biota generating additional production while influencing shore geomorphology by trapping and binding loose sand particles (Hemminga and Duarte 2000).

Within the beds themselves, organisms such as the green sea turtle and the manatee, both federally protected species in the U.S. Caribbean, feed on seagrass leaves and algae. Tidal embayments containing seagrass beds are important habitat for juvenile green sea turtles in the Caribbean (Musick and Limpus 1997) that feed on the grasses, as well as on jellyfish and sponges. Similarly, juvenile hawksbill sea turtles frequent estuarine habitats although their feeding preference is for sponges on red mangrove roots rather than seagrass (Musick and Limpus 1997). In the Caribbean, research indicates that turtle grass dominates the diet of adult green sea turtles, probably because it is the most abundant seagrass. However, manatee grass, shoal grass, and paddle grass were also found in the stomachs of foraging turtles, as were red and green algal species and one species of sponge (Bjorndal 1997). Adult hawksbills have also been found to consume marine plants, including algae and seagrasses, although sponges dominate the diet of these sea turtles in the Caribbean (Bjorndal, 1997).

The role of seagrass habitats as dominant producers in tropical and subtropical systems enables the development of a rich community of organisms. Garcia-Rios (2001) illustrates the numerous groups of organisms associated with seagrass, including coelenterates, bryozoans, ciliates, flagellates, sarcodines, foraminifera, crustaceans, fishes, echinoderms, mollusks, and algae. *Thalassia* beds in Jamaica were found to be populated by a variety of macrofaunal species, including polychaetes (20), crustaceans (39), mollusks (61), fishes (41), and others (40) (Greenway 1995). Studies of the population biology of decapods in seagrass beds in Dorado, Puerto Rico (Bauer 1985a, c, b) suggest diversities similar to those found in the Jamaica study, as 34 crustaceans species, including shrimp and hermit crabs, reaching densities of 72 individuals per m^2, were found. In addition, the interaction between seagrass beds and other marine systems such as mangroves and reefs, facilitates trophic transfers and cross-habitat utilization by fishes and invertebrates (Orth 2007) and has been found to be essential for the health of all the systems involved (Nagelkerken et al. 2002).

Seagrasses provide habitat for numerous highly mobile species. Berrios et al. (1985) performed visual observations of fish populations in seagrass beds dominated by turtle grass and coral reefs in Cayos Berbería and Ratones off the coast of Ponce, Puerto Rico. They found approximately 29 species of juvenile fishes in large numbers, including grunts, parrotfish, and yellowtail snappers. They also found young adults of bluestriped grunt (*Haemulon sciurus*), gray snapper (*Lutjanus griseus*), and schoolmaster (*L. apodus*) to be common in seagrass beds around Cayo Berbería. Adult grunts and snappers were also very common in both reef environments, which Berrios et al. (1985) attributed to the presence of seagrass beds in both reefs. Similarly, Aguilar-Perera (2004) evidenced the importance of seagrasses in La Parguera Natural Reserve as habitat for juvenile populations of species of commercial importance such as grunts and snappers. Jenkins and Hamer (2001) found a relationship between the abundance of post settlement fish and meiofaunal crustaceans in seagrass beds. Due to the greater abundance of prey items in seagrass beds in comparison to bare sandy sediments, post settlement fish were more abundant in seagrass beds.

The complex trophic interactions within seagrasss communities are paramount in sustaining juvenile and adult populations of special interest, including commercially important fishery and protected species. However, because the contribution of any particular seagrass species (evenness) is low within a seagrass bed and associated fauna and algae contribute most of the diversity of seagrass habitats, observers are often wrongly inclined to think seagrass habitats do not sustain significant biodiversity (Hemminga and Duarte 2000). Because the fauna of seagrass beds is cryptic and the grasses themselves are not as evident as coral reefs, Orth et al. (2007) estimate that the publicity given to seagrasses is 10 to 100 times less than that given to other coastal habitats. Despite the fact that the economic value of seagrass meadows has been estimated. The value of marine macrophyte ecosystems (contain macroalgae and seagrasses) has been estimated at $19,000 per hectare per year (Costanza et al. 1997). This is 23 times higher than the average oceanic and terrestrial values, 3 times more than coral reefs and 10 times more than tropical forests. Therefore seagrass meadows are one of the highest valued ecosystems on Earth. The lack of awareness of the ecological and economic importance of seagrass beds on the part of the general public, resource agencies, and lawmakers may explain the lack of protection afforded to this essential habitat.

STATUS

Population growth, increased urbanization and the rapidly expanding agricultural and tourist-industrial sectors in the insular Caribbean have increased pressure on the coastline and their seagrasses (Short and Wyllie-Echeverria, 1996). Increased traffic of ships and recreational vessels are causes of anchor damage, trampling, propeller scarring, detrimental shading by marinas and piers, and damage by dredging (Thorhaug *et al.*, 1985; Creed et al., 2003). In the smaller islands, seagrass beds are damaged by illegal sand mining which suspends sediments and alters local hydrodynamics. In St Lucia, seagrass beds have been destroyed by dynamite fishing (Creed *et al.*, 2003) and seamoss (*Gracilaria spp.*) is cultivated in seagrass areas.

Seagrasses are subjected to nutrient pollution mainly from land-based sources of pollution, particularly sewage and grey water (Short and Burdick, 1996,). Increased sediment loading as a result of deforestation, urbanization and agricultural activities has caused major damage to beds. A comparison of seagrass bed community structure between areas of high and low water quality showed a significant difference in biodiversity and below-ground biomass, with higher biodiversity and higher total biomass in areas with high water quality (Roberson, unpub. data). Extensive seagrass beds in the Archipelago Sabana-Camagüey, Cuba, have been impacted by increased salinity in the inner water bodies, due to anthropogenic changes in the hydrological regime (James and Hart, 1993). Indirect activity such as overfishing of wrasses and triggerfishes off Haiti and the U.S. Virgin Islands results in an explosion of sea urchins which then destroys seagrass bed by overgrazing (Creed *et al.*, 2003). Seagrass beds can also be damaged by hurricanes and from overgrowth of bluegreen alga.

A study of the seagrass system in Guayanilla Bay showed that approximately 17% of the area surveyed was covered with seagrass (Vicente1992). The two major regions within the study area that were devoid of seagrass were the deepest area in the central bay and the western bay, which was very turbid (Vicente et al. 1980). The turbidity in the western bay was due to dredging and filling in the coastal and shallow water areas as part of industrial development in the bay, as well as turbulence from sediment resuspension caused by the power plant’s cooling system discharge (Vicente and Rivera 1982). Thus, industrial development that includes ports and nearshore discharges along portions of the coast of Puerto Rico have led to a loss of seagrass beds.

Similarly, a study in Culebra found that commercial, housing, and recreational development and associated construction of docks and other marine facilities led to decreased water transparency from eroding soils and wastewater discharges and increased water turbidity from boating and a resulting decline in seagrass beds (Vicente 1992). In Puerto Rico, Gonzalez-Liboy (1979) found propeller scarring in areas associated with channels between mangrove cays in La Parguera. Gonzalez-Liboy (1979) reported intense impacts to seagrasses in the channel near Magueyes Island that caused significant patchiness of seagrass beds in this area. More recently, Carrubba et al. (2003) documented major propeller scar impacts in various locations in La Parguera Reserve, including shallows near Magueyes Island, Cayo Caracoles and Cayo Collado backreefs where 43-74% of the area potentially affected by boat traffic showed damage due to propeller scarring. Other studies characterizing the impacts of mechanical damage to seagrass beds in Cordillera Reefs and Culebra showed that boating activities, in particular transit through shallow areas on the west coast of Puerto Rico and anchoring practices on the east coast of Puerto Rico and around Culebra resulted in considerable damage to seagrass beds where boaters were concentrated (Carrubba et al. 2003, Otero and Carrubba 2007, 2008). In Cordillera Reefs, based on impacts to seagrass mainly from anchoring, it was estimated that 7%, 14%, and 21% of the seagrass habitats in Palominito, Palomino, and Icacos were impacted (Otero and Carrubba 2007). Mechanical damage to seagrass beds in Culebra resulted in impacts to at least 54%, 25.2%, 12%, 1.7%, 4.6%, 54.3%, and 0.5% of the shallow seagrass beds in Ensenada Honda (near Dewey), Cayo Verde, near Punta Cabras, Costa Bonita near Mosquito Bay, Malena Bay, Dakity backreef and Bay, Fulladosa Bay, and the south shore of Culebrita (Otero and Carrubba 2008). A related study of the impacts of dock construction on the seagrass beds in Culebra found that dock construction and associated long-term mooring of vessels at docks had resulted in the loss of 3.6% of the continuous shallow water seagrass beds in Fulladosa Bay (Otero and Carrubba 2008).

OUTCOME AND CONSEQUENCES OF CLIMATE CHANGE TO PUERTO RICO'S SEAGRASSES

Climate change represents a relatively new threat, the impacts of which on seagrasses are largely undetermined (Short & Neckles, 1998). Potential threats from climate change may come from rising sea level, changing tidal regime, localized decreases in salinity, damage from ultraviolet radiation, and unpredictable impacts from changes in the distribution and intensity of extreme events (Short & Neckles, 1998). Higher carbon dioxide concentrations may, however, increase productivity thus not all climate change outcomes are negative.

SEA LEVEL RISE (AND ALTERED CURRENTS/TIDAL REGIME)

Rising sea levels could result in changes in light attenuation, wave energy, substrate type, and grazers influencing seagrass beds (Short and Neckles, 1998). Impacts could be positive or negative. A positive impact of sea level rise on seagrass beds would be less disturbance. However, increased sea level could also be negative in terms of less sunlight penetration to the beds. Therefore we would see reduced productivity and growth and eventually a loss of beds and their associated ecosystem services. The growth rates of seagrass rhizones vary greatly, from a few centimetres per year in *Posidonia oceanic* to more than 5 meters per year in *Halophila ovalis*. The ability of Puerto Rico's seagrass beds to cope with rising seas depends on whether the rate of sea level rise is within the rate of vertical growth of most seagrass species present. According to the PRCCC WG 1 Report Puerto Rico has already experienced a 0.14 cm per year increase in sea level since the 1950s with a detected acceleration in recent years of between 0.2 and 0.3 cm sea level rise per year (consistent with global trends of .18 cm per year and 0.3 cm acceleration). The WG 1 report describes projections for future sea level rise through 2100 that generally range from 0.59 meters to 2 meters. Assuming a linear increase in sea level of 0.2 meters by the year 2050 (note: sea level rise is not linear and will continue to experience accelerations and decelerations), per year that would work out to a 0.5 cm increase by 2050; within the vertical growth of most seagrass species (but higher than some, such as *Posidonia oceanic).* However, if assuming a linear increase in sea level at the upper limit of sea level rise projections as determined by the WG 1 a 0.5 m increase by 2050 would amount to 1.28 cm increase per year, thus reducing the number of species that growth vertically enough to acclimate to the change.

Additionally, seagrass canopies may remain at relatively similar depths for long periods of time, provided they can also trap materials to elevate the sediment surface, such as observed in reef-building species. The rise in sea level will cause a greater inland penetration of seawater in estuaries, and this may result in the displacement of less salt-tolerant submerged macrophyte species by seagrasses. Sea level rise will also lead to coastline regression and sediment erosion (see Beaches section in this report). If seagrasses are unable to migrate at new erosion rates, Puerto Rico will experience widespread losses due to sediment erosion. Sediment erosion may be aggravated in areas where coastlines are densely populated. Structures and development activities may contribute to the destabilization of the marine sedimentary environment and cause additional seagrass loss (Hemminga and Duarte 2000). In addition, deeper populations are likely to be lost as many species have a narrow depth limit (Carr et al. 2012). Changes in coastal water quality as a result of increased erosion (increased turbidity) as well as coastal development will further increase the loss of suitable deep water habitat. Lastly, changing current patterns or the tidal regime could either erode seagrass beds or create new areas for seagrass colonization (Harlin et al. 1982).

CHANGES IN PRECIPITATION

Changes in the river flow regimes and sediment transport may lead to increased sediment loading, thereby burying seagrass beds, and in localized salinity changes. Some seagrass species have a narrow tolerance for salinity changes, which can trigger major shifts in species composition (Lirman and Cropper, 2003). Salinity in association with nutrient enrichment can also become a stressor when freshwater inputs are drastically reduced.

INCREASED TEMPERATURE

Temperature stress on seagrasses are expected to result in distribution shifts, changes in patterns of sexual reproduction, altered seagrass growth rates, metabolism, an dchanges in their carbon balance (Short and Neckles 1998). Meadows occurring in thermally stressed environments, like thermal effluence, could be more greatly affected by a 1.5˚C change in temperature than other seagrass beds ((CANARI) 2008). However, temperatures of 35˚C or more can prevent some species, from rooting (Vicente *1992*). Therefore an increase in sea surface temperatures could lead to a decrease or increase in species range depending on the species and temperature. When temperatures reach the upper thermal limit for individual species, the reduced productivity will cause them to die (Coles et al. 2004). The risk of local extinction could be great.

Much more research is required to allow the formulation of specific predictions for seagrasses under higher temperatures. The direct effects of increased temperatures will depend on individual species' thermal tolerances and on the specific temperature dependency of the many processes that determine growth and reproduction. For species currently living near their upper temperature limit, a further increase in temperature may be fatal. Species living in shallow water and enclosed embayments will be more susceptible and it is likely that temperature stress will exacerbate stresses caused by changes in water quality with sea level rise. In less critical environments, a rise in seawater temperature may have an impact on processes like photosynthesis, respiration, nutrient uptake, flowering and seed germination (Short and Neckles 1998). Such effects will have consequences for the vigor and competitive ability of populations. Therefore, it is likely that the species-dependent impact of temperature increase will alter seagrass distribution and abundance in the long-term.

