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# Queensland's biodiversity under climate change:

## terrestrial ecosystems

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# PREAMBLE

This report is one of seven background documents prepared by the CSIRO Climate Adaptation Flagship for the Queensland Government. Content from this report contributed to the synthesis report titled “Queensland's biodiversity under climate change: impacts and adaptation” by Williams *et al.* (2012a).

The seven background reports are:

A. Overview of climate change in Queensland (Williams & Crimp 2012)

B. Ecological scaling of terrestrial environmental change (Ferrier *et al.* 2012)

C. Terrestrial ecosystems (Murphy *et al.* 2012)

D. Freshwater aquatic ecosystems (Kroon *et al.* 2012)

E. Coastal and marine ecosystems (Bustamante *et al.* 2012)

F. Ecosystem services (Williams *et al.* 2012b)

G. Adaptation principles and options (Dunlop *et al.* 2012)

View across coastal estuaries to Hinchinbrook Island, Queensland (credit: Gregory Heath, CSIRO Land and Water, science image BU5468).

# EXECUTIVE SUMMARY

Queensland's terrestrial biodiversity includes globally significant flora, fauna, ecosystems and diverse iconic landscapes, such as the wet tropical rainforests of north Queensland, the Gondwanan forests of the Border Ranges, the coastal dune forests of Fraser Island and the ephemeral floodplains of the Channel Country. Evaluating potential change in these or any other terrestrial ecosystem is challenging, as environmental changes involve atmospheric as well as climatic changes. Specifically, increasing atmospheric concentrations of $CO_2$ are both a driving force of climate change and a primary substrate for photosynthesis. Rising atmospheric $CO_2$ concentrations will influence growth and resource allocation in plants. Where water and essential plant nutrients are limiting, forage may become less nutritious and potentially more toxic for herbivorous animals, potentially leading to further change in food web structures.

Queensland's biodiversity has already undergone significant change with land clearing, grazing and invasive alien plants and animals all contributing to degradation, fragmentation of remaining habitat and declining populations of native species. The top five of the 10 Australian ecosystems identified by scientists as most vulnerable to 'tipping points' and at risk of collapse have occurrences in Queensland. Research has generated a great deal of ecological knowledge about some regions, like the wet tropical rainforests of north Queensland, but much less is known of the biota and their responses to climate change in the savanna and more arid regions.

Areas of terrestrial Queensland at particular risk include the Wet Tropics, high altitude and montane regions, tropical savanna-woodlands, drier rainforests types (including vine thickets), coastal floodplains and wetlands and particular concentrations of species or centres of endemism. An anticipated rise in the cloud layer blanketing coastal mountains will result in significant reductions in water intercepted by vegetation during the dry season and may lead to more sclerophyllous mountain-top communities. The majority of Queensland west of the Great Dividing Range is expected to change rapidly due to its low relief. Species will need to respond relatively quickly if they are to keep pace with the change. Refuge features in the landscape that supported native species during past climate change are likely to be important for future persistence. Rising sea-levels will have consequences for coastal inundation and many littoral ecosystems will be replaced by estuarine systems and saline wetlands.

A number of terrestrial ecological change types or phenomena can be related to climate change. These arise as species respond to their changing environmental conditions. Changes in the seasonal timing of climatic events affect the phenology of species through their life cycles and result in changes in species behaviours and interactions and changes in the match between species and their habitat in their current locations. These changes and responses are natural phenomena that occur at the individual species level and cascade through species interactions to alter the composition of ecosystems, until at larger scales they are reflected in changes to the structure, function and productivity of ecosystems.

Species rarely live in isolation but rather interact with other species that share the same or overlapping habitats. Species-specific responses to various components of climate change have the potential to cause temporal, spatial or functional shifts in the composition of species assemblages that affect species interactions. Preserving interactions among species is critical for maintaining long term production of food and materials in terrestrial ecosystems, while allowing for these ecosystems to change. Seed dispersal, pollination and nitrogen fixation are examples of key processes facilitating ecosystem function that rely on species interactions.

Assessing the consequences of climate change for biodiversity is therefore a multifaceted problem and involves acknowledgement of a large degree of uncertainty.

Disturbance is an integral part of natural ecosystems. However, altered disturbance regimes as a result of climate change lead to rapid ecosystem change. For example, fire is an important terrestrial transformer of Australian ecosystems. Fire can modify habitat structure and directly influence plants and animals through mortality and loss of shelter and food resources. In a warmer climate, fire activity is likely to increase. There have already been increases in an annual index of fire weather (the cumulative forest fire index, ∑FFDI) over the past 30 years at weather stations across the major regions of Queensland. Projected interactions between climate change, $CO_2$ fertilisation effects on vegetation biomass and fire will also alter future fire regimes with major implications for ecosystem function. With increasing temperatures and cycles of flooding rain and drought the subtropical sclerophyll forests of Queensland's central and south-east regions are likely to experience an increase in fire activity and intensity beyond the current familiarity of local managers. Increasingly severe fire weather conditions may limit the opportunity to undertake prescribed burning, which may increase the risk of the spread of unplanned fires.

Changes to climate are likely to interact with disturbance regimes, land use change, invasive species, disease and pathogens and other agents of change resulting in 'threat syndromes'. Threat syndromes occur when a number of threats, both present and future, interact to increase concerns about the continued persistence of certain types of biodiversity. For example, in the tropical savannas and the brigalow, the invasion by exotic grasses and its impact on fire regimes is likely to pose a greater threat to these ecosystems than a change to the fire regimes caused solely by potentially more severe fire weather associated with climate change. The interaction between climate change and existing human-induced pressures on species and ecosystems is a particular concern because these could precipitate rapid ecosystem transformations and reduce their capacity to provide familiar ecosystem services. It is likely that changes to the nature and character of biodiversity in Queensland will come from such threat syndromes, rather than from the operation of any one agent of change.

In healthy diverse ecosystems, species act in complementary ways to optimise the use and recycling of resources, such as nutrients and water, whereby production is maximised across a range of environmental conditions. Biodiversity also confers resilience on ecosystems undergoing change; that is, the capacity of an ecosystem undergoing disturbance to withstand the pressure without the loss of ecosystem function. Long-lived, woody plant species are likely to dominate the processes of landscape modification through structural habitat changes that also influence fire regimes and affect the nature of supporting habitat for fauna and other flora.

Adaptation in the context of managing for climate change is the ability of a system (the environment and human management) to adjust to climate change (including climate variability and extremes), to moderate potential damages, to take advantage of opportunities or to cope with the consequences. Knowledge of the adaptive capacity of ecosystems and their constituent species in the face of climate change is fundamental prerequisite for their management. There is some evidence for adaptation at the individual organism level, for example through behavioural or physiological changes in response to changing climatic cues. By and large, the capacity of individual organisms to adapt to climate change is very poorly understood and may be underestimated for some species and overestimated for others. However, there is consensus among scientists that healthy, intact ecosystems with clean, healthy resources, such as water and soil and diverse habitats and rich biodiversity, are likely to be the most resilient to environmental change.

Two prerequisites must exist for species' and ecosystems' adaptive responses and resilience to be successful in countering the rapid predicted rate of climate change. These are landscape and biogeographic connectivity to allow organisms to reach suitable habitat, climate space or refugia and adequate time to allow movements or adaptive changes. Landscape refuges provide protection from unsuitable or threatening conditions for plants and/or animals and allow them to persist. Refugia habitats are prominent in many of Queensland's reserves and Australia's biodiversity hotspots, such as the Carnarvon Gorge, and coastal mountain regions, such as the Wet Tropics. Some species (including Queensland's endemic frogs and lizards) may minimise the effects of adverse climatic conditions by adapting their individual biology or by utilising cooler or moister micro-habitats (micro-refugia) that buffer them from climate and weather extremes. The distribution of these places in the landscape and their functional value as refuges for different species, both spatially and temporally, is not well understood or documented for many regions.

Research in tropical forests suggests that species might undergo dramatic declines in abundance before changes in distribution ranges occur. In the tropics, where the latitudinal temperature gradient is very shallow, range shifts for terrestrial species can be expected to be either upslope in mountainous areas (i.e., the Wet Tropics) or over long distances in flatter areas (e.g., the northern savannas). This is the case for both native and alien species. When changes in environmental conditions exceed the ability of a species to shift or adapt — for example, through avoidance behaviours and acclimatisation responses — extinction over part or the entire distribution range is a possibility. Adaptation through natural selection is not expected to keep up with rates of environmental change for the majority of higher plants and animals.

The impacts of climate change on terrestrial ecosystems will not be uniform across the state of Queensland. Differences in land use and weather patterns will generate different degrees and types of ecological change over time. Areas with projected climates that are most dissimilar to current conditions will likely undergo the greatest ecological change. For example, coastal and south-eastern Queensland, with relatively high, reliable rainfall and lower temperatures, will be affected by increasing temperatures, rising sea levels and changes in seasonal rainfall patterns. These changes could result in the south eastern regions of Queensland becoming more like northern and western Queensland over the long term (beyond the end of this century).

Regions most vulnerable to change are likely to be those containing species and ecosystems with restricted geographic distributions, ecosystems near the limit of their environmental tolerance (such as dry rainforest) and those subject to synergistic threat syndromes. These include regions with high elevation mountain ecosystems, dry rainforest types, tropical savannas, ecosystems of the Gulf region, ecosystems in arid and semi-arid regions or the Channel Country and areas with concentrations of narrow-range (endemic) species. Ecological models can facilitate understanding and provide a framework for identifying hotspots of change over time, where strategic monitoring could provide early warning signals for adaptive management responses.

Moisture is a fundamental prerequisite of life. Regions subject to significant changes in the balance between rainfall and evaporation, both annually and seasonally, will see a corresponding change in primary productivity and vegetation biomass, affecting the structure and function of supporting habitats for other species. Western, inland regions, already characterised by large temperature ranges and extended hot conditions, are likely to support flora and fauna close to their thermal limits and that may be vulnerable to even modest increases in the number of consecutive increasingly hot days.

The adaptive capacity of ecosystems can be enhanced by reducing human-mediated stresses. Biodiversity management under climate change is everyone's responsibility and will require broad government, industry and community participation. For example, there is a need to improve observation networks through citizen science participation and to enhance research capability in understanding and predicting changes in biological systems, including greater geographical balance in research effort and monitoring. Conservation goals will also need to be reassessed to account for the effects of climate change on ecosystems. For example, climate adaptation measures for biodiversity will need to consider the entire landscape matrix and the mix of uses in which reserves are one aspect.

Conservation measures by resource managers as an adaptation to the impacts of climate change will need to focus on spatial strategies at multiple scales, such as ecological networks, short-term refugia, robust corridors, transmission pathways, off-reserve conservation and potential future protected areas – by broadening existing initiatives. Consideration of climate change forecasts in operational practices, such as fire and biodiversity management, will need to extend beyond reserves to the entire matrix. Climate change considerations in invasive species risk assessment and lists of alert species and sleeper weeds need to form part of biodiversity conservation assessments.

Biodiversity management under climate change will require broad government, industry and community participation. Understanding which actions are reversible and which actions have long-term implications will require dynamic landscape conservation plans and flexible management strategies to avoid foreclosing options. Urgent actions may include securing, enhancing and connecting remnant and regrowth habitats in fragmented ecosystems. Considerable uncertainty regarding costs and unintended impacts surround other actions, such as where and how to restore habitats and whether to translocate keystone species to enhance ecological function and evolutionary capacity. Biodiversity conservation plans therefore need to be integrated with other land use plans, especially reafforestation, to ensure society capitalises on new investments through enhanced landscape resilience.

# 1. INTRODUCTION

Queensland's globally-important terrestrial biodiversity is contained within a wide variety of landscapes that include tropical and subtropical rainforests, savannas, natural grasslands, stony and sandy arid lands, high, cool rocky woodlands and arid floodplains. The geographic distribution of these landscapes broadly corresponds with 13 terrestrial bioregions (Queensland Herbarium 2011) (Figure 1). The largest bioregions are in the arid and semi-arid areas that have relatively shallow climatic gradients and low relief. The eastern coastal areas experience sharp climatic gradients in places of high relief (e.g., Figure 2). These diverse landscapes harbour a high level of endemic wildlife and a large proportion of Australia's total flora and fauna species are represented in Queensland (DERM 2010a).

Biodiversity in Queensland has already undergone significant change due to land clearing, grazing and invasive alien plants and animals. These all contribute to declines and fragmentation of habitat (Guymer *et al.* 2008; Wilson *et al.* 2008a). The remaining remnant vegetation has been classified and mapped into its component regional ecosystems for 83% of the State (Accad *et al.* 2008; Neldner *et al.* 2005). Of the 1384 defined regional ecosystems, 89 (6%) are represented by 10% or less of their original extents. These ecosystems occur mainly in fertile agricultural areas and coastal regions that have been extensively cleared (Queensland Herbarium 2011). As a result of land use change, 10% of the 185 regional ecosystems in the Wet Tropics bioregion are considered endangered (< 10% of their original extent remaining) with another 72% considered 'of concern' (<30% of their original extent remaining) under the *Vegetation Management Act 1999*. Many of these remnant habitats continue to provide services such as timber extraction, grazing, tourism and recreation, and some carry the legacy of degradation from past unsustainable land management practices. Taking into account these impacts on condition or ecological integrity of remnant habitats as defined by the "biodiversity status" of these ecosystems, 42% are considered endangered and 47% are 'of concern' (i.e., vulnerable) (Queensland Herbarium 2011). These degradation processes are reflected in declining abundance of some native species and, where specific information is available about extinction risk, their listing under the *Nature Conservation Act 1992*. Queensland's draft strategy for the conservation of biodiversity therefore encompasses species, ecosystem and landscape level objectives and actions (DERM 2010a; DERM 2010b).

Continental and global reviews have identified terrestrial ecosystems of Queensland at particular risk from climate change impacts: the Wet Tropics of Queensland (Hughes 2011), high altitude and montane regions (Hilbert *et al.* 2001; Steffen *et al.* 2009; Williams *et al.* 2003), tropical savanna-woodlands (Laurance *et al.* 2011), drier rainforests types including vine thickets (Laurance *et al.* 2011), coastal floodplains and wetlands (Laurance *et al.* 2011) and areas of high endemism (Thomas *et al.* 2004), such as the Wet Tropics, Border Ranges and Iron Range/McIlwraith Range area of Cape York for vascular flora (Crisp *et al.* 2001). A recent consensus paper identified the top 10 Australian ecosystems most vulnerable to 'tipping points', defined as a circumstance whereby a relatively modest change in an environmental driver or perturbation can cause a major shift in key ecosystem properties (Laurance *et al.* 2011). The top 5 all occur in Queensland (though not exclusively) and the major environmental drivers of change are climate-related. In order, the top five ecosystems considered most vulnerable are:

elevationally restricted mountain ecosystems, tropical savannas, coastal floodplains and wetlands, coral reefs and drier rainforests (Laurance *et al.* 2011).

