Chapter 7
Mangrove Ecosystems under Climate Change

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7.1 Introduction

From fossil records, evidence suggests that mangroves first started colonizing the intertidal zone around the Tethys Sea in the Upper Cretaceous (Ellison et al. 1999). Since then, they have formed an interface between land and sea in most tropical and sub-tropical regions worldwide, and are now expanding into temperate regions of multiple continents (Saintilan et al. 2014). Environmental impacts on mangrove plants and sediments make them susceptible to climate change outcomes, including sea level rise (SLR), temperature shifts, and alterations of atmospheric gas composition and moisture over long temporal scales. All of these changes, as well as anthropogenic impacts from land use and water resources in the watershed are likely to significantly affect mangrove distribution and performance directly during this century. The consequences for mangrove ecosystems, and the ecosystem services that they provide, are likely to vary on a local to regional scale (Lee et al. 2014). Effective

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V.H. Rivera-Monroy et al. (eds.), *Mangrove Ecosystems: A Global Biogeographic Perspective*, https://doi.org/10.1007/978-3-319-62206-4_7

conservation and management therefore requires a local to regional scale assessment and understanding of mangrove responses to climate change.

Hazards to mangroves not only arise from climate change, but also from human interventions like, for example, land use change, urbanization, alterations to river catchment hydrology, overexploitation of natural resources, and coastal construction. Moreover, since the fourth IPCC (Intergovernmental Panel on Climate Change) report (Solomon et al. 2007), "Anthropocene" climate change and warming of the earth, i.e., during the past approximately 250 years and in particular, since the 1950s, is suspected to be largely related to human interventions, i.e., the burning of fossil fuels, land use change, and other human-induced changes in the atmosphere's composition. Mangroves have an adaptive capability as demonstrated by the fact that they have survived harsh environmental conditions and climate and sea level changes for millions of years, but at the same time, they now face rates of environmental change that are unprecedented in their history.

In recent decades, it has become clear that mangroves and their connectivity to adjacent terrestrial and marine ecosystems provide important ecological functions and ecosystem services (e.g., Ewel et al. 1998; Mumby et al. 2004; Wells et al. 2006). Moreover, in recent years, political and societal awareness of the importance of ecosystem services has increased, mainly due to the activities of global institutions such as the IPCC and the Millennium Ecosystem Assessment (2005). Political decisions in an economy-driven world hardly consider the value of a resource or service that is not directly marketable. Therefore, attempts have been made to put a value on ecosystem services (ES) from coastal wetlands. For example, one such study resulted in a value of 10,000 USD ha^{-1} $year^{-1}$ for mangroves (Costanza et al. 1997). However, giving these ES a direct economic value is difficult and remains debatable.

This chapter will first identify the possible outcomes of climate change that are likely to affect mangrove ecosystems, then report how mangrove forests respond to these outcomes, and from that define the most vulnerable regions. The chapter will also discuss the interaction of climate change with human interventions, the effects on ecosystem services, and adaptation and management options. It will close with an outlook on knowledge gaps and priority research needed to fill these gaps.

7.2 Climate Change Effects

There are well-known direct effects of climate change, which have the potential to alter the structure, functions, and ES of mangroves. However, these effects additionally alter the physical, chemical, biological, and geomorphological setting of the mangrove environment. These, in turn, can indirectly alter the structure, functions, and ES of mangroves through an altered exchange with neighboring ecosystems, including effects of multiple climate change stressors and other anthropogenic and natural stressors on mangrove systems (Table 7.1). There are also potential

Table 7.1 Direct and indirect effects of climate change and direct human effects on mangroves

Direct effects of climate change	Indirect effects of climate change	Direct human impacts
• Sea level rise • Warming of surface waters • Warming of atmosphere • Changing atmospheric moisture transport and precipitation • Changing atmospheric gas composition (higher CO_2)	• Changing surface ocean circulation affecting tidal exchange and geospatial dispersal of mangrove propagules • Changing salinity gradients affecting tidal exchange • Surface water acidification • Changing freshwater inflow • Changing allochthonous sediment input • Changes in extreme weather events • Increased frequency of extreme high water events • Changes in seasonality • Degradation of ecosystems that are functionally linked to mangroves	• Clearing of mangroves • Changing hydrology and tidal flushing by roads and levees, and dredging navigation channels • Changing freshwater inflows by river damming and diversion • Changing riverine sediment inflows by increased erosion and river damming • Pollution (e.g., nutrients, garbage, sewage, dredge spoil, oil spills) • Overharvesting of forest and fisheries • Subsidence due to extraction of water, petroleum, and gas

feedback mechanisms of climate change outcomes that may affect mangrove resistance and resilience (e.g., Cahoon et al. 2006; Krauss et al. 2014a).

7.2.1 Direct Effects

Sea level rise is a prominent direct effect of climatic warming on mangrove ecosystems. However, what is important is not the absolute sea level change, but rather the relative sea level change that mangroves experience locally. The concept of relative SLR has been described in various ways by different scientific disciplines (see Cahoon 2015), but for mangrove vulnerability assessment, relative SLR must account for a number of regional and local factors (e.g., impoundment, harvesting effects on subsidence) not all of which are related directly to climate change (Webb et al. 2013; Lang'at et al. 2014; Wolanski and Elliott 2015; Kiwango et al. 2015). For example, in some regions and settings, subsurface processes, such as mangrove root production and decomposition, can be a primary control of mangrove sediment surface elevation and hence, local-scale trends in relative sea level, within an individual mangrove site (Cahoon et al. 2006; McKee et al. 2007; Krauss et al. 2014a). One significant external factor of regional significance is the glacial rebound that presently affects mangroves differently in various areas of the world. Morphology and relief (i.e., a mangrove site's physiographic setting) certainly play an important role for the survival of mangroves facing relative SLR (Krauss et al. 2014a). Another significant factor is the tidal regime deformation in shallow waters facing large river deltas such as that of the Mekong, which can increase the tidal range and hence tidal flooding (Nhan 2016).

The global mean sea level rose by 3.2 ± 0.4 mm $year^{-1}$ in the past two decades, but with large regional variations (Wolanski and Elliott 2015). For example, sea level fell by 1–2 mm $year^{-1}$ along the western coast of North America and northern coast of South America, but it rose by 5–20 mm $year^{-1}$ in the Southeast Asia/Western Pacific region (Nicholls and Cazenave 2010). A recent IPCC report projects sea level to rise 26–55 cm (average 40 cm) by 2081–2100 for the “best case” scenario RCP2.6 (RCP = representative concentration pathways) and between 45 and 82 cm (average 63 cm) for the “worst case” scenario RCP8.5 (relative to 1986–2005). This rise will not be uniform across regions and approximately 70% of the global coastline will experience a change of ±20% of the global mean (IPCC 2014).

Another important factor directly affecting mangrove wetlands is the rise in air temperature and sea surface temperature. The projected increase of both can vary largely according to the four IPCC scenarios (RCP2.6, RCP4.5, RCP6.0, and RCP8.5) and it differs between land and ocean. The increase in sea surface temperature (SST) will likely be lowest as the ratio of land to ocean warming is 1.4–1.7 (Collins et al. 2013). Moreover, the temperature increase is not spatially uniform (Fig. 7.1); thus the impact on mangroves worldwide will vary according to location. The degree to which changing SSTs affect extreme winter air temperatures may be most important for determining whether mangroves will continue to expand into temperate zones, and supplant salt marsh. In the second half of the twentieth century, the SST increase was generally more pronounced in higher latitudes and in the northern hemisphere. For surface air temperature, the general trend was similar, but with a few low latitude regions exhibiting temperature increases in the same magnitude as in polar regions. In particular, arid to semiarid areas in eastern South America and western North Africa experienced an increase of 1.5–2.5 °C (Hartmann et al. 2013). According to IPCC projections, the increase in SST will be larger in the low latitudes in the coming two decades before it becomes larger in the high latitudes at the end of the twenty-first century. Surface air temperature will generally increase more rapidly and the increase will be larger in high latitude regions, in particular in the northern hemisphere (Collins et al. 2013; Kirtman et al. 2013).

Changes will also occur in atmospheric moisture content and transport and hence in precipitation. Again, this change will not be uniform across the globe. With respect to mangrove distribution, the most relevant projected changes are increases in precipitation in the equatorial Pacific, East Africa and the Middle East, and India. While eastern South America and western Africa will probably become drier, changes in Australasia and Southeast Asia are not that clear (Christensen et al. 2013; Hartmann et al. 2013). As a result of changes in precipitation and evapotranspiration, the runoff will change. The largest increase in runoff is projected for the Southeast Asia/West Pacific region while the strongest decrease is to be expected for Central America and eastern South America (Collins et al. 2013; Fig. 7.2).