Elevated temperatures may also increase the growth of competitive algae and epiphytes, which can overgrow seagrasses and reduce the available sunlight they need to survive. Another potential outcome of increased sea surface temperatures is greater prevalence of disease.

INCREASED CARBON DIOXIDE AND OCEAN ACIDIFICATION

Seagrasses use Carbon Dioxide (CO_2) for photosynthesis rather than bicarbonate like most marine macroalgae. Atmospheric CO_2 has a complex impact on the availablility of dissolved inorganic carbon for seagrass photosynthesis, as the increase in dissolved CO_2 in seawater causes interacting shifts in the equilibrium of the various dissolved inorganic carbon species and pH (Goudriann, 1993; Short & Neckles 1998, Feely et al. 2004, Fabry et al. 2008). The overall consequence of rising atmospheric CO_2 will be an increase in both the actual concentration of dissolved CO_2 and the relative proportion of CO_2 to bicarbonate (HCO_3^-). This could, in turn, enhance the productivity of seagrass beds (Unsworth et al. 2012), which have been shown to experience carbon limitation under present-day conditions (Zimmerman et al. 1995). Similarly, the depth range of seagrasses may be extended, as the higher photosynthetic rates at elevated dissolved CO_2 levels would be conducive to a positive carbon balance at greater depths. Changes may occur in the competition between seagrass species and between seagrasses and macro-algae. The positive effects on seagrass growth may be offset by stimulated growth of epiphytic algae on leaf surfaces. Algal epiphytes respond positively to increased availability of CO_2 but due to their faster uptake rates can form blooms that remove nutrients and block light (Raven et al. 1995) needed by the seagrass host. Overfishing has severely reduced the populations of large herbivores in seagrass habitats that control overgrowth of algae. The loss of herbivorous fishes along with mammalian grazers like sea turtles and manatees will allow algal blooms to go unchecked. Decreased productivity could also occur if there is a loss of algae diversity in the seagrass beds. Furthermore, the algae overgrowth could increase the risk of seagrass removal by waves due to increased drag.

Another affect of climate change on seagrass beds is from ocean acidification. Increasing CO_2 concentrations is expected to have a negative effect on calcification rates of calcifying algae, corals, and other carbonate-dominated systems like seagrass beds, therefore affecting seagrass growth and productivity. Multiple studies have been published that used realistic carbonate chemistry manipulations, results showing decreased recruitment rates under high CO_2 conditions and an increase in non-calcifying algae cover (Kuffner et al. 2008), thus changing population structure and dynamics of submerged aquatic vegetation communities, reduced growth in certain rhodoliths of mixed cructos coralline algae (Jokiel et al. 2008), and reduced coralline algal cover and even complete dissolution when pH is reduced to 7.0 (Martin et al. 2008).

TROPICAL STORMS AND HURRICANES

Climate change is causing tropical storms to increase in intensity and possibly in frequency (see PRCCC WG1 Report). Increased storm and tidal surges, changes in storm intensity and frequency, and subsequent change in river flow regimes and sediment transport will impact seagrass beds (Hassan et al. 2005). Seagrasses grow in low energy environments, and thus, increased turbulence from storms and tidal surges could have disastrous effects by uprooting or burying the plants. Storms also increase turbidity thus resulting in a loss of photosynthesis. During storms not all concerns are from the sea, but also from the land. The increase in rainfall and discharges from rivers can increase sediment loading that may also result in decreased light levels or smothering of seagrasses. Since seagrass species are relatively slow-growing plants (compared to some algae) they need to establish extensive root systems in order to anchor themselves in the sediment, their ability to re-establish after a storm will be dependent on the time interval between storms. Additionally, seagrasses could be further stressed by decreases in salinity brought on by heavy rainfall associated with storms passing through the region.

OTHER IMPACTS OF GLOBAL CLIMATE CHANGE ON SEAGRASSES

Increased ultraviolet (UV) radiation, changes in surface currents, decreases in biodiversity, increased susceptibility to disease, and loss of critical associated habitats like coral reefs can all have negative and far-reaching impacts on seagrass populations worldwide. Increases in UV radiation due to global climate change will increase the likelihood of damages to the photosynthetic apparatus and DNA in seagrasses. The response of UV radiation will vary by species. As temperature gradients change and increase, so will ocean circulation patterns and intensity. Most seagrass species are reliant on water transport of propagules to suitable habitat and some sites are entirely dependent on propagules from upstream sites to ensure continued survival of seagrass communities. High biodiversity has been shown to act as a buffer to disturbance and increase resilience in a variety of systems, including seagrass beds. Not only species diversity but genetic diversity within species has been found to increase resilience in seagrasses (Hughes and Stachowicz 2011). All of these stresses combined, plus the stresses previously described, greatly increase the change of infection by deadly diseases, as occurred in the east coast of the United States in the 1970s. As seagrasses become stressed and biodiversity decreases, their susceptibility to disease will increase.

The loss of critical habitat by the development of marinas and other nearshore construction and the loss of water quality puts seagrass communities at high risk of further losses. Puerto Rico has the highest population density in the Caribbean and is rapidly developing coastal areas across the island. This increases the risk and susceptibility of local seagrass populations for biomass loss and extinction, especially when all stressors are combined.

RESEARCH GAPS AND INFORMATION NEEDS

Effective management of seagrass meadows is essential in order to maintain these valuable ecosystems and their important functions. Controlling human impacts will help maintain healthy seagrasses that are better able to resist or recover from climate change impacts. There are a number of information gaps and research needs that should be addressed so managers can support the resilience of seagrass systems:

1. Information is needed regarding the potential shifts in seagrass habitat with sea level rise and changes in ocean temperatures, including whether or not all species will be able to tolerate changing conditions. This is of particular importance for the dominant species, turtle grass, manatee grass, shoal grass, and star grass, around Puerto Rico, which already exhibit depth zonation to some degree.
2. Long-term observations and monitoring of seagrass meadows are needed. There was a gradual decline of areal distribution noticed during the 1980s and 1990s that was related to elevated sea-surface temperatures observed since the 1980s (Glemarec et al. 1997 as cited in Hemminga and Duarte 2000). Such observations suggest that expansion and regression of seagrass meadows may follow climate changes, and that widespread changes in the areal distribution of seagrasses as a result of current global change phenomena are not unlikely. Regular assessments are also important in order to identify disturbances to seagrass habiats and to determine management efforts. These assessments should include seagrass abundance (cover, density and/or biomass), habitat extent and species compositon (in mixed meadows). Long term observations must also include measurements of coastal water quality and light availability for photosynthesis. These could be of importance as early warnings of disturbances.
3. Productivity and rate of vertical growth measurements of Puerto Rico seagrass. Very little is known about the current status of seagrass beds in Puerto Rico. Density, species composition, biomass and distribution must be measured to provide a productivity estimate per species. Benthic surveys were completed in 2001 as part of a cooperative effort between NOAA's National Centers for Coastal Ocean Science, the U.S. Geological Survey, the National Park Service, and the National Geophysical Data Center, to produce benthic habitat maps and georeferenced imagery for Puerto Rico and the U.S. Virgin Islands, but no detailed studies have been completed since, despite the fact that the surveys showed seagrass communities in more than half of nearshore habitats in Puerto Rico. Improved benthic habitats mapping and marine ecosystems characterizations have been conducted by the NOAA Biogeography branch (Kendall et al. 2006, Zitello et al. 2008, Bauer et al. 2008, Jeffrey et al. 2010).
4. Responses of individual seagrass species to higher temperatures. Large die-offs of beds have occurred in shallow areas at low tide on sunny days where temperatures and UV exposure are high. Community and species responses in Puerto Rico will be critical in understanding the suitability of new habitat added as sea level rises and the impact on the survivorship of seagrass populations.
5. Connectedness between populations. As surface currents shift direction and strength, sources of propagules and genetic diversity will shift or be eliminated. Studies by Hughes and Stachowicz (2004) have shown that in addition to ecosystem biodiversity, genetic diversity within seagrass populations enhances resistance to disturbances and so any limitation in diversity may have a negative impact on seagrass ecosystem resistance and resilience to future disturbances. It should be established where seagrass

meadows are ecologically linked to adjacent habitats (e.g. mangroves or coral reefs). Such linkages could be taken into account when designing protected areas and management plans.

SUMMARY OF CLIMATE CHANGE AND SEAGRASS BEDS

Seagrasses are submerged marine flowering plants forming extensive meadows in many shallow coastal waters worldwide. The leafy shoots of these highly productive plants provide food and shelter for many marine animals (including commercially important species, e.g. prawns), and their roots and rhizomes are important for oxygenating and stabilizing bottom sediments and preventing erosion. The monetary value of seagrass meadows has been estimated at up to $19,000 per hectare per year, thus being one of the highest valued ecosystems on earth.

Many seagrass habitats have already been severely impacted by human activities, and may be additionally affected by global climate change. Seagrass meadows are especially vulnerable to sediment disturbances, eutrophication or the addition of excess nutrients to coastal areas, light reductions, and temperature increases.
Climate change impacts include potential effects of increasing seawater CO_2 levels, decreasing pH (ocean acidification), increasing temperatures, and rising sea level. These factors can have both positive and negative impacts on seagrass photosynthesis and growth and ecosystem function, but more data is needed to better predict how climate change will affect seagrass meadows, especially in areas already impacted by other human disturbances. In Puerto Rico the main impact on seagrasses may result from synergistic interactions between increasing temperature and habitat degradation that will push them over their limit of existence, resulting in widespread seagrass loss. Loss of sea grass habitats will not only impact sea grass communities but will have a negative impact on associated coral reef and mangrove habitats as well as ecosystem services.

COASTAL AND PELAGIC FISH SPECIES

Fish have complex life cycles, comprising several distinct life history stages (egg, larva, juvenile and adult), each of which may be affected in different ways by climate change. Over their lifespan, fish typically increase in body size by a factor of 10^5 (Rothschild, 1986), and successive life-history stages may require spatially separated habitats. A prerequisite for population persistence is connectivity among the required habitats, allowing the survivors to mature and return to the spawning grounds to reproduce successfully (Sinclair, 1988). Within these stage-specific habitats, fish must experience suitable abiotic conditions, find food for growth, and find shelter to avoid predation or disease.

Climate-driven changes in fish populations may result from four, often interlinked, mechanisms: (i) a physiological response to changes in environmental parameters, such as temperature, (ii.) a behavioral response, such as avoiding unfavorable conditions and moving into new suitable areas, (iii) population dynamics, through changes in the balance between rates of mortality, growth and reproduction in combination with dispersal which could result in the establishment of new populations in new areas or abandonment of traditional sites, (iv) ecosystem-level changes in productivity and/or trophic interactions. In addition, (v) commercial exploitation greatly affects the abundance and distribution of fish and may interact with the effects of climate change.

Climate change will affect individuals, populations and communities through their physiological and behavioral responses to environmental changes (Boesch& Turner, 1984). Extremes in environmental factors, such as elevated water temperature, low dissolved oxygen, changes in salinity and pH, can have deleterious effects on fishes (Moyle &Cech, 2004). Suboptimal environmental conditions can decrease foraging, growth, and fecundity, altermetamorphosis, and affect endocrine homeostasis and migratory behavior (Barton & Barton, 1987; Donaldson, 1990;

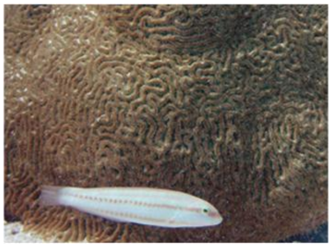
Pörtneret al., 2001). These organism-changes directly influence population and community structure by their associated effects on performance, patterns of resource use, and survival (Ruiz *et al.*, 1993; Wainwright, 1994). Projections of future conditions portend further impacts on the distribution and abundance of fishes associated with relatively small temperature changes. Changing fish distributions and abundances will undoubtedly affect communities of humans who harvest these stocks (Roessig*et al*.).

This section is specific to coastal finfishes (nearshore demersal or bottom-dwelling and nearshore pelagic fishes) and pelagic or offshore oceanic finfishes, and are assessed independently. This assessment does not include sharks or rays. The effects of climate change on sharks and rays are generally unknown but assumed to be similar to those for finfish occupying equivalent habitat types. For example, reef sharks, such as Puerto Rico's Nurse Sharks (Gatas; *Gynglymostomacirratum*), Caribbean Reef Sharks (Tiburon coralino; *Carcharhinusperezii),* Caribbean Sharpnose Sharks (CazonPlayon; *Rhizoprionodonporosus);* Tiger Sharks (Tintorera; *Galeocerdocuvier)* and Whitetip Reef Sharks (Tiburon oceanic; *C. longimanus*), are highly dependent on coral reefs (Opitz 1996) and are likely to be affected negatively by extensive habitat degradation. Similarly, this assessment does not include invertebrates, which are commercially important in Puerto Rico, and should be included in future assessments. Additionally, fisher livelihoods and climate change are discussed in the PRCCC Working Group 3: Society and Economy report.