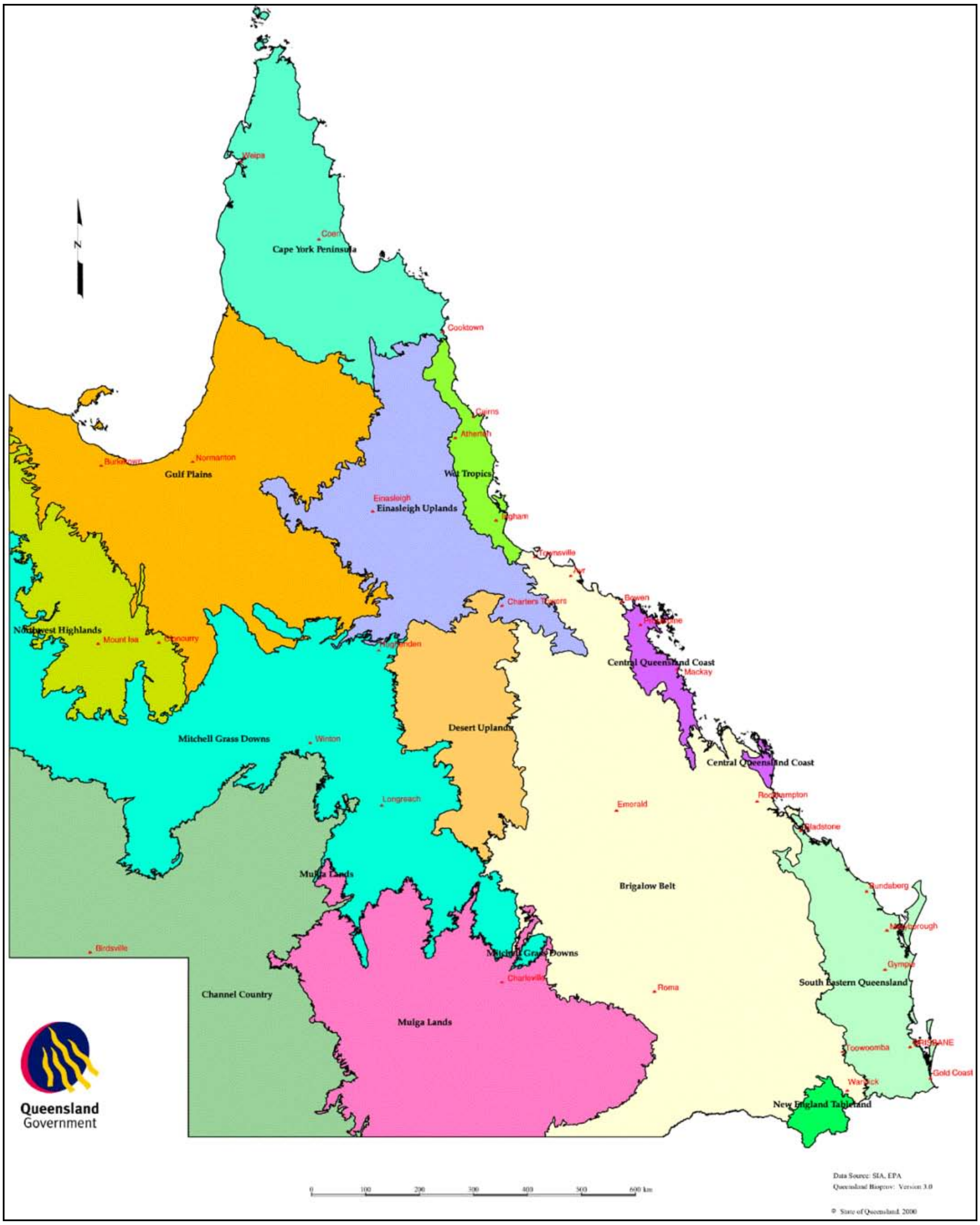


Figure 1: Thirteen of Australia's 85 terrestrial bioregions occur in Queensland. Source: Department of Environment and Resource Management, http://www.derm.qld.gov.au/vegetation/bioregions.html.

Other work has identified particular taxa at high risk; amphibians (Pounds *et al.* 2006; Steffen *et al.* 2009), reptiles (Kearney *et al.* 2009) and endemic fauna of montane regions (Williams *et al.* 2003). The majority of Queensland west of the Great Dividing Range, composed of tropical and subtropical grasslands, savanna and shrublands, is expected to have among the fastest ‘velocities’ of climate change owing to its low relief (Loarie *et al.* 2009), indicating species will need to respond relatively more rapidly to keep pace with climate change than in areas with slower velocities (e.g., mountainous areas). For example, the Channel Country of south-west Queensland is highlighted as a vulnerable region. Many animals are living close to their temperature limits during summer. Catastrophic losses of wildlife could occur during heatwaves and drought. These potentially represent mass mortality events akin to coral bleaching on the Great Barrier Reef (Low 2011). In addition, the Cape York and Gulf regions are predicted to experience novel climates (i.e., climates not currently existing) as well as the disappearance of some extant climates (Williams *et al.* 2007) (see also Ferrier *et al.* 2012). These northern regions represent sentinels of initial change that will flow southward as global warming continues (see Williams & Crimp 2012). Low (2011) postulates a dominant northward movement of Gulf biota toward the coast where temperatures are lower and rainfall higher , however, species movements there will be limited by the Gulf of Carpentaria expansion with sea level rise flooding southward into low lying areas (see Bustamante *et al.* 2012).

Knowledge about biodiversity and the impacts of climate change in some regions is far greater than for others. For example, there has been a relative wealth of research on the potential impacts of climate change on biodiversity in the Wet Tropics region compared with the savanna and more arid areas of the state. Thus, while many of the examples we use here are based on research undertaken in the Wet Tropics, this is due to a research bias rather than a reflection of the true potential impact or importance of climate change on biodiversity. This report focuses on the terrestrial ecological realm. A detailed assessment of the freshwater aquatic realm is presented in Kroon *et al.* (2012) and of the coastal and marine realm in Bustamante *et al.* (2012).

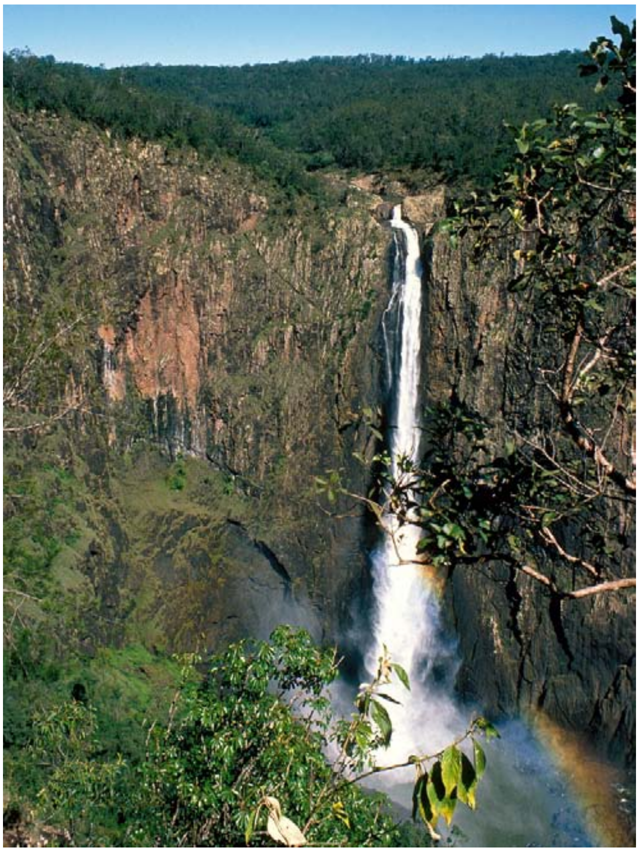


Figure 2: A rainbow hangs on the mist at Wallaman Falls, west of Ingham in north-eastern Queensland. These spectacular waterfalls are the highest in Australia (credit: Paul Peter, CSIRO Land and Water, science image BU6790).

# 2. ENVIRONMENTAL CHANGE

## 2.1 Introduction

Climate change refers to the long-term change in the distribution of weather patterns and includes changes in average weather conditions as well the pattern of extreme events that form part of the statistical distribution of daily observations over time. In the terrestrial ecological realm, changes in average weather conditions due to global warming means there will be permanent changes in thermal living conditions experienced by biota across a diverse range of habitats and scales from arboreal to sand dune, in soil and rocky landscapes. Some habitats will be buffered and others will be exposed to extreme conditions. An overview of the scientific evidence and basis of climate change, and a summary of the changes projected to occur over Queensland, is given in Williams and Crimp (2012). In this section we highlight some climatic and environmental changes (rising $CO_2$, intensity of cyclones, cloud stripping and sea-level rise), that have the potential to directly influence terrestrial biodiversity, the effects of which will flow on to ecosystem services. An additional major emphasis of this section is the interaction between climate change and fire. We describe how fire regimes may change with climate change and the influence of fire on biodiversity, ecosystems and ecosystem services.

## 2.2 Atmospheric $CO_2$ concentrations

While increasing $CO_2$ influences global temperature and climate, it also has both indirect and direct effects on organisms and ecosystem processes. In particular, as $CO_2$ is the primary substrate for photosynthesis, increasing $CO_2$ concentrations in the atmosphere increases the rate of photosynthesis, potentially influencing growth rates and resource allocation in plants.

The influence of elevated $CO_2$ at the ecosystem level is uncertain and even less so when coupled with large changes in climate (see also Box 1). In a comprehensive review on the effects of elevated $CO_2$ on Australian terrestrial species and ecosystems, Hovenden and Williams (2010) found that, at the species level, the most overwhelming response to increased $CO_2$ concentrations was a reduction in plant nitrogen concentration and an increase in the production of secondary metabolites. Increases in plant growth rates were far from ubiquitous and more common in woody plants than in grasses. The impacts of rising $CO_2$ on ecological processes and ecosystems are far less studied than that of individual species. However scaling-up from the response of individual species, it can be expected that many plants will become less nutritious and potentially more toxic for herbivorous animals, resulting in significant changes to trophic interactions (Hovenden & Williams 2010).

## 2.3 Cyclones

On average 4.7 tropical cyclones per year affect the Queensland Tropical Cyclone Warning Centre Area of Responsibility (BOM 2011). There is a strong relationship with eastern Australian tropical cyclone occurrence and the El Niño-Southern Oscillation (ENSO) phenomenon, with almost twice as many impacts during La Niña than during El Niño (BOM 2011). The effect of global warming on the number, duration and intensity of cyclones is not fully clear although most global simulations project an increase in cyclone intensity (Webster *et al.* 2005), while regional models for Queensland project an increase in intensity but potentially an overall decrease in number (CSIRO & BOM 2007b). An increase of 60% and 140% in the intensity of the most extreme storms for 2030 and 2070, respectively, was found using a model with a 15km grid spacing over Australia (Abbs *et al.* 2006). Simulations for Queensland also show more long-lived tropical cyclones and a southward shift in tropical cyclone genesis and decay areas of between 2 and 3 degrees of latitude (Abbs *et al.* 2006; Leslie *et al.* 2007). Fine resolution atmospheric models project a 30% decline in the number of tropical cyclones by the end of the 21st century for the Australian region (Lavender & Walsh 2011). Projections of tropical cyclones for the eastern Queensland coastline for 2055 and 2099 using 11 regional models and A2 emission scenario, also indicate a tendency for a reduction of tropical cyclone activity in the future and a 1° southwards shift in average latitude in activity (Fuentes & Abbs 2010). For more details, see Williams and Crimp (2012) and Bustamante *et al.* (2012).

The main effects of tropical cyclones at landfall include heavy rain, strong wind and ocean storm surges. The destructive force of a tropical cyclone depends on its intensity, extent, location and rate of passage. Tropical cyclones strip forest canopies, remove vines and epiphytes from trees, cause extensive branch and stem damage and tree fall. The storm surge alters the landscape near coastal areas by moving and reshaping dunes and undermining banks. Heavy rainfall destabilises soils in mountainous areas causing landslips. Terrestrial runoff scours land surfaces and river loads of river sediment and nutrients are deposited into swollen wetlands, estuarine systems and plumes extend into coastal-marine habitats. After the cyclone has passed, terrestrial ecosystem recovery can take many years and becomes part of a complex spatial disturbance mosaic. The forest debris, dry beneath stripped canopies, provides fuel for wildfires during the following dry seasons. Forests are at risk of more permanent transformation until the regenerating foliage closes the canopy and restores the humidity of the arboreal environment. Frequent cyclone disturbance regimes can affect the structure and demography of lowland forests resulting in shorter canopy heights and higher stem densities (Gouvenain & Silander 2003; Webb 1958). Whilst northern tropical landscapes are accustomed to tropical cyclones, these are rare events in the central and southern coastal landscapes of Queensland. With tropicalisation, southern cyclones will become more likely and, over time, the structure of coastal and hinterland forests may become more similar to cyclone-disturbed forests of the north.

## 2.4 Occult precipitation – cloud ‘stripping’

Climate change models have shown that the relative humidity surface will shift upwards with a doubling of atmospheric carbon dioxide concentration (Still *et al.* 1999). This likely outcome of climate change will lead to a reduction in cloud interception and critical inputs of mist and water at high altitude (also known as ‘cloud stripping’ or ‘occult precipitation’). Occult precipitation can only occur when the cloud base falls low enough topographically to be intercepted by vegetation, which typically occurs at night and on mountains. A lifting cloud base has already been linked with declines of amphibians and altitudinal shifts in the distribution of birds in the cloud forests of Monteverde, Costa Rica (Pounds *et al.* 1999).

