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Table of Contents

Introduction and Background	4
Climate Change and Invasive Species Interactions	6
Direct Impacts	6
Indirect and Secondary Impacts	9
Ecosystem-Based Adaptation and the Maintenance of Ecosystem Services	12
Case Studies	13
Coastal Protection and Integrity	13
Beach Vitex (<i>Vitex rotundifolia</i>)	14
Nutria, Coypu (<i>Myocastor coypus</i>)	14
Miconia calvescens	14
Fisheries and Marine Ecosystems	15
Lionfish (<i>Pterois volitans</i>)	16
European Green Crab (<i>Carcinus maenas</i>) and Chinese Mitten Crab (<i>Eriocheir sinensis</i>)	17
Freshwater Services and Availability	18
Giant Reed (<i>Arundo donax</i>)	18
Water hyacinth (<i>Eichhornia crassipes</i>)	19
Agriculture, Livestock and Food Security	20
Coffee Berry Borer (<i>Hypothenemus hampei</i>)	22
Bluetongue Virus	23
Human and Wildlife Health	24
Dengue and Malaria	25
Avian Pox (<i>Poxvirus avium</i>) and Avian Malaria (<i>Plasmodium relictum</i>)	26
Chytrid Fungus (<i>Batrachochytrium dendrobatidis</i> – Bd)	27
Climate Change Mitigation	27
Biofuels	28
Mountain Pine Beetle (<i>Dendroctonus ponderosae</i>)	30
Recommendations	31
Policy and Management	31
Science and Research	34
Glossary	37
References	38



S. Burgiel

Introduction and Background

Individually, climate change and invasive species present two of the greatest threats to biodiversity and the provision of valuable ecosystem services. The estimated damage from invasive species worldwide totals more than US \$1.4 trillion annually – 5% of the global economy – with impacts across a wide range of sectors including agriculture, forestry, aquaculture, transportation, trade, power generation and recreation (Pimentel et al. 2001). In environmental terms, islands, by example, with their unique and varied biodiversity have suffered disproportionately from invasive species, which are responsible for half to two-thirds of all species extinctions (Donlan and Wilcox 2008, IUCN 2009b). In comparison, economic projections of global climate change-induced losses may range from 1-20% of gross domestic product, which is equally about 5% of GDP annually (Stern 2006). These projections alone should be enough to make officials responsible for national development to take notice and take action.

Combined, the complexity of the interaction of these two global drivers – climate change and invasive species – increases dramatically, and evidence is rapidly growing on how climate change is compounding the already devastating effects of invasive species. Climate change impacts, such as warming temperatures and changes in CO₂ concentrations, are likely to increase opportunities for invasive species because of their adaptability to disturbance and to a broader range of biogeographic conditions and environmental controls. The impacts of those invasive species may be more severe as they increase both in numbers and extent, and as they compete for diminishing resources such as water. Warmer air and water temperatures may also facilitate movement of species along previously inaccessible pathways of spread, both natural and human-made.

From a food security perspective, there is little point in addressing the impacts of climate change on the productivity of a staple food if the crop has already been decimated by an invasive pest. Similarly, from a conservation perspective, there is little point to addressing climate change if the biodiversity we're trying to protect has already been lost to invasive species. Major agricultural outbreaks or health pandemics could result in significant human suffering and loss.

So what can we do? Ecosystem-based adaptation is gaining attention as a cost-effective means of protecting human and ecological communities against the impacts of climate change (Heller and Zavaleta 2009, Mooney et al. 2009, World Bank 2009). Ecosystem based-adaptation is described as building nature's resilience to the impacts of climate change, while also helping to meet people's basic needs.¹ Invasive species can threaten those basic needs and compromise ecosystem functions by taking advantage of habitat disturbance, species under stress and other chinks in the armor of otherwise healthy systems. This affects the multiple roles of ecosystems in providing provisioning, regulating, supporting and cultural services (Millennium Ecosystem Assessment 2005, Vila et al. 2009). Such ecosystem-based approaches are thereby not simply about saving ecosystems, but rather about using ecosystems to help "save" people and the resources on which they depend. Such an approach can also provide an integrative framework to address impacts from both climate change and invasive species.

¹For a more comprehensive list of terms and their definitions, see the glossary included at the end of this report

Climate change and invasive species are not specific to any one geography or ecosystem, yet their interacting dynamics range from global patterns down to local sites and communities of species. While the science on the complex interactions of such global change processes continues to evolve, action is clearly needed to mitigate against the combined effects of climate change and invasive species. Fortunately, we already know many of the key policy and management solutions to address the threat of invasive species,² as well as some of the broader strategies on how to adjust to global change and the increasing uncertainties in the world around us. These are actions that we should urgently be taking to protect plant, animal and human health along with our natural ecosystems. Adding climate change to the mix increases the urgency for managing invasive species, while also increasing the complexity around their behavior and corresponding management needs. Moreover, climate change can also provide additional direction on how to prioritize management efforts around the most critical ecosystem functions to maintain.

This report is targeted at policy-makers, particularly those responsible for developing climate mitigation and adaptation strategies that address issues like conservation, ecosystem services, agriculture and sustainable livelihoods. It focuses on the primary linkages between invasive species and climate change, as well as the secondary and tertiary interactions of their corresponding impacts. Finally, the enclosed recommendations are intended to provide guidance on the best ways to integrate invasive species prevention and management into the consideration of climate change responses across a range of sectors.

Building on a review of existing scientific and conservation literature (which is frequently centered on well-studied invasive species in developed countries), our research has reaffirmed that there are significant gaps and questions about the intersection of these two major drivers of change. The case studies included below highlight key relationships and questions related to invasive species, climate change and the role of ecosystem-based adaptation.

The three key messages that can be distilled from this report are:

1. Climate change will have direct and second order impacts that facilitate the introduction, establishment and/or spread of invasive species.
2. Invasive species can increase the vulnerability of ecosystems to other climate-related stressors and also reduce their potential to sequester greenhouse gasses.
3. Using an ecosystem-based adaptation approach, these pressures on ecosystems and their ability to provide important services can be offset by preventing the introduction of new invasive species and by eradicating or controlling those damaging species already present.

²For the purposes of this paper, invasive species management is viewed as encompassing prevention, eradication and control of invasive species and their spread. This includes an hierarchical perspective where the preference, pending resources and capacity, is first to prevent, second to eradicate and third to control biological invasions (CBD 2002, Wittenberg and Cock 2001).



S. Burgiel

Climate Change and Invasive Species Interactions

The Intergovernmental Panel on Climate Change (IPCC) estimates that mean surface temperature has increased by 0.6 °C on average over the last century (IPCC 2002). Over the next century, the IPCC predicts that global average warming from pre-industrial times will be in the range of 1.1 to 4.6°C. Increases in ocean temperatures have already been observed with an estimated average increase of 0.10°C from 1961-2001 (IPCC 2007b, Levitus et al. 2009). These shifts are not uniform, as there is significant variation at the regional and national scales with more pronounced temperature increases in the higher latitudes. Temperature shifts have also coincided with hydrological changes (e.g., precipitation patterns, ground-water levels), sea level rise and increased CO₂ concentrations. Additionally, adverse impacts of climate change will vary across ecosystem types with particular vulnerability in freshwater habitats, wetlands, mangroves, coral reefs, Arctic and alpine systems, and cloud forests (SCBD 2009).

These and other ecosystems are also subject to a range of invasive species and a key question now being put to policy-makers, scientists and resource managers is how invasive species will interact with climate change at the site level. A number of researchers and experts have examined the issue from different angles including pathways of invasion (Sutherst 2000, Hellman et al. 2008), freshwater ecosystems (Kolar and Lodge 2000, Rahel and Olden 2008), marine and coastal ecosystems (Carlton 2000, Hershner and Havens 2008), forests (Willis et al. 2010) and more general overviews (Capdevila-Argüelles and Zilletti 2008, Low 2008, Mainka and Howard 2010). The following sections look at the direct and second-order effects/linkages between climate change and invasive species by exploring the broad relationships and by providing some indicative examples. These issues will be addressed in more detail within the context of the case studies reviewed below.

Direct Impacts

The array of anticipated climatic and biogeographic changes has significant implications for species, both native and non-native. One can view the particular set of ecological and climatic conditions or parameters necessary for a species' survival as its bioclimatic envelope. A shift in environmental variables, such as temperature and water availability, will have implications for species, particularly if variables shift outside the range of the species' bioclimatic envelope for survival. This may prompt species to migrate to new areas where conditions may be a better match or to simply go into decline if such movements are not biologically or physically possible. Relationships with symbiotic hosts, presence/absence of predators and other ecological dynamics will also play a significant role in regulating population sizes.

Competition and Range Shift: Invasive species are generally viewed as having a broader range of tolerances (i.e., a bigger bioclimatic envelope) than natives, thereby providing invaders with a wider array of suitable habitats (Walther et al. 2009). A shift in temperature, for example, might then have significant impacts on a native species, but little impact on an introduced species, thereby altering the competitive dynamic between them. In some cases temperature alone may not be a determining factor. For example, with invasive plants, changes in precipitation patterns, elevated CO₂ levels and increased nitrogen deposition may play a greater role (Richardson et al. 2000). Under controlled conditions C₃ plants tend to respond more favorably than C₄ plants

to increased CO₂ concentrations, yet other dynamics, such as temperature and moisture availability, will have additional impacts on plant growth (Dukes 2000). It is therefore necessary to look at the full suite of variables relevant to a particular species' bioclimatic envelope, as well as its broader symbiotic relationships and trophic webs.

Changes in competitive dynamics will not be uniform globally, particularly when considering changes across tropical vs. temperate systems or low vs. high altitude systems. Higher latitudes and altitudes will probably see a shifting range of species as temperatures increase and "new" species migrate from adjacent, previously warmer climates (Parmesan 2006). As tropical systems warm they will not face the same threat as there is no pool of species coming from even warmer climates. However, changes in precipitation and other climatic variables may still stress such ecosystems, thereby increasing their vulnerability to invasive species. In addition to range expansion, there may also be range contraction or diminished impacts of invasive species pending the influence of climatic and other variables (Hellman et al. 2008, Richardson et al. 2000).

Facilitated Movements: Climate change will also increase the severity of extreme weather events. Strong winds, currents and wave action can facilitate the movement of invasive species at regional and global scales. For example, during the 2005 hurricane season, the cactus moth (*Cactoblastis cactorum*) was likely blown from host islands in the Caribbean to Mexico where it poses a significant ecological and economic threat to over 104 species of *Opuntia*, 38 of which are endemic (Mafokoane et al. 2007, March 2008). Red palm mite (*Raoiella indica*), a major pest of fruit-producing palm trees and other ornamental plants, has spread throughout the Caribbean mostly likely by a combination of major storms and hurricanes as well as on infested plants and seeds (*Red Palm Mite Explosion 2007, Welbourn 2009*). Similar phenomena have been observed elsewhere, for example in Swaziland in 1984, Cyclone Dementia blew seeds of *Parthenium hysterophorus* (locally nicknamed the Dementia weed) across the landlocked country. This plant's subsequent spread has had major impacts on agricultural production, indigenous hunting areas and wildlife reserves (IRIN 2002, IRIN 2010).

In some cases, such as the south-central US, flooding has helped to spread invasive species that had been contained in aquaculture farms or captive breeding facilities. Silver and bighead carp (*Hypophthalmichthys molitrix* and *H. nobilis*), which were used to maintain aquaculture and wastewater treatment facilities, escaped into the Mississippi River after major floods in the early 1980s, and now threaten the Great Lakes (Schofield et al. 2005, Sea Grant Pennsylvania 2007). In the Northern Territory of Australia, flooding was also largely responsible for the rapid spread of *Mimosa pigra*, a Class 1 weed of national significance (Lonsdale 1993).

While it will be difficult to directly attribute future events like these to climate change as opposed to *El Niño* or other causes, changes within global circulation of air and water as well as more severe weather events can clearly play a facilitating role in the movement of species.

Another facet of climate change is the movement of species either deliberately for conservation or other purposes. Some scientists and conservationists have proposed the notion of assisted migration for species threatened by climate change as a means to protect endangered biodiversity. Such proposals should consider the broader ecological dynamics of these proposed introductions lest the species targeted for conservation become invasive in another ecological setting.³ Cultivation of species



Cactus moth (*C. cactorum*). Caleb Slemmons



Red palm mite (*R. indica*). Eric Erbe; colorization: Chris Pooley



S. Burgiel

for biomass or biofuel may also rapidly spread invaders to new habitats or increase their propagule pressure where they are already present (Raghu et al. 2006). Tools such as risk and environmental impact assessments should be applied in all of these cases to reduce the risk of biological invasions.