INTRODUCTION TO COASTAL FISHES

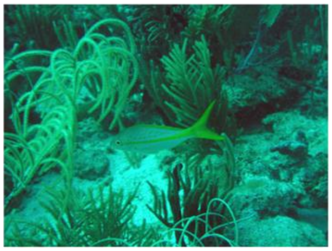
Coastal fish resources in the Caribbean are linked with mangrove, seagrass beds and coral reef habitats. These three coastal habitats are ecologically connected to each other (Beck et al. 2001) and the marine species that depend on healthy habitat connectivity include lobsters, crabs, shrimps, queen conch, a great variety of estuarine and reef fishes, coastal pelagics (clupeids, carangids) and fishes inhabiting the shelf slope (mainly deep-water snappers and groupers). Coastal fisheries in the insular Caribbean have been defined as reef fisheries (Munro, 1983). Estuarine environments are not extensive in the insular Caribbean, with the exception of Cuba and Hispaniola, and commercial fishing is prohibited in the majority of Puerto Rico estuaries. Most of the fishery resources of the island shelves (reef and estuarine fish, lobster, shrimp, conch and others), as well as the deeper demersal resources (mainly snappers and groupers) are considered to have been overexploited since the 1980s, and are coming under increasing fishery pressures throughout the region and are in need of rehabilitation in several islands (Mahon, 1987; 1993; Appeldoorn& Meyers, 1993; Aiken, 1993; Baisre, 1993; 2000; 2004; Claro *et al.*, 2001). Moreover, many of theshallow-water environments that these fishery resources inhabit (e.g., mangroves, seagrass, and coral reefs) have been seriously affected by coastal development, pollution, habitat loss, and overfishing (Rogers, 1985; Hunte, 1988; Bouchon*et al.*, 1987; Mahon, 1993; Claro *et al*, 2004; 2007; Jackson et al. 2001; Hughes et al. 2003; Bellwood et al. 2004). On the other hand, some research suggests that following the Caribbean-wide mass mortality of herbivorous sea urchins in 1983-1984 and consequent decline in grazing pressure on reefs, herbivorous fishes have not controlled algal overgrowth of corals in heavily fished areas although they have restricted algal growth in lightly fished areas. Differences among islands in the structure of fish and benthic assemblages suggest that intensive artisanal fishing has transformed Caribbean reefs (Hawkins & Roberts, 2004). Examples ofcoastal fish species found in Puerto Rico are snooks (robalos; FamiliaCentropomidae), snappers (pargos; FamiliaLutjanidae), grunts (roncos; FamiliaHaemulidae), butterflyfishes (mariposas; FamiliaChaetodontidae), angelfishes (isabelitas; FamiliaPomacanthidae), mullets (lisas; FamiliaMugilidae), damselfishes (damiselas; FamiliaPomacentridae), parrotfishes (loros; FamiliaScaridae), blennies (blenios; FamiliaBlennidae), gobies (gobios; FamiliaGobiidae), and puffers (tamboriles; FamiliaTetradontidae) (figure 16). For more detailed lists see Erdman, Grana 2007; Dennis 2004; Appeldoorn et al. 1987.

- *H. flavolineatum* French Grunt (Condenado)



- *Halichoeres bivattaus* Slippery Dick (Doncella Resbaladiza)

- *Epinephelus guttatus* Red Hind (Mero Cherna o Cabrilla)



- *Ocyums chrysurus* Yellowtail Snapper (Colirrubia o Rabirrubia)

- *Stegastes leucostictus* Beaugregory (Gregorio o Damisela Azul o Burrito)

- *Sparisoma aurofrenatum* Redband Parrotfish (Loro)

Figure 16 Figure 3 Examples of Coastal Fish Species in Puerto Rico by scientific name, English common name, and Spanish common name. Photos courtesy of NOAA CCMA Biogeography Team.

The commercial fisheries of Puerto Rico can be classified as small scale, multispecific, multigear and mostly artisanal in nature. The fisheries are largely demersal, some fishers also target pelagic species. The demersal fishery harvests for reef, bait, ornamental and deep-water species; over 155 finfish groups or species and approximately ten species of shellfish. The most prevalent groups in the reef fisheries include snappers, groupers, grunts, mackerels, parrotfish, trunkfish, spiny lobster and queen conch. The deep-water fishery targets snappers, groupers and tilefish. angelfishes, damselfishes, surgeonfishes, blennies, wrasses, basslets, jawfishes and others are ornamental fish targeted by the aquarium trade industry (Valle-Esquival et al. 2011). Large longline vessels with HMS permits from the continental U.S. only visit the Puerto Rico EEZ for a few days per year at most to target swordfish.

The marine recreational fishery of Puerto Rico has only recently (2000) begun to be systematically evaluated. Data from the Marine Recreational Fisheries Statistics Survey (which is now transitioning to the Marine Recreational Information Program) from the years 2000 – 2011 indicate that the harvest of marine fish by recreational anglers is on average greater than that of commercial fishers in terms of weight. The vast majority of fish harvested in the recreational sector consists of mahi mahi, several tuna species and mackerels. Reef fish are also targeted, but to a much lesser degree. Over half of all recreational anglers fish from shore, but the greatest diversity and abundance of catch occurs in the private boat mode. Charter boat fish harvest is a very minor component of the overall recreational scenario.

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON COASTAL FISHES

The previously described outcomes and consequences of climate change on mangroves, coral reefs, and seagrass beds directly apply to coastal fish species as these habitats provide them with shelter and food. Coastal fishes (and many species of pelagic fishes) are especially reliant on nursery habitat. A given marine habitat is considered a nursery if juveniles occur at higher densities, have lower rates of predation or higher rates of growth compared to other habitats, and also if the habitat contribution, in terms of juveniles per unit area to the production of adults, is greater than that from other habitats harboring juveniles (Beck et al. 2001). Mangroves, seagrass beds, and shallow coral reefs are important habitats functioning as nursery grounds for several reef associated fishes (Thayer at al. 1987; Baelde 1990; Ley et al. 1999; Nagelkerken et al. 2000). These habitats are used as nurseries by several juvenile fishes for the following reasons: (1) shelter from predators (Parrish 1989); (2) high abundance of food (Odum and Heald 1972; Ogden and Zieman 1977; Laegdsgaard and Johnson 2001); and (3) planktonic larvae reception due to extensive habitat coverage (Parrish 1989).

The most important direct effects on coastal fishes are likely to be due to the projected increases in sea surface temperature and ocean acidification and changes in ocean currents (Munday et al. 2008, Brander 2009). Additional environmental changes and severe climatic events, such as increased severity of tropical cyclones, more extreme rainfall events or drought conditions, and sea level rise could have further direct and indirect effects for some species and life stages. The direct and indirect effects of climate change on coastal fish will vary depending on their biology and ecology. For example, recent effects of climate-induced coral loss on reef fish vary depending on their reliance on live corals for food, shelter and/or recruitment (Pratchett 2008). Butterflyfish, and certain species of damselfish, cardinalfish and coral-dwelling gobies have the greatest dependence on live coral and are directly at risk from loss of corals. Other fish may also be particularly susceptible to climate change owing to their sensitivity to changes in temperature or ocean chemistry. The potential impact of projected changes on species is moderated by the species ability to alter its physiology, behaviour and/or distribution to cope with changes. Species with a high sensitivity to changes in environmental conditions and habitats, which as unable to adapt, are most vulnerable and potentially at risk of local extension due to climate change (Adger 2000, IPCC 2001). Examples of ways fish species can adapt include switching to a different prey species, living in slightly different habitats, and evolving increased tolerances. A species-by-species assessment for Puerto Rico is not within the scope of this report, but could be considered necessary for marine conservation and fisheries management.

INCREASING TEMPERATURE

As described in the coral reef section, temperature increases are already degrading coral reefs worldwide. Fish biodiversity is threatened wherever permanent reef degradation occurs. A significant decrease of fish density and biomass in coral reefs as a result of coral cover reduction is expected, as shown to have occurred in the Archipelago Sabana-Camagüey, Cuba, after several bleaching events, and subsequent increase of algal overgrowth (Claro et al. 2007). Similar results have been reported for New Guinea reefs (Jones et al. 2004).

Changes in water temperature have a major influence on most coastal fish because they have limited capacity to maintain an independent body temperature. Temperature tolerance is a determinant of geographic ranges and the general biology and behavior of fish. Temperature increases are predicted to drive species ranges toward the more cold waters (Parmesan &Yohe, 2003) potentially resulting in widespread extinctions where dispersal capabilities are limited or suitable habitat is unavailable (Thomas *et al.*, 2004). Several marine organisms have already been observered shifting their geographic ranges poleward (Holbrook et al. 1997, Parker and Dixon 1998, Perry et al. 2005, Ling et al. 2009). Also, climate change may strongly influence distribution and abundance of fishes (Wood & McDonald, 1997; Brander *et al.*, 2004; Rijnsdorp et al. 2009) through changes in growth, survival, reproduction, or responses to changes at other trophic levels (Brander *et al.*, 2003; Reid, 2003, (Munday et al. 2009b)). These changes may have positive or negative consequences on the productivity and composition of coastal fisheries and thus the nature and value of commercial fisheries. Allison *et al.*, (2005) showed that the distribution of both exploited and non-exploited North Sea fishes have responded markedly to recent increases in sea temperature, with two thirds of species shifting in mean latitude or depth over 25 years. Further temperature rises are likely to have profound impacts on commercial fisheries through continued shifts in distribution and alteration in community interactions (Allison *et al.*, 2005). There is a lack of information on how Puerto Rico's tropical fish will respond to temperature increases.

The main effect of increasing sea surface temperatures on fish is related to increases in their metabolic processes, and their limitations in meeting the associated higher demands for oxygen (Pörtner and Farrell 2008). Metabolic activity for fish increases in direct proportion to moderate increases in ambient temperature (Houde 1989, Jobling 1997, Clark and Johnston 1999, Hunt Von Herbing 2002). However, with increasing SST, there is a decrease in available dissolved oxygen, at the same time a fish's demand for oxygen is increasing. Consequently, oxygen becomes a limiting factor in cardiovascular performance (Farrell 1997), and thus keeping pace with an increased demand for oxygen at higher temperature is the key physiological mechanism controlling fish thermal tolerance (and determining a fish's response to increasing SST). The sensitivity of the aerobic system to increased SST is unknown for most tropical marine species, however, one study found that sensitivity to increasing temperature differs greatly among five species of coral reef fish (Nilsson et al. 2009), suggesting that adults of some species are highly sensitive to small increases in temperatures, whereas other species may be quite tolerant to increases even of 2.5°C to 3.0°C.

Temperature changes could also affect the reproductive performance of some reef fish (Gagliano et al. 2007, Donelson et al. 2010) and are likely to lead to shifts in the timing of spawning, and possibly falling egg production, in some populations. Increases in ambient temperature, and corresponding increases in food demands, are also expected to affect the survival of offspring, especially during pelagic dispersive phases. Larval fish grow and develop more rapidly at higher temperatures, meaning that they may experience lower rates of mortality. However, under higher SSTs, entire cohorts of larval fish may not be able to obtain sufficient food to sustain their rapid growth and development, leading to more frequent recruitment failure (Sponaugle et al. 2006, Munday et al. 2009b).

Life history traits of some species covary in a predictable way with their latitudal ranges and temperature gradients. Although there are many exceptions, reef fish species tend to be shorter-lived and reach smaller maximum sizes in equatorial areas compared with the subtropics (Munday et al. 2008). Juveniles are also expected to reach their asymptotic size at a faster rate at higher temperatures. Consequently, we might expect that as SST increases, life history traits of populations in the subtropics will become more like those exhibited in populations already living at higher temperatures (e.g., shorter lived and smaller maximum sizes). It's important to note, however, that life history traits are also influenced by food supply, and population density and structure, which vary at a range of spatial scales, and might themselves be influenced by increased SST. Therefore, projecting how life history traits may change as a result of changes in SST is difficult and as yet not done for the coastal waters and fishes around Puerto Rico.

An independent global analysis has concluded that the redistribution of fisheries species associated with increasing SST and shifts in climate are expected to have significant impacts on fisheries production of tropical countries (Cheung et al. 2010). Geographical shifts in the distribution and abundance of major fisheries species, combined with projected changes in primary production, are expected to cause a decline of up to 30% in fisheries production in the areas around Puerto Rico (although possible increases could exist according to the analysis). These estimates do not take into account habitat requirements that will limit latitudinal shifts of many reef species; only generalist species that are loosely associated with coral and reef habitats are expected to move poleward to escape rising SST in the tropics.

INCREASING CARBON DIOXIDE

Ocean acidification is the second major threat to fishes as it may affect the food web as well as affect metabolic and development systems of individuals. There is convincing evidence to suggest that acidification will affect the process of calcification, by which animals such as corals and mollusks make shells and plates from calcium carbonate. Ongoing ocean acidification will reduce growth and survival of many calcifying organisms and affect the food webs that depend on them (Fabry et al. 2008, Doney et al. 2009) (Kleypas et al. 2006: figure 17).