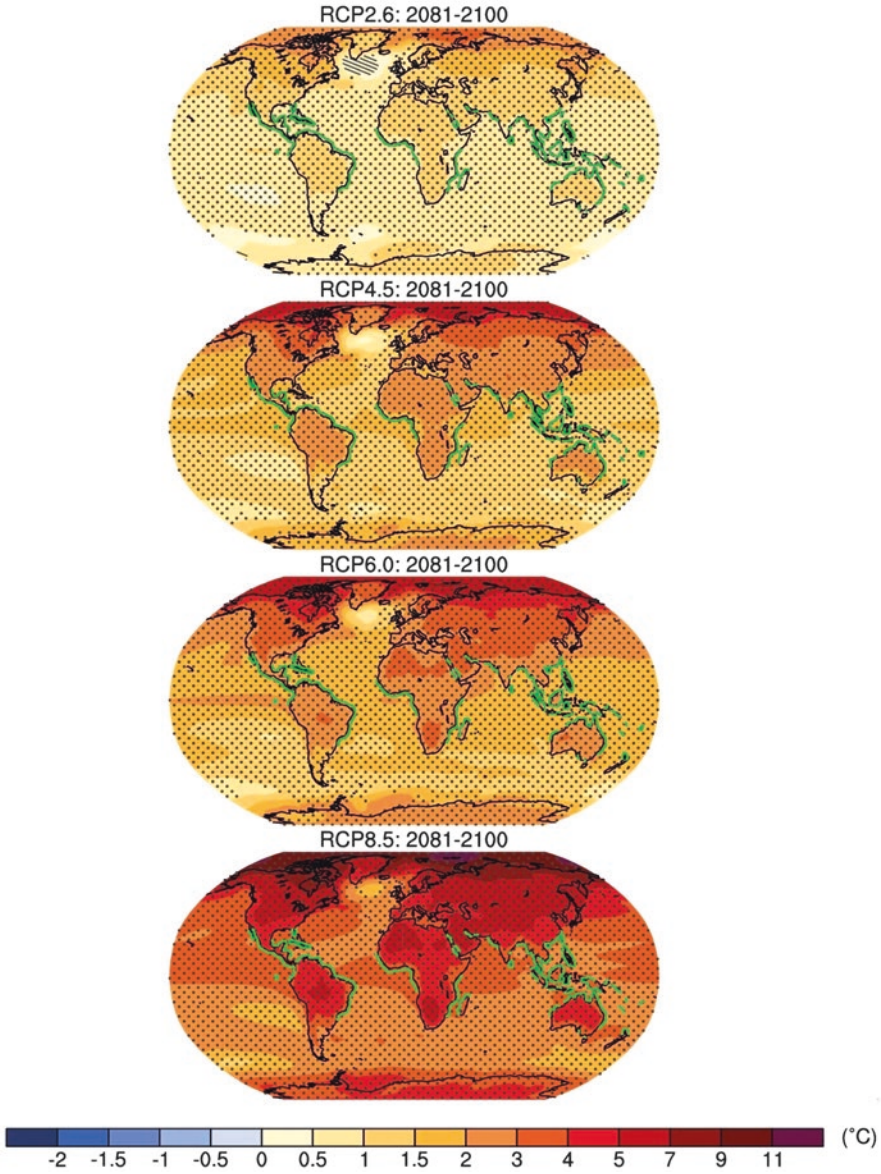

Fig. 7.1 IPCC projections for 2081–2100 of surface temperature changes for four emission scenarios (*RCP* representative concentration pathways; RCP2.6, RCP4.5, RCP6.0, RCP8.5) and the present-day mangrove distribution (*green lines*) according to Giri et al. (2011). Figure modified from Collins et al. (2013)

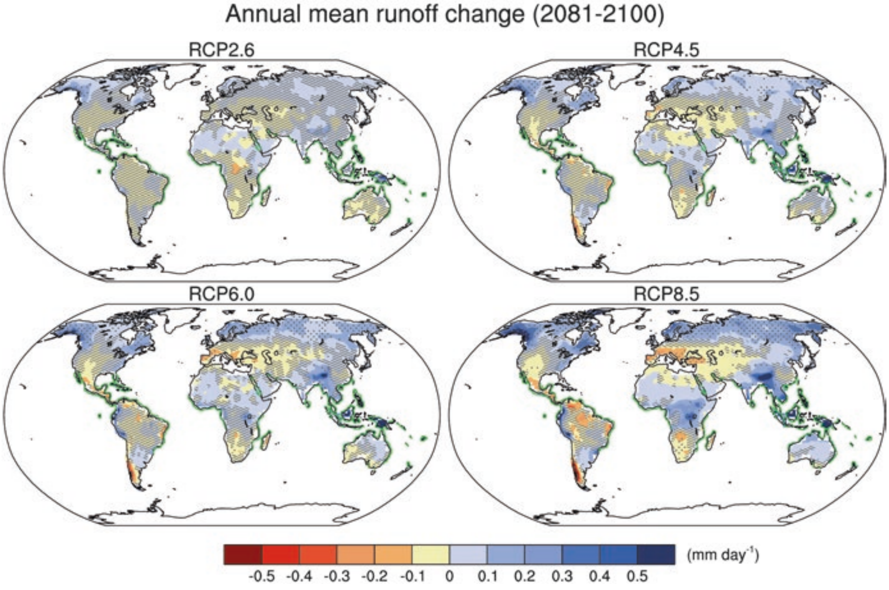

Fig. 7.2 IPCC projections for 2081–2100 of annual mean runoff changes for four emission scenarios (RCP2.6, RCP4.5, RCP6.0, RCP8.5) and the present-day mangrove distribution (*green lines*) according to Giri et al. (2011). Figure modified from Collins et al. (2013)

Last, the atmospheric concentration of greenhouse gases will increase and the increasing CO_2 can directly affect mangrove productivity (Ball et al. 1997; McKee and Rooth 2008). The IPCC projections foresee an increase of atmospheric CO_2 concentration up to 450 ppm in the "best case" scenario (RCP2.6) and of >1000 ppm in the "worst case" (RCP8.5) until 2100 (IPCC 2014).

7.2.2 *Indirect Effects*

Climate change will also indirectly affect mangrove wetlands in several ways (Table 7.1). Changes in the global heat budget and the hydrological cycle can alter the circulation in the coastal ocean, which may alter mangrove propagule dispersal (Duke et al. 1998; Chap. 2) and impact the flushing rate of estuaries and their fringing mangroves. For instance, warming and changes in rainfall and evaporation possibly increase the formation of an estuarine and coastal salinity barrier that prevents the exchange of mangrove water with the coastal ocean. This may result in stagnation that can last several months and formation of hypersaline waters that are stressful to mangroves (Wolanski et al. 1992; Andutta et al. 2011). In places where rainfall increases, mangroves may expand, such as in Moreton Bay, Australia (Dale et al. 2013). In general, changes in precipitation and evaporation will affect the water budget of estuaries and thus, the salinity of their fringing wetlands, including mangroves (Robins et al. 2014). Such changes in salinity are expected to modify the ecology and mangrove species distribution (Osland et al. 2014b).

The expected changes in storminess and coastal hydrodynamics will be crucial for mangroves. An increase in frequency and intensity of tropical storms has been observed in some areas (Knutson et al. 2010), such as the tropical North Atlantic for the period 1983–2005 (Kossin et al. 2007). Identifying general trends is hampered by the scarcity and inconsistency of data. Despite the uncertainties in projections of future changes until 2100, it is likely that the intensity of tropical storms will increase in the North Atlantic and western North Pacific, which already are the regions with highest tropical storm activity (Christensen et al. 2013).

Increasing extreme weather events will lead to physical damage that will be exacerbated if the mangroves are weakened by the formation of hypersaline waters or pollution. These weather events may defoliate the trees, make them more susceptible to attack by wood borers, and erode the soils. Although younger trees may be able to refoliate more rapidly, the process may still take several decades (Salmo et al. 2013). In addition, mangrove sediment elevation can be altered through soil erosion, soil deposition, peat collapse, and soil compression (e.g., Cahoon 2006; Piou et al. 2006; Aung et al. 2013). The frequency of extreme high water events is projected to increase over coming decades. It may affect the position and health of mangroves and associated coastal ecosystems by altering salinity, recruitment, and inundation, in addition to changing the wetland sediment budget. This is due to the same atmospheric and oceanic factors causing global sea level to rise, and possibly additional issues such as variations in regional climate and change in storminess and resulting storm surges (Woodworth and Blackman 2004).

Ocean acidification, a prominent outcome of climate change (e.g., Hoegh-Guldberg and Bruno 2010), will also reach mangrove ecosystems through estuarine flushing and will most likely affect their flora and fauna as well as biogeochemical cycles, although there are no published studies yet to quantify this impact. Mangrove forests are functionally linked to neighboring coastal ecosystems, including seagrass beds, coral reefs, and terrestrial coastal habitats, but the functional links are not fully understood (e.g., Mumby et al. 2004). Coral reefs, seagrass beds, estuaries, beaches, and coastal upland ecosystems may experience reduced area and health from climate change outcomes (e.g., Hoegh-Guldberg and Bruno 2010). For example, one indirect impact of increased temperature and CO_2 on mangroves is the degradation of adjacent coral reefs caused by mass bleaching and impaired growth (Hoegh-Guldberg 1999). The damage to coral reefs reduces their capability to protect mangroves from wave action which, in turn, makes them prone to erosion. On the other hand, mangrove forests may provide a non-reef coral refuge from thermal stress and ocean acidification. On St. John, Virgin Islands, shaded scleractinian corals that grow attached to and under mangrove prop roots do not show signs of bleaching in contrast to unshaded colonies. In addition, it was concluded that the combination of substrate and habitat heterogeneity, hydrographic conditions, and chemical water conditions acts as a buffer against ocean acidification (Yates et al. 2014).