Native vs. Non-native Invasive Species: Outside of the potential for increased competition from introduced species, changes in ecosystems may create conditions that favor a particular native species, change pre-existing population dynamics or shift distribution ranges. In view of the discussion above on the changing nature of competition among species, there is the potential for native species to have increased impacts within their ranges and existing communities. Such phenomena are all too common in aquatic environments, where an increase in salinity or pollution can inhibit some native species while proving advantageous to others. Regardless of the native vs. non-native distinction, management responses will need to focus on all types of damaging species. Existing experience with invasive *alien* species can obviously help in this regard, as will basic prevention efforts to minimize the overall pool of potentially invasive species.

For example, in North America, the native mountain pine beetle preys on species of pine particularly in Colorado and other western states of the U.S. as well as British Columbia and parts of Alberta in Canada. Warmer winter temperatures over the past several years have not been sufficient to induce high levels of mortality, thereby leading to a growing outbreak of pine beetles and significant die-off of pines. In some parts of Africa, malaria may be present, but the parasite is unable to complete its lifecycle within its host (*Anopheles gambiae*) under current climate and altitude conditions. However, warmer temperatures may allow for outbreaks in areas previously regarded as safe. Increased adverse impacts of native species may also be seen in the agricultural context where some native species with weedy traits may have an advantage over cultivated crops and plants.

Classifying native species as invasive is an extremely challenging problem with obvious ramifications spanning from on-site management efforts to the application of legal and policy frameworks. Countries and international agreements have traditionally framed the issue of invasive species as being those that are non-native.⁴ This will become a definitional issue where relevant legislation, regulations, funding and response mechanisms are specifically designated for alien or non-native invasive species (Walther et al. 2009). It will thereby require a shift to incorporate native species that present significant damage into existing policy, institutions and funding structures intended for invasive alien species as response tools and mechanisms will be similar in many cases.

Sequestration Impacts: Invasive species may have a feedback effect that further exacerbates climate change. Invasive species can compromise the ability of intact ecosystems to sequester carbon which helps offset greenhouse gas emissions.

³The pros and cons of such movements are the subject of increased debate within the conservation community. Currently, IUCN has established a Task Force on Moving Plants and Animals for Conservation Purposes convened by IUCN's Invasive Species and Re-Introduction Specialist Groups, to further explore the issue and consider development of guidance.

⁴For example, the Convention on Biological Diversity specifically addresses those species that are alien, i.e., outside their natural past or present distribution (CBD 2002).

The mountain pine beetle, discussed above, and other forest pests, such as *Dothistroma* needle blight (caused by the fungus *Dothistroma septosporum*) a major pest of pine plantations in the Southern Hemisphere, have the potential to increase tree mortality, thereby decreasing the amount of CO₂ that can be sequestered (Woods et al. 2005). Similarly, the combination of invasive grasses and fire in tropical systems can displace native forests and thereby reduce carbon sequestration in those systems.

Of particular management concern are those invasive species that may actually increase sequestration in a system, such as the invasive Chinese tallow tree (*Triadica sebifera*) in the coastal prairie of Texas. Using a singular focus on climate change mitigation, policy-makers might view a species' sequestration benefits without regard for its broader adverse ecological impacts. Further work using a long-term perspective on the sequestration capacity of successional systems (e.g., comparing the sequestration capacity of a regenerated forest vs. a new composition of plants and other species) and consideration of the broad suite of ecological values and services is necessary to inform management decisions in this area.

Indirect and Secondary Impacts

Climate related impacts may also facilitate biological invasions without necessarily being the direct source of their introduction. Particular areas of concern include the role of disturbance events and their impacts on ecosystems, as well as ongoing shifts in species composition and trophic chains responding to climate change. These phenomena may also be linked to broader feedback effects that increase ecosystem vulnerability to the establishment and spread of invasive species (Campbell et al. 2009). Changes in soil composition, flood and drought cycles, fire regimes, and glacial extent and warming permafrost may all provide fertile ground for invasive species. Finally, the range of human responses to climate change, both intentional and unintentional, will influence the impact of invasive species.

Disturbance Events: As previously mentioned, climate change will have a host of impacts including increasing the intensity of severe weather events. As a general rule of thumb, experts suggest that wet regions will likely get wetter and dry regions will likely get dryer (although more precise modeling exists for many areas). This is likely to amplify both flooding and drought particularly if rains are concentrated either seasonally or within individual storms. Both of these processes will stress local ecosystems providing a potential foothold for species that are more tolerant to such extreme conditions or able to thrive with these disturbances.

In addition to the direct movement of species (see section above on facilitated movements), the damage caused by storms will increase disturbance in habitats providing opportunities for the establishment and/or spread of already extant invasive species. For example, after the major tsunami in southeast Asia in 2004, Sri Lanka witnessed a significant expansion of prickly pear cactus (*Opuntia dillenii*), mesquite (*Prosopis juliflora*), lantana (*Lantana camara*) and Siam weed (*Chromolaena odorata*) in degraded coastal areas, as well as of water hyacinth (*Eichhornia crassipes*) and cattails (*Typha angustifolia*) in lagoons and estuaries (Bambaradeniya et al. 2006). On Rarotonga, part of the Cook Islands, invasive balloon vine (*Cardiospermum grandiflorum*) and mile-a-minute vine (*Mikania micrantha*) are strangling native forests, after seeds presumably from ornamental introductions were widely spread in 1987 by Cyclone Sally to areas severely disturbed by the storm (pers. comm. John Waugh 2010).



Relief efforts responding to such natural disasters have the potential to unintentionally introduce invasive species through foodstuffs containing non-native seeds/propagules or on construction, firefighting, military or other vehicles used in other places. Deliberate introductions may occur through restoration or development projects aimed at quickly rebuilding local economies, such as the push to develop non-native biofuel species in Haiti after a major earthquake in 2010.

Changes in hydrological patterns, such as drought, as well as outbreaks of forest pests and pathogens and increased litter/fuel accumulation from invasive plants, will impact fire regimes by potentially increasing their frequency and severity in areas where they naturally occur and by creating fire prone areas where fire was not previously part of the ecosystem dynamics (D'Antonio 2000, Dukes 2000). In some cases, such as invaded wetlands, ecosystem dynamics may shift to the point where fire regimes play a greater role than traditional hydrological cycles in regulating species interactions (Hogenbirk and Wein 1991, as cited in D'Antonio 2000).

For example, in the Caribbean, invasive plants are changing traditional fire regimes and hydrological cycles in both fire tolerant and fire intolerant systems. In Puerto Rico, the incursion of buffelgrass (*Pennisetum ciliare*) into Guanica Dry State Forest has created fuel for frequent grass fires that are adversely impacting native fire-intolerant grasses and creating a feedback loop prompting the further expansion of buffelgrass. Similarly, old world climbing fern (*Lygodium microphyllum*) is increasing the intensity of fires by creating ladders for it to travel into tree canopies in the Bahamas and southern Florida. This increases the mortality of species that are adapted to low-heat, ground level fires, such as Caribbean pine (*Pinus caribaea*), which is native to the Bahamas, Cuba, the Turks and Caicos and parts of Central America (TNC 2002, Caribbean and Florida Fire and Invasives Learning Network 2009). Finally, invasive grasses such as *Arundo donax*, cheatgrass (*Bromus tectorum*), gorse (*Ulex europaeus*) and kikuyu grass (*Pennisetum clandestinum*) are known to increase fire loads and heat intensity, leading to greater mortality in some fire-dependent species and more opportunities for invasion by non-native species.

Changes in Species Composition and Ecosystem Function: The broad categories of climate change impacts on species composition and ecosystems are gradually becoming better defined, however the full implications of these types of changes, particularly at the site level, are still unknown and could be unique to each case. Observed areas of impact include changes in the geographic range of species, their phenology, as well as photosynthetic rates, carbon uptake and productivity (SCBD 2009).

Taken together these dynamics will affect interactions between species and more broadly community composition, trophic webs and corresponding ecosystem functions. For example, earlier flowering dates for plants may not coincide with the emergence of symbiotic pollinators. Similarly, from a management perspective the efficacy of species used for biological control may vary depending on changes in the development, morphology and reproduction of the targeted invasive species (Ziska 2005). Increased herbivory and reproduction rates from insects and mammals due to warmer temperatures and longer seasons may impact plant reproduction. Together these individual interactions may have compounded effects on broader ecosystem services such as groundwater retention and filtering, pollination, disease suppression and carbon sequestration.

The outcome of these changes on the ability of invasive species to establish and spread is closely linked to the threat of invasive species from competition raised in the section above on Direct Impacts. The differentiation is simply to note progressions in ecosystem change (i.e., invasive species being the initial cause of ecosystem change vs. invasive species responding to ecosystem change induced by other drivers).

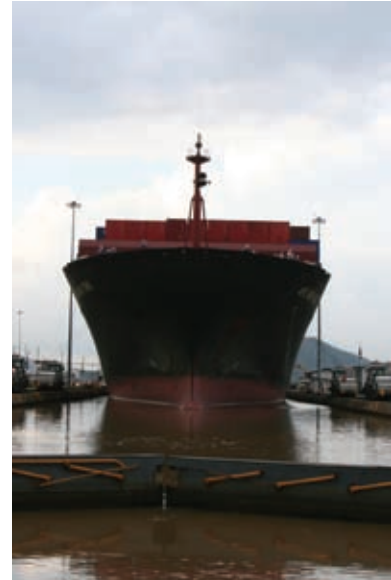
Social Interactions and Responses: Finally, society's response to climate change and its impacts will affect the potential introduction and spread of invasive species. Hardy species that are fast growing, adaptable to harsh conditions, tolerant of disturbance and highly productive will increasingly be in demand for agriculture, forestry, aquaculture, biofuels and other sequestration activities. Coincidentally the traits of such species closely match those of invasive species and in many cases known invasive species have already been proposed or utilized. For example, a significant number (if not majority) of species proposed as biofuels, such as giant reed (*Arundo donax*), castor bean (*Ricinus communis*), pampas grass (*Miscanthus sinensis*), Johnson grass (*Sorghum halepense*) and *Jatropha curcas* are known invasive species in some part of the world (Low and Booth 2007, Barney and DiTomaso 2008, GISP 2008).

At a global scale, changing trade patterns and routes will also increase the potential for the introduction of non-native species into new environments. Higher temperatures and changes in precipitation will have significant impacts on agricultural productivity, and consequently will result in shifts in production. Changes in the trade of agricultural commodities will have impacts on the transport networks used to move such goods and the inherent invasive species risks associated with vectors like ballast water, hull fouling, aviation and ground transport. Receding ice formation in the Arctic is already opening a northwest passage for ships to move cargo and is creating new opportunities for the exploitation of oil, gas and other natural resources. This will significantly increase the exposure of these relatively pristine areas to invasive species introduced through ships' ballast water, as well as hull fouling, drilling platforms and other equipment.

Depleted water supplies, land degradation and sea level rise will likely lead to the mass movement of peoples and even climate refugees. Population migrations, increases in density, as well as poor sanitary conditions may create vectors for the spread of disease, a phenomenon that will likely be compounded by expanded ranges of diseases like malaria, dengue and yellow fever under warming climate scenarios (Reiter 2001). These people will likely bring desirable crops, domestic animals, and ornamental species to their new homes, potentially speeding dispersal of new non-native species.

Managing invasive species must be considered as a front-line strategy in adaptation to climate change. In adapting to climate change, humans will actively build their defenses to a changing climate both through the development of hard infrastructure (e.g., sea walls, water delivery systems) as well as practices designed to enhance the existing role played by natural systems.

The potential for the introduction and spread of invasive species must be considered in the development of national adaptation and mitigation strategies. Alternative energy strategies may consider known invasive plants for use as biofuels, or large-scale wind and solar areas may disturb intact ecosystems thereby introducing new invasive species. Used proactively, proper risk assessment of particular species for deliberate introduction along with broader environmental impact assessments for infrastructural developments can help minimize these concerns.



S. Burgiel

Ecosystem-Based Adaptation and the Maintenance of Ecosystem Services

Society can be proactive in enhancing the resilience and adaptation of ecosystems and their services in the face of both climate change and invasive species. Ecosystem-based adaptation has been defined as:

the use of biodiversity and ecosystem services as part of an overall adaptation strategy to help people adapt to the adverse effects of climate change. Ecosystem-based adaptation uses the range of opportunities for the sustainable management, conservation, and restoration of ecosystems to provide services that enable people to adapt to the impacts of climate change. It aims to maintain and increase the resilience and reduce the vulnerability of ecosystems and people in the face of the adverse effects of climate change (SCBD 2009).

Biodiversity is valued by society for a wide range of reasons from the functional to the aesthetic. Healthy ecosystems thereby provide a wide range of ecosystem services that serve as the background and backbone for the production of necessities like food and fiber, building materials and potable water. Many of our cultural practices and traditions were developed and depend on particular ecological elements or functions and this dependence has been ingrained over countless generations. Invasive species, along with climate change, are a critical threat to many of these fundamental relationships.