A reduction in occult precipitation as a result of a lifting cloud base may have important implications for both flora and fauna, particularly in mountainous regions of Queensland. In the dry season, up to 66% of the total water input to Wet Tropics rainforests above 1000m can be attributed to cloud interception (McJannet *et al.* 2007). A reduction in this water input will potentially affect many taxa requiring high, consistent moisture levels (e.g., microhylid frogs, litter skinks, soil invertebrate fauna, microbes). In the Wet Tropics, these impacts are particularly relevant to the frogs of the family Microhylidae (a diverse group of restricted regional endemics), where species richness is limited by low rainfall in the dry season (Williams & Hero 2001). Furthermore, vegetation communities that have historically been subject to overnight immersion in cloud will be faced with increased evapo-transpiration rates. These effects are likely to indirectly impact insectivores and litter processes, such as decomposition and nutrient cycling (Williams *et al.* 2003), and may have serious implications for vegetation composition and structure with a potential shift from mesic communities to communities with more sclerophyllous components. The Australian National University (2009) has identified the *Nothofagus*-dominated microphyll fern forest to be at particular risk from a rising cloud base in the Gondwana Rainforests of Australia World Heritage Area.

## 2.5 Sea level rise

Sea levels are predicted to rise throughout the 21st century and beyond, with high levels of certainty. Projections based on the IPCC fourth assessment report (2007) conservatively estimate an average 0.8m global sea level rise by the end of this century (Nicholls *et al.* 2011). Rising sea-levels have general consequences for coastal inundation and will influence the interface between marine, freshwater and terrestrial ecosystems. For terrestrial ecosystems in the coastal zone, rising sea levels will be associated with increased coastal erosion processes through the dynamics of interaction between storm-surge and tides, accelerated by extreme events, such as the confluence of intense storms at king tide. Low lying freshwater ecosystems will be inundated with sea water, altering critical water tables in coastal wetlands and precipitating complete turn-over in species composition to salt-dependent mangrove, salt flat and salt marsh communities. Further details about the effects of sea level rise on coastal ecosystems are presented in Bustamante *et al.* (2012).

## 2.6 Fire regimes

Fire is an important terrestrial transformer of Australian ecosystems (Bradstock 2010). The fire regime concept (Bradstock 2010; Bradstock *et al.* 2002; Gill 1975; Gill & Allan 2008) provides the conceptual framework for understanding the patterning and impacts of fire on the landscape. Central to the regime concept is that fires recur and, at any one point, vary in season of occurrence and in frequency (and hence the length of the intervals between fires) and intensity. There may be considerable variation at any point in the landscape in these components of the fire regime. Fire regimes also vary considerably across Australia, because of variation in regional weather systems, the dynamics of biomass growth (to produce combustible fuel) and seasonal variation in fuel moisture (Bradstock 2010; Russell-Smith *et al.* 2007; Williams *et al.* 2009).

Queensland currently experiences a range of different fire regimes with frequent (annual) fires burning in the savannas of Cape York and northern Queensland and less frequent woodland and forest fires occurring in southeast Queensland when conditions are suitable. As fire has the ability to modify habitat structure and directly influence plants and animals through mortality and loss of shelter and food, it is an important aspect to consider in this report. This is further emphasised by the fact that the frequency, intensity and influence of fire is tightly coupled with weather and climatic conditions both at the time of the fire and leading up to the fire. Projected changes in climate, such as increased temperatures, additional plant growth through $CO_2$ fertilisation and changes in rainfall, may alter the risk of fire and alter fire regimes likely to occur in Queensland in future.

# 3. CLIMATE CHANGE AND FIRE REGIMES

## 3.1 Introduction

Climate change will affect individual fires through the effects of changes to temperature, rainfall, humidity, wind – the fire weather components – and through changes to fuel production in response to the impacts of elevated atmospheric $CO_2$ and changes in moisture on vegetation. Because individual fires may change in their characteristics, so too may fire regimes (defined in Section 3.2). There is general debate about the relative importance of drivers, such as weather, fuel, ignitions and related influences of land management, in shaping fire regimes (Marlon *et al.* 2008; Pechony & Shindell 2010; Westerling *et al.* 2006). However, the biophysical components as well as the management opportunity and constraints are all affected in one way or another by climate. Hence climate change is likely to bring potential changes to fire regimes. However, it is also important to understand the relative sensitivities of local weather and fuel factors to climate change and the extent to which climate change impacts on these two critical drivers to act synergistically or antagonistically (Williams *et al.* 2009).

## 3.2 Regional variation in fire regimes

Variation in fire regimes across Australia results from biogeographic variation in four key processes, which can be thought of as hypothetical ‘switches’ (Bradstock 2010). These are: 1) biomass amount, 2) availability of fuel to burn, 3) fire weather and 4) ignition. The different ‘switches’ must all be ‘on’ simultaneously for fire to occur. They are activated at different rates in different places and different places have a different limiting switch – the switch that is activated least often. Thus, in the tropical savannas of northern Australia, fire regimes are generally high frequency and relatively low intensity (e.g., 1-5 year recurrence intervals and intensity <10,000 kilowatts per metre, kW/m). This is a consequence of the annual production and curing of grassy fuels and an annual hot, dry season (Williams *et al.* 2002). In contrast, in the eucalypt-dominated forests of the temperate regions of southern Australia, fire regimes are of relatively low frequency and high intensity (e.g., multi-decadal recurrence intervals and intensity >10,000 kW/m). This regime is a consequence of the infrequent co-occurrence of severe fire weather and prolonged drought (Bradstock 2008; Bradstock 2010; Gill & Catling 2002). Fires also occur in the arid zone at decadal frequencies. This is because several years with higher-than-average rainfall are required to produce enough grassy biomass for fire to propagate across the landscape (Allan & Southgate 2002; Russell-Smith *et al.* 2007).

These findings can be applied to Queensland by considering the broad climatic and ecological characteristics of biomes. To demonstrate this spatially, we developed a landscape-ecological grouping of the agro-climatic classification developed by Hutchinson *et al.* (2005; 1992) and Hobbs and McIntyre (2005). A correspondence between the 18 global agro-climatic zones that occur in Australia and the Australian biogeographic regions (Hutchinson *et al.* 2005) was developed as an aid to assessing the generality of landscape management guidelines (Hobbs & McIntyre 2005). The 13 agro-climatic zones that occur in Queensland fall into 6 broad landscape-ecological types (Figure 3).

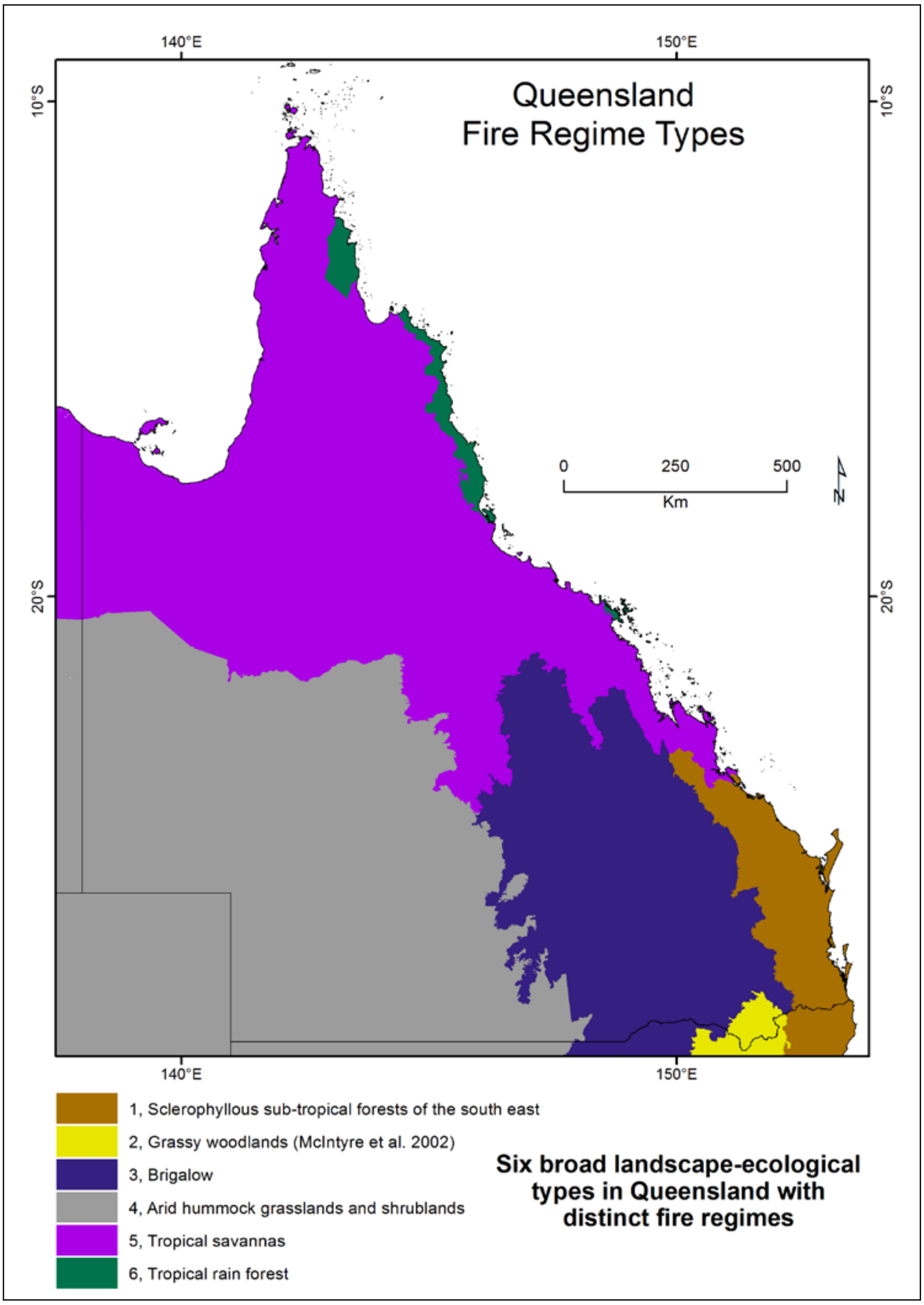


Figure 3: Six broad landscape-ecological types in Queensland with distinct fire regimes: 1) eucalypt open-forests of the south-east and eastern seaboard with spring-summer rain in the south-east, which may have shrubby or grassy understoreys (classes E7 and F4 of Hutchinson *et al.* 2005); 2) An extensive arc of woodland dominated by eucalyptus with variously grassy understorey in the sub-humid region of the Great Dividing Range (classes D5 and E3 of Hutchinson *et al.* 2005); 3) an extensive belt of woodland dominated by *Acacia harpophylla* (brigalow) in inland and eastern Queensland from Townsville in the north to northern New South Wales (class E4 of Hutchinson *et al.* 2005); 4) hummock grasslands or shrublands with a hummock grassland understorey of the arid interior (classes E6 and G of Hutchinson *et al.* 2005); 5) tropical savanna woodlands in the wet-dry tropics of the Gulf and Cape regions (classes H, I1, I2 and I3 of Hutchinson *et al.* 2005); and 6) tropical rain forest of the eastern seaboard in north and far north Queensland (classes J1 and J2 of Hutchinson *et al.* 2005). Boundaries of landscape ecological types follow sub-bioregions based on IBRA 5.1 (Environment Australia 2000).

The fire regimes within these types vary from biome to biome and can be generalised as follows.

1. Sclerophyllous sub-tropical forests of the south east. These ecosystems have inter-fire intervals of 5 to 25 years. The fire regimes are limited by a mix of weather and fuel. However, there is very strong interaction between fire frequency and understorey type. Places with a shrubby understorey are generally associated with low fire frequencies. A grassy understorey is typical where fire frequency is high (Henry & Florence 1966; Watson 2001).

2. Grassy woodlands (McIntyre *et al.* 2002). These ecosystems have inter-fire intervals greater than 10 years (Lunt *et al.* 2006). Fire regimes are strongly dependent on production of herbaceous biomass; hence fire regimes will vary as a consequence of annual variation in rainfall and its subsequent effects on grass production.

3. Brigalow. In general the frequency of fire in Acacia-dominated woodlands in Australia is low (Hodgkinson 2002) with intervals of 10 to 50 years. Brigalow stands may have sparse understorey and closed canopy and hence may not be flammable. The fire regimes of brigalow ecosystems are poorly understood (Fensham 1997).

4. Arid hummock grasslands and shrublands. These ecosystems have inter-fire intervals of 10 to 80 years. Fire regimes here are strongly dependent on grass production following irregular, heavy rainfall (Allan & Southgate 2002).

5. Tropical savannas. Inter-fire intervals in these ecosystems are around 1 to 10 years. Fire weather is generally non-limiting and, like the grassy woodlands of the eastern seaboard and tropical savannas elsewhere in Australia, savanna fire regimes are influenced by variation in annual rainfall and ignition patterns (Dyer *et al.* 2001; Russell-Smith *et al.* 2009; Williams *et al.* 2002).

6. Tropical rain forest. Here, fire is rare and inter-fire intervals may be measured in centuries to millennia (Russell-Smith & Stanton 2002). Fire affects rainforest communities by progressing into the forest from the adjacent non-rainforest grassy or sclerophyll vegetation.

Figure 4: Most fires in the tropical savannas of northern Australia are lit by land managers in the early Dry Season (May-June). Fires at this time of the year tend to be very low in intensity, patchy and limited in extent (credit: CSIRO science image, DA0220).

In Queensland, the fire regimes of all the major biomes have been affected substantially by extensive clearing for agricultural and pastoral development (e.g., see Figure 4), forestry and/or urban development over the past 200 years (Dyer *et al.* 2001; Myers *et al.* 2004; Russell-Smith *et al.* 2009).

## 3.3 Climate change and fire weather

Climate change will have impacts on fire regimes through its effects on fire weather. One widely-used measure of fire weather in Australia is the Forest Fire Danger Index (FFDI; Luke & McArthur 1978). It is a measure of temperature, humidity and wind speed, coupled with a drought index. FFDI can be calculated at various temporal scales, from hourly to annually.

One measure of FFDI that has been used in relation to climate change is the change in cumulative annual FFDI – the sum of the daily values over the course of the year (∑FFDI). Lucas *et al.* (2007) analysed the patterns of ∑FFDI from 2001-2007 and compared them with the values for 1980-2000. ∑FFDI rose by 10-40% at many sites in south-eastern Australia from 2001-2007 relative to 1980-2000. These data, plus additional data not considered by Lucas *et al.* (2007) can be summarised for seven sites in Queensland and northern NSW (Table 1 and Figure 5). These, sites are representative of the six broad Queensland agro-climatic classes (Figure 3). An upward trend in ΣFFDI is apparent at all locations, as has been detected in most other areas of Australia where such analyses have been undertaken (Williams *et al.* 2009).