7.3 Response to Climate Change

The response of mangroves to climate change effects will be manifold (e.g., Alongi 2015). The most important expected responses are shifts in (i) distribution (latitudinal and keeping pace with SLR), diversity, and community composition, (ii) physiological processes of flora and fauna, (iii) water budget, (iv) productivity and respiration, (v) carbon storage in biomass and sediments, and (vi) filter function for elements beneficial or harmful to life. These responses will likely vary from region to region, at least between the two large biogeographical regions, the Atlantic East Pacific (AEP) and the Indo West Pacific (IWP), which display profound differences in the diversity of mangrove trees and key faunal species (Tomlinson 1986; Lee 2008).

7.3.1 Distribution, Diversity, and Community Composition

7.3.1.1 Geographic Distribution and Shoreline Position

Global warming and SLR will directly lead to shifts in mangrove distribution. Using the Tropics of Capricorn and Cancer at 23.5 degrees northern and southern latitude is a common way to delimit the tropical regions. However, there are other climatological and geographic features, mainly temperature, precipitation patterns and atmospheric circulation that allow for other classifications and define slightly different geographical limits. Taking these into account, climate observations from 1979 to 2004 indicate a poleward shift of the tropical belt by 2–4.5 degrees latitude. For comparison, model simulations for the same period found an expansion of the tropical belt by about 2 degrees latitude (Seidel et al. 2008 and references therein). Changes in surface temperatures have been observed to be affecting the latitudinal limits of mangrove distribution at least in the North Atlantic and North Pacific, where range is limited by temperature only (e.g., Saintilan et al. 2014).

Osland et al. (2013) discovered that a 0–2 °C increase in mean annual minimum temperatures would facilitate mangrove expansion onto salt marshes on an additional 740 km^2 of coastline in southeastern United States, while a 2–4 °C increase could give rise to an additional 9860 km^2 of mangrove vegetation. Such temperature-mediated shifts in mangrove expansion are becoming widely documented (Cavanaugh et al. 2014; Saintilan et al. 2014), and a strong global effort is underway to understand the impact of temperate-zone marsh-to-mangrove habitat shifts on carbon storage, nutrient processing, surface elevation change, and structural provisioning for wildlife (e.g., Perry and Mendelssohn 2009; Osland et al. 2014a). New efforts have even started to identify the influences of rainfall regime in areas having active temperature-mediated mangrove encroachment (Osland et al. 2014b).

A global-scale examination of temperature variation and latitudinal limits of the two mangrove genera *Avicennia* and *Rhizophora* did not find common isotherms characterizing the poleward limit of their distribution. *Rhizophora* did not expand towards the colder *Avicennia* limit along the same coast although all other environmental conditions were similar (Quisthoudt et al. 2012). Nevertheless, a 28-year time-series of mangrove distribution at the northeast coast of Florida revealed a large increase in mangrove area at its northern latitudinal distribution limit related to a decreased frequency of extreme cold events (days colder than −4 °C; Cavanaugh et al. 2014).

Mangrove poleward proliferation corresponds well with the extension of temperature thresholds observed on five continents in the past 50 years (Saintilan et al. 2014). However, the response is not evenly distributed; a detailed study of the southern limit of mangroves in the western South Atlantic showed no expansion to the south (Soares et al. 2012). Similarly, at the southern limit of mangroves along the western Pacific Ocean in Peru, the cold and arid conditions hinder mangrove expansion south of a small stand of mangrove vegetation in the Piura River Estuary (Clüsener and Breckle 1987). Obviously, species-specific responses to warming in combination with other abiotic and biotic factors play a role in the poleward expansion of mangroves. Although temperature is a major control of mangrove distribution, it appears that there is no consistent response of poleward mangrove expansion to temperature increase (e.g., Bianchi et al. 2013).

The estuarine and coastal ocean circulation may also be altered, in a manner not necessarily reflecting the atmospheric change. This could have an impact on the variability of flushing and salinity of estuaries and their fringing mangrove vegetation because of changes in rainfall and evaporation. Mangroves trap sediment at rising tides, especially so during river floods (Victor et al. 2004); thus a change in the river hydrology will affect the sedimentation rate in mangroves and their ability to keep up with SLR.

The first and simplest response of mangrove distribution to SLR would be to keep up with the sea level and/or migrate landward where not obstructed. Mangroves keep pace with changing sea level when the accretion of the mangrove sediment surface is at least similar to the rate of change in relative sea level. The change in mangrove sediment surface elevation is controlled by both surface and subsurface processes (Krauss et al. 2014a). Whether mangroves can keep up with the sea level depends, in part, on the net sediment budget, which includes both the organic matter build-up through the growth of mangrove roots and deposition of litter, as well as the inorganic sediment inflow from the estuary and coastal waters (Lovelock et al. 2015a). These processes depend strongly on human activities in the catchment (Table 7.1) and on the local sea level change generating a prograding or an eroding coast. However, the "true" landward migration and possible change in area covered also depends on a number of other factors determining the environmental setting, including rainfall (Woodroffe 1992; Eslami-Andargoli et al. 2009; Woodroffe et al.2016). The relatively few data available show that sediment accretion rates in healthy mangrove forests are often higher than current rates of SLR (Alongi 2008; Lovelock et al. 2015a), but this assessment can be highly variable among forests. In those environmental settings that have sufficient allochthonous sediment input and/or production and accumulation of organic matter and a suitable gradient of land surface elevation, SLR should not impose a risk on mangrove distribution. This generally holds true for environmental settings that are dominated by rivers and tides with abundant sediment supply on prograding coasts only (Woodroffe 1992; Woodroffe et al. 2016).

However, mangrove surface elevation is not exclusively determined by sediment accretion, but also by (i) land uplift or subsidence, (ii) groundwater influx, (iii) vegetation and soil processes, and (iv) whether the coast is prograding or eroding (Krauss et al. 2003, 2014a; McKee et al. 2007; Cherry et al. 2009; Lovelock et al. 2015a). Taking these factors into account, a review of the relatively sparse global database concluded that surface elevations in many mangrove forests will not be able to keep pace with SLR for very long under the projected highest IPCC scenario RCP8.5 until the end of 2100. This holds true for mangrove areas in small island locations with little allochthonous sediment input, e.g., in the Caribbean, East Africa and parts of the Indo-Pacific region, even for the lowest scenario RCP2.6 (Sasmito et al. 2015); although there are noteworthy exceptions (McKee et al. 2007). Using surface elevation change records at 27 sites, a recent study assessed that the elevation gain is significantly correlated with sediment accretion in Indo-Pacific mangrove forests to SLR (Lovelock et al. 2015b). The surface elevation gain in 69% of the records was lower than the long-term rate of SLR. Model simulations based on these data sets and a moderate SLR (0.48 m by 2100, RCP6 scenario) indicate submergence of mangrove forests by 2070 in the Gulf of Thailand, the southeast coast of Sumatra, the northeast coasts of Java, Papua New Guinea and the Solomon Islands. In general, mangrove forests may be submerged by 2080 in regions of the Indo-Pacific with low tidal range and low allochthonous sediment supply under the moderate IPCC RCP6 scenario, while mangroves receiving higher sediment loads and larger tidal ranges are more buffered from submergence (Lovelock et al. 2015b). Thus, in their current shoreline positions, some mangroves will be converted to deeper water habitats, and where not obstructed, they may expand landward of their current positions as relative sea level rises.

The landward migration response is determined by a complex setting of local and regional environmental aspects modulating the effects of climate change. For example, mangrove forest area along the coastline southeast of the Amazon mouth expanded by 718 km^2 over a 12-year period (1996–2008), which seems to be typical for this region (Nascimento Jr. et al. 2013). Also, along the semiarid coast of northeast Brazil, mangroves are expanding rapidly in response to reduced rainfall due to climate change and land use alteration in the local watersheds, in particular damming. For example, in the estuary of the Jaguaribe River, the largest river in the state of Ceará, the mangrove-covered area increased by 24 ha between the years 1992 and 2003. A combination of land use-induced increased sedimentation and a decrease in rainfall over the river basin allowed mangroves to rapidly colonize newly formed islands in the estuary (Godoy and Lacerda 2014).

The complexity of factors controlling mangrove distribution makes it difficult to assess the potential net loss in areas related to climate change in this century. For example, based on the IPCC maximum SLR scenario, Gilman et al. (2006) predicted an up to 13% loss of Pacific island mangrove until the year 2100. Similarly, Alongi (2008) concluded that a global loss rate of mangroves related to climate change on the order of 10–15% over a long time frame is a realistic projection.