Protecting forests, wetlands, coastal habitats and other natural ecosystems can provide social, economic, and environmental benefits, both directly through more sustainable management of biological resources and indirectly through protection of ecosystem services. Protected areas, and the natural habitats within them, can protect watersheds and regulate water flow and water quality; prevent soil erosion; influence rainfall regimes and local climate; conserve renewable harvestable resources and genetic reservoirs; and protect breeding stocks, natural pollinators, and seed dispersers, which maintain ecosystem health. Floodplain forests and coastal mangroves provide storm protection and act as safety barriers against natural hazards such as floods, hurricanes, and tsunamis, while natural wetlands filter pollutants and serve as nurseries for local fisheries. Better protection and management of key habitats and natural resources can benefit poor, marginalized and indigenous communities by maintaining ecosystem services and maintaining access to resources during difficult times, including in times of drought and disaster. (The World Bank 2009)

In addition to the broader value of ecosystem services for society, there is also an inextricable link between poverty and the loss of ecosystem services and corresponding biodiversity. Subsistence livelihoods, particularly those relating to farming, animal husbandry, fishing and forestry, are the most immediate beneficiaries of healthy ecosystems and their services (TEEB 2008). Efforts to combat invasive species have a long history in these areas. The critical realization for those outside the invasive species community is that the impact of invasive species often deemed as exclusively agricultural or environmental in scope is now having broader repercussions on other sectors critical for maintaining human societies. This crisis is fostered by globalization involving increased levels and volumes of trade, faster modes of transport, and the diminishing distance between communities around the world as well as between

human settlement and less accessible centers of biodiversity. Unfortunately, overlaying climate change provides an additional suite of pressures, challenges and fears.

Ecosystem-based adaptation is essentially the development of management activities to enhance the resilience of ecosystems providing critical services in the face of climate change. A key element of those management efforts is the reduction of other major threats, which when compounded with the effects of climate change would push a system beyond its ability to function properly. Major international environmental policy is beginning to recognize this connection. The Convention on Biological Diversity is prioritizing management of the major drivers of global change as critical for halting the present rate of biodiversity loss. Similarly, discussions under the U.N. Framework Convention on Climate Change are also highlighting the role of mitigating and adapting to the effects of climate change by protecting the natural systems around us.

Invasive species are clearly one of those major stressors and are recognized as a direct driver of biodiversity loss. Thus existing methods and efforts to manage invasive species can potentially serve a major benefit by increasing the ability of species and ecosystems to withstand climate related impacts. The identification of key ecosystems and the services they provide can also inform the risk assessment, planning and prioritization processes for regulating existing invasive species and the pathways by which they are introduced.

Case Studies

The following section looks at the intersection of these three areas – invasive species, climate change and ecosystem services – within a broader framework of critical ecosystem processes of importance to a range of human concerns and sectors. As mentioned, our scientific knowledge of the intersection of these elements is limited, and we have tried to identify examples that can highlight critical facets and threats, while also raising the key questions that need to be addressed by the scientific and policy-making communities. More specifically, we have focused on those ecosystem goods and service that we want to protect and maintain (e.g., freshwater availability, food security). In some cases the lack of available research has forced us to include more hypothetical analyses to illustrate the problem. Work around both climate change and invasive species has traditionally operated with some degree of uncertainty, and this level of uncertainty will likely increase. However, we can still make “good” management decisions based on existing knowledge and practice to address the threat of invasive species.

Coastal Protection and Integrity

Two major consequences of climate change are the likely increase in storm severity and sea level rise. Taken together these phenomena can have major impacts on coastal systems, communities and infrastructure by increasing erosion, salinity levels and storm damage from winds, flooding and storm surges. Healthy coastal ecosystems play a role in buffering many of these effects and thereby protect both biodiversity and human settlements. For example, experts contend that the degradation and destruction of low-lying island systems and wetland areas off the coast of Louisiana, U.S., allowed Hurricane Katrina to hit the city of New Orleans in 2005 with significantly more impact than it otherwise would have (Shafer et al. 2009). Similarly, an examination of areas impacted by the southeast Asian tsunami of 2004 shows that areas with more intact ecosystems fared better than areas where coastal ecosystems had been developed or otherwise transformed (Mascarenhas and Jayakumar 2008, Kaplan et al. 2009).



S. Burgiel

While neither the effects of Hurricane Katrina nor the tsunami may have been attributed to climate change, climate change may contribute to similar types of damage from future incidents. Additionally, where invasive species have had significant impact on the integrity of coastal systems, this may increase ecosystem vulnerability to weather-related phenomena and sea level rise. The section below includes examples of invasive species in coastal ecosystems – beach vitex, nutria and *Miconia calvenscens* – and their interactions with climate change.



Beach vitex (V. rotundifolia) (Hilo, Hawaii, US).
Forest & Kim Starr

Beach Vitex (*Vitex rotundifolia*)

Beach vitex is a perennial shrub of coastal sand dunes and is native to Asia and the islands of the Pacific. It was introduced into the coastal areas of the southeastern U.S. in the mid-1980s both for ornamental horticulture and dune stabilization through erosion control. Unfortunately, beach vitex caused significant loss of dunes and coastal habitat in North and South Carolina because its root system causes high rates of erosion (Westbrooks and Madsen 2006). The erosion has had a twofold effect of increasing the vulnerability of homes and other coastal infrastructure to storms and gradual erosion, as well as causing habitat loss for native species. The plant can also deter female sea turtles from laying their eggs and entangle newly emerged sea turtle hatchlings. The environmental impacts of beach vitex may interact with climate change to further compromise the integrity of dune systems under heightened storms and sea level rise. Storms and other rough weather can aid in its spread as the plant disperses through runners and plant fragments (Gresham and Neal 2004, Cousins et al. 2010). Significant management efforts are being made in the U.S. by available chemical and mechanical control methods, yet despite its impact the plant is still being sold commercially in some regions of the U.S.



Nutria (*M. coypus*). Or Hiltch

Nutria, Coypu (*Myocastor coypus*)

Nutria/coypu (*Myocastor coypus*) are a species of aquatic rodent living in marshes, swamps and other coastal brackish water systems with abundant vegetation. Native to South America, nutria were introduced into parts of North America, Europe, Asia and Africa primarily for fur production. The animal feeds on the rhizomes and young shoots of aquatic vegetation which significantly impacts plant cover and the bird, fish, plant and invertebrate species depending on this habitat (Carter and Leonard 2002). In coastal areas of Louisiana, Maryland and Mississippi, U.S., nutria have converted large sections of marshland into open water, thereby increasing saltwater encroachment, decreasing natural processing of rainwater runoff and increasing inland vulnerability to storm surges and erosion (Carter et al. 1999, pers. comm. Steve Kendrick). Current management consists of harvesting nutria through hunting, trapping and use of dogs, which are time consuming and costly methods (Panzacchi et al. 2007). The spread of nutria in combination with predicted severe storms and sea level rise could compromise the ability of marshlands to provide important ecosystem services, like coastal protection, natural habitat and spawning grounds.

Miconia calvenscens

Miconia calvenscens is a tree species native to Central and South America that is now proving to be a major invasive plant in the forests of Tahiti, Hawaii, Sri Lanka and other parts of the Pacific, Asia and the Caribbean. Originally introduced in 1937 as an ornamental plant into Tahiti, by 1996, *M. calvenscens* had spread to 65% of Tahiti with mono-specific stands on approximately 25% of the island (Meyer 1996). In Tahiti, the introduction and establishment of *M. calvenscens* is dramatically transforming native habitats. The roots of *M. calvenscens* destabilize soils, giving rise to landslides and erosion on steep slopes in the presence

of substantial rainfall. In Tahiti and Hawaii this has subsequently increased levels of silt in riverine, coastal and reef communities. In regions like the South Pacific Ocean, precipitation is projected to increase under climate change in certain seasons, which will likely exacerbate such landslides and their environmental impacts on soil erosion, reef fishery viability, and watershed functions (Christensen et al. 2007).

The impact of the *M. calvescens* invasion indirectly affects coastal biodiversity and ecosystem services and creates public hazards. Experts calculate that 40-50 endemic Tahitian plant species could be at risk of extinction by its spread. In terms of management, a fungal pathogen, *Colletotrichum gloeosporioides forma specialis miconiae* (Deuteromycotina), has been introduced in Tahiti and shows some positive signs of enabling the regeneration of endemic plant species (Meyer and Florence 1996, Meyer et al. 2007).

Fisheries and Marine Ecosystems

The marine environment provides important resources and services that support people living in coastal areas and beyond through fisheries, temperature regulation and protection against coastal erosion and storm surges. Yet, many of us are unaware of the sensitivity of marine ecosystems to invasive species and climate change. Recent research indicates that some marine communities may be affected by climate change more rapidly than terrestrial communities, due to the spread of invasive species (Sorte et al. 2010). As coastal marine and estuarine habitats are being impacted by invasive species, climate change is altering ocean temperatures, chemical processes, currents and sea levels (Bindo et al. 2007).

One of the main intersections between invasive species and climate change in the marine environment has been shifts in ranges of invasive species towards the poles. Increases in ocean temperature of less than 2°C have enabled species that were limited by temperature to expand their ranges by considerable distances. In comparison to terrestrial invasive species, marine invasive species generally spread over an order of magnitude faster (Sorte et al. 2010). For example, more than a decade ago Carlton had already reviewed significant expansion of ten marine species, including mollusks, crustaceans, sea squirts, hydroids and bryozoans, along the Pacific coast of North America (Carlton 2000). Occasionally, these range shifts or expansions can also be caused by extreme weather events, like *El Niño*, which is characterized by stronger and more persistent currents that can transfer a species several hundred kilometers (Huyer et al. 2002).

In addition, there are other stressors on the marine environment that can increase its susceptibility to biological invasion. The structure of marine and coastal ecosystems will likely be altered by:

- increased turbidity;
- changes in primary production caused by variations in UV light penetration and precipitation levels;
- changes in salinity, particularly in estuarine systems;
- the impacts of ocean acidification on corals; and
- the loss of framework species essential for maintaining ecological patterns (Smith 2010, Carlton 2000).

For example, climate change is causing an increase in the acidity of the ocean, which threatens all marine organisms that rely on calcium carbonate for their physical structure. The potential ramifications for corals of ocean acidification in tandem with disease and other stressors could be large-scale mortality across coral reef communities. Such disturbance events and the empty niches they create will likely facilitate the establishment and spread of invasive species, particularly algae and sea grasses.

The intersection between invasive species and climate change impacts may have unexpected and irreversible consequences for fishery stability, and, therefore, the economy, food security and local livelihoods (Harris and Tyrrell 2001, Stachowicz et al. 2002). The examples below illustrate the potential impacts of marine invasive species on fisheries and marine systems due to their expanded ranges under climate change.



Lionfish distribution in the Caribbean, Gulf of Mexico and Atlantic as of January 2009 (as included in Freshwater et al. 2009).



Lionfish (*P. volitans*) (the Bahamas).
Willy Volk

Lionfish (*Pterois volitans*)

The lionfish is native to coral reefs in the sub-tropical and tropical regions of the South Pacific, Indian Ocean and the Red Sea. Outside its native range, the lionfish is a voracious predator with venomous spines that negatively impacts other fishes, especially native coral reef and mangrove species, shrimp and crabs (Fishelson 1997). Because of their venomous spines, lionfish have no predators in their invaded range and this helps their populations to continue spreading (Meister et al. 2004, Whitfield et al. 2007).

After releases of aquarium specimens into waters of southern Florida in the mid-1980's, the lionfish quickly became invasive (Whitfield et al. 2002, Albins and Hixon 2008). In the following decade, lionfish were sighted in the Caribbean Sea and northward along the eastern seaboard of North America where they threaten coastal habitats and fisheries (Whitfield et al. 2007, Freshwater et al. 2009). Initially, lionfish were not thought to survive winter temperatures in the northern Atlantic Ocean, but warming ocean temperatures have enabled the lionfish to establish and impact local ecosystems (Kimball et al. 2004, Meister et al. 2004, Whitfield et al. 2007, Albins and Hixon 2008). Small changes of 1°C in winter bottom water temperatures have already shifted the species balance in some marine ecosystems from temperate towards tropical communities (Parker and Dixon 1998). In certain areas of the invaded southern Atlantic, lionfish populations are now as abundant as the native groupers (Whitfield et al. 2007). Their physical aggression and overcrowding may eventually displace native species, thereby negatively impacting commercial and subsistence fishing of native groupers (Taylor et al. 1984, Moulton and Pimm 1986).

The harmful impact of lionfish is particularly pronounced in the Bahamas where the species is impacting coral reef and mangrove ecosystems. There, lionfish populations are five times denser than in their native range which compounds their higher rates of consumption on reef fish, crustaceans, invertivores, herbivores, invertebrates and planktivores (Fishelson 1997, Green and Cote 2009, Morris and Akins 2009, Barbour et al. 2010). The invasion of lionfish in the Bahamas has shown a documented reduction in the recruitment of native reef fish and may significantly affect fisheries (Albins and Hixon 2008). Lionfish may also threaten the resiliency of coral reefs as they deal with other climate change induced stressors, such as ocean acidification and increased storm frequency.