The potential impacts of climate change on fire weather in south-eastern Australia were also assessed by Lucas *et al.* (2007). Their modelling was based on historical daily weather patterns from various stations in south-eastern Australia, coupled with the outputs from a range of IPCC climate change scenarios, using two CSIRO climate models. The results suggested an increase in both the cumulative annual fire danger and in the incidence of days of high fire weather danger. That is, days when the FFDI is in the ‘Very High’ category or above (i.e., >25 FFDI). By 2020, the increase in ΣFFDI is generally 0–4% for the low emissions scenarios and 0–10% for the high scenarios. By 2050, the increase is generally from 0–8% (low) and 10–40% (high). With respect to the incidence of days where FFDI >25 the modelling shows an increase of 5–65% in the incidence of such days by 2020 and 5–110% by 2050. Thus, further increases in the frequency of extreme fire-weather events in the whole of south-eastern Australia are likely. As a rule, projected changes under the high emissions scenario are substantially larger than those of the low emissions scenario. The results for the Queensland stations are within this projected envelope, both with respect to ∑FFDI (Table 2) and the incidence of severe fire weather (Table 3). Increases in these measures of fire weather risk are apparent for sites within the south-east, the arid interior and the coastal tropical and sub-tropical seaboard of Queensland.

This does not imply that elevated risk of fire occurrence or severity, as measured by increases in measures of fire weather, translates evenly across biomes into changes in fire regime. In the arid zone fire regimes are limited by infrequent rainfall events. The recurrence rate of these major driving events under climate change scenarios is unknown. On the other hand, regions where fire regimes are more under the influence of fire weather and the recurrence rates of major droughts, such as was the case in the south east and the eastern seaboard, may experience an increase in the risk of more extensive and/or intense fires. Climate change will also affect the opportunities for prescribed burning during the cooler, winter months. Under some circumstances, the opportunities for undertaking prescribed burning safely may increase under climate change scenarios (Williams *et al*. 2011).

Table 1: Average change in annual cumulative Forest Fire Danger Index (∑FFDI) from 1981 to 2001 and from 2002 to 2010. The index is the sum of daily FFDI values for a given year. The percentage change between the two periods is also shown. Source: Lucas *et al.* (unpublished data) according to methods described in Lucas (2010). The figures differ from those quoted for the analysis presented Lucas *et al.* (2007) because these data are for a slightly different sequence of times and the data are also corrected for wind direction.

| **Site** | **1981–2001 ∑FFDI** | **2002-2010 ∑FFDI** | **% change** |
|---|---|---|---|
| Amberley | 3354 | 3790 | 13 |
| Brisbane Airport | 2025 | 2153 | 6 |
| Charleville | 6493 | 6983 | 8 |
| Longreach | 7979 | 8896 | 11 |
| Rockhampton | 3240 | 3851 | 19 |
| Cairns | 1848 | 2074 | 12 |
| Bourke (NSW) | 5274 | 7139 | 35 |
| Coffs Harbour (NSW) | 1378 | 1496 | 9 |

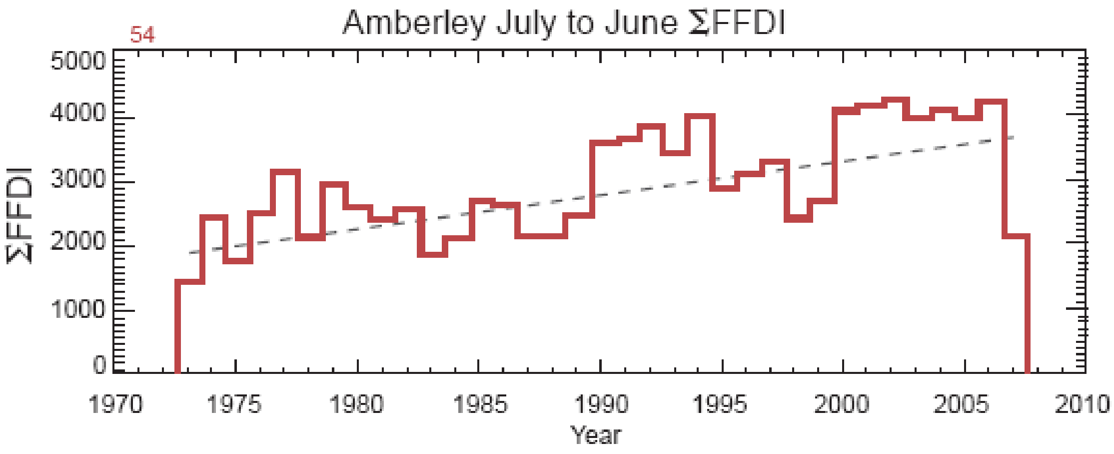


Figure 5: An example illustrating the rise in annual cumulative Forest Fire Danger Index (∑FFDI) from 1973 to 2007 for Amberley Bureau of Meteorology station (near Ipswich/Brisbane). The dashed line represents the line of best fit; the trend is 54 FFDI points per year over the period (red number on the upper left). Source: Lucas *et al.* (2007).

Table 2: Projected changes to annual cumulative forest fire danger index ( ∑FFDI)for selected stations in Queensland and northern New South Wales. Annual (July to June) average cumulative FFDI ( ∑FFDI) for ‘present’ conditions (based on 1973–2006 data) and for projected changes (% change) under selected 2020 and 2050 climate change scenarios, relative to 1990. The outputs are for two climate change models: CCAM (Mark 2; ‘Mk2’) and CCAM (Mark 3; ‘Mk3’). Source: Lucas *et al.* (2007) and Lucas et. al. unpublished data (*) NA = data/analyses not available.

| Site | Present ∑FFDI | 2020 Low Mk2 % Δ | 2020 High Mk2 % Δ | 2020 Low Mk3 % Δ | 2020 High Mk3 % Δ | 2050 Low Mk2 % Δ | 2050 High Mk2 % Δ | 2050 Low Mk3 % Δ | 2050 High Mk3 % Δ |
|---|---|---|---|---|---|---|---|---|---|
| Amberley | 2919 | 2 | 7 | 1 | 6 | 4 | 24 | 3 | 19 |
| Brisbane airport | 1990 | 1 | 5 | 0 | 4 | 3 | 19 | 2 | 16 |
| Charleville | 6127 | 4 | 11 | 2 | 8 | 7 | 37 | 5 | 25 |
| Rockhampton | 3166 | 2 | 6 | 2 | 7 | 4 | 21 | 4 | 22 |
| Cairns* | 1580 | NA | NA | NA | 6 | NA | NA | NA | 16 |
| Bourke (NSW) | 4758 | 4 | 10 | 3 | 8 | 7 | 33 | 5 | 26 |
| Richmond (NSW) | 2152 | 1 | 6 | 2 | 8 | 3 | 20 | 5 | 26 |
| Coffs Harbour (NSW) | 1255 | 1 | 3 | 1 | 6 | 2 | 11 | 3 | 18 |

Table 3: Projected changes to number of days per year with a fire danger rating of ‘very high’ or ‘extreme’ (FFDI ≥ 25) for select stations in Queensland. Figures are the number of days per year under present circumstances (Current) based on records from 1973–2007; and the percentage change (% Δ) from the current value for various model scenarios to 2020 and 2050. Outputs for high and low emissions scenarios and CCAM (Mark 2): ‘Mk2’ and CCAM (Mark 3) ‘Mk3’, as per Table 2. Source: Lucas *et al.* (2007).

| Site | Current No. days | 2020 Low Mk2 % Δ | 2020 Low Mk3 % Δ | 2020 High Mk2 % Δ | 2020 High Mk3 % Δ | 2050 Low Mk2 % Δ | 2050 Low Mk3 % Δ | 2050 High Mk2 % Δ | 2050 High Mk3 % Δ |
|---|---|---|---|---|---|---|---|---|---|
| Amberley | 13.3 | 8 | 6 | 23 | 18 | 15 | 11 | 70 | 57 |
| Brisbane airport | 5.2 | 4 | 2 | 14 | 12 | 9 | 7 | 63 | 45 |
| Charleville | 89.0 | 7 | 5 | 22 | 15 | 14 | 9 | 66 | 42 |
| Rockhampton | 11.2 | 7 | 6 | 17 | 20 | 10 | 14 | 66 | 73 |
| Bourke (NSW) | 57.2 | 9 | 7 | 25 | 20 | 16 | 13 | 81 | 60 |
| Richmond (NSW) | 13.3 | 4 | 6 | 14 | 23 | 9 | 13 | 53 | 77 |
| Coffs Harbour (NSW) | 1.5 | 6 | 6 | 22 | 20 | 20 | 18 | 57 | 71 |

## 3.4 Climate change and fuels

Climate change will have complex effects on fuels. On one hand, elevated $CO_2$ may enhance vegetation production (Wang 2007) and thereby increase fuel loads. On the other hand, declines in annual moisture availability, which is already occurring in southern Australia and projected to decline further (CSIRO & BOM 2007a), may decrease long-term vegetation production. It follows that fuel loads may also decrease because the productivity of terrestrial ecosystems is strongly positively correlated with gross measures of climate including annual rainfall (Begon *et al.* 2006; Donohue *et al.* 2009; Moles *et al.* 2009). Furthermore, drought may decrease the moisture of fuel that is present, thereby increasing potential rates of fire spread. However, research into these interactions is in its infancy. Williams *et al.* (2011) presented data on research undertaken on the impact of climate change on fuel mass and fire rate of spread in temperate eucalypts forests of the Sydney basin of NSW. This study linked climate change projections for rainfall to fuel production models and found that a warming and drying climate produced lower fuel amounts but greater availability of fuel to burn due to lower moisture contents. Changing climate increased potential rate of fire spread. Importantly, reduced fuel loads ameliorated weather induced changes in rate of fire spread for one model. These findings highlight an important principle identified by Williams *et al.* (2009) that the projected outcomes of climate change – in this case warming and drying – on the determinants of fire regimes may act synergistically or they may act antagonistically.

An example of synergistic interaction between the determinants of fire regimes would be more severe weather acting on a landscape within which there is more fuel and/or the fuel is more available to burn due to declines in fuel moisture. This combination of circumstances would, all other factors being equal, increase fire intensity and/or area burnt. An example of an antagonistic interaction would be a decline in the long-term fuel mass due to climate change; for example, due to declining moisture; which may act to counter the effect of more severe weather. Such antagonistic or synergistic scenarios are both plausible in future climate change scenarios but the long term prognosis based on the outcome of such interactions is highly uncertain. For example, “interval squeeze” may be induced by lower moisture under climate change. Intervals between fires may shorten but, under a regime of reduced moisture, the interval-fire needed for species to achieve critical life stages may increase. This has the potential to cause major changes to species and vegetation composition and structure (Williams *et al.* 2009). Thus we need to understand the interplay between disturbance regimes, variability in resources, climates and resulting biotic responses. Research into such complexity is in its infancy. There have been no detailed studies of this type in Queensland, although the FACE (Free-Air $CO_2$ Enrichment) experiment in the savannas of the Townsville region (Stokes *et al.* 2003; Stokes *et al.* 2005) has indicated that grass fuel loads can increase under elevated $CO_2$.

The impacts of climate change on fuel dynamics and subsequent impacts on fire regimes are uncertain and require more detailed region-specific research. The implications of climate change for fire and biodiversity management is outlined in Section 8.5, in the context of climate adaptation responses for terrestrial biodiversity. In forthcoming sections we consider climate change and fire regimes further from the point of view of the ecological adaptive capacity of ecosystems (Section 5.4) and possible regional variation in ecosystem responses (Section 6.5).

# 4. ECOLOGICAL CHANGE

## 4.1 Introduction

In this report we define ecological change as the response of living systems to the environmental or physical changes arising from climate change and the drivers of climate change. Ecological change cascades from the effects on an organism's individual biology, through individual ecology to changes in species populations, to communities and ecosystems (Box 1). In this section we outline the ecological changes expected at each of these levels and emphasise the complexity involved in predicting the consequences of climate change at the ecosystem level. The complexity is driven by uncertainty about idiosyncratic species responses, the role of changing ecological interactions and interactions with other stressors (i.e., threat syndromes, see Section 4.5) that modify responses and how these effects cascade through each level of biological organisation.

Climate affects a suite of ecological, physical and physiological factors that all may act on population processes (rates of birth, death, dispersal and so forth) to shape the distribution of abundance within a species range and the limits of species ranges (Brown *et al.* 1996; Carter & Prince 1981; Zacherl *et al.* 2003). While the magnitude of climate change (since the onset of rapid warming by the 1970's) presently being experienced by an organism is relatively low, the first responses observed are likely to be changes in individual biology (including physiology, behaviour, phenology and so forth). Over time and as the magnitude of climate change increases in line with climate projections (see Williams & Crimp 2012) these changes in individual biology will modify demographic parameters (e.g., recruitment success and mortality, dispersal effectiveness). Changes in demographic rates may result in shifts in abundance of distributions within a species' geographic range. In fact, since most species will not or cannot intentionally migrate to track climate change. Shifts in distributions must be preceded by changes in demographic rates in response to improving or deteriorating climatic conditions throughout the species' range and/or changes in recruitment success and dispersal effectiveness beyond current range boundaries.

At the landscape scale, species range contractions, expansions and shifts are considered the most likely responses to relatively rapid climate change. Shifting species ranges are in fact the complex result of extinction of metapopulations (a group of spatially separated populations of the same species which interact at some level) at their geographic/environmental range limits, where local conditions become increasingly unsuitable, and colonisation and growth of metapopulations into adjacent or accessible regions that become climatically suitable for persistence. So, to understand the potential effects of climate change on a species, we must consider the dynamics of populations throughout a species geographical range, in conjunction with the spatial configuration of suitable and unsuitable habitat that influence the function and permeability of landscapes for effective dispersal. The important influence of spatial features of the landscape on a species response to climate change is highlighted in Section 5.3. When a species is unable to achieve an adequate response of any other kind, local, regional or species extinction is the likely result. Climate change interactions with other stressors and threats, such as invasive species, fire and the prevalence of disease, will also shape species' responses. Other ecological factors are also important, such as where the interface between two strongly competing species can limit the range of both (Bull & Possingham 1995).