7.3.1.2 Diversity and Community Composition

The future distribution of 12 common mangrove trees under climate change were projected by applying species and community distribution models (Record et al. 2013). A range of SLR projections and a set of climatic, hydrological, and geomorphological variables associated with mangrove distribution patterns were used in these models. For half of the modeled species (*Avicennia germinans*, *Laguncularia racemosa*, *Rhizophora mangle*, *R. racemosa*, *R. mucronata*, *Lumnitzera littorea*), a poleward shift of 2 degrees of latitude or more was projected. In addition, losses in the total area of suitable coastal habitat available were predicted (Record et al. 2013). Four species (*A. marina*, *Ceriops tagal*, *Lumnitzera racemosa*, *R. apiculata*) will gain suitable coastal habitat with mean latitudinal gains of less than 2 degrees regardless of the amount of predicted SLR. *Sonneratia alba* and *R. stylosa* were forecasted to gain coastal habitat and experience a decrease in absolute mean latitude. Regarding species richness, a gain is predicted across much of southeastern Asia, southern Brazil, northern Chile, eastern Australia, southeastern Africa, parts of northern Africa, and parts of northwestern Mexico. By contrast, a loss in species richness is projected for the Caribbean islands, parts of Central America, and parts of northern Australia (Record et al. 2013).

For tropical and subtropical regions, species-specific changes in growth, productivity, and competitive ability will probably lead to changes in tree species composition (Pernetta 1993). It has been suggested that mangrove crabs and molluscs expand poleward and that the length of inactive periods of native species in certain locations may be reduced in response to a shorter cold season (Nordhaus 2008).

7.3.2 *Physiology of Flora and Fauna*

Global warming, the altered hydrological cycle, and the increase of atmospheric CO_2 will directly affect the physiology of mangrove trees and animals which, in turn, will entail changes in productivity and respiration, community composition, distribution, and biogeochemical transformations.

7.3.2.1 Flora

Mangrove trees are very plastic in their response to shifts in salinity and inundation, as they are able to adjust photosynthetic water use efficiency and leaf morphological characteristics. As the environment changes over time, there may be a shift in species composition to more stress-tolerant trees within a forest (Reef and Lovelock

2015; Lovelock et al. 2016). While the majority of studies investigating adjustments in water use efficiency in mangroves along environmental gradients focus on seedlings, recent evidence suggests that mangroves can also adjust their water use efficiencies and CO_2 uptake capabilities at the stand level (Barr et al. 2013). This ability of mangrove stands to take up CO_2 and facilitate growth and maintenance despite reductions in rainfall or increases in salinity positions mangrove stands for resiliency.

The few sap flow studies that are available from mangrove trees, generally confirm their water use efficiency values documented from the seedling leaf–gas exchange literature (Lovelock et al. 2016): Water use values for individual trees are typically <31 L H_2O d^{-1}, which is approximately one-third the water use of non-mangrove trees of similar size (Lovelock et al. 2016). Thus, water use by most mangrove stands is rather conservative relative to evapotranspiration (Krauss et al. 2015), and can rival the water use reductions documented by C_4 salt marsh grasses in some settings (Krauss et al. 2014b). The ability to conserve water would buffer mangroves against harmful rainfall variability and fluctuations in salinity associated with both climate and land use alterations. How water is partitioned is also strongly dependent upon forest structural characteristics. For example, stand water use of trees ≥ 5 cm in diameter from two mangrove forests occurring within 2 km of each other in Rookery Bay, Florida, ranged from 373–481 mm $year^{-1}$ over a 2-year period (or 36–47% of evapotranspiration), with the higher water use being related to larger individual trees of one species (*Laguncularia racemosa*) (Fig. 7.3; Krauss et al. 2015). Thus, any environmental, land use, or climate factor that alters forest struc-

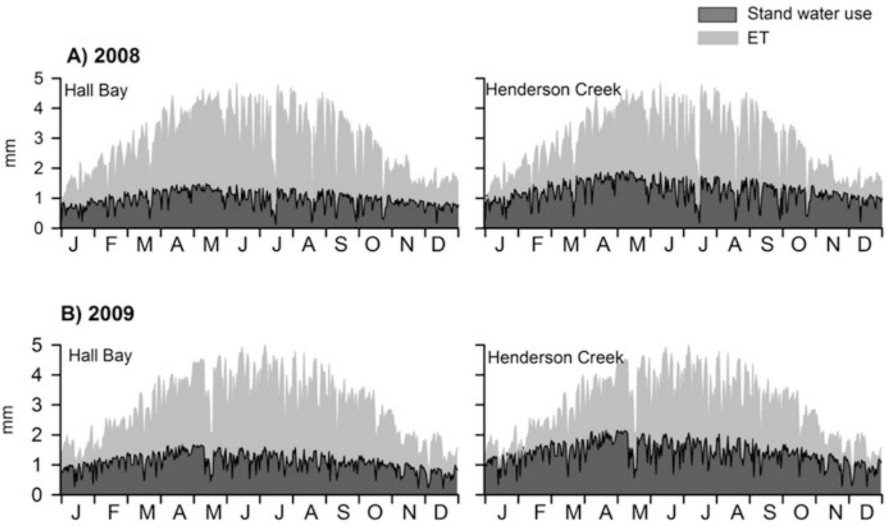

Fig. 7.3 Stand water use by dominant mangrove forest vegetation versus regional evapotranspiration (ET) for two forests (Hall Bay, Henderson Creek) differing slightly in forest structural attributes in Rookery Bay, Florida for the modeled year (**a**) 2008 and (**b**) 2009 (Data from Krauss et al. 2015)

ture also has the potential to alter water use requirements of mangroves significantly. In fact, while mangrove forest vegetation seemed to restrict losses to evapotranspiration from those two locations in Rookery Bay, water use by mangrove forests with larger trees in a different location was 872 mm year^{-1}, and accounted for a higher percentage of regional evapotranspiration (63–66%; Krauss et al. 2015).

7.3.2.2 Fauna

Benthic invertebrates in mangrove forests experience harsh environmental conditions during their life phases. Juveniles and adults of crustaceans and mollusks have to cope with daily fluctuations of temperature, salinity, and pH associated with the tidal cycle. However, most have planktonic larvae that are less exposed to these fluctuations. This is beneficial because larvae can have increased susceptibility to environmental stress (Pechenik 1999) and are often regarded as the bottleneck for success of marine species with regard to ocean warming and acidification (e.g., Byrne and Przeslawski 2013).

Considering distribution patterns, thermotolerant species with a wide latitudinal range of distribution are often robust to warming (Byrne and Przeslawski 2013). By contrast, many tropical species are considered to live close to their upper thermal tolerance (McMahon 2001; Nilsson et al. 2009) and may thus respond with reduced respiratory scope to even small temperature increases as reported for coral reef fishes (e.g., Nilsson et al. 2009). The heat coma temperature of intertidal gastropods from mangroves, rocky shores, and salt marshes displayed an overall negative quadratic relationship with latitude. While there was no trend with latitude in tropical regions, heat coma temperature decreased strongly with increasing latitude in extratropical regions. The highest thermal tolerance was found for gastropods of the group Littorinoidea from mangroves and salt marshes (McMahon 2001).

Two crab species occurring in tropical and subtropical mangrove forests demonstrated species- and population-specific responses to temperature increases. *Uca urvillei* is a thermal generalist and tolerates a broad range of temperatures across its geographical distribution. By contrast, the thermal tolerance of *Perisesarma guttatum* is much lower, but populations showed adaptations to local conditions. Tropical populations of *P. guttatum* had higher tolerances to acute heat stress than subtropical populations and are likely less vulnerable to global warming (Fusi et al. 2015). Another example is the mangrove oyster *Crassostrea rhizophorae,* which survived aerial exposure at 45, 42, and 35 °C for 2, 5, and 24 hours, respectively. Thus, adaptation to high temperatures typically occurs in mangrove forests (Littlewood 1989).

So far, the combined effects of warming and acidification on mangrove invertebrates have not been investigated, but information is available for other marine and estuarine species. A recent review on multistressor impacts of ocean warming and acidification on marine invertebrates found effects to vary across life stages and species (Byrne and Przeslawski 2013). Pre-larval stages are particularly sensitive to slight warming (+2 °C), whereas larvae of some species may tolerate near-future warming and acidification (+2 °C/pH 7.8). By contrast, deleterious effects on larvae, e.g., reduced sizes and survival rates, were recorded in experiments with higher deviance from present conditions (ca. ≥4 °C/pH <7.6). Calcifying organisms, e.g., corals, mollusks, and the larvae of echinoderms will be most negatively impacted by ocean acidification (Kroeker et al. 2013). Calcifying larvae need more energy for calcification as a result of decreased saturation of carbonate minerals (Byrne and Przeslawski 2013). For instance, the exposure to near-future elevations of pCO_2 and temperature had deleterious effects on the oysters *Magallana gigas* (formerly *Crassostrea gigas*) and *Saccostrea glomerata*, including a reduction in the fertilization success of gametes, a reduction in the development of embryos, size of larvae and spat, and an increase in abnormal morphology of larvae (Parker et al. 2010). More active organisms, such as mobile crustaceans and fish, may be less sensitive to acidification (Kroeker et al. 2013). Crustacean species that inhabit fluctuating environments are considered to be the most tolerant to ocean acidification as they are capable of compensating for acid–base disturbances via ion exchange mechanisms (Whiteley 2011). However, experiments with coastal prawns (*Palaemon pacificus*) from Japan revealed effects of ocean acidification on their molting frequency and growth (Kurihara et al. 2008).