Control methods for lionfish currently consist of only mechanical harvest by divers, although the poisonous barbs can make this a difficult task. Nonetheless, local harvesting and other monitoring efforts may allow for early detection and rapid response to deter the

further spread of the lionfish invasion, particularly into sites of ecological or socioeconomic value. Currently, eradication is unlikely given the ability of the lionfish to quickly spread and establish (Albin and Hixon 2009, Morris and Akins 2009).

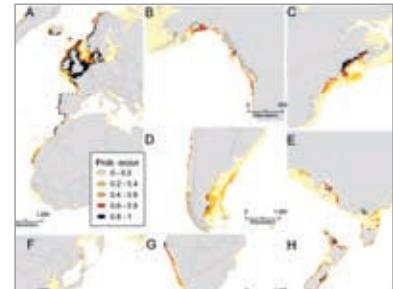
European Green Crab (*Carcinus maenas*) and Chinese Mitten Crab (*Eriocheir sinensis*)

Natural ecosystems are often threatened by more than one invasive species at a time, increasing the complexity of impacts and adding more stress to the system as it also faces climate change and other stressors. An overarching challenge therefore becomes determining how climate change will interact with individual species and how that will scale up to complex marine communities (Williams and Grosholz 2008). San Francisco Bay, one of the world's most invaded aquatic environments, is a virtual laboratory for studying the behavior of invasive species, and in this case the European green crab (*Carcinus maenas*) and the Chinese mitten crab (*Eriocheir sinensis*).

European green crabs, native from northwestern African to northern Europe, were first detected in San Francisco Bay in 1989 (Cohen et al. 1995). The crabs are thought to have arrived via ballast water or from discarded seafood and baitworm packaging (Behrens Yamada and Gillespie 2008). Just three years later, the Chinese mitten crab, native to the rivers and estuaries of Eastern Asia, was introduced into the same estuary via ballast water (Dittel and Epifanio 2009). Estimates have shown that, since their initial introductions, both species have spread over 1,200 km northward along the Pacific coast and to over several thousand km² around San Francisco Bay (Rudnick et al. 2003, See and Feist 2010).

Together in their new invaded range, these crabs are posing a suite of threats to the native ecosystems of the Pacific coast of North America. The European green crab threatens the native bivalve and crab community by being a significant predator of native clams and mussels and a significant competitor with the native Dungeness crab (*Cancer magister*) (See and Feist 2010, see citations therein). The Chinese mitten crab has substantial impact on coastal fisheries since they prey on salmonids and damage fishing nets, bait and overall operations of shrimp and crayfish fisheries (Veldhuizen and Stanish 1999, Hui et al. 2005, Dittel and Epifanio 2009). In areas where Chinese mitten crabs are particularly abundant, juveniles can burrow into banks to the extent that they undermine structural stability, leading to slumping or collapse, erosion of marsh sediments and a decrease in vegetation (Dutton and Conroy 1998, Hui et al. 2005, Rudnick et al. 2005, Dittel and Epifanio 2009). When consumed, this species can pose a health threat to wildlife and humans because it hosts the Oriental lung fluke and bio-accumulates heavy metals, including mercury (Hui et al. 2005).

Given the profound environmental and economic impacts of these two invasive crab species in the San Francisco Bay Delta region, there is much concern about their potential to spread to other areas of the Pacific coast (Dittel and Epifanio 2008). While it is difficult to differentiate the exact relationships between climate change and *El Niño*, it is most likely that the combination of warming waters from climate change and circulation changes from *El Niño* are working in tandem to facilitate the spread of these crabs. The European green crab has been spreading northward into British Columbia as pelagic larvae, with the most significant migration occurring during *El Niño* events (Grosholz and Ruiz 1995, Behrens Yamada and Hunt 2000). Similarly, the Chinese mitten crab has rapidly established northward into British Columbia because of its tolerance for a wide range of conditions and its dynamic population cycles (Rudnick et al. 2003).



Probable range expansion for the European green crab (*C. maenas*): (a) northwestern Europe, (b) western North America, (c) eastern North America, (d) Patagonia, (e) southeastern Australia, (f) Japan, (g) South Africa and (h) New Zealand. High probabilities of occurrence are indicated by dark shading. Predictions are constrained to depths of 200m or less (as included in Compton et al. 2010).



European green crab (*C. maenas*).
Luis Miguel Bugallo Sánchez

Each of these crab species has the potential to disperse long distances and to establish in a relatively wide range of habitats, with a high-risk of invading the coastal estuaries of Alaska. The invasive populations of the European green crab in British Columbia could be the source for an invasion into southern Alaska by *El Niño* cycle, which could quickly transfer larvae several hundred kilometers further north (Huyer et al. 2002, Behrens Yamada and Gillespie 2008). Based on current temperatures, only a few coastal sites in Alaska would be suitable for the European green crab. However, if winters remain mild and the ocean temperature increases 2°C, the number of Alaskan sites at risk of invasion by the European green crab would double (deRivera et al. 2007, Compton et al. 2010). Similarly, a 2°C increase in water temperature would also allow larval survival of Chinese mitten crabs (Hanson and Sytsma 2008).

Alaskan fisheries are integral to the state's economy, ecosystem and cultural heritage, yet control options are currently limited. Anticipating potential range expansions and monitoring for introductions may serve to guide response efforts particularly in key sites as further management options are examined.



Giant reed (*A. donax*). Chuck Bargeron

Freshwater Services and Availability

Climate change is expected to have major impacts on precipitation levels and timing, as well as on broader hydrological cycles. Projections of reduced precipitation and intense drought in some regions will have broad implications for ecosystem function and the people that rely on sustainable water availability (Christensen et al. 2007). Africa is one of the most vulnerable continents to climate change and climate variability, a situation aggravated by a low adaptive capacity and a population that already experiences high water stress (Boko et al. 2007). Global warming and the creation of arid and semi-arid lands in sub-Saharan Africa will be more extreme than the global mean, especially in portions of southern Africa (Boko et al. 2007, Christensen et al. 2007). Throughout the world there are a number of invasive species known to affect freshwater availability and services, including giant reed (*Arundo donax*) and water hyacinth (*Eichhornia crassipes*) – addressed below – as well as *Melaleuca quinquinervia* and species of eucalyptus, acacia, *Tamarix* and *Prosopis*.

Giant Reed (*Arundo donax*)

Giant reed (*Arundo donax*) is native to riparian habitats of eastern Asia and has been introduced throughout the world, where it has often had detrimental impacts on water services and water availability, especially in arid climates. Giant reed was introduced to South Africa for its use as building material and has since become a very widespread and common invasive plant in riparian habitats of rivers and streams (Milton 2004, Nel et al. 2004).

Recognizing the importance of water availability, South Africa adopted water legislation in 1996 requiring the maintenance of the ecological integrity of river ecosystems to protect their capacity to deliver goods and services to people on a sustainable basis. This policy has generated more attention to invasive plants, many of which have a significant impact on water resources (Mgidi et al. 2004). In a South African study, the total incremental water use of invasive plants was estimated to be 3,300 million m³ of water annually (LeMaitre et al. 2000).

Giant reed is having significant impacts on the hydrology of South Africa. As the species invades South African riverbanks, it becomes dominant in dense, monotypic stands that replace native vegetation and decrease wildlife diversity (Co man et al. 2004, van Wilgen et al. 2007). These tall stands of grass have above average water usage (based on per leaf area

transpiration) which can alter stream hydrology and sedimentation, while increasing the risk of flooding (Mgidi et al. 2004). Additionally, giant reed can increase fire incidence and subsequently regrows three to four times faster than native South African riparian plants, thereby ensuring its continued invasion (Coman et al. 2004).

Climate change will likely exacerbate the invasion of giant reed and its impacts on water availability. In general, invasive grasses are projected to increase in South Africa, and this plant is no exception (Milton 2004). Giant reed is capable of tolerating a wide range of environmental conditions and is already climatically suited to 76% of the South Africa region (Mgidi et al. 2004, Quinn and Holt 2008). Experiments in California, U.S., which also has a Mediterranean climate and lists giant reed as a prohibited invader, have shown that temperature is the most influential factor in its growth and survival (Decruyenaere and Holt 2005). Rooting success of stem fragments was 100% at temperatures of 17.5°C and above (Wijte et al. 2005). This experimental evidence suggests that giant reed may become more invasive under climate change.

Climate change may also change the management options for giant reed, particularly given its perceived beneficial uses. In the U.S., where some are trying to control the grass by mechanical and biocontrol means, others are proposing to cultivate it for biofuel (Mack 2008, Goolsby et al. 2009). The species is an attractive candidate for biofuel because of its rapid growth and ease of propagation. However, because it is highly aggressive and known to be invasive in numerous other settings, there is some basis to deny its propagation for biofuel development (Mack 2008).

Water hyacinth (*Eichhornia crassipes*)

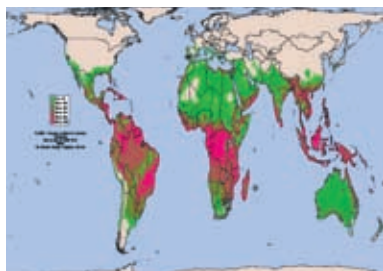
For two decades, the Lake Victoria Basin of Kenya, Tanzania and Uganda has been invaded by water hyacinth (*Eichhornia crassipes*), one of the world's worst aquatic weeds (Ogutu-Ohwayu 1997, Villamagna and Murphy 2010). Water hyacinth, originally perceived as a practical problem for fishing and navigation, is now also considered a threat to water availability, biodiversity and the approximately 30 million people that depend upon the lake in some way (Luken and Thieret 1997, Njiru et al. 2008). Recent research on how water hyacinth may respond to climate change has implications for the continued spread and management of a known invasive weed that is present in 50 countries across five continents (Villamagna and Murphy 2010).

Native to the tropical and subtropical regions of South America, water hyacinth has been present in the African Great Lakes since the late 1980s, and was first reported in Lake Victoria in 1990 (Harley 1991, Twongo 1991). Presumably, the free-floating perennial aquatic weed thrived and spread over time due to its fast growth rate and surrounding anoxic and high-nutrient water conditions (Reddy et al. 1989, Zhang 2010). Boats, machinery and water currents also aided the distribution of water hyacinth throughout Lake Victoria. By 1998, water hyacinth had multiplied to its peak coverage of more than 17,000 ha (Albright et al. 2004).

Impacts of invasive water hyacinth are far-reaching, as it interferes with fishing activities, boating, irrigation, water treatment, hydroelectric power, human health, tourism, and last, but certainly not least, the lake's natural ecosystem (Witte et al. 1992, Ogutu-Owayu 1997, Opande et al. 2004, Williams et al. 2005, Odada and Olago 2006). For example, dense mats of water hyacinth interfere with boat movement, the catch-per-unit-effort for fishermen, and the intake for hydroelectric power generators and the filters for municipal water



Water hyacinth (*E. crassipes*) (Kampala, Uganda). Sarah McCans



Potential distribution of water hyacinth (*E. crassipes*) (as included in EPPO 2008).

supplies (Opande et al. 2004, Villamagna and Murphy 2010). Human health can be impacted when water hyacinth acts as a breeding ground for mosquitoes that transmit malaria and for freshwater snails that transmit bilharzia (schistosomiasis) (Ogutu-Ohwayu 1997, Masifwa et al. 2001, Plummer 2005). One of the most important impacts of water hyacinth is water loss – this aquatic plant significantly increases water loss by high rates of evapotranspiration: 2.7-3.2 times greater than water loss in open water (Penfound and Earl 1948, Lallana et al. 1987).

The water hyacinth invasion in Lake Victoria has varied in its extent, and the use of biocontrols have shown some success (Matthews and Brandt 2004, Julien 2008, Villamagna and Murphy 2010). In particular, two biocontrol weevils, *Neochetina eichhorniae* and *N. bruchi*, were released in Lake Victoria in 1995 (Wilson et al. 2007). It's widely thought that these biocontrol agents were successful at limiting the water hyacinth invasion, whose cover had dramatically decreased by 2000 (Ogwang and Molo 2004, Wilson et al. 2007). Others, however, contend that the climatic *El Niño* conditions played a greater role in decreasing water hyacinth, as associated cloudy and wet conditions would not be ideal for water hyacinth growth (Williams et al. 2005). In all likelihood, the observed decline was due to a combination of various factors, most notably biocontrol and *El Niño* climate conditions, and to a lesser extent, mechanical removal (Albright et al. 2004, Williams et al. 2007). It should be noted that there has recently been a resurgence in particular sites, although that is likely due to continued influx of nutrient-rich waters from specific rivers feeding into the lake.