Much progress has been made in understanding the potential impacts of climate change at the individual species' population level, particularly with increasing computing power to enable modelling of potential changes in species distributions through correlation with environmental factors. However, the underlying assumptions of these models are the subject of regular debate and their ability to predict ecological outcomes is unproven (Dawson *et al.* 2011). Scaling from individuals to populations through to communities and ecosystems will be very challenging as the population response for many species will not be additive (Walther 2010). Furthermore, assessing the consequences of climate change on biodiversity is complicated by uncertainty about the rate and magnitude of climate change, the potential for novel and disappearing climates (see Ferrier *et al.* 2012), the diversity of individual species responses to different and interacting climate variables and the interacting effects of climate change with other stressors (Dawson *et al.* 2011).

Predicting ecological or ecosystem change requires a detailed understanding of many fields of research including biology, ecology, hydrology, climatology, to name a few. It also relies on adequate sampling of observations through field research and monitoring data. As outlined in the example presented in Box 1, the leaf-scale physiological processes of photosynthesis scale up to impacts on landscape function through the canopy function of populations of trees. This example demonstrates the complexity of understanding required in order to infer potential change in stands of trees. For example, while the leaf-level responses of plants to $CO_2$ concentrations are well established, the ecosystem consequences are very hard to predict; a generality from global FACE experiments is that the initial stimulation of a step change in $CO_2$ at the leaf and whole-plant level decreases with time and that the population and ecosystem levels respond much less than a naïve extrapolation from the leaf level would predict. However, scaling up processes to whole landscapes, this example demonstrates the potential cascades of the effects of climate change from the composition of individual tree stands, to habitats and the biodiversity they support, with a flow-on of effects to other ecosystem services valued by people.

Box 1: An example of the scales of change and processes involved from the leaf to the landscape based on an understanding of vegetation dynamics (including extracts from Huxman & Scott 2007).

*Ecological Processes*

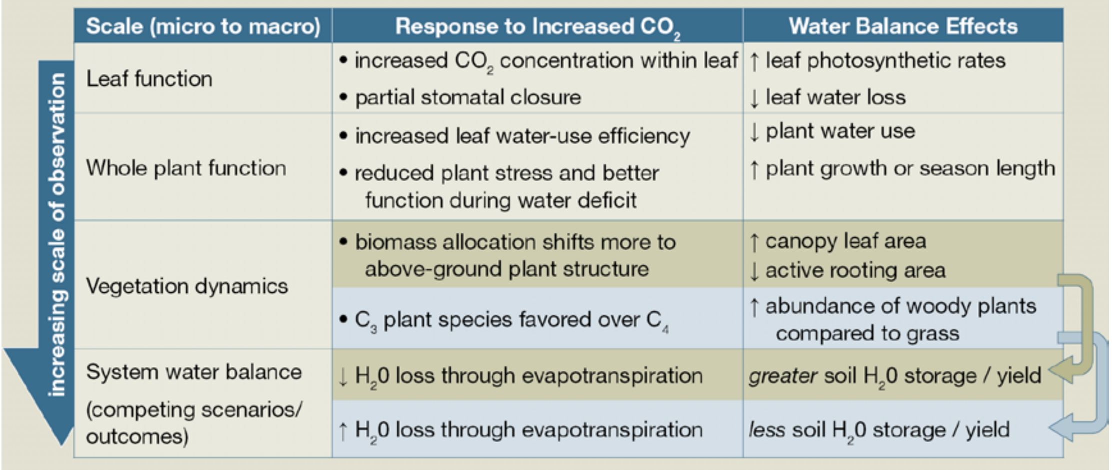
Carbon-dioxide ($CO_2$) in the atmosphere has increased by over 100 parts per million since the beginning of the industrial revolution. Rising $CO_2$ concentration has implications for atmospheric temperature change due to its greenhouse gas characteristics but it also has significant ramifications for how vegetation functions on landscapes and its ecohydrological consequences.

$CO_2$ is the primary substrate for photosynthetic energy acquisition by life. The process uses light energy to combine $CO_2$ and water to produce organic compounds and releases oxygen. Since photosynthesis is an unsaturated biochemical reaction in plants, rising $CO_2$ concentrations increase photosynthetic rates under current conditions. Studies have shown that this change in photosynthetic rate results in greater above- and below-ground plant growth, especially in water-limited regions. Also, at higher than current ambient $CO_2$ concentrations, plants reduce the apertures of the small pores in their leaves – stomates – that permit $CO_2$ and water vapour exchange with the atmosphere. These changes result in a decrease in whole plant water use, which influence the storage of water in the soil and scale up to affect the landscape water balance. Thus, through its impacts on plant water use and surface soil water storage, rising $CO_2$ is predicted to increase recharge and streamflow when scaled to landscapes. This is likely to be most profound in areas where evapo-transpiration and thus vegetation characteristics, dominate the behaviour of the hydrological cycle.

A second major finding is that the composition of plant communities changes at greater than ambient $CO_2$ concentrations due to differential growth and resources use by major plant types in a region. This is because several biochemical types of photosynthesis are found in terrestrial plants, resulting in different responses to changes in atmospheric $CO_2$ concentration. $C_4$ photosynthetic species (which initially form four carbon-atom molecules) tend to be less responsive to rising atmospheric $CO_2$ concentration than $C_3$ species. In Australia, the deep-rooted woody species are predominantly of the $C_3$ photosynthetic type while summer-active and many tropical grasses are dominated by $C_4$ photosynthetic type. Elevated $CO_2$ concentrations favour woody plants over grasses and may accelerate woody-plant thickening or encroachment. Changes in the ratio of woody plants to grasses can influence the landscape water balance by affecting recharge and streamflow: larger woody vegetation populations would be expected to increase the amount of water leaving landscapes as evapo-transpiration.

Thus, rising $CO_2$ concentration suggests two contrasting water resource scenarios. On one hand, we expect greater plant efficiency with respect to water use that will increase landscape yield but, on the other hand, changes in vegetation will influence how much water is returned to the atmosphere through evapo-transpiration.

***Figure**: Vegetation impacts on water balance due to rising atmospheric CO2 concentration. At larger scales, two different scenarios emerge, one that would provide greater water and the other providing less water (Huxman & Scott 2007).*



| Scale (micro to macro) | Response to Increased $CO_2$ | Water Balance Effects |
|---|---|---|
| Leaf function | • increased $CO_2$ concentration within leaf | ↑ leaf photosynthetic rates |
| | • partial stomatal closure | ↓ leaf water loss |
| Whole plant function | • increased leaf water-use efficiency | ↓ plant water use |
| | • reduced plant stress and better function during water deficit | ↑ plant growth or season length |
| Vegetation dynamics | • biomass allocation shifts more to above-ground plant structure | ↑ canopy leaf area<br>↓ active rooting area |
| | • $C_3$ plant species favored over $C_4$ | ↑ abundance of woody plants compared to grass |
| System water balance (competing scenarios/ outcomes) | ↓ $H_2$0 loss through evapotranspiration | *greater* soil $H_2$0 storage / yield |
| | ↑ $H_2$0 loss through evapotranspiration | *less* soil $H_2$0 storage / yield |

*Ecological Outcomes*

At the physiological level, trees need to open the pores on their leaves (stomata) to allow $CO_2$ to enter for photosynthesis to happen and assimilate carbon (C) as mass and energy. In opening these stomata, water vapour lost as evapo-transpiration is replaced with water absorbed through the roots from soil water stores.

Increases in the amount of $CO_2$ in the atmosphere means that trees do not need as many stomata or that they may open their stomata for shorter periods to assimilate the same amount of carbon as they did with lower $CO_2$ concentrations. This increase in water use efficiency results in less water used per tree, with potentially for more plants to grow using the available environmental water or more growth for a given tree utilising the same amount of water.

If a plant or tree runs out of water in the soil, and is unable to compensate through tolerance and avoidance behaviours in the face of continuing atmosphere demand for evaporation, the plant will die. Different species have different sensitivity to low soil water availability, different abilities to store water and different mechanisms to tolerate or avoid drought stress. It is generally expected that the dynamic mixtures of species native to any particular region have co-evolved through inter-dependence and competitive interactions to efficiently utilise available water supplies. Therefore, the hydrological processes of soil water infiltration, deep drainage, runoff and storage, that depend on the characteristics of the substrate, as well as rainfall regimes of amount, seasonality and intensity, are critical in determining the potential standing biomass of plants that any given region can support.

As trees themselves are effectively hydrological pumps—moving water from the soil to the atmosphere—they are critical to local, regional and landscape water balance. This movement of water by trees has many affects including humidifying the air to form clouds that return the moisture as rain (e.g., in montane rainforest and in semi-arid shrubby landscapes). Vegetation biomass in balance with its regional potential helps suppress the height of water tables, limits salinisation from rising ground water, influences deep drainage into ground water and runoff into stream flows.

Plants and associated litter also act as filters protecting soil from erosion, slowing the flow of water across the land and allowing time for infiltration. This service builds soil profiles, cleanses water and allows runoff to move into creeks, rivers and wetlands with reduced sediment loads.

Any change to these processes will result in changes to the ecological dynamics that influence the retention of tree stands. For example, tree clearing in some regions removes the process of evapo-transpiration in which soil water is returned to the atmosphere, resulting in a rise in water tables that bring mineral salts to the land surface, change the soil environment and prevent establishment by plants other than the most specialised salt-resistant types, often lower plants, herbs or shrubs.

Trees are relatively long-lived species (many years, usually decades and sometimes centuries) with considerable capacity to tolerate adverse conditions, up to a point. It is often difficult to know the current status of a stand of trees in the natural landscape water/tree balance, without conducting detailed physiological studies of water and carbon flux. The water requirement of the tree stand may be less than that available from the current rainfall regime and soil stores. In this case, one conjecture might be that the stand is recovering from a previous disturbance (fire/drought/cyclones/storms). Alternatively, the stand requirements for water may be exceeding availability, in which case stem death and stand thinning are likely consequences. The dynamic status of a stand of trees in this delicate balance between total biomass production and environmental water availability cannot be measured over a single year. It depends on the seasonal dynamics of rainfall and the inter-annual variability of rainfall.

*Land Management*

Many land management issues relate to changes in these hydrological processes combined with the outcome of competitive interactions between tree/grass systems which result in increased or decreased tree biomass. Factors that influencing any single aspect of a trees' lifecycle from establishment, growth, flowering, seeding, germination and survival to senescence can result in changes that cascade to whole stands of trees. As deep-rooted woody plants become more abundant in the landscape, photosynthesis and evapo-transpiration becomes more closely coupled with longer term processes associated with groundwater than with surface water and short-term rainfall patterns. The loss of water to the atmosphere and sequestration of carbon are both controlled by vegetation such that the two processes are negatively related: systems with greater water loss through evapo-transpiration also sequester or accumulate more $CO_2$ from the atmosphere. As such, land managers likely will need to consider the potential trade-off between these two exchanges at the landscape scale in making decisions on vegetation management.

Climate change will alter temperature, $CO_2$ concentrations and rainfall volume and seasonality. These changes will result in shifts in the landscape water/tree balance and may cause shifts in tree species composition, density and structure at any location in Queensland. At a broad scale, high rainfall supports the northern rainforests; fire-prone savannas dominate as rainfall becomes more seasonal inland and in the monsoonal north; and where rainfall is low and variable between years, grasslands and semi-arid vegetation dominate in the south west of the State.

Typical tropical savanna in Northern Australia (near Palmerston) demonstrating the high tree density and regular spacing characteristic of many savannas (source: Bidgee, 28/11/2008; Wikipedia Commons).

Coastal rainforest near Townsville, far northern Queensland (credit: John Coppi, CSIRO Land & Water, science Image BU12191).

Changes to tree stands will ultimately impact the services provided by trees, such as habitat for biodiversity, erosion control, timber and forestry products, genotypes for horticulture, shade and carbon storage. This process from the leaf to ecosystems and landscapes illustrates the complex pathway of processes that flow from climate change interacting with vegetation and that result in dramatic deviations from the current character of an ecosystem.

## 4.2 Changes in individual biology

### 4.2.1 Introduction

This section outlines the types of changes in individual biology which may occur as climate changes throughout a species geographic range. Biological responses to climate change include the potential for changes to growth and photosynthetic rates in plants (Cunningham & Read 2002, summarised for Queensland by Low 2011), changes in metabolic rates (Dillon *et al.* 2010), changes in phenological patterns (Parmesan 2006) and changes in behaviour, morphology and physiology (Shoo *et al.* 2010; Telemeco *et al.* 2009). In some respects, though onerous to measure, changes in individual biology provide some of the clearest evidence of climate change responses because these are generally less directly influenced by anthropogenic activity and management than are measures of distribution or abundance.

Phenotypic plasticity is defined as the ability of an organism to change its morphology, development, biochemical or physiological properties or behaviour in response to changes in the environment. Plasticity can both provide a buffer against rapid climate change and assist rapid evolutionary adaptation (Chevin *et al.* 2010; Hof *et al.* 2011; Nicotra *et al.* 2010). Thus, many species may minimise negative impacts resulting from climate change by adapting their individual biology or by utilising micro-habitats (micro-refugia) that buffer against climate change and extreme events (Shoo *et al.* 2010). These strategies are also likely to have allowed many species to persist through past rapid climatic changes (Hof *et al.* 2011). We describe some of this adaptive capacity in this section.

### 4.2.2 Phenology

Climatic patterns are critical in setting the timing of phenological events (Walther *et al.* 2002). Phenological changes that have been linked to climate change over the last 30 years include earlier flowering of plants, early arrival of migrant birds and earlier spawning in amphibians (Bertin 2008; Chapman *et al.* 2005; Parmesan & Yohe 2003; Walther *et al.* 2002). Some of the best documented effects of phenological responses to climate change have been shifts in leaf phenology and flowering time (Parmesan & Yohe 2003). These changes may reflect both genetic and plastic changes (Nicotra *et al.* 2010). However, the vast majority of measured changes in phenology in response to climate change come from the northern hemisphere due to a long tradition in recording the timing of spring events (e.g., Ibáñez *et al.* 2010; Thackeray *et al.* 2010). Recently in Australia, researchers have begun to pay particular attention to phenological changes in species' responses to climate change (Howden *et al.* 2003; Hughes 2000). As a result we generally only have relatively short-term datasets to draw from and any longer-term datasets have usually not been collected with enough regularity to detect subtle changes (Beaumont *et al.* 2006; Chambers 2009; Gallagher *et al.* 2009; Hovenden *et al.* 2008a; Hovenden *et al.* 2008b; Kearney *et al.* 2010b).