The tropical and subtropical corals are expected to be among the worst affected, with implications for the stability and longevity of the reefs that they build and the organisms that depend on them. Other calcifying organisms that may be affected are components of the phytoplankton and the zooplankton, and are a major food source for fish and other animals. Regional variations in pH will mean that by 2100 the process of calcification may have become extremely difficult for these groups of organisms particularly in the Southern Ocean. Some studies suggest that growth and reproduction in some calcifying and non-calcifying marine species could be reduced due to the projected changes in ocean chemistry. With two duplicate six-month manipulative experiments, Shirayama&Thorton (2005) demonstrated that a 200 ppm increase in carbon dioxide adversely affects the growth of both gastropods and sea urchins. Thus, even moderate increases in atmospheric carbon

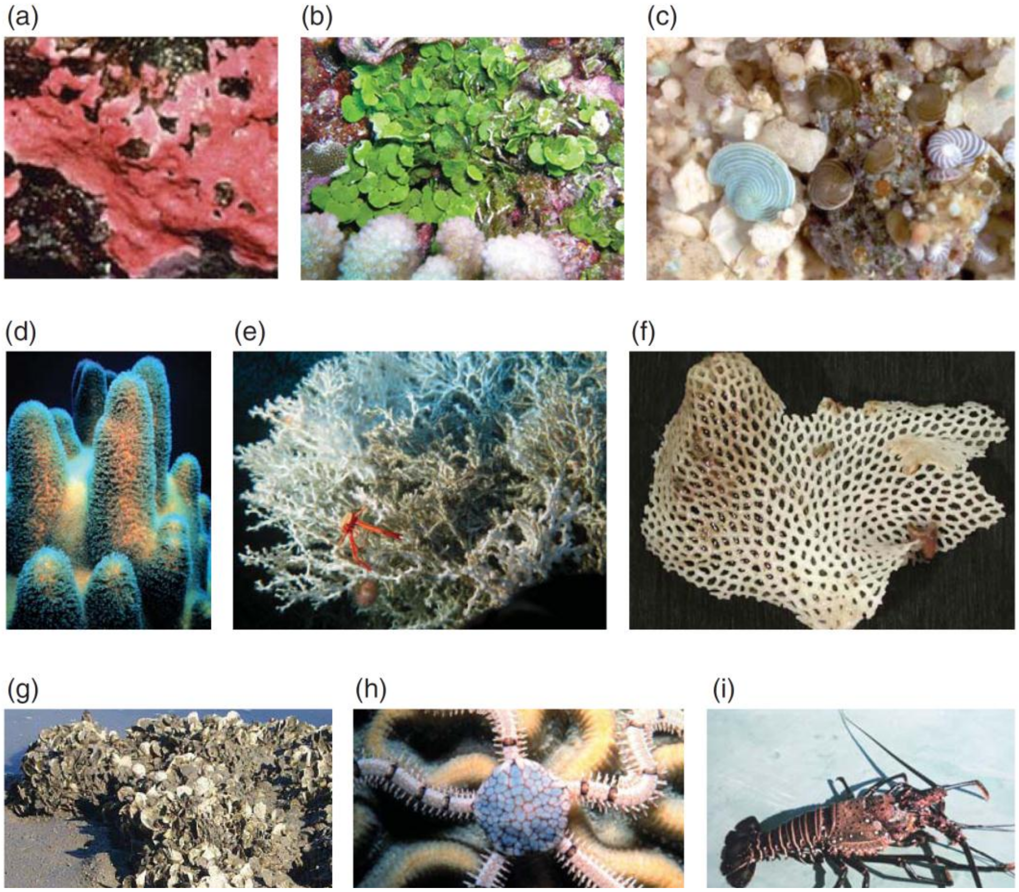


Figure 17 Representatives of major benthic calcifiers: **(a) coralline algae** (photo by Nancy Sefton; courtesy NOAA/CORIS); **(b**) *Halimeda* (photo by James Watt; courtesy NOAA/NMFS); **(c)benthic foraminifera** (courtesy P. Hallock); **(d) reef-building coral** (*Dendrogyracylindrus*; Cmdr William Harrigan, NOAA Corps; courtesy Florida Keys National Marine Sanctuary); **(e) deep-water coral** (*Lopheliapertusa*; from 413 m depth off North Carolina. Large red crab is *Eumunidapicta*; urchin below it is Echinus tylodes; courtesy S.W. Ross, K. Sulak, and M. Nizinski); **(f) bryozoan** (courtesy NOAA/Ocean Explorer); **(g) mollusk** (oyster reef; courtesy South Carolina Department of Natural Resources); **(h) echinoderm** (brittle star; Larry Zetwoch; Florida

Keys National Marine Sanctuary); **(i) crustacean** (lobster; Dr. James P. McVey, NOAA Sea Grant Program). The calcification responses of many of these groups have not been investigated. Figure printed with permission from Kleypas et al. 2006.

dioxide that could well be reached by the middle of this century will adversely affect shallow water marine benthic organisms.

The reduction in ocean pH could also have a range of direct effects on coastal fish, from increased metabolic demands to developmental problems (Pörtner et al. 2004). For instance, reduced calcification could affect the process by which fish develop otoliths (ear bones) (Munday et al. 2011) as they are composed of aragonite, whereas the rest of the fish skeleton iscomposed of calcium phosphate. Otoliths are important for individual orientation and hearing, especially during the dispersive larval stages. More research is necessary to know whether acidification affects otolith growth or development. Another developmental concern of ocean acidification is the effect that elevated carbon dioxide levels could have on the sensory ability of larvae. Clownfish larvae exposed to waters with reduced pH lose the ability to distinguish olfactory cues from preferred settlement habitat, or to detect and avoid the smell of predators (Dixson et al. 2010), at the end of their larval phase. In other words, their sense of smell is altered by ocean acidification according to studies (Munday et al. 2009a, Dixson et al. 2010). Their sense of smell is what they use for a wide range of important behavior decisions, such as navigation to reefs and selection of settlement sites (Atema et al. 2002, Kingsford et al. 2002, Gerlach et al. 2007). Impairments of this process by ocean acidification could have serious implications for the replenishment of adult populations and patterns of population connectivity in coral reef ecosystems.

From the evidence available, it is not certain whether marine species, communities and ecosystems will be able to adjust or evolve in response to changes in ocean chemistry, or whether ultimately the services that the ocean’s ecosystems provide will be affected. Ocean pH has changed very little over the past 800,000 years (Luthi et al. 2008), and so it might be expected that marine organisms lack genetic variation necessary for rapid adaptation to changes in seawater chemistry (Hoegh-Guldberg et al. 2007).

TROPICAL STORMS AND HURRICANES

Increasing storm intensity will likely further reduce available fish habitat, e.g. mangroves and seagrasses, which are the main nursery areas for many coastal fishes and invertebrates, as well as coral reefs (Rogers *et al.*, 1982; Gelardes& Vega, 1999; Anonymous, 2005). The loss of habitat is greater in areas already affected by pollution and unsustainable use of the coastal zone.

OTHER CLIMATE CHANGE OUTCOMES FOR COASTAL FISHES

The Caribbean coastal waters periodically experience extensive blooms of algae that impact living resources, local economies and public health. Impacts of harmful algal blooms include human illness and death from ingesting contaminated shellfishes or fish, mass mortalities of wild and farmed fish, and alterations of marine food chains through adverse effects on eggs, young, and adult marine invertebrates (e.g. corals, sponges), sea turtles, seabirds, and mammals. Harmful algal blooms are increasing worldwide in frequency, distribution and impact, with significant threats for the insular Caribbean (Sierra-Beltrán*et al.*, 2004). Recently, blooms have occurred in new coastal areas and new species have appeared (GEOHAB, 2001; 2005). Harmful algal blooms are usually associated with upwelling systems, and wind is the main driving force in upwellings. So, variations in the wind regime due to climatic changes could cause short-term variation in upwelling-downwelling cycles (GEOHAB, 2005).

With temperature increases, mass mortalities due to disease outbreaks are expected, as some have affected major taxa in the oceans. For closely monitored groups like corals and marine mammals, reports of the frequency of epidemics and the number of new diseases have increased recently; this has been discussed in detail in the previous sections. The role of coral community structure and diversity in maintaining productive fish and invertebrate populations is well documented, but links between these aspects and coral diseases are generally unstudied

INTRODUCTION TO PELAGIC FISH SPECIES

Offshore fish resources are considered to be those caught off the islands shelves. These include small pelagics, mainly sardines, flying fish and ballyhoo, which are also found in coastal waters, some of which are associated with upwellings and highly productive waters more common off continental shelves; medium or large size pelagic predators and migratory fishes (tunas, mackerel, swordfishes, billfishes, sharks and rays); demersal fishes (mainly groupers and snappers); and bathypelagic fishes inhabiting offshore banks and oceanic cays (fished mainly by Puerto Rico and Hispaniola fishers). Demersal and pelagic species differ in their habitat requirements. Pelagic species remain within the three-dimensional pelagic habitat (figure 18) (although some species, such as herring, require specific demersal habitats for spawning), whereas demersal species display a distinct transition from the three-dimensional pelagic realm of (eggs and) larvae to the two-dimensional habitat of the demersal stages.

The fish resources of deep slopes and banks, which are based on fewer species than those on the island shelves, are in a better situation and in some cases, like in Cuba, they are underexploited. Large offshore pelagic fish resources are generally considered to hold the greatest potential for development in the islands. The (mostly recreational) catch consists of several species with a wide variety of life histories: tunas, billfishes, dolphin fish, wahoo, king mackerel, sharks. Most of these species show a marked seasonal availability in

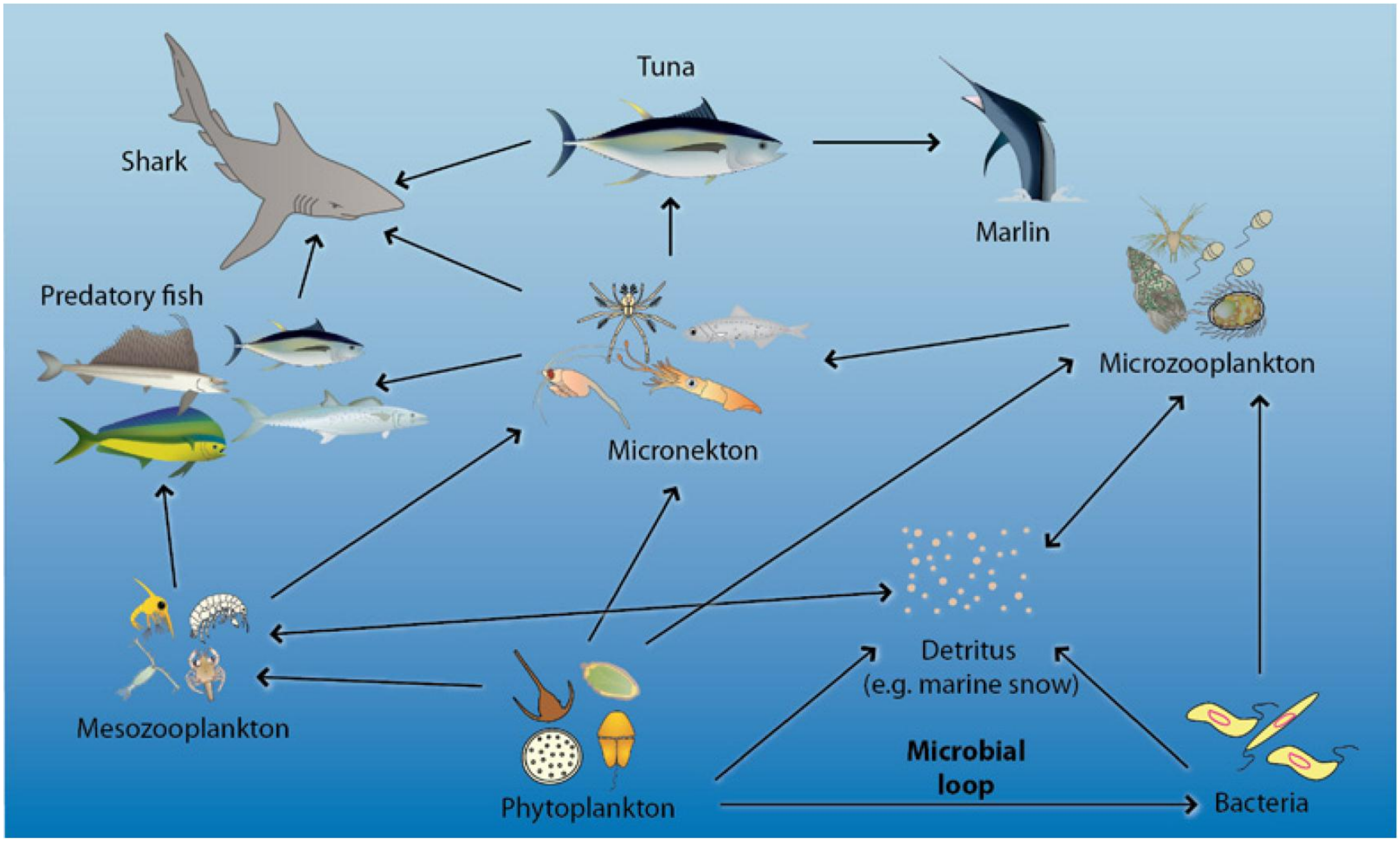


Figure 18 Generalized food web supporting tuna and other large pelagic fish. Note that in the lower levels of the food web, the classical and microbial pathways are linked through formation of 'marine snow' and other detritus. Printed with permissions from Bell JD, Johnson JE and Hobday AJ (eds) (2011) Vulnerability of Tropical Pacific Fisheries and Aquaculture to Climate Change. Secretariat of the Pacific Community, Noumea, New Caledonia.

the whole region. Dolphin fish are the principal recreational fishing species for private boat anglers in Puerto Rico, and thus have considerable socioeconomic value. Flying fish are also exploited in some of the islands of the Lesser Antilles. The status of offshore pelagic fish resources is highly variable depending on the species. Some of the most important resources (skipjack and other small tuna, and swordfish) are heavily exploited in most areas, but little information is available for stock assessment. The existing information about resource assessment and management of the main fisheries in the Caribbean islands is patchy and much of the information is extrapolated from studies on the same species and types of fisheries elsewhere in the Wider Caribbean.

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON PELAGIC FISHES

TEMPERATURE

Perhaps the biggest concern with pelagic fish is plankton abundance, community structure, timing of seasonal abundance, and geographical range (Hays et al. 2005). Plankton consists of microscopic plants (phytoplankton) and animals (zooplankton) that drift in the ocean and form the base of the marine food web. All species of fish feed on plankton during their larval (early) stage and some species continue to depend on plankton into adulthood. Long-term changes in plankton such as those predicted with climate change could have a significant impact on commercial fish stocks (Hays et al. 2005). This could include commercial species like Blue Runner, Wahoo, Yellow Fin Tuna, Dolphin Fish (MahiMahi) and Swordfish. Rising sea surface temperatures could also have a significant affect as fish species are very sensitive to slight changes (a few degrees) in ocean temperature and warming could cause migration to new areas or depths (Mimura et al. 2007). The habitat for Dolphin Fish, for example, will become significantly less favourable with just a 1°C (1.8 °F) increase in the average temperature of the Caribbean Sea (Trotz 2009) (figure 17). Pelagic fish distribution can also be impacted by shifts in ocean currents and other oceanographic conditions that help to determine where fish settle out in their larval stages and influence their migration patterns and other dynamics in their adult stages (Johnson and Marshall 2007). While these changes may result in the loss of established fisheries, they may also give rise to new fisheries. See figures 19 and 20 as examples of migration from studies completed by the Caribbean Community Climate Change Centre (5C's).