The uncertainty over the role of *El Niño* raises additional questions about the implications for climate change on water hyacinth. Some research suggests that warmer temperatures in the region will have adverse impacts on water hyacinth as its growth rate is retarded above 30°C (Sato 1998, Julien 2008). Others show that increased CO₂ concentrations can increase the biomass of water hyacinth under controlled conditions (Williams et al. 2005). Finally, the effectiveness of the biocontrol agents themselves is in question, as their efficacy has varied with climatic factors (Hill and Olckers 2001). Regardless of how climate change impacts water hyacinth in Lake Victoria, there is a great need to keep this weed under control given its impact across a range of economic and environmental priorities.

More broadly, based on projections of various climate change factors, water hyacinth is likely to expand its global distribution (EPPO 2008). For example, water hyacinth is currently established in parts of southern Europe but could readily expand to the rest of the Mediterranean Basin and further northward into Europe pending rates of global warming (EPPO 2008). It is likely that such newly invaded regions would suffer impacts similar to those experienced in and around Lake Victoria.

Agriculture, Livestock and Food Security

The effects of climate change will add stress to agricultural systems, specifically by increasing invasive species, including weeds, pests and diseases, that impact crop and livestock production. Agriculture is particularly important, first and foremost because it is critical for providing food for human consumption. A loss in agricultural productivity would also be devastating to the global economy as it directly supports the livelihoods of farmers (36% of the world population) and countless more (2.5 billion people in developing countries alone) through the international market for agricultural goods (ILO 2007, FAO 2008a, Nelson et al. 2009). Moreover, climate change is very likely to result

in price inflation. A decrease in food security – food availability, food access, a stable food supply and stable food utilization – will only intensify as the world's population expands as there is predicted to be a 50% rise in demand for food by 2030 (FAO 2008b, Rangi 2009). Maintaining food security in part requires reducing losses from invasive species, and, more recently, adapting to climate change.

Climate change is predicted to directly affect agricultural production by altering the suitability of present locations to certain crops and thus, crop yield. Climate factors, like variation in rainfall, can determine the physical and economic viability for crop production depending on how sensitive the crop is to climatic changes and how significant those changes are for the region (Liverman and O'Brien 1991, Conde et al. 1997, FAO 2008b). Experts warn that climate change will result in less food security, especially in developing countries and for the resource poor who cannot meet their food requirements (FAO 2008b). Note, however, that climate change is also predicted to benefit production in some agricultural areas by creating climatic factors that benefit plant growth. Even so, the overall impacts of climate change on agriculture are expected to be severely negative (Chang 2002, Nelson et al. 2009).

Indirectly, climate change will impact agriculture by increasing the incidence and intensity of invasive species (Petzoldt and Seaman 2005, Ziska 2005, Rangi 2009). Invasive species, in the form of plants, animals, insects and diseases, are already arguably the largest impediment to global food security and agricultural productivity (FAO 2008a, Rangi 2009). For example, in many countries of Africa, where nearly half of crops are lost to invasive species, the parasitic plant, *Striga hermonthica* causes annual losses in maize of US \$7 billion, adversely affecting 300 million Africans. The maize weevil (*Sitophilus zeamais*), a common pest in most African countries, can destroy up to 40% of stored crops. Similarly, the larger grain borer (*Prostephanus truncatus*) can destroy 70% of dried stores, resulting in crop losses of up to US \$800 million in West Africa alone (Rangi 2009).

Increased outbreaks in invasive pathogens will also result in further economic strain on exporting countries due to trade bans and costs of meeting sanitary and phytosanitary requirements. Evidence shows that the ranges of several important crop insects, weeds and plant diseases have already expanded poleward (Rosenzweig et al. 2000). Earlier onset of warm temperatures could result in an earlier threat from potato/late blight (caused by the invasive *Phytophthora infestans* in many regions) with the potential for more severe epidemics and increases in the number of fungicide applications needed for control (Kaukoranta 1996, as cited in Petzoldt and Seaman 2005). Wheat rust, grey leaf spot, cassava mealy bug, and cactus moth are just a few of the other major agricultural problems whose increased invasion is a key question under future climate scenarios (Zimmermann et al. 2004, Rangi 2009).

As food security is challenged by climate change and invasive species, the management of these threats will also become more difficult. There is an increasing amount of evidence that demonstrates a decline in chemical efficacy of herbicides on invasive plants with rising CO₂ (Ziska 2005). Similar changes in the success of other management strategies, such as fungicides, insecticides and biological control, are also possible under climate change (Chakraborty et al. 2000). An increase in chemical application or a required switch to another strategy may have corresponding economic costs that may be prohibitive to small-scale farmers (Gay et al. 2006).



Coffee berries. Andreas Balzer

There is a substantial lack of research investigating the interactive effects of climate change and invasive species on agriculture, especially on projected rates of change and basic climate data for major agricultural and forested areas (FAO 2008b). While a number of technical studies have been performed on the separate effects of climate change and invasive species, the examples selected below have been selected given their additional attention to the interaction between these two important global changes on agriculture.

Coffee Berry Borer (*Hypothenemus hampei*)

Coffee (*Coffea arabica* and *C. canephora*) in monetary value is second only to oil in globally traded commodities and is a major cash crop in several regions of the world. Coffee is also subject to losses in production and quality because of climate change and invasive species. Coffee is a fragile investment, vulnerable to several pests and sensitive to changes in temperature and precipitation. According to the International Coffee Organization, production is decreasing and prices will likely increase due to the effects of climate change (Schwartz 2010). In combination with pests, like the coffee berry borer, climate change is making coffee production difficult to sustain.

Coffee, a native plant of Eastern and Central Africa, is vitally important to the economy of numerous regions. Millions of people, including small-scale farmers, in eastern Africa, southern and southeastern Asia, and Central and South America are in some way reliant on coffee for their livelihood (Gay et al. 2006, Oxfam 2008, CABI 2010c). This valuable crop is affected by many pests – berry borers, leaf rust, coffee berry disease, bacterial blight, nematodes, leaf miners – that can significantly diminish crop yields. The coffee berry borer (*Hypothenemus hampei*) was transported around the world in contaminated coffee seed and is now the most damaging pest of coffee crops globally (Jaramillo et al. 2009, CABI 2010b). This beetle, native to Africa, causes premature fruit-fall and reduces the weight and quality of the coffee bean. In Indonesia, the invasive borer causes an annual production loss of 15-20% (CABI 2010b).

Under climate change, coffee pests like the coffee berry borer will be more difficult to manage. Over the last decade the incidence of the borer, along with other coffee pests, rose dramatically (CABI 2010a). The borer has a broad thermal tolerance and has been shown to experience greater population growth as temperature increases. Seeing that the borer is restricted by both temperature and the availability of their host coffee plants, it is predicted to follow plant distribution. In Uganda and Indonesia it has already expanded its altitudinal distribution range to attack coffee plantations at higher elevation. Changes in precipitation due to climate change will also change the impact of the borer on coffee. It will likely become more problematic in countries like Colombia where precipitation is well-distributed throughout the year, but less so in regions like eastern Africa where there are prolonged droughts (Jaramillo et al. 2009).

More generally, the growth of coffee itself and its associated industry is predicted to be severely impacted by climate change (Jaramillo et al. 2009). Native to the humid tropics, coffee only thrives in a mildly warm and wet climate that varies within certain parameters throughout the growing season (Gay et al. 2006, Oxfam 2008). Changes in the temperature and precipitation regime can reduce the yield and quality of the coffee harvest. For example, too much or unpredictable rain will reduce flowering, the ability to dry the beans and soil fertility (Oxfam 2008). Foreseen changes in temperature and precipitation would make many of the areas now seen as ideal for growing coffee as

unsuitable (Gay et al. 2006). Studies performed in Brazil, Mexico and Uganda show that even minimal increases in temperature will have disastrous consequences, in some cases reducing areas suitable for coffee production by up to 95% (GRID-Arendal 2002, Assad et al. 2004, Gay et al. 2006). In Kenya the unpredictable and unreliable rainfall has made crop management and disease control extremely difficult; coffee berries mature at different times, thereby requiring a larger investment in labor throughout more of the year to harvest a reduced crop (Obulutsa and Fernandez 2010).

Farmers are already adapting to climate change and the effects it is having on coffee production and management of crop pests, like the coffee berry borer (Oxfam 2008). Options for cultivating coffee, some of which are already being implemented, include planting at higher densities, genetically engineered tolerant strains and arborization (Camargo 2010). In particular, arborization, growing coffee plants under shade-providing trees, is thought to mitigate microclimatic extremes and buffer changes in air temperature, humidity and wind. Other added benefits of arborization are better quality coffee and habitat for predatory arthropods that naturally control the coffee berry borer (Camargo 2010, Jaramillo et al. 2009). A drawback to arborization is a comparatively lower yield and the possible movement of coffee plantations into higher elevation, cooler sites where they would displace forests (Gay et al. 2006). Fine-tuning these management systems in the face of both climate change and invasive species, such as the coffee berry borer, is critical for local and global economies, as well as for our morning cup of coffee.

Bluetongue Virus

One invasive livestock disease whose emergence is well tied to climate change is bluetongue, a livestock disease now common in many parts of Europe. Bluetongue historically occurred in Europe, but never at its current range and abundance. Evidence has implicated climate change in the spread of bluetongue virus by having a positive effect on its insect vector (Wilson and Mellor 2009). Between 1998 and 2005, bluetongue was the cause of death for over 1.5 million sheep and the culling of thousands of animals (Purse et al. 2008, Wilson and Mellor 2009). This disease has a direct economic link; the outbreak of bluetongue in 2007 alone had a direct cost exceeding US \$200 million to the farming industry in affected countries (Hoogendam 2007, as cited in Wilson and Mellor 2009).

Bluetongue is considered to be one of the most important diseases of domestic livestock and has received A-list status by the World Organisation for Animal Health (OIE) (Mellor and Wittmann 2002). This disease is caused by the bluetongue virus (*Orbivirus*), of which there are 25 immunologically distinct serotypes (Hendrickx 2009). All domestic and wild ruminants are susceptible to infection, but certain breeds of sheep, especially the fine wool and mutton breeds common in Europe, are more sensitive (MacLachlan 1994). Bluetongue results in numerous symptoms, including death. Tolerant animals contribute to the presence of the virus in "silent" disease-resistant hosts, most often cattle, that is difficult to manage (Mellor and Wittmann 2002, Purse et al. 2005). The bluetongue virus is considered native to Africa, Australia, and parts of the northern hemisphere and Asia (Tabachnick 2010). Though nearly global in its distribution, bluetongue was only historically found for brief periods at the southern and eastern fringes of Europe. Throughout this period, the potential for the bluetongue virus to enter Europe has long existed because of the trade of infected ruminants or by the wind-dispersal of infected midges (Purse et al. 2008).

Culicoides biting midges (Ceratopogonidae) are the arthropod vectors of bluetongue virus. The vector responds positively to climate change-induced temperature increases with greater population size, survivorship and faster disease transmission (Wittmann and Baylis 2000, Mellor and Wittmann 2002). In the geographical range of *Culicoides* midges, which is limited by cold winter temperatures, there is seasonal transmission of bluetongue virus only in the milder months (Wittmann and Baylis 2000, Purse et al. 2005). Climate change may also increase the availability of brackish breeding sites for *Culicoides* midges due to a predicted increased frequency of storms, flooding and sea-level rise (Wilson and Mellor 2009).

Recently the virus has extended its range northwards into areas of Europe never before affected to cause the greatest epizootic of the disease on record. The reasons for this dramatic change in bluetongue epidemiology are complex, but are easily linked to the expanding range of its major vector, *Culicoides imicola*, and to the involvement of novel vectors (Mellor and Wittmann 2002). It was thought that the cold winters would protect temperate regions of Europe from the disease. But the bluetongue virus has been able to overwinter in Europe for several years (Mellor and Whittman 2002). In 2006, major outbreaks of bluetongue occurred in countries with high population densities of small ruminants, such as Germany and France, and then subsequently in the Czech Republic, Switzerland and the United Kingdom in the following years (Purse et al. 2005, Tabachnick 2010). There is also a remarkable spatial congruence between the European regions that have experienced the most warming and the highest incidences of bluetongue in Europe. The *C. imicola* vector has spread into regions with obvious warming, but not into those showing cooling or no temperature change (Purse et al. 2005, Purse et al. 2008). Some have argued that climate change is likely not the sole determinant for this spread and that changes in husbandry and the habitat of *Culicoides* could also have contributed to its spread (Tabachnick 2010).

As a result of its impact on livestock, national sanitary and phytosanitary measures have been developed to restrict livestock trade between bluetongue-infected countries and bluetongue-free countries. Current management solutions are to quarantine infected livestock and to vaccinate livestock when possible (Tabachnick 2010). However, there are numerous serotypes of bluetongue virus, and new ones being introduced into Europe so vaccination is unlikely to stop the transmission of all bluetongue virus (Hendrickx 2009, Mellor and Wittmann 2002).