A serious shortcoming of these laboratory experiments is that adaptations to the more gradual change of temperature and pH in the ocean over several decades cannot be evaluated. Species with short generation times may be able to tolerate near-future oceanic change through acclimatization and/or adaptation (Byrne and Przeslawski 2013). However, considering the data available, a clear signal of acclimatization is lacking (Kroeker et al. 2013). In addition, most experiments were conducted with single species only. Responses were more pronounced when species were exposed to acidification in multispecies assemblages, which led to the conclusion that indirect effects also have to be considered (Kroeker et al. 2013).

Based on the few studies available, a high thermal tolerance can be expected of the adults of a number of benthic species in mangrove forests. However, not only species-specific but also population-specific responses and adaptations (Fusi et al. 2015), varying responses of the different life history stages (Byrne and Przeslawski 2013) and species interactions (Kroeker et al. 2013) have to be considered to predict the consequences of climate change. The negative effects of warming and acidification on many planktonic larvae may result in a decrease of benthic populations and shifts in community composition if species are not able to adapt accordingly.

7.3.3 Water Budget

The water budget and associated variations in salinity are also important controls of mangrove distribution. The demise of the large mangrove area in the arid Indus Delta region illustrates this well (Kidwai et al. 2016). Damming of the Indus in the middle of the last century and the use of water for numerous purposes reduced the freshwater flow to about one-fifth of its original flow of 180 km^3 $year^{-1}$ and the sediment load from 400 to 100 million tons $year^{-1}$. In combination with the arid climate, the salinity in the Indus Delta region increased to 50 and as a consequence, the number of mangrove species decreased from 8 to 3; the community is now dominated by the highly salt-tolerant and dwarfed *Avicennia marina* (Amjad et al. 2007). Although caused by human interventions, this example demonstrates how the combination of reduced freshwater flow and increasing temperature as a consequence of climate change can endanger the survival of mangroves. A similar case was observed in the Tanzanian Wami River. There, a decrease in precipitation during the second half of the last century in combination with increasing human uses of river water for agriculture, irrigation, industry, and drinking water led to increasing dryness in the estuary during dry seasons (Kiwango et al. 2015). Even with no anthropogenic changes in the hydrology of rivers and estuaries, the combination of increasing temperature and changing atmospheric moisture from climate change will affect the precipitation/evaporation ratio and the freshwater flux and salinity of mangrove environments. The largest threat to mangroves in terms of the water budget is the combination of increasing temperature with decreasing rainfall.

Higher temperatures and reduced rainfall may lead to a reduction in mangrove area because upper tidal zones are converted to hypersaline flats (Gilman et al. 2008). Despite the water conservation strategies of mangroves discussed previously (Sect. 7.3.2.1), increasing pore water salinity and decreasing water availability will reduce productivity, growth, and seedling survival (Field 1995; Ellison 2000) and the reduced inflow of nutrient-laden freshwater will reduce overall ecosystem productivity. In areas of low rainfall, mangrove forests are characterized by dwarfed trees, narrower margins, and interspersed salt flats mainly because of salt stress (Ellison 2000). Species-specific responses will also entail changes in community composition and most probably, a reduction in biodiversity in these areas. In contrast, mangroves in regions with increasing rainfall will benefit as the increased water availability will reduce salinity and physiological stress of plants and increase sediment and nutrient inflow. Usually, mangroves are more productive, taller, and more diverse in regions with high rainfall compared to those with lower rainfall (Field 1995; Duke et al. 1998). Moreover, enhanced rainfall may lower salinity and allow mangroves to outcompete salt marshes and hence, increase mangrove area (Rogers et al. 2005).

7.3.4 Productivity and Remineralization

The rise in atmospheric CO_2, air temperature, and water temperature will likely increase the productivity and respiration of mangroves (e.g., Field 1995; Ball et al. 1997). A globally uniform trend, however, is unlikely because of the complex interplay of the aforementioned factors with other regional conditions and environmental settings. In general, mangrove productivity will probably not change much in the inner tropics, while the increase will be largest near the poleward temperature limits (Osland et al. 2013; Cavanaugh et al. 2014). However, in arid and semiarid regions where temperature may exceed a threshold of 33 °C, the photosynthetic rate of mangrove plants declines (Cheeseman 1994; see also Osland et al. 2014b). This will mainly concern the Red Sea and Persian Gulf regions.

A linear response of mangrove productivity to CO_2 increase is not likely because of the interplay of photosynthesis with other factors, namely salinity, water use efficiency, and nutrient availability (Lovelock et al. 2016). In an experiment on Caribbean mangroves *Rhizophora mangle*, *Avicennia germinans*, *Conocarpus erectus,* and *Laguncularia racemosa* under exposure to instantaneous pulses of increased CO_2 (361–485 ppm), trees displayed decrease in stomatal conductance and transpiration and increase in transpiration efficiency. However, net primary productivity decreased in *L. racemosa* and did not change in the other species. Therefore, it was suggested that a global rise in atmospheric CO_2 may result in a competitive disadvantage to *L. racemosa* in mixed mangrove communities and may lead to alteration of the community composition in these locations (Snedaker and Araujo 1998).

Rhizophora mangle trees displayed enhanced growth rates and biomass in a one-year experiment under doubled CO_2 and became reproductive much earlier than usual in mangrove forests of Belize, Central America (Farnsworth et al. 1996). Growth of *Rhizophora apiculata* and *Rhizophora stylosa* did not increase under doubled CO_2 over 14 weeks in high salinity areas, but increased in low salinity areas during an experiment in Australia (Ball et al. 1997). In general, exposure to elevated CO_2 enhanced mangrove seedling growth by 12–47% (up to a maximum 71%) and enhanced water use efficiency by 2–218% across a range of studies (Krauss et al. 2008; Lovelock et al. 2016). It was concluded that an increase of CO_2 may enhance mangrove productivity when carbon uptake is limited by the evaporative demand of the leaves, but not when it is limited by salinity at the roots (Ball et al. 1997; Gilman et al. 2008). The partly contradicting results of these experiments suggest species-specific responses to elevated CO_2 and the need for further research.

7.3.5 Carbon Storage in Biomass and Sediments

The well-known high carbon storage potential of mangrove ecosystems (Twilley et al. 1992; Jennerjahn and Ittekkot 2002; Bouillon et al. 2008; Alongi 2014; Chaps. 5, 6) gained scientific and public attention in past years when the climate change debate started focusing on the identification and conservation of natural carbon sinks in the ocean, the so-called “blue carbon” concept (Nellemann et al. 2009). Mangroves are part of the vegetated coastal habitats with very high relative carbon storage rates per unit area, much higher than on shelves or in the open ocean (Alongi 2014). Being among the most productive ecosystems on earth, mangroves store a larger amount of carbon in their above- and belowground biomass than terrestrial forests. In addition, they accumulate carbon-rich sediments composed of dead plant material and, depending on the environmental setting, large amounts of allochthonous mineral sediments and carbon (e.g., Jennerjahn and Ittekkot 2002; Donato et al. 2011).

Climate change can affect the carbon storage of mangrove biomass and sediments in manifold ways, first and foremost through changes in area, second through related changes in gain and loss terms of autochthonous (photosynthesis) and allochthonous (exchange with land, sea, atmosphere) carbon. Mangrove ecosystems gain carbon mainly through photosynthesis of (i) mangrove plants and (ii) macro- and microalgae colonizing roots and the sediment surface and (iii) through import from adjacent terrestrial and marine systems. Major carbon losses occur through respiration and tidal export, while secondary production and respiration by mangrove macrofauna are of minor importance (Alongi 2014).

Starting off with the first global mangrove carbon storage estimate by Twilley et al. (1992), efforts to quantify global mangrove carbon storage have increased in the past decade. Interestingly, when normalized to an area of 138,000 km^2 (Giri et al. 2011), the annual total global mangrove carbon storage estimates increased from 14–16 Tg year^{-1} (Twilley et al. 1992; Jennerjahn and Ittekkot 2002) to 22–24 Tg year^{-1} (Alongi 2012; Breithaupt et al. 2012) over one decade. In the light of the ongoing mangrove area loss per year, such an increase is surprising and the possible reasons are manifold. First, despite the advance in identifying and quantifying gain and loss terms of carbon, there are still large uncertainties, for example, in the loss through dissolved inorganic carbon (e.g., Bouillon et al. 2008). Second, mangrove carbon accumulation rates can vary by one to two orders of magnitude both among and within sites (e.g., Breithaupt et al. 2012). Global-scale extrapolation from the relatively small database therefore can have a quite large degree of uncertainty. Third, it is possible that the increasing atmospheric CO_2 and eutrophication of coastal waters enhance mangrove productivity and burial of carbon in sediments. Several studies have shown that nutrient enrichment can enhance mangrove growth (e.g., Feller et al. 2002, 2003; Lovelock et al. 2007) and possibly also increase carbon burial in sediments. The increase in global estimates by a factor of two over relatively short time is probably a result of all of these factors.