A large number of Australian bird species are partial migrants (Chan 2001), typically migrating along a north-south axis and many exhibit complex migratory pathways (Griffioen & Clarke 2002). In a comprehensive study on phenological response of Australian birds to climate change, Beaumont *et al.* (2006) found that, as with Northern Hemisphere birds, migratory birds visiting or breeding in south-east Australia have undergone phenological shifts in migration since the 1960s. Of the 24 species included in their study, 50% have shown a significant trend toward earlier arrival since 1960, while only one species is arriving significantly later. One of the many implications of this is that, if breeding, non-breeding and stop-over sites vary in the extent to which climate is changing, species may find their arrival at migratory sites to be out of synchrony with weather conditions and peak food supplies. In addition, research in the northern hemisphere has shown that advancing arrival dates that become out of synchrony with life-history adaptations, such as egg laying dates, have major impacts on reproductive success (Both & Visser 2001).

A recent analysis of the timing of breeding in some common and widespread Australian birds could detect no consistent fingerprint of climate change, though some species showed significant trends (Chambers 2007). For example, Willie Wagtails (*Rhipidura leucophrys*) bred earlier in south-east Australia in years with higher average temperatures (Chambers 2007). Masked lapwings (*Vanellus miles*) have also been shown to breed earlier in north-east Australia (at a rate of about 2 days per year) over the last 40 years (Chambers *et al.* 2008).

Many species already appear to use a combination of climatic cues and other environmental cues to determine phenology which may help avoid temporal mismatching of climate and life-histories. For example, the painted honeyeater (*Grantiella picta*), an Australian mistletoe specialist, appears to use photoperiod to time its overall breeding season and then incorporates information from the local environment on mistletoe fruit abundance to fine-tune initiation of breeding (Barea & Watson 2007).

### 4.2.3 Behavioural and physiological changes

Recent research using a diverse array of physiological and biophysical approaches indicates that tropical ectotherms (i.e., 'cold-blooded' organisms that regulate their body temperature largely by exchanging heat with their surroundings; e.g., reptiles) may be particularly vulnerable to climate change (Deutsch *et al.* 2008; Dillon *et al.* 2010; Kearney *et al.* 2009). Ectotherms are experiencing increased metabolic rates which will increase need for food, increase vulnerability to starvation, allow less discretionary energy for reproduction and alter demographies (Dillon *et al.* 2010).

Kearney *et al.* (2009) showed that for most terrestrial ectotherms, the primary thermal challenge is not to attain high body temperatures (although this is important in temperate environments) but to stay cool (particularly in tropical and desert areas, where ectotherm biodiversity is greatest). A major mechanism of behavioural thermoregulation in ectotherms is an altered daily and seasonal timing of activity. For tropical and arid system ectotherms, rising temperatures will require reducing activity in the summer months and potentially increasing activity in the drier winter months, a strategy which may require changes in life-history. Buffering via behavioural thermoregulation is also tied strongly to the availability of shade and hence to vegetation cover. Kearney *et al.* (2009) described how altered seasonal activity and shade availability under climate warming also may interact across a landscape to affect rates of energy acquisition. Although ectotherms can avoid overheating by reducing activity during warm periods of the year, their resting body temperatures and hence metabolic rates may be unavoidably high. Thus, thermal constrictions on potential foraging time under climate warming may have an adverse impact on an ectotherm's energy budget.

Nest temperature determines offspring sex for many reptiles; climate warming may skew the sex-ratio of these animals unless they are able to buffer climate change by adapting some aspect of nesting behaviour or through rapid shifts in the thermal threshold that triggers sex determination (Telemeco *et al.* 2009). Researchers suggest the latter strategy is unlikely because the genetic trait is highly conserved (Doody *et al.* 2006). Telemeco *et al.* (2009) monitored nesting behaviour in three-lined skinks (*Bassiana duperreyi*) in cool-climate habitat in eastern Australia over a 10 year period where air temperatures during the reproductive season increased by approximately 1.5-1.7°C. They noted that females adjusted their nesting behaviour by digging deeper nests and ovipositing earlier. However, these adaptive measures were not sufficient to compensate for increased temperatures later in the incubation period and resulted in the crossing of a biological threshold in which offspring sex determination shifted from being thermo-regulated to genetically-regulated. The effects of such a shift on population viability remain unclear (Telemeco *et al.* 2009).

Tropical ectotherms can exploit complex microclimatic mosaics to regulate their body temperatures behaviourally (Kearney *et al.* 2010a). Rocky landscapes (litho-refugia) have been identified repeatedly as important long-term climate refugia for Australian rainforest flora and fauna (Shoo *et al.* 2011). It is thought that rocky landscapes offer stable, long-term habitat that is relatively buffered from short- and long-term variation in temperature and moisture (e.g., see Figure 6 and Figure 7).

Figure 6: Boulder fields and rocky landscapes in high elevation montane landscapes in the Wet Tropics (credit: Andrew Ford, CSIRO).

Figure 7: Boulder fields and rocky landscapes at Iron Range National Park, Cape York (credit: Dan Metcalfe, CSIRO).

Many other animals already use micro-refugia to assist with thermoregulation. Echidnas use a variety of sites including burrows, caves, logs and tree stumps to provide protection against unfavourable conditions (Brice *et al*. 2002a; Brice *et al.* 2002b). Two lizards (*Carlia scirtetis*, *Nactus galgajuga*) and a frog (*Cophixalis saxatilis*), endemic to Black Mountain on Cape York Peninsula avoid high temperatures and low humidity by retreating further beneath boulders (Low 2011). Shoo *et al.* (2010) showed that mountain-top boulder fields in the Wet Tropics can be as much as 10°C lower than near surface conditions. They suggest that species, such as the critically endangered beautiful nursery frog (*Cophixalus concinnus*), may persist longer than expected by sheltering during extreme or prolonged heat in these boulder fields.

### 4.2.4 Genetic adaptation and phenotypic plasticity

Species-level adaptation occurs via genetic evolution to new environments or phenotypic plasticity. Which species are able to adapt via evolution depends on many factors including the ecological breadth of individual genotypes, the distribution of genetic variation for relevant traits and the extent of gene flow between populations (Davis *et al.* 2005). For some organisms with long generation times, the pace of current climate change may be too fast for evolutionary adaptation to occur; only species with relatively rapid life-histories are likely to have adequate time to allow evolutionary adaptive response (Jump & Peñuelas 2005). There is currently little empirical data to suggest that climate change will cause absolute climatic tolerances of a species to evolve sufficiently to allow it to conserve its geographic distribution and thereby inhabit previously unsuitable climatic regimes (Jump & Peñuelas 2005; Parmesan 2006). However, as we have described above, some species will have the capacity to adapt phenotypically to minimise impacts from climate change over the shorter-term, perhaps also buying more time for evolutionary adaptation over the longer term (Jump & Peñuelas 2005).

Rarely, in Australia, heritable genetic changes have been documented as species adapt to changing climate. Researchers have found rapid shifts in the particular polymorphisms of the common fruitfly (*Drosophila melanogaster*) along the eastern coast of Australia, reflecting an equivalent shift of 4° in latitude over the last 20-25 years (Umina *et al.* 2005). Populations in southerly areas now have the genetic constitution of more northerly populations, suggesting an adaptation to drier, warmer conditions. Species may not need to have generation times as fast as Drosophila to adapt rapidly to climate change; rapid evolutionary adaptation has been documented in empirical research on vertebrates as well as insects, in response to anthropogenic changes in the environment, such as heavy metal and air pollution and insecticides and herbicides, as well as changes in food sources and climate (Reznick & Ghalambor 2001). Reznick & Ghalambor (2001) found that almost all cases of contemporary adaptation were associated with ‘colonization’ events, where individuals became established in a previously unoccupied habitat. Thus, colonisation of previously unoccupied but newly suitable habitat as a result of climate change may lead to rapid adaptive evolution.

Hybridisation among some species may enhance capacity to adapt *in situ* to climate change. For example, eucalypts show relatively high levels of gene flow between species (Shepherd & Raymond 2010). Under climate change, new genetic combinations may prove adaptive and spread widely, for example, tolerance to aridity (Low 2011).

## 4.3 Changes in abundances

Changes in survival (death rates), reproductive output and dispersal (birth rates and emigration and immigration) are likely to result in changes to the population size or abundance of individuals. This process can be local (individual sites) or regional (across many sites) and thus influence population dynamics at a range of scales. Changes in a species’ abundance, within current geographic distribution ranges, may be the most obvious and observable signature of climate change. This is because changes in biological and demographic responses require long-term observation and/or detailed experiments. Potential range shifts, contractions or expansions in a species distribution may take a long time to demonstrate through meta-analysis of multiple, repeated observations. Point-in-time sampling can also reveal dramatic changes but attribution to climate change is more difficult.

Changes in individual biology as a result of climate change ultimately manifest as changes in demographic rates of survivorship, reproduction and dispersal. For example, breeding in Australian Magpies (*Gymnorhina tibicen*) has been shown to decrease with increasing temperature, suggesting that if northern, inland and desert areas become hotter and drier, Magpie populations in these regions could undergo rapid declines (Gibbs 2007).

Recent climate change is linked to both increases and declines in the population size of organisms (McCarty 2001). The direction of population size change depends on the species biology (as discussed above), the position of a population within the species’ geographic range and the interactions of the species with other organisms. Rapid declines in population size are of direct concern but the effects of further increases in already abundant, aggressive and ecosystem transforming species are also likely to result in significant changes to ecosystem function (McCarty 2001). For example, the ecosystem transforming high-biomass invasive species gamba grass (*Andropogon gayanus*) is adapted to the seasonally dry tropics and is limited in the south by cold minimum temperatures. Therefore, gamba grass may have the capacity to increase in abundance towards the south with rising temperatures (Csurhes & Hannan-Jones 2008). The potential impacts of gamba grass and its interaction with fire are described in Section 6.5.

Some Queensland research in tropical forests suggests that species might undergo dramatic declines in abundance before changes in distribution area occur (O'Grady *et al.* 2004). Shoo *et al.* (2005) examined the altitudinal abundance patterns of 12 endemic rainforest birds of north-eastern Australia and highlighted the implications of a potential decoupling of population size and distribution area. They showed that nine out of the 12 endemic birds will exhibit disproportionately large losses in population size following an initial reduction in distribution area. Using a modelling approach, Li *et al.* (2009) also demonstrated that the grey-headed robin (*Heteromyias albispercularis*, Figure 8), may go extinct in north Queensland under some climate change scenarios long before the complete loss of its habitat. Of concern is that declines in population size and distributions for organisms, such as long-lived trees may lag rapid climatic change. It may be difficult to detect deteriorating conditions from records of population size or density until the conditions are past those that will guarantee local or regional extinction (Abrams 2002; Jump *et al.* 2006). Importantly, some research has shown that species may become functionally extinct; that is, they cease to successfully perform a key ecological function; long before they actually become very rare or extinct. For example, McConkey and Drake (2006) showed that flying foxes cease to function effectively as seed dispersers on Pacific Islands long before becoming rare. Part of the reason for this is behavioural: several species of flying fox defend feeding territories in fruiting trees. But when flying fox abundance is low, sub-dominant individuals are not chased out of fruiting trees, so they tend to remain in one tree, dropping seeds directly beneath it (McConkey & Drake 2006).

Figure 8: The grey headed robin, *Heteromyias albispercularis* (credit: Adam McKeown, CSIRO)

### 4.3.1 Extinction

When environmental conditions consistently exceed the ability of a species to shift or respond to the change, for example, through avoidance behaviours or acclimatisation, extinction over part or the entire distribution range is a possibility (Parmesan 2006; Thomas *et al.* 2004). Low abundance and low population density have often been associated with higher levels of extinction over a range of timeframes (Lavergne *et al.* 2006; Sutton & Morgan 2009). Species having small ranges are predicted to have greater vulnerability to extinction due to range displacement tied to climate change because they have a narrower tolerance for environmental conditions and more readily show decrease in abundance over even minor changes in environmental gradients (Schwartz *et al.* 2006; Wilson *et al.* 2004). Therefore, areas of endemism are particularly noteworthy in terms of their biodiversity value and the potential for climate change to have the most serious implications for species extinction rates; narrowly endemic species are by definition rare, have small ranges and therefore are potentially highly threatened (Crisp *et al.* 2001). Major centres of endemism for vascular flora in Queensland occur in the Wet Tropics, Border Ranges and Iron Range/McIlwraith Range area of Cape York.

Other characteristics typically considered to predispose species to a higher probability of extinction include ecological specialisation, large body size, slow generation times and high trophic level (Isaac *et al.* 2009; O'Grady *et al.* 2004). Tropical species may be especially sensitive to climatic fluctuations because their narrow thermal tolerances and elevation ranges can restrict their ability to persist in, or disperse across, alternate habitats (Bell *et al.* 2010).

## 4.4 Change in biotic interactions

Species rarely live in isolation but interact with other species that share the same or overlapping habitats. Some species have direct relationships with others, such as predators and prey, and facilitation, such as a plant and its specialised pollinator. Other species may rely on others though a chain of relationships. Therefore, a change in abundance of one species will likely have direct and indirect, predictable and unpredictable consequences and subtle and not-so-subtle cascades of effect on a range of other species. Species-specific responses to various components of climate change have the potential to cause temporal, spatial or functional shifts in the composition of species assemblages that affect species interactions. Idiosyncratic shifts in response to past warming and cooling glacial cycles can be detected in pollen cores which show a shuffling of plant communities in mid-elevation tropical forests; albeit with the expected net upward shift in warm periods and downward shifts in cool periods (Bush 2002; Bush & Hooghiemstra 2005). Even closely related species appear to respond idiosyncratically to climate change (Moritz *et al.* 2008). These idiosyncratic changes make it more likely that species interactions will change. Untangling the impact of changing biotic interactions on biodiversity is extremely complex and requires detailed knowledge about species dependencies, some of which may be very subtle.

There are numerous ecosystem services that rely on, sometimes tightly coupled, species interactions. Seed dispersal, pollination and nitrogen fixation are key processes facilitating ecosystem function that rely on mutually beneficial (mutualistic) ecological interactions. These interactions can be disrupted in at least two ways; through temporal (phenological, Section 4.2) and spatial (distributional, Section 4.6) mismatches that may change the availability of mutualistic partners.