Human and Wildlife Health

Diseases, crippling by their very nature, are predicted to increase in prevalence and frequency under climate change. Environmental change and new ecological conditions may allow the proliferation of invasive species that directly and indirectly influence emerging infectious diseases, such as avian influenza, plague, babesiosis, Rift Valley fever, cholera, sleeping sickness, dengue fever, tuberculosis, ebola, yellow fever, Lyme disease, West Nile virus, and malaria (McMichael and Bouma 2000, WCS 2008). Climate change combined with global trade and transport networks may significantly increase the threat of such pandemics, just as the margins between transmission of disease across humans, livestock and wildlife are decreasing.

In general, vectorborne, zoonotic and waterborne diseases will increase worldwide (Portier et al. 2010). Vectorborne diseases, which are transmitted through a separate host, may increase or decrease their range in response to ecological and climatic changes (McMichael and Bouma 2000). For example, mosquitoes, which vector many infectious

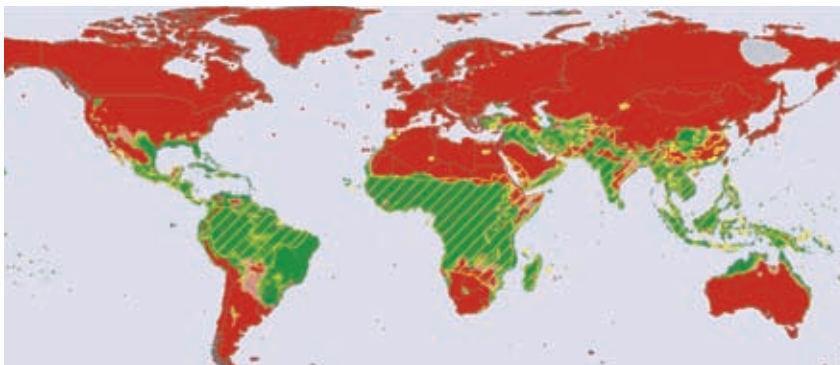
diseases in hot and wet climates, may increase their ranges and disease transmission rate under climate change scenarios (McMichael and Bouma 2000, Halstead 2008, Gething et al. 2010).

Additionally, diseases transmitted by mosquitoes may increase with more flooding, a weather phenomenon related to the outbreak of Rift Valley fever in Kenya and Somalia after the 1997 *El Niño/La Niña*-Southern Oscillation event (McMichael and Bouma 2000). As discussed in the food security section, the susceptibility of livestock and aquaculture animals to disease is a growing concern. There is a potential for the spread and exacerbated impact of blue tongue and rift valley fever in livestock, and epizootic ulcerative syndrome and algal blooms on aquaculture. Freshwater, one of our most critical resources, will likely witness more waterborne diseases in the near future. Such diseases will be magnified by increases in water temperature, precipitation frequency and severity, evaporation-transpiration rates and changes in coastal ecosystem health (Portier et al. 2010). Finally, movement of people from climate change, as well as increased crowding of climate change refugees, could further exacerbate the spread of disease through closer human contact and unsanitary conditions.

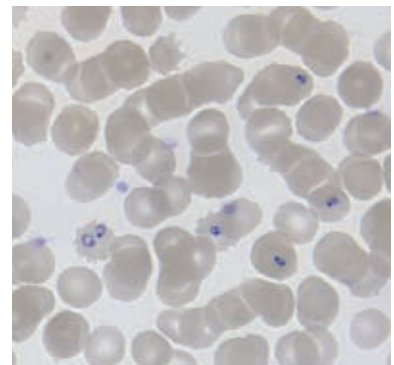
The examples below illustrate how diseases interact with humans and wildlife and how they may be worsened by climate change.



Mosquito (*A. aegypti*). Muhammad Mahdi Karim



Predicted limited spread of *falciparum* malaria under climate change (as included in Rogers and Randolph 2000)



Malaria (*P. falciparum*) parasite in blood. M. Zahner

Dengue and Malaria

Dengue fever and malaria, two diseases that can be fatal to humans, are common throughout the tropics and their disease incidence has now spread to more than 100 countries as their native mosquito vectors have become more pandemic. Dengue (caused by *Flavivirus* viruses) and malaria (caused by *Plasmodium* parasites) are spread by mosquito vectors, mainly *Aedes aegypti* and *Anopheles gambiae*, respectively. Combined, these diseases affect from 50 - 311 million people annually, leading to more than one million deaths worldwide (CDC 2010a, CDC 2010b). Dengue is especially fatal to children and is the leading cause of serious illness and death among children in some Asian countries (WHO 2009). Malaria was estimated to have killed almost one million people in 2008, with 89% of the world's malaria-caused deaths occurring in sub-Saharan Africa, where it is the second leading cause of death after HIV/AIDS (CDC 2010b). Research and modeling also suggests that malaria could spread into areas of North America, Europe and central Asia (Martens 1999). Because of the severe impacts of these diseases, understanding the role of climate change in altering the rate of infection and geographic extent of these diseases is a critical matter for public health (Larter 2009).

As dengue and malaria are both transmitted by a mosquito vector, on the surface, it would seem logical that infection rates are affected by the same climatic variables that affect mosquito populations. As mosquitoes benefit from warmer and wetter temperatures, some location-specific data on dengue and malaria has shown longer-term relationships between temperature, precipitation and disease incidence (Halstead 2008, Johansson et al. 2009, Gething et al. 2010). Research using long-term global weather patterns does not necessarily predict an increase in the number dengue and malaria outbreaks under climate change conditions. It appears as though prevalence of dengue and malaria is affected by climate variables in short-term, small-scale scenarios, but long-term, global disease epidemics are also influenced by other human-induced variables (Johansson et al. 2009, Larter et al. 2009, Rogers and Randolph 2000). Public officials will therefore have to balance these short and long-term implications of climate change, potential expansion of outbreaks to new areas and other intersecting factors influencing the spread of these diseases.



Scarlet honeycreeper (*V. coccinea*).
D. Hutcheson

Avian Pox (*Poxvirus avium*) and Avian Malaria (*Plasmodium relictum*)

Avian pox (*Poxvirus avium*) and avian malaria (*Plasmodium relictum*) are invasive diseases that threaten the remaining species of endemic Hawaiian honeycreepers. Under climate change predictions, it is unlikely that these rare birds will be able to escape disease-carrying mosquitoes. The endemic honeycreepers (Drepanidinae) were once abundant in the Hawaiian Islands from sea level up to tree line (van Riper et al. 1986). Because of several environmental threats, the honeycreepers are now confined to certain elevation ranges and face one of the highest rates of extinction in the world – 17 species are thought to be extinct and 14 are considered endangered (FWS 2006).

Since their introductions via nonnative bird introductions to Hawaii in the late 1800's and early 1900's, respectively, avian pox and avian malaria have become a major factor in the decline and extinction of endemic forest birds (Warner 1968, van Riper et al. 1986, van Riper et al. 2002, Atkinson and LaPointe 2009). Both diseases, spread by the invasive mosquito (*Culex quinquefasciatus*), have eliminated sensitive honeycreeper species from low and mid-elevation forests where mosquitoes are prevalent (van Riper et al. 1986, Atkinson and LaPointe 2009). The honeycreepers only find refuge at high elevation (above 1500m), where cooler temperatures limit the distribution of mosquitoes and, thereby the diseases (van Riper et al. 1986, Benning et al. 2002). Despite the lethality of avian pox and malaria in honeycreepers and other native bird species, non-native birds are often resistant to these diseases and act as a disease reservoir (Warner 1968, Stone and Anderson 1988, Benning et al. 2002, Atkinson et al. 2005).

The honeycreepers, diseases and mosquitoes all respond to elevational gradients (van Riper et al. 1986, Ahumada et al. 2004). High-elevation forests are the last remaining disease-free refuge for eight species of endangered forest honeycreepers. Under climate change, landscape analyses predict that mosquito-free honeycreeper habitat will decrease – Hawaiian forest reserves with areas of low malaria risk (below 13°C) will be reduced to less than 300 ha on each of two Hawaiian mountain reserves (Benning et al. 2002). If climatic changes occur slowly, the upward migration of forest habitat might be able to keep pace with warming temperatures, allowing the lower limits of refugia to remain above the elevation for disease transmission. However, disease transmission is influenced by more than temperature. A predicted decrease in precipitation may prevent expansion of the forest into higher elevations, thus squeezing the remaining high-elevation birds between the upper limits of suitable habitat and expanding disease transmission from

mosquitoes at a lower elevation (Atkinson and LaPointe 2009). Without successful management, such as habitat protection or the introduction of disease-resistant honeycreepers, the interaction between climate change and invasive disease could push remaining populations of honeycreepers to extinction (Atkinson and LaPointe 2009).

Chytrid Fungus (*Batrachochytrium dendrobatidis* – *Bd*)

In recent decades, amphibian populations have declined across the globe at an alarming rate, with over one-third of species threatened or extinct (Global Amphibian Assessment 2008). *Batrachochytrium dendrobatidis* (*Bd*) has been implicated in this massive extinction of primarily tropical frog species (Lips et al. 2005, Pounds et al. 2006).⁵ *Bd* is a rapidly spreading chytrid fungus that causes the highly virulent skin disease chytridiomycosis by inhibiting electrolyte transport across amphibian skin (Voyles et al. 2009). In the early 1990s, scientists began documenting losses of harlequin frogs (*Atelopus*) in Latin America, which are an important component of global amphibian biodiversity and an attractive feature for ecotourism in the region. Highland and lowland harlequin frog species have disappeared by 75 and 38%, respectively (LaMarca et al. 2005). Scientists agree that the invasive *Bd* has played a role in the decline and some have also cited climate change and an indirect increase of vulnerability to *Bd*.

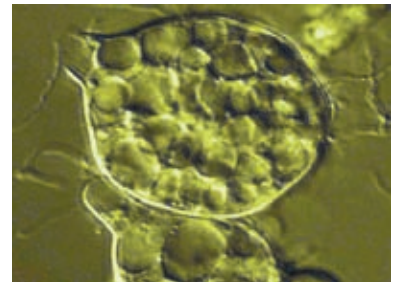
Those asserting a link to climate change note that large-scale global warming might provide optimal environmental conditions for the fungus *Bd*. According to their research, temperature shifts are causing convergence of daytime cooling and nighttime warming, which promotes disease transmission. Moreover, changes in cloud cover, and thereby temperature and moisture, are suggested to create habitats where *Bd* thrives, reducing habitats where the frogs can seek refuge (Pounds and Puschendorf 2004, Pounds et al. 2006). Others suggest that the dry *El Niño* conditions, which force amphibian populations to coalesce around a few wet microhabitats, result in the transfer of *Bd* to remaining populations (Anchukaitis & Evans 2010). Yet another group of scientists contend that there is no climate-related basis for the decline in amphibians from *Bd*, citing inadequate testing of causal links between climate change and the prevalence and spread of *Bd*. Instead, they suggest that the amphibian declines in Central and South America are best explained by the invasive *Bd* spreading through upland populations in accordance with a simple spatiotemporal pattern (Lips et al. 2008). Whether or not *Bd* is directly influenced by climate change, amphibian scientists all agree that frog biodiversity is threatened because frogs are simultaneously suffering from *Bd* and climate change (LaMarca et al. 2005, Lips et al. 2008).

Climate Change Mitigation

One basic approach to climate change mitigation is reducing overall emissions of greenhouse gasses. This can be done by reducing the amount of greenhouse gasses released into the atmosphere (reducing the *sources* of emissions) or by maintaining or increasing the sequestration of greenhouse gasses (through carbon *sinks*) including, for example, those that are already embodied in forests, peat swamps and other biological material. Efforts to reduce sources of greenhouse gasses have focused on areas such as increased fuel efficiency, energy conservation and the use of alternative energy sources from those based on fossil fuels. One renewable energy resource that has gained significant attention, research and investment over recent years is the use of biofuels, particularly second generation biofuels derived from ligno-cellulosic crops. While the sector holds economic promise, it does raise questions in terms of the selected biofuel crops and their fast-growing, weedy characteristics.



Limosa Harlequin Frog (*A. limosus*) female.
Brian Gratwicke



Batrachochytrium dendrobatidis. AJ Kann



Significant attention has also focused on the potential of intact natural ecosystems to sequester greenhouse gasses (increasing carbon sinks). Native forests, peatlands and other ecosystems are particularly recognized for their role as CO₂ sinks. Unfortunately, destruction or degradation of those ecosystems can compromise their integrity and may even shift the balance towards release of greenhouse gasses. Thus, under the U.N. Framework Convention on Climate Change, reducing emissions from deforestation and forest degradation is a core area of focus. For example, the 1997 peat swamp fires in Indonesia released approximately 1,098 teragrams of CO₂, and the 2002 Biscuit Fire in Oregon, U.S., that burned approximately 200,000 ha released somewhere between 3.5 and 4.4 teragrams of CO₂ (Heil et al. 2006, Campbell et al. 2007). Even where fires are a natural part of the ecosystem, changes in fuel availability, fire intensity and species composition are fundamentally altering the regime and patterns of carbon sequestration and release.