The average of all available budgets results in an annual global mangrove carbon storage of 22 ± 6 Tg year^{-1}. Approximately 75% of this carbon is stored in sediments, the rest is above- and belowground biomass (Donato et al. 2011; Alongi 2014). Interestingly, there are differences between the two major biogeographic regions. Based on the global set of data used by Breithaupt et al. (2012), the average carbon accumulation rate is 265 ± 161 g C m^{-2} year^{-1} in the IWP and 194 ± 155 g C m^{-2} year^{-1} in the AEP. Despite the large variability in numbers, this difference is consistent with the finding of a significantly higher aboveground biomass in the IWP than in the AEP mangrove forests at the same tree height. It potentially reflects a higher primary productivity in the IWP region, the reasons of which can be manifold, e.g., differences in climate, tree density, stand age, species architecture, and anthropogenic disturbances (Lee 2008 and references therein). The low latitudes of the Asian/western Pacific region receive the highest fluxes of dissolved nutrients and suspended particulate matter worldwide and are also strongly affected by human interventions (Smith et al. 2003; Syvitski et al. 2005; Milliman and Farnsworth 2011; Jennerjahn 2012). Accordingly, the high nutrient input may promote high productivity and, in combination with the higher allochthonous organic matter input, also a higher carbon accumulation in IWP than in AEP mangroves.

Overall, the total annual average mangrove carbon sequestration of 22 ± 6 Tg year^{-1} accounts for approximately 0.2% of the 9200 Tg year^{-1} (estimate for the period 2002–2011) of anthropogenic carbon emissions (Ciais et al. 2013). Assuming that climate change related loss of carbon is equal to the estimated 10–15% loss of mangrove area by the year 2100 (Alongi 2008), the annual carbon storage will be reduced to 19 Tg year^{-1}. In this context, mangroves appear to be a negligible sink for anthropogenic CO_2. However, the carbon that is released from the destruction of mangroves is quantitatively more important. A recent estimate of the total mangrove carbon stock worldwide, accounting for the uncertainties of available data, arrived at the broad range of 4000–20,000 Tg (Donato et al. 2011), which adds significantly to the total tropical wetland forest carbon storage of 82,000–92,000 Tg (Page et al. 2011). Climate change-related 10–15% mangrove loss until 2100 would mean an annual release of 4.7–35.3 Tg C year^{-1} and sum up to a total of 400–3000 Tg C. Accordingly, a complete loss, though unlikely, would mean an annual release of 47–235 Tg C year^{-1}. Compared to the annual anthropogenic carbon emissions, it remains a small, but at 2.5% not a negligible amount (Fig. 7.4). Because of the large uncertainties in fluxes, it remains unclear whether mangroves in the year 2100 will

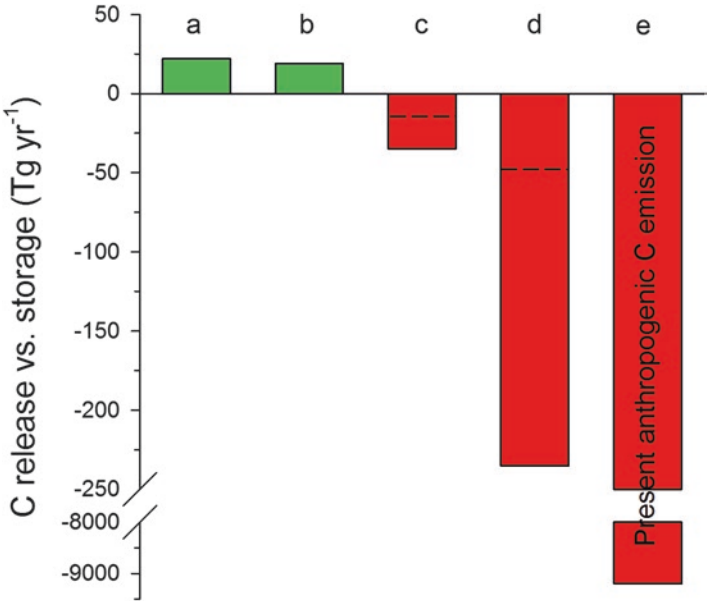

Fig. 7.4 Annual mangrove carbon storage (*green*) and release (*red*) today (*a*) and under 10–15% loss (*b* and *c*) and total loss scenarios (*d*) until the year 2100 compared to the present-day (i.e., 2012) anthropogenic carbon emissions (*e*). *Dashed lines* denote the lower limit of carbon release from mangroves as reported in the text (Data sources: Ciais et al. (2013), Donato et al. (2011) and data sources in the text. Note the break in the *Y*-axis)

be carbon sinks or sources. However, from these calculations, it is conceivable that the consequences of climate change alone can turn mangroves from sinks into sources of carbon by the end of the century.

7.3.6 Filter Function for Elements Beneficial or Harmful to Life

The many characteristics making mangroves such a unique ecosystem in the land–ocean transition also make them an important filter for land-derived substances that are beneficial or harmful to life. They trap mineral sediments, organic matter, and nutrients. Climate change related SLR, temperature increase, altered hydrology, and ocean acidification will alter these properties. As a consequence, land-derived substances retained and processed in mangroves will decrease if the mangrove area decreases 10–15% by 2100 (Alongi 2008).

For example, eutrophication, i.e., the exposure of coastal waters to excess nutrients, is a major man-made phenomenon. In a global-scale analysis of estuarine systems, Valiela and Cole (2002) showed that salt marshes and mangroves can protect seagrass meadows from land-derived nitrogen loads. Fringing wetlands are capable of absorbing up to 20 kg N ha^{-1} year^{-1}. Nitrogen loads between 20 and 100 kg N ha^{-1} year^{-1} are known as a critical range for seagrass meadows (Valiela and Cole 2002). Climate-related loss of mangroves will therefore reduce the retention capacity and increase the exposure of downstream ecosystems to land-derived nutrients.

Tidal exchange of suspended matter (TSS), as well as total (TP), particulate (Part-P), and soluble reactive phosphorus (SRP) in northeast Brazil showed that the retention of phosphorus in mangroves varies with anthropogenic inputs. A mangrove site receiving effluents containing 1.2–5.2 kg h^{-1} of total phosphorus from nearly 3000 ha of shrimp ponds could trap about 40%, 45%, 47%, and 70% for TSS, TP, SRP, and Part-P, respectively, of the incoming phosphorus flux. However, a non-impacted mangrove site receiving phosphorus from only 10 ha of ponds (0.2 kg h^{-1}) tidal balances retained 92% of the total input of TSS and 100% of all other P fractions. This suggests that mangrove phosphorus retention capacity decreases with increasing nutrient input and limits the potential of mangroves as a natural nutrient barrier (Fonseca et al. 2014).

Similarly, mangroves display an elevated capacity for storing calcophylic metals as sulfides precipitates. Metal sulfides formed as a consequence of bacterial dissimilatory sulfate reduction is efficiently buried in anoxic mangrove sediments due to high accretion and waterlogged conditions. Mangrove sediments immobilize 40.6 g m^{-2} of Zn, 3.3 g m^{-2} of Cu and 43 mg m^{-2} of Hg in the highly polluted Guanabara Bay, Rio de Janeiro, southeastern Brazil (Machado et al. 2002). Some metals, such as Hg, may suffer a different fate in mangrove environments due to their high affinity to organic complexes. Inorganic Hg forms the major pool of Hg entering mangroves through tides or river transport. It will efficiently accumulate in sediments, but a significant fraction, rather than precipitating as sulfides will be sequestered by the large amount of dissolved organic complexes (DOC) present in pore waters due to the incomplete oxidation of organic matter. Formation of complexes with DOC then will increase the amount of highly bioavailable organic Hg complexes being exported by mangroves relative to the incoming fluxes of these chemical species. A 5-year study in the Jaguaribe estuary showed an increasing export of dissolved Hg to adjacent waters over time. It was enhanced in dry periods, which are more frequent because of an overall decreasing annual rainfall, and because of damming in the river basin (Lacerda et al. 2013).

The impact of global climate change on the aforementioned processes will depend on local geomorphological dynamics. In areas with expected mangrove expansion, such as those in northeastern Brazil and in the southeastern USA, accumulation and storage of carbon and nutrients as well as of pollutants may be increased accordingly. However, along constrained coastlines where mangroves cannot expand landward or on low-lying islands, erosion of the deposited sediments may result in the release of nutrients and pollutants to the adjacent waters.

7.4 Vulnerability of Regions

The four most important factors determining the vulnerability of mangrove forests to climate change are SLR and associated increase in frequency/intensity of storms, temperature, and aridity. In this context, mangrove forests are best protected in river- and tide-dominated settings where allochthonous inputs from land and ocean can help keep pace with the rising ocean, i.e., along macrotidal coasts, in river estuaries and in wet coastal areas (e.g., Woodroffe 1992; Woodroffe et al. 2016). Applying these criteria allows for an identification of the most vulnerable mangrove regions to climate change worldwide as depicted in Fig. 7.5. While low-lying islands in the equatorial Pacific are mainly threatened by SLR, areas further north and south are possibly additionally threatened by an increased intensity of storms. The same holds true for the Caribbean. In the already arid regions of the Red Sea and the Persian Gulf, IPCC projections foresee increasing temperature and aridity (Christensen et al. 2013; Collins et al. 2013), which may go beyond ecological thresholds and therefore pose an additional risk to the existence of mangroves. Other vulnerable areas include mangrove forests along coasts where narrow coastal plains are restricted by mountains, such as in southeastern Brazil and on mountain islands. It appears, though, that more mangrove systems are vulnerable to the adverse effects of climate change in the IWP than in the AEP region (Fig. 7.5). A recent assessment of climate change impacts on mangrove forests, based on IPCC projected regional changes in salinity, precipitation, and SLR (Alongi 2015), came to a similar conclusion.