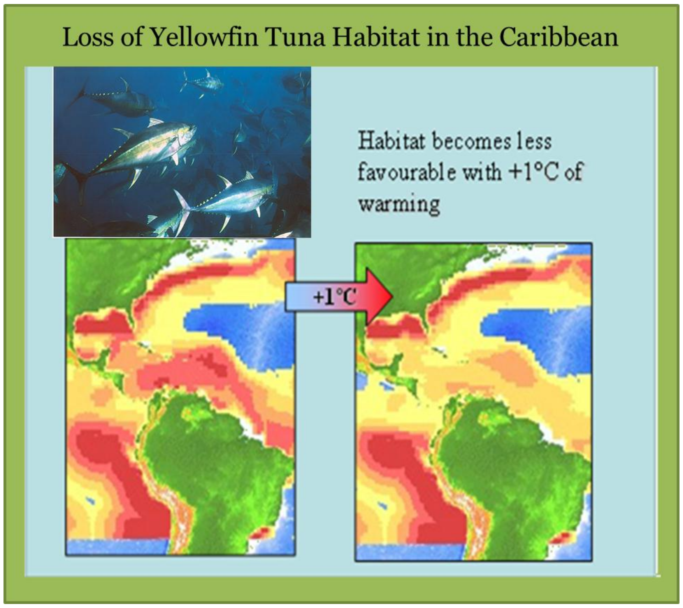


Figure 19: Loss of Yellow Fin Tuna fish habitat in the Caribbean due to +1°C of warming, courtesy of the Caribbean Community Climate Change Center

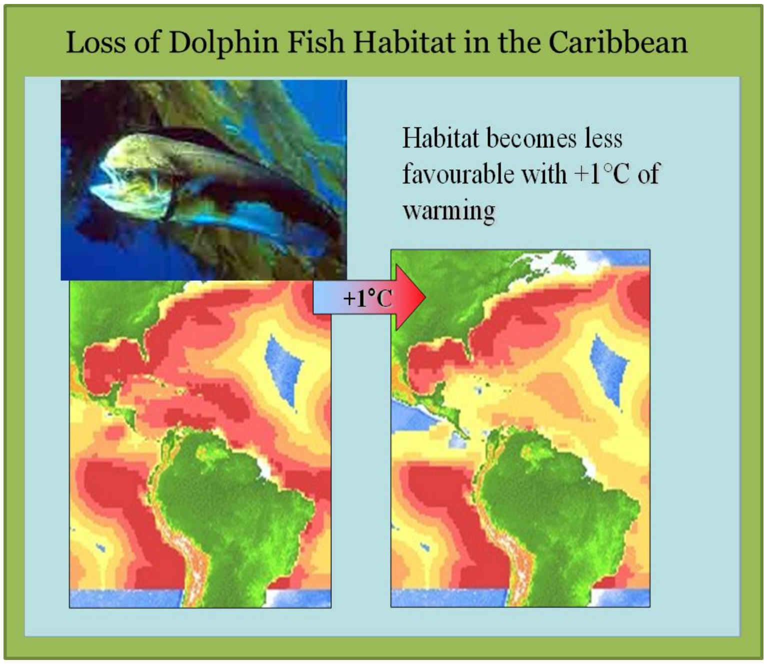


Figure 20: Loss of Dolphin Fish habitat in the Caribbean due to +1°C of warming, courtesy of the Caribbean Community Climate Change Center

Temperature changes in the ocean could also create more favorable conditions for the establishment of invasive species as natural ecosystem processes are disrupted (IUCN, n.d.). Invasive species tend to out-compete or prey on native species. Although not attributable to climate change, the recent invasion of Lionfish (*Pteroisvolitans*) in the Caribbean Sea and the effects this has already had on fisheries in the Turks and Caicos Islands demonstrates the damage that just one marine invasive species can do (Schofield 2009). Some positive effects on pelagic fishes may occur; for example, warmer waters may increase fish larval growth rate and swimming ability while decreasing the age of metamorphosis; all of which could improve the survival of larval fish (Johnson and Marshall 2007).

OCEAN ACIDIFICATION

The physiological responses to ocean acidification are the same for pelagic fish and coastal fish. However, the ecosystem-level effects of ocean acidification will more greatly affect the coastal ecosystems that support coastal fishes than that of the open ocean. Ocean acidification is expected to reduce the shells of calcareous phytoplankton and zooplankton, making them more fragile and therefore vulnerable to predation. Changes in the complexity of the food web may result as some of these organisms disappear. In order to accurately assess the affect ocean acidification may have on the plankton of Puerto Rico's waters, the percentage of calcareous plankton versus non-calcareous plankton in Atlantic Ocean and Caribbean Sea waters would need to be known. The PRCCC WG 2 was unable to find a reliable source with this information.

RESEARCH GAPS AND INFORMATION NEEDS

1. More research is needed to test the effect of elevated CO_2 on the early life stages of coastal fish across a broader range of species, and to examine possible synergistic effects of elevated temperature and CO_2.
2. Despite increasing concerns about designing and implementing an integrated approach in coastal conservation and management, the ecological relationships (*i.e.* habitat connectivity) between mangroves, seagrass and coral reefs have received relatively little attention. Fundamental ecological research on the habitat connectivity is urgently needed for the development of sound coastal conservation and management (Aguilar-Perera 2004).
3. Nearshore habitat responses: Additional research is need on the response of mangroves, seagrasses and coral reefs to climate change stressors, including rising sea levels, increased SST and beach erosion.
4. Fish and invertebrate responses: Relatively little research has been carried out on the physiological and ecological response of tropical fish and invertebrates to climate change stressors such as effects on species distributions, migratory patterns, otolith development, spawning periods, recruitment success, etc.

5. Estuarine responses: Changes in weather patterns which affect rainfall total amounts and intensity will result in changes in salinity in estuarine nursery areas for fish and invertebrates. Research is needed to understand the effect of this on commercially and recreationally important fish and invertebrate species as well as non-target species.
6. Information on composition of planktonic communities surrounding Puerto Rico and the percentage of calcareous plankton versus non-calcareous plankton.

SUMMARY OF THE OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON COASTAL AND PELAGIC FISHES

Climate-driven changes in fish populations may result from four, often interlinked, mechanisms: (i) a physiological response to changes in environmental parameters, such as temperature, (ii.) a behavioral response, such as avoiding unfavorable conditions and moving into new suitable areas, (iii) population dynamics, through changes in the balance between rates of mortality, growth and reproduction in combination with dispersal which could result in the establishment of new populations in new areas or abandonment of traditional sites, (iv) ecosystem-level changes in productivity and/or trophic interactions. In addition, (v) commercial exploitation greatly affects the abundance and distribution of fish and may interact with the effects of climate change.

This section is specific to coastal finfishes (nearshore demersal or bottom-dwelling and nearshore pelagic fishes) and pelagic or offshore oceanic finfishes, and are assessed independently. This assessment does not include sharks or rays. The effects of climate change on sharks and rays are generally unknown but assumed to be similar to those for finfish occupying equivalent habitat types. For example, reef sharks, such as Puerto Rico's Nurse Sharks (Gatas; *Gynglymostomacirratum*), Caribbean Reef Sharks (Tiburon coralino; *Carcharhinusperezii)*, Caribbean Sharpnose Sharks (CazonPlayon; *Rhizoprionodonporosus);* Tiger Sharks (Tintorera; *Galeocerdocuvier)* and Whitetip Reef Sharks (Tiburon oceanic; *C. longimanus*), are highly dependent on coral reefs and are likely to be affected negatively by extensive habitat degradation. Similarly, this assessment does not include invertebrates, which are commercially important in Puerto Rico, and should be included in future assessments.

Coastal fish resources in Puerto Rico are linked with mangrove, seagrass beds and coral reef habitats. These three coastal habitats are ecologically connected to each other and the marine species that depend on healthy habitat connectivity include lobsters, crabs, shrimps, queen conch, a great variety of estuarine and reef fishes, coastal pelagics (clupeids, carangids) and fishes inhabiting the shelf slope (mainly deep-water snappers and groupers). The previously described outcomes and consequences of climate change on mangroves, coral reefs, and seagrass beds directly apply to coastal fish species as these habitats provide them with shelter and food. Coastal fishes (and some species of pelagic fishes) are especially reliant on nursery habitat. The most important direct effects on coastal fishes are likely to be due to the projected increases in sea surface temperature and ocean acidification and changes in ocean currents.

Additional environmental changes and severe climatic events, such as increased severity of tropical cyclones, more extreme rainfall events or drought conditions, and sea level rise could have further direct and indirect effects for some species and life stages. The direct and indirect effects of climate change on coastal fish will vary depending on their biology and ecology. For example, recent effects of climate-induced coral loss on reef fish vary depending on their reliance on live corals for food, shelter and/or recruitment. Butterflyfish, and certain species of damselfish, cardinalfish and coral-dwelling gobies have the greatest dependence on live coral and are directly at risk from loss of corals. Other fish may also be particularly susceptible to climate change owing to their sensitivity to changes in temperature or ocean chemistry. The potential impact of projected changes on species is moderated by the species ability to alter its physiology, behaviour and/or distribution to cope with changes.

Species with a high sensitivity to changes in environmental conditions and habitats, which as unable to adapt, are most vulnerable and potentially at risk of local extension due to climate change. Examples of ways fish species can adapt include switching to a different prey species, living in slightly different habitats, and evolving increased tolerances. A species-by-species assessment for Puerto Rico is not within the scope of this report, but could be considered necessary for marine conservation and fisheries management.

Offshore fish resources are considered to be those caught off the islands shelves. These include small pelagics, mainly sardines, flying fish and ballyhoo, which are also found in coastal waters, some of which are associated with upwellings and highly productive waters more common off continental shelves; medium or large size pelagic predators and migratory fishes (tunas, mackerel, swordfishes, billfishes, sharks and rays); demersal fishes (mainly groupers and snappers); and bathypelagic fishes inhabiting offshore banks and oceanic cays. Perhaps the biggest concern with pelagic fish is plankton abundance, community structure, timing of seasonal abundance, and geographical range. Long-term changes in plankton such as those predicted with climate change could have a significant impact on commercial fish stocks. This could include commercial species like Blue Runner, Wahoo, Yellow Fin Tuna, Dolphin Fish (Dorado) and Swordfish. Rising sea surface temperatures could also have a significant effect as fish species are very sensitive to slight changes (a few degrees) in ocean temperature and warming could cause migration to new areas or depths. The habitat for Dolphin Fish, for example, will become significantly less favourable with just a 1°C (1.8 °F) increase in the average temperature of the Caribbean Sea. Pelagic fish distribution can also be impacted by shifts in ocean currents and other oceanographic conditions that help to determine where fish settle out in their larval stages and influence their migration patterns and other dynamics in their adult stages. While these changes may result in the loss of established fisheries, they may also give rise to new fisheries. The physiological responses to ocean acidification are the same for pelagic fish and coastal fish. However, the ecosystem-level effects of ocean acidification will more greatly affect the coastal ecosystems that support coastal fishes than that of the open ocean. Ocean acidification is expected to reduce the shells of calcareous phytoplankton and zooplankton, making them more fragile and therefore vulnerable to predation. Changes in the complexity of the food web may result as some of these organisms disappear. In order to accurately assess the affect ocean acidification may have on the plankton of Puerto Rico’s waters, the percentage of calcareous plankton versus non-calcareous plankton in Atlantic Ocean and Caribbean Sea waters would need to be known.

BIOLUMINESCENT BAYS

INTRODUCTION TO BIOLUMINESCENT BAYS

Bioluminescent Bays are rare worldwide ecosystems that have intrigued researchers and the community in general due to the display of light. In the world there are five widely recognized bioluminescent systems and three of them are in Puerto Rico: Bahía Fosforescente in Lajas, Laguna Grande in Fajardo and Puerto Mosquito in Vieques. There are also other systems in Puerto Rico where some bioluminescence has been observed (e.g. Monsio José, Lajas; Laguna Joyuda, Cabo Rojo; Bahía de San Juan, San Juan).

Bioluminescence refers to the emission of light by living organisms and is caused by a photochemical reaction involving the oxidation of a protein (*i.e.* luciferin) in the presence of an enzyme (*i.e.* luciferase) (Hastings 1983). During this reaction, energy is liberated resulting in the liberation of light. However, to make these systems “glow” the water needs to be mechanically stimulated.

In Puerto Rico the bioluminescence displayed is due to high concentrations of the bioluminescent dinoflagellate *Pyrodinium bahamense* var. *bahamense* (Margalef, 1957; figure 21). However, other bioluminescent dinoflagellates, such as *Ceratium fusus* and *Protoperidinium divergens*, have been also observed (Glynn et al. 1964, Gold 1965). Hernández-Becerril (1995) studied the thecate dinoflagellates present at Bahía Fosforescente and observed the coexistence of *P. bahamense* with others species like *Ceratium furca* var. *hircus*, *Dinophysis caudata* var. *ventricosa*, *Gonyaulax spp.*, *Prorocentrum spp.* and *Protoperidinium sps.* among others. Also, high densities of *Cochlodinium polykrikoides* have been observed in the northwest channel of Bahía Fosforescente (Margalef 1961, Burkholder et al. 1967, Cintrón 1969, Cintrón et al. 1970, Seliger et al. 1971, Seixas 1983, 1988, Cedeño-Maldonado 2008). Among these species *P. bahamense* and *C. furca* have been the most abundant dinoflagellate species in this bay.

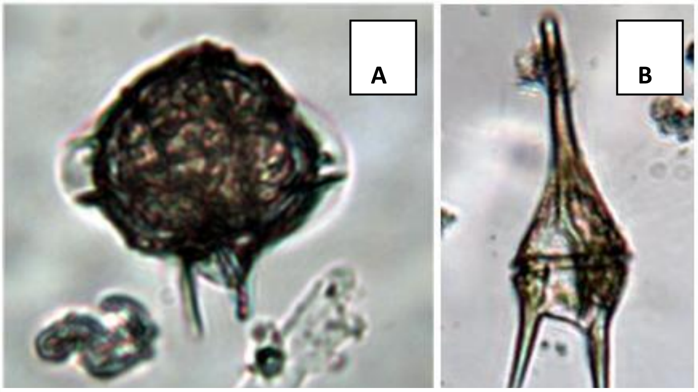


Figure 2 a) *Pyrodinium bahamense* b) *Ceratium furca*. Scale bar = 20µm.