Invasive species and other pests can also drive such ecosystem change by increasing plant mortality, replacing native herb or shrub-dominated systems with trees or vice-versa, changing fire or hydrological regimes or altering species composition in other ways which may in turn increase susceptibility to other drivers of change. Research has documented that some invasive species reduce the sequestration ability of ecosystems, such as invasion of woody shrublands by grasses as well as changes in soils and microbial biomass in forests (Bradley et al. 2006, Strickland et al. 2010). On a more dramatic scale, the mountain pine beetle (*Dendroctonus ponderosae*), a species native to North America that feeds on pines, is playing a major role in transforming forest ecosystems in a warming climate. Managing these threats, whether of introduced or native species that are creating extensive ecosystem damage and change, needs to be a key focus of maintaining carbon sinks and climate change mitigation strategies.

Biofuels

With recent fluctuations in energy prices and growing concerns about greenhouse gas emissions associated with the use of fossil fuels, many countries are currently looking at growing high-yielding crops for the production of biofuels. Domestic production of biofuel crops can also increase national energy security as well as support local economic development. Rapid investment and development in this sector over the past few years has raised a range of issues including potential displacement of regularly grown food crops, increased food prices, additional pressures for land conversion and interest in using fast-growing, non-native species with histories of biological invasion (Sawyer 2007, Koh and Ghazoul 2008).

A 2008 review by GISP of almost 40 crops being considered for use as biofuels found that approximately 75% of them had some record of being invasive (GISP 2008). Additionally, this trend towards consideration of known invasive plants is practically global in extent including North, South and Central America, Africa, Australia, Europe and Asia (Low and Booth 2007, GISP 2008, Howard and Ziller 2008, IUCN 2009a). Potential biofuel species are frequently fast-growing, tolerant of dry conditions or poor soils, and resistant to (or having few) pests (Barney and DiTomaso 2008, Raghu et al. 2006). Additionally, land conversion for biofuel development, particularly of native forests and other ecosystem types, has been questioned in terms of its efficacy in reducing greenhouse gas emissions over a range of time-scales (Danielsen et al. 2008, Gibbs et al. 2008).

Experts have recommended the use of weed risk assessments to analyze the risk associated with particular species before their introduction – a lesson of historical importance given that a significant proportion of the world's most invasive plants were intentionally introduced for fodder, soil stabilization, agriculture and ornamental purposes (Barney and DiTomaso 2008, Howard and Ziller 2008, Mack 2008, Simberloff 2008). Research has also focused on the use of native species which have shown effectiveness in helping with climate change mitigation (Tilman et al. 2006). Ultimately, preventing the introduction of known invasive species (or ensuring that any risk of spread or further damage is managed) will reduce the potential pool of invasive species and pressures on ecosystems and their associated services.

Jatropha curcas: One of the most controversial and widely touted species proposed for use as a biofuel has been *Jatropha curcas*. *J. curcas* is a tall shrub that produces oil-rich seeds that is native to Central America. The species has been widely promoted for use as a biofuel in both small and large-scale production particularly on degraded lands in parts of Africa, South America and Asia (especially China, India and Sri Lanka) despite a record of invasiveness in many areas of the world such as Brazil, Australia, the U.S. and the Pacific (Low and Booth 2007, Brittain and Lutaladio 2010). Given infestations in northern and western Australia, the state government of Western Australia has declared the species a weed and currently the plant is banned from import into the country (Low and Booth 2007, Randall 2004). In addition to its properties as an environmental weed, consumption of the seeds, which contain toxins, can be injurious to humans and livestock. Finally, despite the promises of economic development, experiences, particularly in Kenya, have shown that cultivation of *J. curcas* is not economically viable for small-scale farmers (Endeleu Energy et al. 2009).

Reed canarygrass (*Phalaris arundinacea*): Fast-growing grasses are another popular area for investigation of potential biofuel species, such as giant reed (see section on Freshwater Services and Availability), switchgrass (*Panicum virgatum*) and *Miscanthus* spp. One of the highest-yield, cool season, perennial grasses is reed canarygrass (*Phalaris arundinacea*), a species commonly found in wetland and riparian areas of central North America, Europe and Asia. Reed canarygrass is being considered for biofuel use in northern Europe and can be grown in a range of poorly drained or compacted soils (Wrobel et al. 2009). However, it has also been identified as an invasive weed in parts of the U.S. and Australia by forming dense masses that can dominate wetlands and block waterways (Marten 1985, Kercher and Zedler 2004, Molofsky 1999). The fact that this species and associated cultivars can be invasive in areas close to their original distribution is also an issue for particular attention in the consideration of what may be viewed as native species.

Mesquite (*Prosopis juliflora*): Mesquite (*Prosopis juliflora*) and other *Prosopis* species are fast-growing woody shrubs with low nutrient requirements and the ability to survive in arid and semi-arid lands. Mesquite is native to Central and South America but has been introduced in Africa, Asia and Australia for use as fuelwood, fodder, shade, soil improvement and erosion control (ENS 2010). This wide range of existing uses has made it a popular candidate for biofuel development along with the fact that it grows well in arid habitats. Unfortunately, when left to propagate naturally (for example, when failed plantations are abandoned) the species has proven invasive by creating dense thickets that outcompete native species and place a higher demand on sub-surface waters (IUCN 2009a). In eastern Africa, infestations have hampered local agriculture as well as livestock grazing which has been a central part of local pastoralist livelihoods. In South Africa, *Prosopis* spp. have invaded at least 1.8 million ha of alluvial plains and seasonal watercourses (Van Wilgen et al. 2001).



Infested ponderosa pines (Rocky Mountain National Park, Colorado, US).
Quinn Dombrowski

Mountain Pine Beetle (*Dendroctonus ponderosae*)

Outbreaks of alien and native pests can severely alter forest health and composition with consequent changes in their release and sequestration of carbon. In North America, the mountain pine beetle (*Dendroctonus ponderosae*) is a native insect that feeds on a variety of pine species. The beetle's population is naturally regulated by temperature, including cold spells in the fall or spring during the most vulnerable stages of the beetle's development or during sustained periods of sub-zero temperatures during the winter (Regniere and Bentz 2007). In recent years, winters in British Columbia and parts of Alberta in Canada, and in Colorado and other western states in the U.S. have been mild thereby contributing to a population explosion of the beetle with significant loss of pines. Mountain pine beetle also serves as a vector for the transmission of pine blister rust (*Cronartium ribicola*), a fungus that is also detrimental to pines and which has also experienced an increase (Logan et al. 2003).

In British Columbia, the outbreak of mountain pine beetle has been exacerbated not only by mild winters, but also warmer summers and less summer precipitation. These conditions have facilitated beetle expansion into northern and higher elevation forests. By the end of 2006, the cumulative forest area affected totaled approximately 13 million ha, with major losses in timber. Some areas incurred mortality of up to 95% of the pine canopy. Estimates for release of carbon in British Columbia are around 270 megatonnes, which is an order of magnitude larger than releases from all previously recorded outbreaks (Kurz et al. 2008). Experts have directly attributed the extent of this outbreak to ongoing climate change. The current status of mountain pine beetle also corresponds to a trend since the 1970s in the expansion of climatically suitable habitats for the beetle, which will likely allow for further expansion north, east and into higher elevations (Carroll et al. 2004).

Research has also shown that pine beetle infestations increased the probability of forest fires which can be further exacerbated by drought and other climatic factors (Lynch et al. 2006). Large-scale forest destruction from such infestations can have further consequences in loss of habitat for local biodiversity, as well as increased soil erosion, runoff and siltation of water bodies. In British Columbia, forest mortality caused by the beetle will likely have significant impacts on populations of caribou (*Rangifer tarandus caribou*), fisher cat (*Martes pennanti*), martens (*Martes americana*), pygmy nuthatches (*Sitta pygmaea*) and woodpeckers (Ritchie 2008).

This experience with the mountain pine beetle, a native species, is also raising serious questions about the potential impacts of non-native invasive species, such as the emerald ash borer (*Agrilus planipennis*), the Asian long-horned beetle (*Anoplophora glabripennis*) and other major forest pests and pathogens which have the potential to eliminate entire tree species (and release their stored carbon) from North American and other forests.

Recommendations

Perhaps the key lesson stemming from this report is that while we lack full knowledge of the compounded impacts and interactions between invasive species and climate change, their actual and future impacts are severe enough to warrant action. Additionally, in the area of invasive species, our existing experience and practices are one clear path forward on reducing this overall threat. Even with better research, longer-term data and site-level specifics, we will be faced with gaps in knowledge or uncertainty regarding future scenarios and will still be faced with need to act. The following discussion addresses recommendations for policy and management as well as for science and research, which will require ongoing integration given the increasingly short timeframes for change.

Policy and Management

From a policy perspective, invasive species and climate change issues have largely been kept separate, however it is increasingly apparent that they will require integration particularly around the common priorities identified below. It should be clearly stated that the management of invasive species can be a key tool for ecosystem-based adaptation under climate change as it can reduce one of the major stressors on ecosystems and their services in and of itself, while also reducing the potential for additional greenhouse gas emissions caused by the impact of invasive species.

As Pyke et al. state "the design and implementation of climate-change policy ... should specifically consider the implications for invasive species; conversely, invasive-species policy should address consequences for climate change" (Pyke et al. 2008). From a general perspective, policy development should:

- Characterize the direct and second order interactions between climate change and invasive species;
- Identify areas where climate change policies could negatively affect invasive species management; and
- Support potential synergies between climate change and invasive species policy, such as ecosystem-based adaptation and enhancing ecosystem resilience.

From the perspective of **climate change policy**, key recommendations include:

Climate change adaptation management plans and activities, particularly those focused on ecosystem-based adaptation, should incorporate invasive species management as a key tool to reduce pressure on key ecological services and to enhance ecosystem resilience.

More specific actions can include:

- Prevent the introduction and establishment of new non-native species to minimize the possibility of future invasions and their subsequent impacts;
- Eradicate or control priority existing invasive species (including damaging native species) with the potential to fundamentally alter ecosystem composition and services, thereby enhancing ecosystem resilience; and
- Assess the potential for biological invasion associated with the development and construction of adaptation practices particularly those designed to meet key human needs (e.g., water distribution systems, aquaculture facilities, shifts in agricultural practices).



S. Burgiel

Climate change mitigation plans and activities should not create new or exacerbate existing biological invasions. Key activities can include:

- Assess the potential for biological invasion in the cultivation of particular species (e.g., intentional introductions for biofuels) or the development and construction of particular sequestration technologies (e.g., unintentional introduction or spread through energy infrastructures, ocean fertilization practices); and
- Eradicate or control damaging species with the potential to reduce the carbon sequestration ability of ecosystems to sequester carbon.

These efforts will have to take into account the spatial and temporal aspects of range shift under climate change, as well as event-based phenomena, such as storms, that have the potential to move species or further facilitate invasions through ecosystem disturbance.

From the perspective of **invasive species management**, we need to ensure that policy facilitates the use of resources and tools across disciplines and types of damaging species. Current management techniques honed on non-native invasive species are a critical piece of conservation knowledge that will be needed to manage all types of damaging species in novel ecosystems and communities—including both native and non-native species. It is also critical to recognize that we will need to continuously re-examine the efficacy of current practices given that climate change may alter mechanisms around the transport and introduction of invasive species; the climatic constraints, distributions and impacts of those species; as well as the effectiveness of our management strategies (Hellmann et al. 2008).

Current laws and regulations focusing on prevention of invasive entry and establishment are vital to minimizing the pool of potentially invasive species. Such prevention efforts can focus on identifying and minimizing the risk of transporting invasive species at their source as well as through pathways of introduction (Sutherst 2000). For example, off-shore quarantine programs are being used more frequently to facilitate trade in agricultural goods where the source/exporting country conducts a series of certified inspections and/or treatments to expedite customs and quarantine procedures in the receiving country. International and national laws addressing known pathways of introduction (e.g., ballast water, hull fouling, solid-wood packaging) should also be re-examined for their efficacy in the context of climate change, including new trade routes and partners.

Additionally, invasive species policy and funding focused on eradication and management will need to change to reflect the possibility of native invasives. Greater funding and institutional coordination will also be needed to ensure that policy development and implementation, response efforts and research activities can accommodate these additional dimensions and work towards the facilitation of ecosystem-based adaptation and the maintenance of ecosystem services. More specific recommendations include:

Prevention of new introductions

- Close pathways for the unintentional introduction of non-native species;
- Conduct risk assessments of proposed introductions of non-native species that include biogeographical factors and potential climate scenarios; and
- Develop early detection and rapid response systems targeting likely pathways and points of introduction, taking into account climate change dynamics.