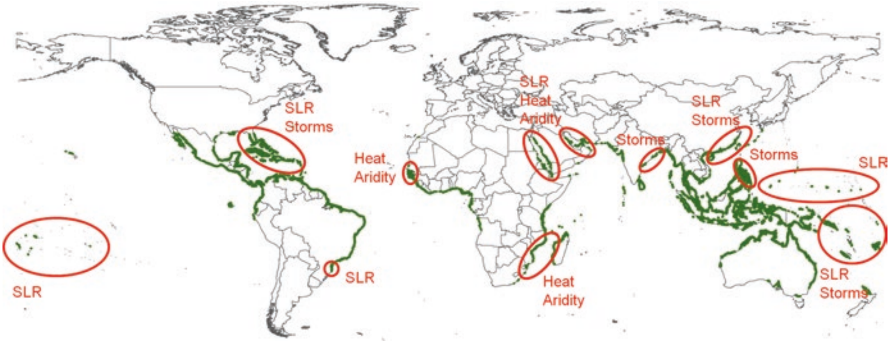

Fig. 7.5 Mangrove regions most vulnerable to consequences of climate change (sea level rise, increased frequency/intensity of storms, increasing temperature and aridity) are marked in *red*. Regions were chosen according to most severe changes projected by the IPCC AR5. Mangrove distribution (*green lines*) (According to Giri et al. (2011); data set taken from UNEP World Conservation Monitoring Centre, URL: data.unep-wcmc.org/datasets/21. World borders data set taken from Thematic Mapping Engine, URL: thematicmapping.org)

7.5 Interaction with Human Interventions

Human activities play an important, if not the dominant, role in the decline of mangrove forests (Valiela et al. 2001), although mangrove loss has slowed over the last decades (Spalding et al. 2010). It is still alarmingly high. Assuming that deforestation continues at the usually cited loss rates of 1–2% per year (Alongi 2002), mangroves will be lost before the end of the century, rendering climate-related changes almost negligible (Alongi 2008). However, recent studies indicate that the average loss rate was an order of magnitude lower since the year 2000. Between 2000 to 2012 global mangrove deforestation varied between 0.16% and 0.39% per year (Hamilton and Casey 2016) and was on average 0.18% per year in Southeast Asia (Richards and Friess 2016), the region which contains the greatest diversity of mangrove species and roughly one third of the global mangrove area. Taking this into account mangrove vulnerability to climate change threats may be larger than previously thought.

The major factor determining mangrove resilience to climate change related SLR is landward migration as shorelines retreat. The survival of the ecosystem as a whole is then mainly determined by hinterland topography and/or sediment accretion allowing the system as such to migrate landward and to maintain a suitable surface elevation. Simulations of landward migration in Gazi Bay, Kenya, until the year 2100, have shown that mangroves under low to medium SLR scenarios can manage without significant losses. However, under a maximum SLR scenario, further landward migration is obstructed by a strong increase of the topographical gradient (Di Nitto et al. 2014). What becomes even more important in this context is coastal development, i.e., settlements and hard engineering coastal protection (e.g., seawalls, revetments), that hinders landward migration of mangroves. Coasts suffering from mangrove clearing can become destabilized even when coastal engineering measures are taken. In Guyana, for example, the coastal dikes constructed after mangrove clearing were not as efficient in stabilizing the muddy coast as the mangroves were (Anthony and Gratiot 2012). However, with respect to SLR and an increasing storminess, coastal engineering measures and mangroves may even complement each other as shown by an example from the Mekong Delta. There, keeping or restoring mangroves helps to keep the necessary dikes lower and hence less expensive than without a seaward mangrove belt (Albers and Schmitt 2015).

Other important human factors are the regulation of hydrology and land use change, which affect fluxes of sediments and all other dissolved and particulate constituents from terrestrial runoff. Land use change, mainly deforestation, settlements and infrastructure, leads to an increase of erosion and higher river fluxes of suspended sediments, nutrients, and other substances. The opposite happens in the case of river damming and other hydrological regulations (e.g., Vörösmarty et al. 2003; Walling and Fang 2003; Jennerjahn 2012). A global analysis of river fluxes showed that human activities have increased the suspended sediment transport to the ocean by 2.3×10^9 t year^{-1} through soil erosion, and simultaneously decreased the transport by 3.7×10^9 t year^{-1} through trapping in reservoirs behind dams, resulting in a net loss of 1.4×10^9 t year^{-1} suspended sediment during the Anthropocene (Syvitski et al. 2005). However, large regional variations in the net effect of both processes and increases in the sediment load during the Anthropocene are almost exclusively observed in the tropical belt. While the coasts of Africa, large parts of South America, and West Asia suffer from reduced sediment input, sediment fluxes increased in parts of South and Southeast Asia (mainly Indonesia and the Philippines) and in Central America and the Amazon region (Syvitski et al. 2005). In the latter regions, the increased sediment input can probably to some extent offset the adverse effects of SLR and may even promote an increase in mangrove area. In the former coastal regions, sediment starvation may aggravate the loss of mangrove areas to SLR.

Eutrophication is another anthropogenic threat to coastal ecosystems. Human discharges of nitrogen and phosphorus into the ocean have increased by a factor of three from the 1970s to the 1990s (Smith et al. 2003). The Mississippi–Gulf of Mexico and the Danube–NW Black Sea are two prominent large-scale examples of how human interventions drastically change the amount and composition of nutrients with associated effects on biogeochemical cycles and food webs (e.g., Humborg et al. 1997; Rabalais et al. 2000). However, the inputs and ecological consequences can be very different in tropical regions (e.g., Jennerjahn et al. 2008; Jennerjahn 2012). Flooding with nutrient enriched coastal waters and increased input of fluvial nutrients may enhance productivity of mangrove forests, but may also cause changes in community composition of flora and fauna. However, nutrient enrichment may also increase mangrove mortality (Reef et al. 2010). Since nutrient enrichment favors the growth of shoots over roots, the higher water demand of the shoots has to be met by the roots. This, in turn, increases the vulnerability of mangrove trees to environmental stress such as high salinity, low humidity, and low rainfall (Lovelock et al. 2009). Seaward fringing forests are less vulnerable than landward scrub forests due to differences in inundation frequency and these differences become larger along gradients in aridity. The higher ratio of above- to belowground biomass could also make mangrove forests more vulnerable to windthrow and waves (Alongi 2008; Lovelock et al. 2009). It is therefore conceivable that eutrophication will exacerbate climate change–related mangrove dieback particularly in areas exposed to tropical storms and where temperature and aridity increase.

The increasing settlement and use of coastal zones also changes the physical conditions. Large deltas of major world rivers have always been preferred locations for human settlement. Tropical deltas also host the most luxurious mangrove forests because of the high sediment and nutrient supply, and wide intertidal zones. The subsidence of deltas has increased dramatically in the past decades, mainly due to the reduced sediment supply caused by river damming, extraction of oil, gas, and water from delta sediments, and floodplain engineering (Syvitski et al. 2009). Subsidence due to natural compaction is generally ≤3 mm year^{-1} (Syvitski 2008). Accelerated compaction due to human activities can be orders of magnitude higher as, for example, in the Chao Phraya, where it ranges between 50 and 150 mm year^{-1} (Saito et al. 2007). An analysis of 33 representative world deltas revealed that many of them are sinking relative to the global sea level mainly because of human activities making them particularly vulnerable to flooding and storm surges. Many of the major world river deltas in the tropics are at great risk, particularly in Asia. Relative rates of SLR in those regions are generally >2 mm year^{-1} and can be as high as 150 mm year^{-1}. Moreover, the northern hemisphere Asian deltas are exposed to the highest frequency and intensity of tropical storms (Syvitski et al. 2009). Taken together, it appears that mangrove areas that were supposed to be least vulnerable to SLR and other consequences of climate change because of high allochthonous inputs of sediment and nutrients, may be more vulnerable than previously thought due to the human-induced sinking of deltas (Lovelock et al. 2015b).

7.6 Effects on Ecosystem Services

The ecosystem services (ES) of mangroves will change during this century because of climate change and associated SLR, but probably much more by other human interventions. In particular, the "provisioning" and the "cultural" ES, which are directed towards sustaining livelihoods will be much more affected by increases in exploitation. The climate change related overall loss of mangrove areas will also lead to an overall loss of ES, but apart from that the consequences of climate change will be more relevant for the "regulating" and the "supporting" ES and probably display large regional variations (Table 7.2).