Increases in abundance of some species that are major competitors, predators, pathogens or invasives are likely to have major implications for species interactions. For example, Low (2009b) outlined a scenario whereby noisy miners (*Manorina melanocephala*) which already occur in high abundance in urban and peri-urban areas, could benefit from climate change due to an increase in suitable habitat as tree deaths increase in fragmented forests. Noisy miners defend nectar sources from migratory honeyeaters (Kath *et al.* 2009) which in turn reduces cross-pollination of eucalypts which may reduce their capacity to respond to climate change through hybridisation (Low 2009a; Low 2009b).

Tropical communities provide some of the best examples of close co-evolved plant-pollinator relationships and in absolute terms contain a higher number of plants with specialised pollination systems than in temperate systems (Ollerton & Cranmer 2002). In fact, for virtually all taxonomic groups, species richness has been found to increase towards the tropics and the role of biotic interactions has often been proposed as a mechanism responsible for the origin and maintenance of the higher species diversity (Schemske *et al.* 2009). In support of this hypothesis, biotic interactions have consistently been found to be more important in the tropics (Schemske *et al.* 2009). This suggests climate may not limit current species ranges in the tropics in the same way it might in more temperate areas or on the higher latitude aspect of a species range. It also suggests that tropical species may not be influenced as much by changing climates directly, as by changing interactions with other species as an indirect result of climate change. Thus, predicting changes in ecosystem functions that rely on species interactions as a result of climate change in the tropics is especially complex.

## 4.5 Interactions with other stressors – threat syndromes

The change in abundance or relative importance of particular stressors, such as fire, introduced plants and animals and diseases, has particular impacts on natural ecosystems. These provide a special case of threat interaction and will be discussed in this section. Changes to climate are likely to interact with disturbance regimes, land use change and other agents of change, exacerbating the impacts on biodiversity and becoming an additional problem for conservation management (Laurance *et al.* 2011). Burgman *et al.* (2007) have termed these interacting processes ‘threat syndromes’. From a management perspective, it is likely that the major threats to biodiversity conservation in Queensland will come from such threat syndromes, rather than from the operation of any one agent of change. For example, Laurance *et al.* (2011) suggested that, while climatic change is likely to play a key role in predisposing Australian ecosystems to tipping points, synergisms among different environmental drivers of change can be extremely important in pushing a system towards a tipping point, resulting in major environmental change.

### 4.5.1 Fire regimes

Fire is likely to interact with particular elements of climate change, for example, drought and with other stressors, such as invasive species (Williams *et al.* 2009). In the savannas and the rainforests, an example may be the interaction between cyclones and fires, given that substantial tree fall (e.g., Cook & Goyens 2008) may be followed by fire. In the savannas, the current threats posed by invasive grasses are likely to add to any threats to biodiversity posed by climate change and the threats of invasive grasses in the future are unlikely to abate. The importance of interactions between drought and exotic grass species in determining the future fire regimes in the brigalow was highlighted in Section 6.5.

### 4.5.2 Invasive alien species

As with native species distributions, weed species and feral animals will also respond to climate change. As a general rule, suitable habitat for most tropical and sub-tropical weed species will shift polewards (Kriticos *et al.* 2003; Kriticos *et al.* 2005; Murphy *et al.* 2009; Scott *et al.* 2008; van Klinken *et al.* 2009). For weed species currently restricted to tropical north-east climates this may mean an expanded distribution into southern Queensland, for others it likely means a contraction of the climatically suitable distribution in Queensland, as in the case of candyleaf – *Stevia ovata* (Murphy *et al.* 2009), with expansion into southern Australian states. Scott *et al.* (2008) modelled the potential future distribution of 41 sleeper and alert weed species which shows that under climate change scenarios for 2070, sleeper and alert weed species from the far north of Australia can be expected to be displaced southwards by over 1000km. For example, under the current climate, only south-east Queensland contains suitable habitat for the holly-leaved senecio (*Senecio glastifolius*), whereas under future scenarios, suitable climatic space for this species contracts southward and the species may cease to exist in Queensland. For some species, large areas of Queensland still remain climatically suitable but potential distributions contract towards the east, as in the case of praxelis (*Praxelis clematidea*).

Shifting of weed species distributions to the south, while sounding promising for future weed expansion, actually opens up a serious issue for far north Queensland. A polewards shift of suitable habitat for existing sleeper and alert species as well as species already considered invasive in Queensland, creates the opportunity for new invasive threats from as yet un-identified species. Part of the challenge of adapting to climate change in both environmental and agricultural sectors will be managing newly emerging weeds. Under changed climatic conditions, emerging invasive plant species may come from: (a) species already present in Australia in low abundance or which have restricted distributions for which climatic conditions become more favourable; (b) species native to, or known to be naturalised in, adjacent countries or countries that have similar climatic conditions to the predicted future climates of Queensland but which are not yet present in Australia; or (c) native Australian species that undergo significant changes in range or abundance in response to climate change or other anthropogenic stressors (invasive natives).

### 4.5.3 Pathogens and disease

Low (2011) provides an example of the potential flow-on effects of changes in abundance of the plant root pathogen *Phytophthora cinnamomi*, a soil-borne water mould that is expected to benefit from higher temperatures and increases in rainfall seasonality. *Phytophthora cinnamomi* typically occurs in higher-altitude areas that support many endemic species. Rainforest dieback associated with Phytophthora has been recorded from several Wet Tropics locations, including Mt Lewis, Mt Bartle Frere, Mt Windsor and Mt Spec and also from the Clarke Range near Mackay (Gadek 1999). Infection exacerbates the impacts of rising temperatures as it causes loss of the shady rainforest canopy which then compounds the threat of climate change for animals by limiting shade and exposing them to higher temperatures and increased desiccation.

There is some evidence to suggest that climate change in Queensland may provide conditions that facilitate invasion of the chytrid fungus (*Batrachochytrium dendrobatidis*) (Alford *et al.* 2007; Laurance 2008), which is frequently cited as a potential casual agent in sudden declines and regional extinctions of amphibians in tropical areas (Hero & Gillespie 1997; McDonald &

Alford 1999). Recent research suggests that chytrid-related extinctions of South American frogs of the *Atelopus* genus occurred immediately after unusually warm years, which provided favourable growth conditions for the fungus (Pounds *et al.* 2006). Researchers studying frog declines in eastern Australia have suggested a modified-version of the warm-preceding-year hypothesis with research here indicating that a multi-year warming effect is more likely (Alford *et al.* 2007). Laurence (2008) analysed temperature data for eastern Australia where at least 14 upland-rainforest frog species have also experienced chytrid-related declines or extinctions and also found evidence consistent with a multi-year warming effect. In particular, Laurance (2008) found upland frog declines coincided with rising annual minimum temperatures in tropical latitudes. This contrasts with other researchers in South America who have not found evidence for climate change driven outbreaks of the fungus (Lips *et al.* 2008). Harvell *et al.* (2002) suggests this pathogen may be one of a few for which climate warming could disrupt disease spread, because it requires cool, moist, high-altitude conditions.

Recent research in the Wet Tropics suggests that climate warming may increase the prevalence of mosquito borne avian diseases at increasingly higher elevations (Hilbert 2010). Hilbert (2010) undertook an extensive survey of rainforest mosquitoes in the Wet Tropics and found that both abundance and diversity of mosquitoes increased with increasing mean annual temperature. Furthermore, they found infection rates of avian malaria (*Plasmodium relictum*) were highest in warmer sites, declining to an average of approximately 3% at mean annual temperatures below about 22.5°C. In Hawaii, there is strong evidence to suggest that avian pox and avian malaria have had a major impact on forest bird populations. Avian diseases are often cited as the major factor in the demise of the Hawaiian endemic forest bird fauna (Atkinson & LaPointe 2009). High-elevation refugia from mosquitoes are considered to be the only reason some susceptible bird species have not become completely extinct in Hawaii. Climate change modelling predicts, however, that these refugia may be significantly reduced in future and remaining populations of threatened and endangered forest birds will be pushed to extinction (Atkinson & LaPointe 2009). Hawaiian researchers have already recorded a more than doubling of the prevalence of malaria in forest birds at elevations of 1900 m in the last decade (Atkinson 2010; Freed *et al.* 2005) .

Changing climates may also stress organisms, weakening body condition, immune response and adaptive capacity, through habitat loss, extreme weather events or prolonged periods of resource scarcity (Epstein 2001). These conditions can then permit the expression of disease in hosts that may otherwise have been resistant. For example, nutritional stress enhances the risk of Hendra virus infection in little red flying foxes (Plowright *et al.* 2008).

## 4.6 Macro-scale distribution shifts

As defined in Section 4.1, shifting species ranges are the complex result of extinction of metapopulations at the ‘trailing’ geographic range edge (i.e., usually at the low latitude or low elevation limit) and colonisation and growth of metapopulations at the ‘leading’ edge (i.e., the high latitude or high elevation limit). These extinctions and colonisations result from the individual (biological) responses to changing climate and the interaction with other species. We would expect these changes to significantly modify current species distributions either through expansion into new areas or contraction of current range boundaries. Macro-scale species distribution changes are considered the most likely response to relatively rapid climate change

(Huntley *et al.* 2010). For example, using model simulations and some simplifying assumptions, Iverson *et al.* (2004) demonstrated that ‘source’ strength or the abundance of species at the expanding edge of their range could be the most important parameter driving migration rates for North American trees; more important even than the amount of suitable habitat available to occupy beyond the current range boundary. Organisms, whose abundance tapers towards the leading edge of their range boundary, may therefore have limited capacity to track climate change (Murphy *et al.* 2010b).

Species distribution modelling for a wide range of Queensland fauna shows a southerly contraction, with the distributions of some species moving out of Queensland entirely with increasing magnitude of climate change. For example, a large part of the current geographic range of the Koala (*Phascolarctos cinereus*) in western arid and semi-arid regions in Queensland is expected to be no longer suitable by 2070 (Adams-Hosking *et al.* 2011). Greater gliders (*Petaurus volans*) are expected to undergo dramatic range contraction from northern Queensland with 3°C warming (Kearney *et al.* 2010c). Richie and Bolitho (2008) used bioclimatic models to show an average reduction in macropodid distributions in northern Australia of 48% in response to 2°C of warming. Richie and Boltiho’s (2008) work predicted severe range reductions for all macropodids (average 96%) with 6°C of warming and the possible extinction of the antilopine wallaroo (*Macropus antilopinus*). They note that the extinction of the antilopine wallaroo would constitute a significant cultural and economic loss to indigenous people of northern Australia. Importantly, the eastern Cape York area of Queensland is predicted to be the only remaining large contiguous area of suitable climatic space for this species with 2°C of warming.

Recognition of the potentially enormous impacts of climate change on the unique biodiversity of the Wet Tropics has led to a large amount of research on potential changes in species distributions there over the last 10 years. A consistent finding is that the geographic distributions of Wet Tropics endemic faunal species will decline (Hilbert *et al.* 2004; Meynecke 2004; Williams *et al.* 2003). For example, Williams *et al.* (2003) predicted that the ranges of all 65 regionally endemic vertebrate species will decline dramatically with a temperature increase of 3.5°C. An increase of 7°C would result in loss of the entire area of core habitat for these species indicating extreme vulnerability to extinction. Bioclimatic modelling for the very restricted microhylid frog *Cophixalus neglectus*, which is limited to altitudes between 1300 and 1600 m, shows a complete loss of its current range with just 1°C of warming (Meynecke 2004).

The rainforests of the Wet Tropics will be highly sensitive to climate change within the range likely to be experienced over the next 50-100 years (Hilbert *et al.* 2001). Hilbert *et al.* (2001) demonstrated the potential for large changes in the distribution of some rainforest types with even minor climate change. In particular, highland rainforest types (simple nototphyll and simple microphyll forests) are predicted to decrease by 50% with only a 1°C warming. However, the potential for lowland biotic attrition has also been highlighted for the tropics. This is based on the fact that, whereas in temperate regions, poleward or upslope range shifts may be compensated for by migration from species more adapted to warmer temperatures, no community of species now living in hotter places is available to replace lowland tropical species (Colwell *et al.* 2008).

Bioclimatic modelling indicates that under current climate change projections, a large proportion of eucalypt species will have their entire distributions subjected to average temperature and rainfall combinations that no individuals are currently exposed to (Hughes *et al.* 1996). For example, for 53% of eucalypt species, the populations at the present southern boundary are expected by 2070 to be exposed to warmer temperatures than any currently tolerated by their most northerly populations. Despite this, Low (2011) outlines a body of research that suggests eucalypt species' ranges are not strongly limited by climate and presents evidence that pollination, hybridisation and the mallee growth form sometimes facilitated survival under past climate change. Inter-specific hybridisation is an evolutionary strategy that has conferred a large amount of adaptability to eucalypts and other species over geological time (Byrne *et al.* 2008; McKinnon *et al.* 2004). In response to fluctuations in climates over the last few tens of thousands of years, eucalypt species' ranges have contracted and expanded (e.g., Dodson 1989). Hybridisation between species is quite common within eucalypt complexes where flowering is synchronous and geographic range overlap occurs but less common among related species that maintain breeding separation through asynchronous flowering. Climate change could differentially affect flowering time patterns so that interbreeding among compatible species becomes feasible. Molecular studies have also demonstrated significant differentiation between populations within species that may have occurred without hybridisation (Butcher *et al.* 2009; Sale *et al.* 1996). These evolutionary adaptations, at the 'normal' pace of climate change, are well demonstrated for eucalypts. However, climate projections point to significant change within just one or two eucalypt breeding cycles. Low (2011) concludes that while some species will show considerable resilience to climate change, large population declines may be expected for species vulnerable to drought. In particular, eucalypts of eastern Queensland could show declines in range sizes if drought events become more frequent or severe. Droughts are more prevalent in drier bioregions where species are more likely to have adaptations, such as deeper tap root systems and water efficient physiology.