Eradication and control

- Eradicate invasive species already present in a system where feasible;
- Control known invasive species and as necessary damaging native species if eradication is not feasible; and
- Monitor known invasive species as well as suspect non-native and native species with the potential for biological invasion.

Monitoring, early detection and rapid response systems can be used in the management of both existing and potential introductions of invasive species.

- Monitor existing species, including known invasives, suspect non-native species and potentially damaging native species, for possible *range shift* and in *post-disturbance surveys* looking for new or expanded infestations; and
- Develop early detection and rapid response capacity to prevent the establishment and spread of new biological invasions focused on key pathways for introduction (including movement by storms, strong *El Niño* events as well as man-made vectors like ships, airplanes and construction equipment) and on areas that might be particularly vulnerable to new invasions (e.g., areas experiencing glacial retreat, warming coastal areas, disturbed areas).

Risk and vulnerability assessments relating to the introduction and spread of potential invasive species and the health of ecosystems will need to integrate climate change in order to serve as a useful tool for managers and decision-makers. The probability that new species will be introduced and their potential impacts will directly impact ecosystem health, and similarly an ecosystem's health will help determine how it responds to incipient biological invasions. The combination of these assessments can overlay analyses of habitat suitability for new species, climatic matching and other factors that might facilitate introduction and spread, as well as the potential threat posed by introduced species to priority habitats (PRI 2008, Sutherst 2000).

Human responses to adapt to and mitigate the effects of climate change may also facilitate the introduction and spread of invasive species. Selection of species for reforestation or biofuel development needs to consider the potential impacts of non-native species. Development of wind farms and other alternative energy infrastructure may disturb habitats or create new vectors for the introduction of invasive species. Such activities need to incorporate risk assessment processes that include pathway evaluation and potential impacts of invasive species associated with activities designed to respond to climate change. Similarly, risk assessments of proposed species introductions, which are commonly done in the agricultural and aquaculture context, should take future climate change scenarios into account. Risk assessment methods for all taxa should be developed and implemented by all nations.

Managing complexity will be an increasingly important issue for policy-makers and practitioners as they struggle to deal with the uncertainty and risk inherent in both the climate change and invasive species fields. Generally, we know the broad outlines of climate change at a regional level or how invasive species might be introduced into an ecosystem, yet we often don't have the ability to make fine-scale predictions on responses at specific sites given available data and the precision of our models.

We therefore need to be able to use and act upon the knowledge that we do have (e.g., about broad climate trends, ecosystem pressures and invasive species management), while conducting research and learning by doing as we go along.

Climate change challenges conservation practice with the need to respond to both rapid directional change and tremendous uncertainty. Climate change adaptation therefore requires implementation of a range of measures, from short to long-term and from precautionary and robust to more risky or deterministic, but specifically anticipatory (Heller and Zavaleta 2009).



S. Burgiel

Adaptive management embodies this iterative approach and underscores the need for action in the face of change, while recognizing that we will continue to learn and adjust our practices as we gain further information from both success and failure. Finally, these efforts should embrace the precautionary approach which is compatible with adaptive management, recognizing the need for action now to enhance the resilience of ecosystems and their services and that invasive species are a key threat. The number of invasive species with linkages to climate change will likely just be a subset of the whole, so we'll need to target the clear cases and be overly inclusive to address those with probable or high risk impacts. Developing a suite of preventive measures, for example through improved national quarantine and sanitary/phytosanitary requirements, could be such a broad measure that serves both the climate change-invasive species relationship specifically, as well as the broader set of invasive species concerns more generally. Ultimately, reducing the overall threat of invasive species and corresponding efforts to support ecosystem health can be considered a low risk strategy with ancillary benefits regardless of the level of climate change-induced impacts.

Science and Research

Within the past several years, the research community has increasingly addressed the intersection between invasive species and climate change. What began nearly a decade ago with early publications by Duke and Mooney (1999), Mooney and Hobbs (2000), and Rogers and McCarty (2000) has now developed into a sub-discipline with publications addressing empirical research, synthetic reviews, and management and policy analyses as cited throughout this report. Experimental approaches, pattern detection, and mathematical modeling have all contributed to our understanding of the implications of changing conditions and our subsequent management approaches (Kritikos et al. 2007, EPPO 2008, Gjershaug et al. 2009). Despite these significant advancements, this field is replete with anecdotal evidence which has led to vague predictions or hypotheses instead of well-substantiated evidence of a link between these phenomena. The specific study on the intersection between these invasive species and climate change requires significant progress, especially to guide management strategies.

We suggest that broader-scale conclusions have not been widely generated for several reasons. Firstly, where progress has been made, research has focused on select biological systems that are already highly studied. These often include economically important species (e.g., agricultural weeds, such as cheatgrass and spotted knapweed), which are easy to manipulate in experimental contexts (e.g., laboratory and greenhouse designs) and are located in areas with active research (e.g., North America) (see Smith et al. 2000,

Bradley and Mustard 2006, Broennimann et al. 2007, Broennimann and Guisan 2008).

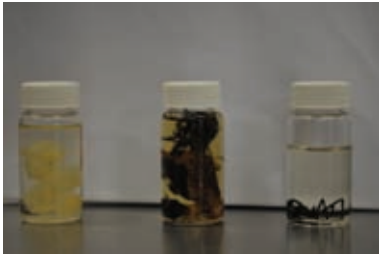
Data to inform predictions on invasive species and climate change interactions is largely nonexistent in most developing countries, where basic baseline information on invasive species impacts themselves is often lacking (Parmesan 2006).

Secondly, much of the evidence-based science to date has examined species range shifts based on temperature, while many laboratory and field experiments have examined species responses to increased CO₂ concentrations. Therefore we lack information about the link between invasive species and other climate change factors, such as variations in precipitation, sea level rise, ocean acidification, simultaneous climate factors and broader global change factors (e.g., land use, population growth) (Walther et al. 2009).

Thirdly, there are significant limitations to climate change projection models, which affect their utility for studying invasive species. For application to certain locations and species, climate change models have been well scaled to the regional level, such as the oriental fruit fly in the Pacific and water hyacinth in Europe (EPPO 2008, Kriticos et al. 2007). However, climate change models often have unreliable data and limited predictive power, particularly when they are downscaled from a coarse landscape level to the site level where environmental management is practiced (EPA 2008). In undeveloped regions of the globe the lack of finer-scaled climate models compounds the shortage of invasive species information. Additionally, the spatial and temporal scales of research on climate change and conservation strategies, such as invasive species management, are not well matched, which limits our understanding of their interactions and our strategies for their management (Wiens and Bachelet 2009). Further work is necessary to fine-tune climate-matching tools like CLIMEX and those used to evaluate climate change impacts under the U.N. Framework Convention on Climate Change to assist in the management of invasive species.⁶

Management strategies are often stymied by a lack of research, specific to the applied need of management (EPA 2008). This includes research that would inform risk assessments, vector transport and invasive species control methods. Risk assessments used to estimate the invasibility of a species or the vulnerability of a critical habitat do not regularly incorporate climate change although this is critical for understanding risks over time (PRI 2008). Pathways or vectors for the introduction of invasive species are increasingly the focus of management. However, this approach generally lacks the interaction of invasive species with climate change, especially how current priority pathways will change under climate change, what new pathways will emerge, and how vector analysis can be modified to account for climate change. Another seemingly obvious, but often overlooked, sector of research on the intersection of invasive species and climate change is control methods. There is an immediate need to understand the performance of control methods, be they mechanical, chemical or biological, under climate change. Even for widespread well-studied invasive species, we do not know which current control methods will be most adaptable, remain robust or change under climate change (EPA 2008). In summary, although the need for fast solutions is great, we need to improve our body of knowledge with tried and tested management options under changing climatic conditions.

⁶For further information on CLIMEX, see <http://www.climatemodel.com> and for further information from the UNFCCC on their Methodologies and Tools to Evaluate Climate Change Impacts and Adaptation, see http://unfccc.int/methods_and_science/impacts_vulnerability_and_adaptation/methods_and_tools_for_assessment/items/596.ph.



S. Burgiel

This effort reveals that a significant body of work and longer-term data on the topic has yet to be developed. Moving forward, we need to ensure that research and management efforts help to build this knowledge base. Evidence-based science is likely the best way to garner policy support for both climate change mitigation and adaptation as well as invasive species management. Some items for consideration by the scientific community include:

Existing research and data can be better utilized by the researchers. For example, research on population extent and the edges of distribution might elucidate the ranges of tolerance of species for various climatic factors. Additionally, phenological shifts may be evident from herbarium records or other datasets.

Different species and climate change dimensions can be the focus of new experiments addressing less-traditional invasive species and factors of climate change. For example, further research can focus on micro and macro-scales from invasive micro-organisms and changing soil compositions to community level interactions; laboratory experiments can incorporate ocean acidification; greenhouse experiments can include varying precipitation regimes; and field experiments can evaluate interactions between sea level rise and the biotic environment.

Experiment complexity, incorporating multiple variables, will need to become commonplace. Scientists can begin studies that incorporate multiple invasive species, multiple climate change factors, multiple latitude or sites, and multiple environmental stressors (land-use, resource extraction, pollution, etc.) over the long-term. These studies are more complex, but their results reflect real ecosystem scenarios and diverse human impacts and will help us to understand interactions between invasive species and climate change (EPA 2008, Heller and Zavaleta 2009, Walther et al. 2009).

Within this process there are also some clear information priorities, including:

- Ecological knowledge about potentially invaded environments, ecologically dominant species and trends in habitat disturbance and degradation;
- Detailed biological and population information for particular species;
- Ongoing data generation and collection to inform predictive models of both invasive species and climate change;
- Lessons learned in ecosystem restoration and the maintenance of ecosystem services;
- Climate data and methodologies sufficient for downscaling to make projections at the site level; and
- Strategies to incorporate uncertainty into evaluations and decision-making (Capdevila-Argüelles and Zilletti 2008, EPA 2008, PRI 2008).

Research into these areas obviously requires funding, yet climate change science is arguably receiving the lion's share of funds compared to other environmental issues, including invasive species. The scientific and management communities therefore also need to make the case for linking invasive species and climate change research within the context of broader funding priorities.

Glossary

Assisted migration: Human-aided, intentional dispersal of a species into an area where conditions are more favorable for its conservation (McLachlan et al. 2007).

Bioclimatic envelope: The realized and projected distribution of a species based on the relationship between the current distribution and climate factors.

Climate change adaptation: "Adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities" (IPCC 2007a).

Climate change mitigation: Technological change and substitutions that reduce greenhouse gas emissions (sources) and enhance sequestration processes (sinks).

Disturbance event: An event that causes a change in environmental conditions that interfere with ecosystem function.

Ecological resilience: The ability of an ecosystem to maintain its integrity in the event of internal change and external stressors and disturbances.

Ecosystem-based adaptation: The use of biodiversity and ecosystem services as part of an overall adaptation strategy designed to maintain and increase the resilience and reduce the vulnerability of ecosystems and people in the face of adverse impacts of climate change (SCBD 2009).

Ecosystem services: The benefits derived from ecosystems, which can be categorized according to their provisioning, regulating, supporting and cultural functions (Millennium Ecosystem Assessment 2005).

Emerging infectious disease: An infectious disease whose incidence is increasing following its appearance in a new host population or whose incidence is increasing in an existing population due to long-term changes in its epidemiology (Cleaveland et al. 2007).

Invasive species: A species whose introduction or spread into a new ecosystem threatens environmental, human or economic health and well-being.

Invasive species management: The prevention, eradication and/or control (preferably in that order of priority) of invasive species.

Pathogen: An agent, especially a micro-organism such as a bacterium, virus or fungus, that causes disease.

Pest: "Any species, strain or biotype of plant, animal or pathogenic agent injurious to plants or plant products" (IPPC 2010).

Phenology: The study of periodic biological events, such as flowering and migration.

Range shift: A change in the geographic coverage of a species as determined by environmental and bioclimatic factors.

Risk analysis: A process for determining the potential threat posed by an organism, event or development, including: risk assessment – a process to characterize and predict the probability, nature and magnitude of present and future risks; risk management – evaluation and selection of measures to reduce risk; and risk communication – conveying information on risk assessment and management to decision-makers and other stakeholders.

Trophic web: The set of interactions between species in a community that define the exchange of energy between producers, consumers and decomposers.

Vectorborne disease: An infectious disease whose transmission cycles involves animal hosts or vectors. Vectors carry the pathogen from one host to another.

Weed: A plant regarded as noxious, undesirable or detrimental.

Zoonotic disease: An infectious disease transmitted from animals to humans by either contact with animals or by vectors that can carry zoonotic pathogens from animals to humans.

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