At the lower end of the temperature range, global warming may positively affect almost all ES. The "provisioning" ES may benefit from the temperature-induced increased productivity, i.e., at the northern and southern latitudinal limits of mangrove distribution, it could lead to an enhanced supply of fuelwood and building materials. It could also enhance carbon sequestration ("regulating") and even tourism and recreation ("cultural") could benefit from more comfortable temperatures. Such a proliferation at latitudinal limits will often occur at the expense of salt marshes. As both these ecosystems have almost similar ES, also in monetary values (Barbier et al. 2011), this ecosystem change may not change the overall ES of such a coastal zone. However, there are clear exceptions, such as Hawaii, where mangroves are non-native introduced species. Changes resulting from mangroves replacing other coastal habitats have been interpreted as reducing ES, including reduced stream flow with concomitant flooding risk in adjacent inhabited areas, eliminating open water habitats of native waterbird species, and obstructing culturally valuable fish ponds (Allen 1998; Chimner et al. 2006).

At the upper end of the temperature range, further temperature increase can exceed species-specific thresholds and result in less productivity, hence reducing "provisioning" ES. This is of particular importance in areas that may suffer from reduced freshwater supply in the future, e.g., in the Persian Gulf and the Red Sea. In general, increasing aridity and decreasing freshwater inflow probably impair the "regulating" ES such that, for example, the filter function for nutrients, sediments, and other elements will be reduced. In areas that are projected to expect an increasing freshwater and allochthonous sediment input, the "regulating" and "supporting" ES may benefit. According to the IPCC AR5, this is mainly the case for the Southeast Asia/West Pacific region and the west coast of South America (Fig. 7.2). However, in the former region, the large number of dams and other regulations of hydrology will counteract such enhanced freshwater and sediment inflow into the coastal zone while increased erosion due to deforestation may support it. Species-specific responses of organisms to warming and changing water availability may lead to changes in the community composition, which can have consequences for ES. For example, changes in tree community composition can directly affect sediment trapping, water quality maintenance, and nutrient cycling functions, while a change in benthic fauna through altered burrowing and feeding activities may affect nutrient cycling and carbon sequestration (Table 7.2).

Rapid SLR will mainly impair the "regulating" ES by increasing beach and soil erosion and, in turn, also lower the coastline protection. This is of particular importance in regions with low freshwater and allochthonous sediment input and where increasing heat and aridity are projected to be most vigorous like, in the Red Sea and the Persian Gulf. Increases in the frequency and intensity of storms will probably impair all the "regulating" ES and tourism may also suffer strongly in affected regions (Fig. 7.5). During floods, storms, and extreme high water events exacerbated by climate change, the increased turbidity and direct hydrodynamic damage will change the "supporting" ES in a way that will alter nutrient cycling, probably destroy nursery habitats and change biodiversity (Table 7.2). The most deleterious effects on ecosystem services can be expected in areas that are most vulnerable to the combined effect of SLR and storms like, for example, in the Caribbean, on Pacific islands and along the Chinese and Japanese coasts (Fig. 7.5).

Table 7.2 Climate change-related effects on mangrove ecosystem services: Ecosystem services of mangroves defined by the UNEP World Conservation Monitoring Center (Wells et al. 2006; second column) are grouped in the four categories defined by the Millennium Ecosystem Assessment (2005; first column)

Category	Mangrove ecosystem service	Response to climate change
Provisioning	(a) Subsistence and commercial fisheries (b) Aquaculture (c) Honey (d) Fuelwood (e) Building materials (f) Traditional medicines	• Area loss decreases all ES • Shift in community composition may lead to change of particular ES value (c) to (f) (one species may have a higher value than another) • Increasing temperature and reduced freshwater inflow may impede (a) and (b) • Increasing productivity (because of increasing temperature and CO_2) increases (d) to (f)
Regulating	(a) Protection of beaches and coastlines from storm surges, waves, and floods (b) Reduction of beach and soil erosion (c) Stabilization of land by trapping sediments (d) Water quality maintenance (nitrogen and pollutant filter) (e) Climate regulation (carbon sequestration)	• Area loss decreases all ES • Rapid SLR decreases (a) to (d) • Increasing intensity of storms decreases (a) to (e) • Reduced freshwater inflow decreases (d), increased freshwater inflow may increase (d) (if freshwater not laden with anthropogenic nutrients and pollutants) • Increased/reduced allochthonous sediment input related to increased/reduced freshwater input may increase/decrease (a) to (e) • Species-specific responses and changing community composition may alter (c) to (e)
Cultural	(a) Tourism and recreation (b) Spiritual—Sacred sites	• Area loss decreases all ES • Changes in freshwater and sediment inflow may impair (a) • Increases in storm intensity decrease (a)
Supporting	(a) Cycling of nutrients (b) Nursery habitats (c) Biodiversity	• Area loss decreases all ES • Reduced/increased freshwater inflow decreases/increases (a) • Increases in storm intensity decrease (a) and (b) • Species-specific response will alter, probably reduce (c)

The degree of resource exploitation is another critical factor that may accelerate the climate change related loss of mangroves. Natural resources are a major supplier of mangrove ecosystem services, i.e., the "provisioning" ES (Table 7.2), directly sustaining livelihoods of people, but at the same time also a major threat to mangrove survival, if not controlled in a sustainable way. The "regulating" ES of mangroves are to a large extent determined by the physical setting and its interaction with the flora and fauna. The vegetation, in particular the roots, largely determines the dissipation of tidal and wave energy in mangrove forests, which promotes the settling of particles and hence, the formation of sediments. It stabilizes the whole system and contributes to an increase in area (e.g., Mazda et al. 2007). As such, the mangrove vegetation provides protection from waves and storms and supplies the substrate and preconditions on which the "provisioning" ES build. Similarly, ecosystem engineers such as crabs also have important functions like, for example, retaining nutrients in the system (e.g., Robertson 1986; Robertson and Daniel 1989; Nordhaus et al. 2006). Overuse of the "provisioning" ES by, for example, extraction of too much mangrove wood, probably leads to a destabilization of the substratum and eventually to the loss of sediments and nutrients, i.e., a weakening of the "regulating" and "supporting" ES. The overexploitation-induced destabilization can therefore accelerate the loss of mangrove areas related to SLR and increased intensity of tropical storms.

7.7 Adaptation and Management Options

Being located at the interface between land and ocean in the high temperature range of the globe and exposed to large gradients in physicochemical properties of water, mangrove forests as a whole and their assemblage of organisms are by nature successful "adaptors." They have developed strategies that allow them to survive in such harsh environments and even to move in response to sea level oscillations as long as topographic gradients and surface elevations allow. Large-scale oscillations of sea level on glacial–interglacial timescales led to major disruptions of mangrove distribution during the Pleistocene. However, since around 7000 years ago, when the speed of SLR slowed down and sea level almost reached its present position, mangroves have colonized a large part of the tropical intertidal zone (Woodroffe et al. 1985; Woodroffe 1992).

Considering SLR, it is obvious that systems with little or no allochthonous sediment input are generally the most vulnerable. From Holocene records, it was concluded that mangrove ecosystems can keep pace with rising sea level on the order of 8–9 cm in 100 years, are under stress at rates of 9–12 cm in 100 years, and may collapse at higher rates (Ellison and Stoddart 1991). This view has been challenged by examples where mangrove swamps kept pace with higher rates of SLR (e.g., Snedaker et al. 1994 and references therein). Nevertheless, the aforementioned rates appear to be valid for carbonate settings and/or low-lying islands where mangrove growth almost exclusively builds on the autochthonous production. The various scenarios of the recent IPCC report suggest an average global SLR of 40–63 cm until the year 2100 (Church et al. 2013), clearly higher than the threshold value for mangrove settings without allochthonous sediment input. Increasing temperature, atmospheric CO_2, and nutrient input may increase mangrove productivity and growth in carbonate settings and/or low-lying islands and hence, raise the aforementioned thresholds. However, such an increase will not at all match even the lowest projected rate of SLR. As a consequence, many of the mangroves mainly in the western Pacific and the Caribbean have little chance to survive long term.

In the numerous regions where human exploitation dominates, direct management options mitigating potential adverse consequences of climate change are limited.

An overview of adaptation options to augment mangrove resistance and resilience to climate change (Gilman et al. 2008) suggests that it is rather the human-induced degradation that can be readily mitigated. For example, reducing or eliminating the main non-climate related stressors on mangroves and functionally-linked adjacent coastal ecosystems would contribute to reducing mangrove susceptibility to climate change outcomes. Managing coastal land uses to facilitate gradual landward mangrove migration at sites where this is predicted to occur in response to relative SLR is an additional adaptation method.

In combination with these and other options for augmenting mangrove resistance and resilience to climate change, there is a need for (i) regional monitoring networks to provide a stronger basis for inferring whether global versus local stressors are causing observed changes in mangrove position, structure, and processes and (ii) outreach and education activities to increase awareness of the value of mangrove ecosystem services among the public and decision makers. The future of mangroves therefore to a large extent depends on the degree of human interventions and their interactions with climate-related changes.

7.8 Knowledge Gaps and Future Directions

- The physiological response of mangrove plants and animals as well as the response of communities to climate change are not well understood and require further study, in particular on the response to interacting multistressors.
- The carbon storage potential has large uncertainties and needs to be improved.
- There are large knowledge gaps on how the interaction of human interventions and climate change will affect mangrove ecosystem structure, functions, and services; multi- and interdisciplinary studies on these are required.
- Further efforts to increase the awareness among the public and decision makers of the value of mangrove ecosystem services will contribute to decreasing the risk of mangrove loss related to climate change.

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