Some attempts have been made to understand the dinoflagellates population dynamics in these systems and the mechanisms that sustain their high cell densities. The water circulation pattern and physiographic conditions have been proposed as important factors in determining the high abundances of species (Margalef and Gonzalez 1958, Coker and González 1960, Margalef 1961, Smayda 1970, Seixas, 1988). It is suggested, in the case of Bahía Fosforescente, that less dense offshore surface water enters the bay driven by the prevailing winds of the region, while denser water exit close to the bottom as a consequence of high evaporation rates (Margalef 1961). The combination of the above current pattern and the shallow and narrow entrance of these types of bays, the absence of riverine discharges and the small tidal range make the water mixing rates with the exterior low; conditions considered necessary to maintain high concentrations of dinoflagellates within them. This agrees with reports that have found that increased levels of nutrients and vitamin B_{12} are important factors that trigger dinoflagellates blooms at Bahía Fosforescente (Burkholder and burkholder 1958, San Juan and González 2000). Furthermore, the diel distribution patterns driven by phototactic responses of these motile organisms may also contribute to their accumulation in the bays (Seliger et al. 1971).

STATUS OF BIOLUMINESCENT BAYS

In recent years, scientists, federal, state and private organizations, and the general public have expressed concerns in relation to the apparent decrease in bioluminescence at Bahía Fosforescente in Lajas. This decline and the associated reduction of *P. bahamense* abundance are accompanied by high densities of a non-bioluminescent dinoflagellate, *Ceratium furca var. hircus* (Walker 1997, Seixas 1988, Soler-Figueroa 2006) (Figure 21b and 22a). In contrast, at Puerto Mosquito in Vieques, *P. bahamense* is the predominant dinoflagellate throughout the year (Walker, 1997; Soler-Figueroa, 2006) with more than 100,000 cells L^{-1} (Soler-Figueroa, 2006; figure 22b).

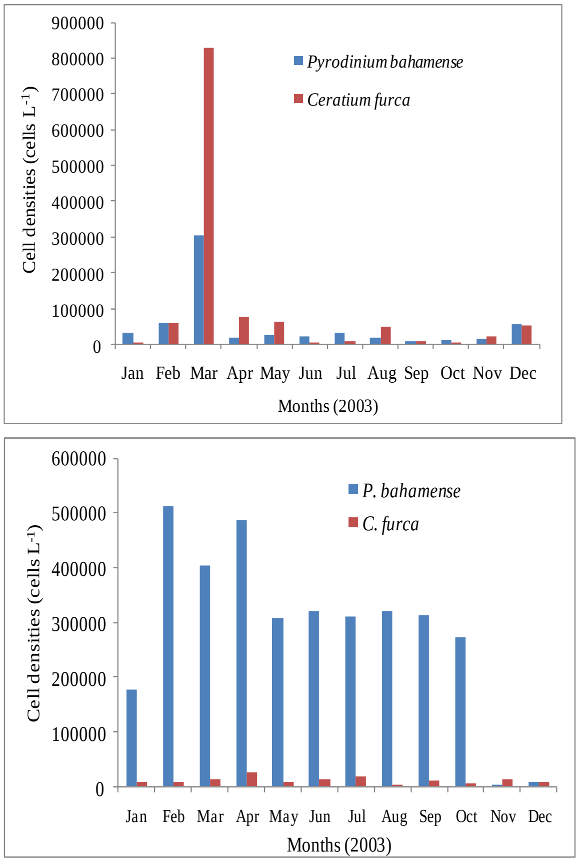


Figure 22 ***Pyrodinium bahamense*** **and** ***Ceratium furca*** **cell densities at a) Bahía Fosforescente in La Parguera, Puerto Rico and b) Puerto Mosquito in Vieques, Puerto Rico (Source: Soler-Figueroa, 2006).**

The apparent shift of dinoflagellate species dominance observed at Bahía Fosforescente could be considered an indicator of change of coastal ecosystem triggered by natural and/or anthropogenic causes. Increases in boat traffic and their associated pollution, changes in watershed management trends and in nutrient regimes have been quoted as potential stressors (Smart et al. 1985). However, research that focuses on the mechanisms at work is needed in order to understand the dynamics of dinoflagellate populations in relation to their fluctuating environment.

The lack of short-term time scale monitoring studies has been one of the major barriers in deciphering what mechanisms drive the fluctuations of phytoplankton species and bioluminescence, as most are monthly sampling efforts (Seixas, 1988; Walker, 1997; Soler-Figueroa, 2006). In addition, most of the studies have been focused on the study of few phytoplankton species (mainly *P. bahamense* and *C. furca*) and very limited information is available on the phytoplankton community structure and other co-occurring species. Considering that short term changes in the environment modulate the generation times of phytoplankton species within time scales of days to weeks (Smetacek and Cloern 2008), approaches that increase the resolution of phytoplankton sampling, assessment of taxonomic composition and monitoring of water quality are expected to yield vital information to develop a robust link between potential environmental change, whereas natural or anthropogenic, and the response of the planktonic community in this unique and economically important coastal system.

POTENTIAL OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON BIOLUMINESCENT BAYS

The dinoflagellates in bioluminescent systems are part of the phytoplankton community. Phytoplankton organisms are one of the main primary producers, providing half of the primary production in the ocean (Falkowski et al. 1998), and represent the base of the trophic food webs mediating the transfer of energy to higher trophic levels. Therefore, any changes in their abundances and distribution, will lead to changes in the trophic interactions of the ecosystem (Stenseth and Mysterud 2002, Winder and Schindler 2004). Also, phytoplankton organisms play a key role in regulating atmospheric carbon dioxide concentrations (CO_2) and are important components of the biogeochemical cycles.

Do to their small size, specific growth requirements, rapid nutrient uptake and growth rates, and vulnerability to grazing, phytoplankton are susceptible to environmental variability (Reynolds 1984). These organisms have been considered a sensitive index of speed and severity of global climate change (Widdicombe et al. 2010) and it has been proposed that their year-to-year variability is a response of disturbances from anthropogenic activity or shifts in the climate system (Cloern and Jassby 2010).

However difficult it may be to determining the impacts of global climate change on phytoplankton organisms, it should be a first priority. The responses of these organisms to changes in the environment can be observed as shifts in their physiological condition, reproduction and mortality, and community species composition (Miranda-Azeiteiro and Marques 2003). Therefore, the expected climate changes such as increases in temperature, enhanced surface stratification, alteration of ocean currents, stimulation of photosynthesis by elevated CO_2, increased heavy precipitation and storm events causing changes in land runoff and micronutrient availability may all produce intertwined responses at the individual, population and community levels (Hallegraeff et al. 2009). In general, the discrimination of anthropogenic and climate impacts or among their interacting effects have been difficult or uncertain. Nevertheless, changes in phytoplankton primary production and composition due to changes in climatic forces such as hurricanes and rainfall (Paerl 1995, Pearl et al. 2006, Adolf et al. 2006) have been reported.

The determination/predictions of the impacts of global climate change on bioluminescent systems and their phytoplankton community is complex. Not only there are few studies related to the phytoplankton community dynamics in these ecosystems, but also there is a lack of consistent long-term monitoring studies that would be essential to evaluate changes due to global climate trends (Hays et al. 2005, Moore et al. 2008, Marić et al. In press).

When considering the origin of bioluminescent bays, it is important to keep in mind that these systems resulted from the interaction of factors, such as, climatology, physiography, hydrology, geology, chemistry and finally biology over periods of decades to hundreds of years, if not millennia. Any alteration of the above, due to anthropogenic causes or climate changes, may trigger ecosystem shifts that may cause unexpected or even irreversible effects (Interior 1986).

CHANGES IN PRECIPITATION AND STORMS

The main potential stressors to bioluminescent systems due to climate changes are the increases in heavy precipitation, storms and hurricanes. The expectation of increases in heavy precipitation and storms, lead to increases on land runoff to these systems that can trigger increments in sediment and nutrient loadings affecting water quality due to changes in the sedimentation (turbidity), productivity and frequency and extension of salinity shifts. In the case of Bahía Fosforescente this problem might be compounded do to the decreased vegetation cover along the adjacent watershed (Gilbes-Santaella 2003); unpublished data).

One of the principal determinants of phytoplankton species composition in an ecosystem is the nutrient pool – including ratio, concentration and forms of available nutrients. It can be anticipated that nutrient alterations in the bioluminescent systems due to hurricanes, precipitation and land runoff will have a significant effect by altering its phytoplankton community diversity and composition (Smayda 1997, Buyukates and Roelke 2005, Spatharis et al. 2007). However, the predictions of changes in community composition are complex since the rate of nutrient assimilation by phytoplankton species is controlled by their nutritional preferences, uptake capabilities, and physiological or nutritional status (Anderson et al. 2002) and also depend on other climate related environmental factors, such as light and temperature.

A possible scenario for bioluminescent bays is that the high nutrient loading due to land runoff will lead to increases in diatom groups. Margalef (1978) states that the main phytoplankton life-forms can be sequence ranging from organisms dominating in turbulent, fertile environments (*i.e.* diatoms) to those found in stratified, nutrient-poor waters (typically dinoflagellates). Also, it has been reported, from microcosms experiments, that the growth of marine diatoms has been correlated with large and frequent additions of nitrate (Lomas and Glibert 2000) and high nutrient concentrations (Carter et al. 2005). However, others have concluded that blooms of dinoflagellates occur when humic substances and other nutritional factors enter coastal waters after heavy rainfall (Prakash and Rashid 1968). A scenario similar to the last was recently observed at Bahía Fosforescente, when high cell densities of *Pyrodinium bahamense* were observed one day after 97 mm of cumulative precipitation (Soler-Figueroa and Otero 2011); figure 23). Studies based on field observation and micro- and mesocosms experiments are needed to evaluate the links between weather patterns and the abundance, growth and phytoplankton community composition in bioluminescent bays.

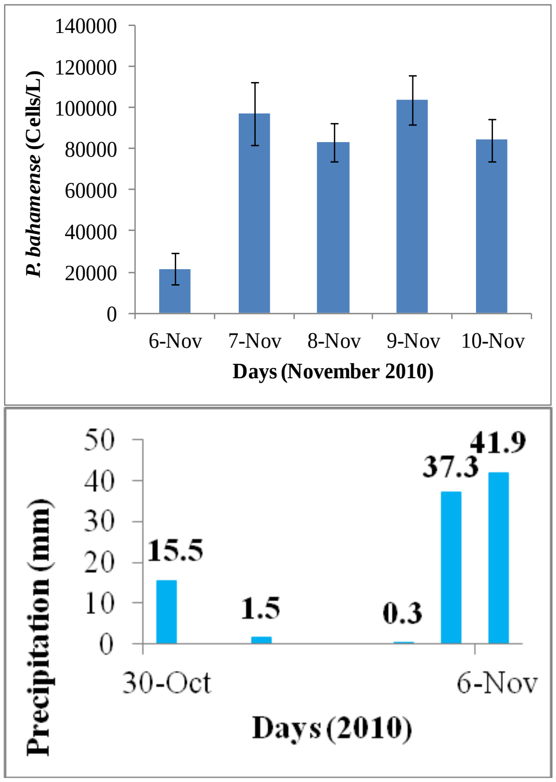


Figure 23 (top) *Pyrodinium bahamense* cell densities during five days on November, 2010. (bottom) Precipitation patterns six days previous the sampling schedule and two hours before the onset of the first sampling day.

INCREASED TEMPERATURE

Phytoplankton organisms can also be affected due to the warmer temperatures expected from global climate change. Temperature is considered one of the most important physical factors that affect the growth of phytoplankton species by regulating the metabolic processes of these organisms (Morris 1981). It has been proposed and observed that as temperature of the oceans gets warmer, small cell size phytoplankton dominate the community (Daufresne et al. 2009, Moran et al. 2010). However, others have reported that dinoflagellates populations (Lassen et al. 2010) and diatoms (Lionard et al. 2012) will be favored with increases in temperature. Moreover, all these studies have been conducted in the open ocean, and different results can be observed in coastal ecosystems such as bioluminescent bays.

Laboratory experiments are necessary in order to evaluate the impacts of increases in temperature over the phytoplankton community in bioluminescent systems. These experiments are essential to model distinct changes in ecophysiology of phytoplankton species (*i.e. Pyrodinium bahamense*) as conditions change. Growth rates and temperature tolerance under different temperature regimes are species specific. Therefore, different responses are expected to occur based on the different capabilities of phytoplankters to genetically adapt to a warmer ocean (Huertas et al. 2011).

RESEARCH AND INFORMATION NEEDS

Much research is urgently needed in order to understand the impacts of global climate change on the phytoplankton composition of bioluminescent systems in Puerto Rico, with special emphasis on *Pyrodinium bahamense*. Among the studies needed are:

1. Short-term studies in order to evaluate the response of phytoplankton organisms to different environmental and meteorological conditions. This type of study have been recently proposed to a limited extend (Puerto Rico Sea Grant, Project number: R-I 01-1-J2). Sites other than Phosphorescent Bay, La Parguera should also be studied to expand knowledge related to geographical difference among sites.
2. Long-term monitoring programs that document the phytoplankton community composition as well as the water quality of these systems. These studies are important in order to relate ecosystem alterations the due to climate change, bioluminescence and phytoplankton community shifts.
3. Controlled experimental studies to evaluate the effects of increasing nutrient concentrations on the phytoplankton community composition. Other highly-valuable experiment-based studies would also focus on the effects of increases in sedimentation and turbulence on the phytoplankton composition of bioluminescent systems.
4. Experiments based on cultures are basic in order to assess the impacts of different temperatures, salinities and nutrient concentrations to the growth rates and physiology of *Pyrodinium bahamense*. Also, the effects of pH and CO_2 should be evaluated.
5. Modeling efforts based on species and population interaction with the environment could serve as the basis for experimental verification of the status of bioluminescent bays. Either empirical or mechanistic models should integrate weather variables, chemical, physical and biological cues.
6. Finally, efforts that integrate research and collaboration among different disciplines and agencies will be crucial in order to protect, conserve and manage these unique ecosystems.

SUMMARY OF CLIMATE CHANGE IMPACTS ON BIOLUMINESCENT BAYS

Bioluminescent Bays are rare worldwide ecosystems. Their displays of light have intrigued people for thousands of years. In the world there are five widely recognized bioluminescent systems and three of them are in Puerto Rico: Bahía Fosforescente in Lajas, Laguna Grande in Fajardo and Puerto Mosquito in Vieques. There are also other systems in Puerto Rico where some bioluminescence has been observed (e.g. Monsio José, Lajas; Laguna Joyuda, Cabo Rojo; Bahía de San Juan, San Juan). In Puerto Rico the bioluminescence displayed is due to high concentrations of the bioluminescent dinoflagellate *Pyrodinium bahamense* var. *bahamense.* However, other bioluminescent dinoflagellates, such as *Ceratium fusus* and *Protoperidinium divergens*, have been also observed. In recent years, scientists, federal, state and private organizations, and the general public have expressed concerns in relation to the apparent decrease in bioluminescence at Bahía Fosforescente in Lajas. This decline and the associated reduction of *P. bahamense* abundance are accompanied by high densities of a non-bioluminescent dinoflagellate, *Ceratium furca var. hircus.* The apparent shift of dinoflagellate species dominance observed at Bahía Fosforescente could be considered an indicator of change of coastal ecosystem triggered by natural and/or anthropogenic causes. Increases in boat traffic and their associated pollution, changes in watershed management trends and in nutrient regimes have been quoted as potential stressor. However, research that focuses on the mechanisms at work is needed in order to understand the dynamics of dinoflagellate populations in relation to their fluctuating environment. The dinoflagellates in bioluminescent systems are part of the phytoplankton community. Phytoplankton organisms are one of the main primary producers, providing half of the primary production in the ocean, and represent the base of the trophic food webs mediating the transfer of energy to higher trophic levels (grazers of phytoplankton). Therefore, any changes in their abundances and distribution, will lead to changes in the food web of the bio bays. Do to their small size, specific growth requirements, rapid nutrient uptake and growth rates, and vulnerability to grazing, phytoplankton are susceptible to environmental variability. The responses of these organisms to changes in the environment can be observed as shifts in their physiological condition, reproduction and mortality, and community species composition. Therefore, the expected climate changes such as increases in temperature, enhanced surface stratification, alteration of ocean currents, stimulation of photosynthesis by elevated CO_2, increased heavy precipitation and storm events causing changes in land runoff and micronutrient availability may all produce intertwined responses at the individual, population and community levels.

Predicting the effects of global climate change on bioluminescent systems and their phytoplankton community is complex. Not only there are few studies related to the phytoplankton community dynamics in these ecosystems, but also there is a lack of consistent long-term monitoring studies that would be essential to evaluate changes due to global climate trends. However, when considering the origin of bioluminescent bays, it is important to keep in mind that these systems resulted from the interaction of factors, such as, climatology, physiography, hydrology, geology, chemistry and finally biology over periods of decades to hundreds of years, if not millennia. Any alteration of the above, due to anthropogenic causes or climate changes, may trigger ecosystem shifts that may cause unexpected or even irreversible effects.

The main potential stressors to bioluminescent systems due to climate changes are the increases in heavy precipitation, storms and hurricanes. The expectation of increases in heavy precipitation and storms, lead to increases on land runoff to these systems that can trigger increments in sediment and nutrient loadings affecting water quality due to changes in the sedimentation (turbidity), productivity and frequency and extension of salinity shifts. In the case of Bahía Fosforescente this problem might be compounded do to the decreased vegetation cover along the adjacent watershed. A possible scenario for bioluminescent bays is that the high nutrient loading due to land runoff will lead to increases in diatom groups and possible decreases in bioluminescence. Much research is urgently needed in order to understand the impacts of global climate change on the phytoplankton composition of bioluminescent systems in Puerto Rico, with special emphasis on *Pyrodinium bahamense*.

MARINE MAMMALS

INTRODUCTION

The marine mammal fauna of the Wider Caribbean Region (WCR) is diverse, and marine mammals have significant ecological, aesthetic and economic value to the countries and territories of the region. Regional success in managing and conserving marine mammals will ultimately depend on the countries' commitments to building capacity in order to implement a regional model by establishing conservation priorities, standards, and strategies for marine mammal conservation and education.

At least 32 species of marine mammals have been documented from the region—six species of baleen whales (Mysticeti), 24 species of toothed whales (Odontoceti), one sirenian (the West Indian manatee), and three pinnipeds (the Caribbean monk seal, the hooded seal, and the California sea lion). For many of these species, waters of the region serve as primary habitat for critical activities that include feeding, mating and calving. Although some species have been studied extensively elsewhere, data are scarce concerning the biology, life history, distribution and behavior of most cetacean (whale and dolphin) and manatee populations in the Caribbean Sea and Gulf of Mexico. The WCR is the one of only two regions in the world to have experienced the extinction of a marine mammal species (the Caribbean monk seal) in the past 250 years.

REGIONAL MARINE MAMMALS STRESSORS

FISHERIES INTERACTIONS

Marine mammals interact with active or abandoned fishing gear that could result in fatalities or accidents. This is something reported for all the Caribbean (Creswell, 2002; Romero et al., 2002). In some islands, like Barbados, when the incidental entanglement occurs the animal is not returned to the sea, but consumed or sold (Creswell, 2002). In Puerto Rico fisheries interactions with marine mammals have been reported for a variety of fishing gear. Humpback whales have been reported entangled with fish trap lines. Dolphin fatalities have been reported from gill net entanglement. Manatees have been caught accidentally on beach seine nets and fatalities have been reported due to infections caused by the abrasion and injures of monofilament line entanglement for the extremities.

POLLUTION AND MARINE MAMMALS HEALTH

Although evidence for links between chemical pollutants and the health of exposed marine mammals remains largely circumstantial, there is a growing concern that exposure to contaminants can increase susceptibility to disease and affect reproductive performance in marine mammals.

HUNTING

Whaling has been an activity occuring in the Caribbean since the 1700's. In some areas it has developed artisanally and in others as an industry. The end of the Barbadian whale fishery is similar to that of Grenada's, Trinidad's, and Bequia's (Creswell, 2002; Romero et al., 2002). Although the whaling industry at each place developed under different influences, used different equipment and methods, and operated in a different time period, each one was ultimately shut down due to overexploited whale populations (Creswell, 2002; Romero et al., 2002). At present Barbados is under pressure to join the International Whaling Commission in order to promote commercial whaling (Creswell, 2002). Puerto Rico has never engaged in whaling activity.

Manatees throughout the Caribbean have been greatly impacted by hunting activities. Trinidad and Tobago are the only islands of the Lesser Antilles where manatees are still observed (Romero et al., 2002). They are locally extinct in Grenada and Guadaloupe. In many countries of Central America the populations has declined drastically. In Puerto Rico hunting of manatees occurred until the mid-1990's.

VESSEL STRIKES

Marine mammals in the WCR are all susceptible to potential vesssel strikes that could be fatal or leave the animal injured. Cruises, sailboats, motor boats and jet skies have been identified as causing these incidents. Puerto Rico has a report of a humpback whale injured by sailboat strike. Puerto Rico's manatees have been sighted with scars from boat propellers. Fatalities have been reported for manatees due to motor boat and jetski strikes.

HABITAT DEGRADATION

Although no study has documented the impacts of habitat degradation on marine mammals, it is logical to assume that there is a direct impact on a variety of species. In the case of manatees, which are coastal, the effects are more evident. Sedimentation, caused by a variety of activities, smothers seagrasses, their main food source. Manatees also seek calm areas for resting grounds, particularly used for calving. As these areas are developed the properties of a resting ground are lost.

NOISE POLLUTION

Marine mammals are susceptible to noise pollution; beaked whales in particular. Military activities using naval sonar have been associated to mass mortalities of these whales. There is a report of one of these incidents for Puerto Rico. Manatees are also affected by noise pollution. The noise generated by boats, chains and dredging activities provoke manatees to leave preferred or secure habitat areas.

STATUS OF CARIBBEAN MARINE MAMMALS

The following is an excerpt taken from Reeves (2005) which provides a summary status of marine mammals in the Caribbean region by major groups:

PINNIPEDS

Althought the West Indian monk seal (Monachus tropicalis) is considered to be extinct, there is evidence that hooded seals (Cystophora cristata) occasionally wander from their Subarctic and Arctic range to occur as vagrants in the Wider Caribbean Region.

BALEEN WHALES

It appears, on present evidence that only the Bryde's whale depends on the WCR for food resources. It is still unknown if they migrate seasonally into and out of the region. Two other species are considered seasonal migrants, the humpback whale (Megaptera novaeangliae) and the minke whale (B. acutorostrata). They come during the winter for breeding, althought there is still incertainty if this is true for the minke. The other balaenopterids that have been recorded – blue, fin, and sei (B. musculus, B. physalus, and B. borealis) – are probably "just visiting," but the possibility cannot be ruled out that their visitations are more regular and frequent in unstudied or poorly studied parts of the region, or that the WCR is a significant portion of the normal range of one or more populations of these species. Similarly, the rare reports of North Atlantic right whales seem to suggest that they are, in some sense, "extralimital." However, in view of recent occurrences of right whales in the region, this conclusion deserves careful reconsideration.

TOOTHED CETACEANS

In terms of science and conservation, research on two species or species groups of cetaceans in the Wider Caribbean Region has particular prominence globally: the sperm whale (Physeter macrocephalus) and the beaked whales (Ziphiidae). Ongoing studies of sperm whales around Dominica by Hal Whitehead and his students from Dalhousie University promise to add important insights on the species' social structure and behavior, particularly when used in comparative analyses with their results from the eastern Pacific. Of more immediate applied importance is the diverse program of research on sperm whales in the northern Gulf of Mexico that includes satellite telemetry, photo-identification, monitoring of dive behavior and physiology, and controlled-exposure experimentation to assess the effects of underwater sound (e.g., Miller et al. 2004a, 2004b; Thode et al. 2002). This work's significance derives from global concern about the effects of acoustic disturbance on sperm (and other) whales, especially that from offshore oil and gas development.

Long-term studies of beaked whales have been undertaken in just two areas of the world

– The Gulf off Nova Scotia, Canada (northern bottlenose whale, *Hyperoodon ampullatus*), and the Bahamas (Blainville's beaked whale, *Mesoplodon densirostris*). In the latter area, the work of Claridge and Balcomb (2001) was key in documenting the vulnerability of beaked whales exposed to military sonar under particular circumstances. Given the general rarity of opportunities for observational studies of beaked whales, and the high profile of the controversy surrounding the effects of high-intensity underwater sound on this group of whales, it is important that research effort in the Bahamas be maintained and, if possible, expanded.

Another point worth highlighting, as noted by Ward et al. (2001), concerns the habitat requirements of beaked whales. Contrary to the usual descriptions of various species' distributions as "cosmopolitan" or "worldwide" in deep offshore waters, it appears that their distributions are considerably more specialized and thus limited. The Bahamas studies provided the first good evidence of this for the mesoplodonts. Blainville's beaked whales generally occur only in depths between 200 -1,000 m and near areas with particular bottom topography. It should be noted in this context that similar analyses of habitat preferences of other odontocetes have been conducted in the northern Gulf of Mexico (e.g., Baumgartner et al. 2001). Thus, the Wider Caribbean Region has become a focal region of research on several species, including the beaked whales, the sperm whale, and several tropical to warm temperate delphinids.

A 2004 workshop (Reeves et al. 2004) ranked taxonomic uncertainty as high for killer whales (*Orcinus* sp.) and called for a global inventory of specimen material for that genus. It seems clear from the records summarized by Ward et al. (2001) that at least some of the killer whales in the WCR are mammal and sea turtle predators, but in the absence of rigorous analyses of morphological, dietary, and other types of data from the region, it remains impossible to judge whether more than one ecotype (and thus possibly eventually subspecies or species) is present. A regional inventory of sighting, stranding, and take records, perhaps followed by a directed effort to obtain and analyze biological materials from killer whales in specific parts of the Wider Caribbean Region (via biopsy), could be a significant contribution to ongoing global analyses. The same can be said of Blainville's beaked whales and Cuvier's beaked whales (*Ziphius cavirostris*), both of which the workshop described as "warranting special attention to resolve taxonomic uncertainty". Their vulnerability to underwater noise as well as entanglement in drift gillnets was noted, as was the fact that Dalebout (2002) found strong evidence of population structure in Cuvier's beaked whale, suggesting that there could be isolated regional populations. The apparently widespread and regular occurrence in the Wider Caribbean Region makes Cuvier's beaked whale a candidate for focused effort along the same lines as mentioned above for killer whales.

The subject of population structure of odontocetes within the Wider Caribbean Region has hardly begun to be addressed, but judging by findings elsewhere, considerable structure is likely present (e.g., island-associated near-shore populations and farther-ranging offshore populations). Given the long history of exploitation of some near-shore species in some parts of the WCR (Caldwell and Caldwell 1975; Price 1985; Reeves 2002), this could mean that a number of populations have already been substantially depleted or even entirely eliminated. In the absence of even rudimentary survey data for most of the region (except U.S. waters of the northern Gulf of Mexico), uncertainty concerning population structure, abundance, and conservation status is the norm.

OUTCOMES AND CONSEQUENCES OF CLIMATE CHANGE ON MARINE MAMMALS

For the most part the literature, regarding climate change and marine mammals, focuses on marine mammals inhabiting the polar regions and indicate that arctic marine mammals are more likely to be affected (Tynan and DeMaster 1997, IPCC, 2001, Harwood, 2001, Clarke and Harris 2003). However, other literature indicates that the affect of climate change on the marine environment has the potential to have, and in some cases has already had, a considerable impact on marine ecosystems and species (Defra 2005). It is often difficult to judge the response of marine mammals to long-term environmental change because of the mismatch in scale between scientific research (months to decades, in localized areas) and marine mammal natural history (decades to centuries, often over large areas). The direct effects of climate change on marine mammals are from increasing temperatures and habitat degradation. They include changes in abundance, distribution, timing and range of migration, community structure, the presence and species composition of competitors and predators, prey availability and distribution, timing of breeding, reproductive success and, ultimately, survival (Defra, 2005). The following provides some information on climate change outcomes as they relate to the Caribbean region.

POPULATION ABUNDANCE AND DISTRIBUTION

Literature in this area is scarce; however, in general, information indicates that migratory marine species, by traveling large distances and being subject to a wide range of environmental influences, are particularly likely to be affected by climate change at some point of their life cycles (CANARI 2008). While some species may increase in abundance or range, climate change will increase existing risk of extinction of some more vulnerable species (Defra, 2005).

FOOD SUPPLY

One of the greatest threats to marine mammals probably comes from changes in their food resources, as a result of climate change. Many prey species such as fish, cephalopods and plankton appear to rely on, and are influenced by, particular sets of environmental conditions (Harwood, 2001). Any changes in the geographic distribution of these oceanographic conditions as a result of climate change will affect the abundance and distribution of prey species (Defra, 2005). Notably, apart from species residing within the region, it appears that other long ranging migratory species may depend on the region for food supply. Reeves (2000) states that at least one species of baleen whale depends on the wider Caribbean region for food resources, namely the common Bryde's whale, *Balaenoptera* cf. *brydei*; however, whether it migrates seasonally into and out of the region remains to be determined (CANARI 2008).

DISEASES

Changes in disease prevalence and, in turn, increased host morbidity or mortality may result from increases in host susceptibility, pathogenicity of the disease agent, or transmission of pathogens to hosts. A number of discrete or interacting factors can influence each of these processes, such as: nutrition, population density, host movement and distribution, host genetics, predation and fisheries bycatch, increased urbanization of the coast, contaminants, and stress (Reynolds III 2005). While information is scare on specifics to how climate change affects marine mammal diseases it can be assumed that as climate change affects these processes, so will disease prevalence. In particular marine mammal stress from adverse oceanic conditions can challenge homeostatic processes.The subtle physiological changes involved in the stress response of marine mammals are still poorly understood but are likely to decrease the immune response through a number of endocrine changes and direct effects on the lymphoid system (Reynolds III 2005). For instance, Wilson et al. (1999) found that epidermal lesions in bottlenose dolphins were more common in animals particularly stressed by low water temperatures and salinity.

Information on diseases and marine mammals in the Caribbean is limited; however, information pertaining to other regions suggests that there is an increase in diseases affecting marine mammals and that this may be, in part, due to temperature changes (Marine Mammal Commission, 2005).

CONTAMINANTS AND METABOLISM (WELLS, 2010)

High summer metabolic rates and mortality rates suggest current thermal challenges, as water temperature approaches body temperature; this situation may be exacerbated through climate change. Thermal stresses may combine with toxicological stresses to increase mortality under warm water conditions. Lipids released from thinning blubber as waters warm can transport associated toxic environmental contaminants (e.g., PCBs, DDT and metabolites) to target organs or to organs where biotransformation can modify toxicity, leading to compromised immune function. Warmer waters are likely to support a variety of old and new pathogens, reduce dolphin host resistance, and/or increase the duration of exposure. Transfer of contaminants via lactation has been suggested as one cause of the increased mortality documented for first-born calves in the area. Taken together, these factors suggest that seasonal warming appears to lead to health challenges for Sarasota Bay bottlenose dolphins, perhaps approaching a tipping point, potentially leading to cascading declines in individual health. Information is needed to identify and detect signals of climate change, predict where and how impacts on marine mammals and their habitat are likely to occur, and prioritize pre-emptive management actions for providing these animals with as much capacity as possible to respond to climate change.

KNOWLEDGE GAPS AND RESEARCH NEEDS

The cumulative effects of climate change stressors and multiple anthropogenic sources of environmental change on marine mammals are unknown, but each examined separately suggests that these animals must contend with an increasingly stressed ecosystem and will affect marine mammals negatively. More research is needed to determine to what extent. Addressing marine mammal responses to climate change requires research focused on predictive modelling of ecosystems, based on data from a suite of species selected for their life history characteristics. The current research that exists focuses almost exclusively on responses of marine mammals to non-climate change stressors, like boat strikes, noise pollution, etc. Information on long-term environmental changes, such as regime shifts and ENSO, are not well known and could help with understanding potential outcomes of climate change. Also, our understanding of diseases in marine mammals is poor compared to our knowledge of such processes in terrestrial animals. As a result, we do not know which diseases are normal components of a healthy marine ecosystem and which are novel and a consequence of anthropogenic factors. We also do not know if there is a real increase in diseases in marine mammals or how to control/prevent these diseases. Finally, there is an inability to properly assess effects of contaminants on marine mammals. The main reasons are the general lack of understanding of marine mammal health and physiology, the hiostoric overreliance on extrapolation from unrelated species and unrelated tissues for toxicological assessments, and the logistical difficulty and hestitation to perform studies using capture-release and captive animals to relate contaminant exposure to effects. Furthermore, monitoring is required to determine whether management changes (such as restrictions to chemical use, changes in oil consumption or shipping, changes in disposal processes) result in decreased, unchanged, or increased burdens in tissues. Nevertheless this monitoring and studies related to effects of contaminants on marine mammals is critical for management and conservation. In summary, if we are to measure responses to climate change we must understand marine mammal life histories, contaminant exposure, how to predict and prevent diseases, and how marine mmals respond to long-term environmental changes.

SUMMARY OF OUTCOME AND CONSEQUENCES OF CLIMATE CHANGE ON MARINE MAMMALS

Marine mammals have significant ecological, aesthetic and economic value to the countries and territories of the wider Caribbean region, including Puerto Rico. At least 32 species of marine mammals have been documented from the region—six species of baleen whales (Mysticeti), 24 species of toothed whales (Odontoceti), one sirenian (the West Indian manatee), and three pinnipeds (the Caribbean monk seal, the hooded seal, and the California sea lion). For many of these species, waters of the region serve as primary habitat for critical activities that include feeding, mating and calving. Although some species have been studied extensively elsewhere, data are scarce concerning the biology, life history, distribution and behavior of most cetacean (whale and dolphin) and manatee populations in the Caribbean Sea and Gulf of Mexico. The Caribbean is the one of only two regions in the world to have experienced the extinction of a marine mammal species (the Caribbean monk seal) in the past 250 years.

Marine mammals suffer from many threats. In Puerto Rico fisheries interactions with marine mammals have been reported for a variety of fishing gear. Humpback whales have been reported entangled with fish trap lines. Dolphin fatalities have been reported from gill net entanglement. Manatees have been caught accidentally on beach seine nets and fatalities have been reported due to infections caused by the abrasion and injures of monofilament line entanglement for the extremities. Pollution, hunting, vessel strikes, habitat degradation, and noise pollution also plague marine mammals around Puerto Rico.

Population abundance and distribution are expected to be affected by climate changes, specifically from increasing sea surface temperatures and ocean acidification, however more research is needed to determine which species ranges may expand or contract and how. Marine mammal food supplies (*i.e.*, fish, cephalopods like squid, and plankton) are directly affected by environmental conditions and as their food supplies are affected by climate change so too will marine mammals. It is anticipated that there will be more reports of disease in Puerto Rico's marine mammals as increasing sea surface temperatures increase the prevalence of disease. High summer metabolic rates and mortality rates suggest current thermal challenges, as water temperature approaches body temperature; this situation may be exacerbated in the future. Thermal stresses may combine with toxicological stresses to increase mortality under warm water conditions. Lipids released from thinning blubber as waters warm can transport associated toxic environmental contaminants (e.g., PCBs, DDT and metabolites) to target organs or to organs where biotransformation can modify toxicity, leading to compromised immune function. Warmer waters are likely to support a variety of old and new pathogens, reduce dolphin host resistance, and/or increase the duration of exposure. Transfer of contaminants via lactation has been suggested as one cause of the increased mortality documented for first-born calves in the area. Taken together, these factors suggest that seasonal warming appears to lead to health challenges. Information is needed to identify and detect signals of climate change, predict where and how impacts on marine mammals and their habitat are likely to occur, and prioritize pre-emptive management actions for providing these animals with as much capacity as possible to respond to climate change.

INTERSECTING ISSUES

In November 2010 the PRCCC founding members stressed the importance of also assessing the intersections between certain ecosystems, species, and key anticipated climate change outcomes and consequences. Changes in climate stressors will not affect Puerto Rico's ecosystems or species independently. Many systems are so connected that what happens in one may alter the structure and function of another. A good understanding of these interconnections can assist in planning our urban, agricultural, rural and conservation areas appropriately to be able to adapt to a changing climate. Due to time limitations, however, a formal assessment of these intersecting issues was not possible. Throughout the report the PRCCC attempted to include many such issues, however, for example the coupled nature of mangroves, seagrasses, and coral reefs, the dependence of commercially-important finfish on nursery habitats, or the interconnectedness of mountain and agricultural areas to coastal systems. Future PRCCC reports should prioritize the assessment of these types of intersections.

Looking at the Caribbean Natural Resources Institute 2008 report (CANARI 2008) on climate change and coastal and marine ecosystems, we can expand on the first theme. The high biodiversity found in Caribbean coral reefs is strongly influenced by the presence of adjacent mangrove forests and seagrass beds. These three ecosystems form strongly coupled habitat complexes, which are not completely understood along the coastal seascape (Koch & Madden, 2001; McKee *et al.,* 2002; Mumby, 2006). There is a continuum across these ecosystems in which complex nutrients exchanges define the spatial and temporal distribution of mangroves, seagrasses and coral reefs and negative impacts in one ecosystem can cascade across the coastal seascape, affecting other areas.

Some of the interactions across the seascape are as follows:

- Nurseries: Mangroves and seagrass beds are considered important juvenile habitats for a variety of fish and invertebrate species that spend their adult life on coral reefs or offshore habitats (Ogden *et al.,* 2005);

- Foraging movements and migration: Diurnal and nocturnal feeding migrations among habitats are a common feature of juvenile and some adult fish (Nagelkerken *et al.,* 2000a, 2000b; Beets *et al.,* 2003). As a result of these migrations, fish can function as vectors of organic material from seagrass beds to reefs, enhancing the growth rates of corals (Meyer *et al.,* 1983);

- Physical interactions: Healthy coral reefs act like hydrodynamic barriers dissipating wave energy and creating low energy environments conducive to mangrove and seagrass colonization while at the land-sea boundary, coastal forests, mangroves and seagrasses act as buffers, which intercept freshwater discharge, stabilize salinity, and trap and bind sediment (Ogden *et al.,* 2005);

- Exchange of particulate and dissolved organic matter: Mass balance studies in a range of mangrove systems generally support the assertion that mangroves export organic matter in both particulate and dissolved forms (Lee, 1995; Robertson & Alongi, 1995).

The buffering capacity of coastal ecosystems is threatened by the projected rate of sea level rise under scenarios of global warming (Ogden *et al.,* 2005). While healthy coral growth may keep pace with sea level rise, weakened reefs may be unable to grow sufficiently to enable them to continue their coastal protection function. These zones will become inundated and subjected to erosion by progressively larger waves. Seagrass and mangrove communities will be eroded and will become less effective buffers, releasing nutrients and sediment and further slowing coral reef growth rate and negatively impacting coral reef health.

CONCLUSION

The Working Group 2 report highlights that the Puerto Rico Climate Change Council members and organizations are increasingly engaging in research, monitoring and observing initiatives to increase our understanding of climate change effects and support effective, informed natural resources management. This first-pass assessment lays the groundwork for future PRCCC reports to build upon and identifies areas where more research and expertise is needed.

The main effect of climate change on Puerto Rico's ecosystems and species will be synergistic in that already stressed systems will be exposed to additional stressors that push them over their limit of existence, resulting in widespread loss of habitat, unfavorable changes to structure and function, or diminished services to Puerto Rico's society. Some ecosystems and species will acclimate to changing environmental conditions better than others. While some species may potentially migrate to more favourable conditions in Puerto Rico (e.g., Colorado trees, swamp cyrilla) species already bumping up against the upper limits of their range may not be so fortunate, and could be diminished or lost from Puerto Rico altogether due to the lack of suitable environmental conditions (e.g., dolphinfish, yellowfin tuna). Others may not have the ability to relocate and may become globally extinct, like the Coquí Duende, the Cricket Coqui, and the forest-dwelling Puerto Rican Upland Sphaero. On the other hand, new species or community assemblages could occur in Puerto Rico that may benefit society as they might provide new ecosystem services. More research is needed on potential future ecosystem states and the ability of future ecosystems to sustain human populations. Healthy ecosystems are our life support system, providing us with essential goods and services that would be extremely expensive or impossible to replace. Ecosystems purify air and water, and provide flood control. They supply us with marketable products and sequester carbon and build soils. They provide recreation, hunting and fishing, and wild places in which to enjoy nature for our spiritual and mental well-being. Puerto Rico's culture is greatly connected to the islands' ecosystems and species, perhaps to a greater extent than many other areas of the world. Human disruption of ecosystems, through climate change and other factors such as habitat destruction and pollution, can reduce ecosystems' ability to provide us with these valuable services.

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