Importantly, bioclimatic modelling indicates that remaining habitat for a large number of species becomes much more fragmented as range contractions occur (Ritchie & Bolitho 2008). Habitat patches shrink with likely subsequent reductions in population size and increased isolation of populations; threats in themselves to population persistence (Beaumont & Hughes 2002; Hilbert *et al.* 2004). This effect can also be visualised for ecosystems using models of the compositional turnover in individual species distributions (Ferrier *et al.* 2002; Ferrier *et al.* 2007). Ferrier *et al.* (2012) applied the outputs of a model for vascular plants, developed using over 12,000 species observations from more than 100,000 sites for continental Australia (Williams *et al.* 2010a; Williams *et al.* 2010b), to show how the pressures of environmental change in Queensland can be scaled in terms of potential compositional change. This process is termed "ecological scaling of environmental change" or "biotically-scaled environments" and represents the relative amounts of change that could occur. The actual change in biological composition resulting from climate change is much more complex than present models are able to depict. Likely factors shaping the actual outcome for any location include biotic interactions, indirect effects of changed fire regimes, dispersal ability, lag effects, adaptation capacity and plasticity; major topics of discussion in this report.

The model of vascular plants has been used to scale environmental change expected for Queensland in 2030 and 2070 under two climate scenarios (A1FI and A1B) for two levels of sensitivity to emission drivers of climate change (high and medium). Selected examples of the 6 combinations of output are presented in Ferrier *et al.* (2012). The outputs depict locations (1km grid-cells in this analysis) with varying potential for change in the species composition of the plant community currently occurring at that location. The key messages from this work are that the potential for compositional change is not evenly distributed across the State and the overall potential for change is high, particularly by 2070. Some regions and environments exhibit greater potential for change than others. These patterns arise from a combination of differences in the amount of change predicted for climate itself (such as temperature, precipitation and evaporation) across different parts of the State (from the global climate models) and of differences in the amount of change in species composition expected for a given change in climate across different environments.

The outputs of bioclimatic species distribution models and models of species compositional turnover, discussed above, are subject to a range of scientific caveats associated with limitations in the ecological assumptions used in developing the models, the data types and their quality and the processing routines used to parameterise the models (Allouche *et al.* 2006; Austin & Van Niel 2011; Elith *et al.* 2010; Elith & Leathwick 2009; Heikkinen *et al.* 2006; Liu *et al.* 2009; McPherson *et al.* 2006; Newbold 2010; Sinclair *et al.* 2010; Webber *et al.* 2011b; Yates *et al.* 2010). Although the level of uncertainty and assumptions associated with distribution models sometimes limits their applicability to management decision-making (Sinclair *et al.* 2010), they continue to be useful in assessing potential trends and risks associated with alternative global-change and policy-response scenarios, thereby informing high-level policy development (Alkemade *et al.* 2009; Leadley *et al.* 2010; sCBD 2010). However, they need to be used in combination with ecological knowledge to provide examples and interpretation of the results to be useful for decision makers and in land management and planning.

How these macro-scale shifts might relate to the adaptive capacity of species and ecosystems is discussed in more detail in Section 5.3.

## 4.7 Changes in response to disturbance regimes and extreme events

Occasional or rare events might have a greater effect on long-term community structure than average environmental conditions; for example, empirical evidence shows that large-scale and infrequent disturbances, such as cyclones or hurricanes, may dominate tropical forest dynamics and shape forest diversity, composition and structure (Burslem & Whitmore 1999; Tanner & Bellingham 2006). They may also have major impacts on fauna through destruction of habitat, reduction of food resources, disruption of breeding and through direct mortality. For example, 100 percent of the habitat occupied by the critically endangered Mahogany Glider was severely impacted by Category 5 Tropical Cyclone Yasi in February 2011 (WTMA 2011). Gifford and Howden (2001) suggest that natural systems in Australia are always in a state of recovery since the last big disturbance event. Some examples of rare events and disturbances, with the potential to dramatically alter natural landscapes, are provided in this section.

### 4.7.1 Severe storms and cyclones

An increase in intensity of cyclones may have major implications for biodiversity (Figure 9). In particular, interactions between intense cyclones and other threats will determine the extent of impact on tropical biodiversity. However, some research conducted post-Cyclone Larry (March 2006) suggests that fauna in the tropics are relatively resilient to severe cyclones. For example, Kanowski *et al.* (2008) found no significant change in abundance for arboreal folivorous marsupials endemic to upland rainforest in the Wet Tropics post-cyclone. In addition, arboreal folivores did not significantly increase their home range area or show a significant change in body condition compared with pre-cyclone measurement (Wilson *et al.* 2008b). Cyclone Larry appeared to cause short-term changes in the abundance of some bird species in rainforest fragments, however, after only 7 months post-cyclone, bird communities recovered in abundance to approximately their pre-cyclone state (Freeman *et al.* 2008). However, more subtle and long-term effects may impact faunal communities post cyclone. The impact of cyclones on breeding behaviour and breeding success may have more significant longer-term impacts. For example, the tooth-billed bower-bird breeding season was significantly affected by the cyclone, with a later season and fewer courts active than in the previous two seasons (Freeman *et al.* 2008). Furthermore, Wilson *et al.* (2008b) note that if climate change results in drier periods following cyclones, greater impacts on wildlife are more likely.

Figure 9: Mt Mackay at Tully following Category 5 Tropical Cyclone Yasi (February 2011) (credit: Dan Metcalfe, CSIRO).

Impacts of cyclones on plant community composition and structure manifest in several ways. The extremity of structural damage to the rainforest following a cyclone strongly influences community composition as well as richness and evenness of successional communities (Murphy *et al.* submitted). The enormous levels of litter and debris also impact species recruitment and may particularly limit species with small seeds (Metcalfe & Turner 1998). Recruitment monitoring of native species following Cyclone Larry has demonstrated a rapid divergence in community composition between sites in the most and least severely damaged areas resulting in the development of persistent statistically distinct community states across relatively small scales. Murphy *et al.* (submitted) suggest this effect may be a major driver of the higher species diversity observed at larger scales in cyclone impacted forest compared with intact forest. An increasing intensity of cyclones, however, may lead to reductions in species diversity as shade-tolerant (and typically slower growing) species experience increasingly rare windows of opportunity to persist long enough to reproduce as well as fewer opportunities to establish.

The widespread and massive disturbances caused by cyclones provide ideal conditions for rapid recruitment and spread of invasive species. Invasive species can alter the successional trajectory of forest recovering from large-scale disturbance by reducing colonization success of native species. Monitoring of invasive species establishment post-Cyclone Larry indicates that while many herbaceous weed species are relatively transient in tropical forests following cyclones, woody weeds grow more quickly (Murphy *et al.* 2008a), show low mortality and persist over much longer time frames (Murphy *et al.* 2010a; Murphy *et al.* 2008b). Importantly, given climate change predictions of more intense cyclones, persistent woody trees, vines and shrubs, may form the vanguard of the next wave of invasion should an area experience further severe cyclones; as has recently occurred in North Queensland with Cyclone Larry (severe Category 4, 2006) and Cyclone Yasi (Category 5, 2011) impacting a similar area (e.g., see Figure 10).

Figure 10: Damage to littoral rainforest with melaleuca north of Cardwell following Category 5 Tropical Cyclone Yasi (February 2011) (credit: Dan Metcalfe, CSIRO Ecosystem Sciences).

The littoral rainforest and coastal vine thickets of the eastern coast of Queensland are listed as critically endangered under the Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999* (Threatened Species Scientific Committee 2007). The listing advice notes that while regular exposure to storm events contributes to the unique structure of these communities, an increasing intensity of cyclones as well as associated events, such as storm surge, is considered to constitute a major threat to their persistence (see Figure 10). Foreshore vegetation and natural dune structures provide protection to coastal communities, beaches and infrastructure, such as roads, marinas and agriculture and aquaculture industries, during storm events and storm surge. Riparian and littoral vegetation are special cases of native vegetation that occur at complex interfaces between terrestrial and aquatic systems, where they protect areas from erosion, filter sediments, nutrients and pollutants, mitigate the effects of flooding and storm events and provide supporting habitats for aquatic biodiversity. Severe storms and cyclones result in major structural changes to foreshore and riparian vegetation, sometimes completely removing their ability to perform these ecosystem service functions. For more details of these ecosystem service functions, see Williams *et al.* (2012b).

### 4.7.2 Flooding

An increase in severe flooding events has the potential to cause significant shifts in vegetation across large areas of Queensland. In the Gulf Plains region, sustained rainfall can produce long-lasting floods as occurred in 2009 where rainfall associated with cyclones in January and February inundated 5.8 million hectares of land (Gobius 2010; in Low 2011). Due to its low relief and limited drainage, water remained in some catchments over large areas for more than six weeks (Low 2011). Widespread pasture death was recorded particularly in low-lying areas, while upper catchments experienced severe erosion from water flow. An analysis of vegetation changes following the flooding indicates a shift towards dominance of pastures by sedges and annuals, with perennial grasses recovering much more slowly (Low 2011). An increase in severe cyclones and frequency and intensity of rainfall events has the potential to cause permanent vegetation shifts and seriously impact productivity of the region.

Of note is that flooding may significantly impact termites. Gobius (2010; in Low 2011) noted the striking loss of mound building termites following the 2009 Gulf region flooding. Recent research has shown that ants and termites provide valuable ecosystem services in arid climates by increasing soil water infiltration and improved soil nitrogen (Evans *et al.* 2011). In Western Australia they have been shown to increase crop yields by 36%. Ants and termites are also well known to move tonnes of soil per hectare, increase aeration and influence availability of several nutrients (Evans *et al.* 2011).

### 4.7.3 Drought

Severe, extended periods of drought have the potential to kill large stands of trees and reduce the health of vegetation and animals. Drought induced die-back in north Queensland has been recorded since the early explorers (Fensham 1997; Fensham *et al.* 2009) and may affect some species more than others, modifying plant composition and landscape function. Drought often results in ecosystem change when rainfall returns by allowing new species to dominate the recovering system, especially introduced or naturalised weeds and feral animals. Any change in the frequency or severity of droughts in Queensland has the potential to affect natural ecosystems. This is especially true if the current biota is adapted to tolerate the present regime of droughts but not adapted to an increase frequency, intensity or duration of droughts.

### 4.7.4 Fire

Fire is a special case as it interacts with other events, such as floods and cyclones that produce copious amounts of fuel and subsequent droughts that provide the dry conditions needed to cure fuels and allow rapid spread of flames. Therefore, predicting the changes in fire regime requires an understanding of both fire weather under climate change and other disturbances leading to altered fuel dynamics (for details, see Sections 2.6 and 6.5).

# 5. ADAPTIVE CAPACITY OF SPECIES AND ECOSYSTEMS

## 5.1 Introduction

All organisms are expected to have some intrinsic capacity to adapt to climate change through ecological (i.e., physiological and/or behavioural plasticity) or evolutionary adaptations (i.e., through natural selection) (Williams *et al.* 2008). Adaptive capacity, which is often used to refer to the set of preconditions that enables species and systems to respond to climate change, is a synonym for many characteristics of resilience. To be resilient, species, communities and systems must generally have the ability to buffer disturbance, reorganise and renew after disturbance and to learn and adapt (Carpenter *et al.* 2001).

Some types of species or ecosystems are predicted to have reduced capacity to adapt to climate change because of their particular life-history traits, physiology, dispersal ability, geographic range location and/or distribution and degree of habitat specialisation. For example, high reproductive rates, fast life history and short life spans are traits that are predicted to promote resilience and reduce extinction risk following change (McKinney 1997). As noted earlier, species with large geographic ranges also tend to have reduced extinction risk and may exhibit higher resilience, while species with ranges located in tropical regions tend to have smaller distributions and higher degrees of habitat specialisation and often are described as having lower resilience to changing conditions (Thuiller *et al.* 2008). The ability to disperse over long distances is also expected to increase a species' resilience in the face of climate change, as species with high dispersal capacity may more readily track rapidly shifting suitable climate space.

Two prerequisites must exist for adaptive responses and resilience to be successful in countering the rapid predicted rate of climate change. These are landscape and biogeographic connectivity to allow organisms to reach suitable habitat/climate space/refugia and adequate time to allow adaptive changes (Williams *et al.* 2008). Human activities have already reduced the capacity of many species and ecosystems to cope with disturbance and change, in particular through modification of the natural landscape in which species evolved (Elmqvist *et al.* 2003).

In a previous section we discussed how adaptive change in individual biology may buffer some species against climate change and increase resilience (Section 4.2). In this section we discuss resilience and the role of diversity at the community or ecosystem level, describe how landscape pattern influences a species' capacity to respond and adapt to climate change and highlight the role of refugia in enhancing species persistence.

## 5.2 Resilience

Resilience is usually defined as the capacity of an ecosystem undergoing disturbance to withstand the pressure or reorganise and renew itself without the loss of ecosystem function (Carpenter *et al.* 2001; Elmqvist *et al.* 2003). As a general rule, ecosystems are most resilient to disturbance where diversity, including both number and composition of the genotypes, species, functional types and landscape units in a given system is maintained (Diaz & Cabido 2001; Fischer *et al.* 2006). Diversity allows ecosystem function to be maintained under environmental flux (Traill *et al.* 2010). The diversity of responses to climate change among species contributing to ecosystem function is critical to resilience (Elmqvist *et al.* 2003). Sustaining ecosystem function in the face of climate change requires that functional groups of species remain available for renewal and reorganisation (Elmqvist *et al.* 2003). An example of an ecosystem service that currently has low redundancy and little insurance in the face of climate change is the reliance of global agricultural systems on the domesticated honeybee (*Apis millifera*) to provide complex and variable pollination services. This honeybee is currently faced with a number of disease and parasite threats (Diaz *et al.* 2005). The maintenance of a matrix of healthy natural ecosystems, interspersed and adjacent to human settlements and agricultural fields, can provide significant insurance that pollination services remain intact through provision of a large and diverse suite of pollinators which assure continued and reliable delivery of this vital service.

Most plant communities comprise a few abundant species and many low-abundance, minor species (Whittaker 1965). The presence of multiple species, whether abundant or rare, within each functional type increases functional redundancy and may have important implications for ecosystem stability (Díaz *et al.* 2005; Hooper *et al.* 2005). Functional redundancy occurs when several species in a community carry out the same process or function, such as nitrogen fixation. It is generally considered that the abundant species in a community largely determine the rate of ecosystem processes (Aarssen 1997; Walker *et al.* 1999). However, the results of experimental work indicate that the role of minor or rare species in communities may be more important than their relative abundance suggests, particularly over longer timeframes in the face of changing climates (Lyons & Schwartz 2001; Smith & Knapp 2003; Zavaleta & Hulvey 2004). Walker *et al.* (1999) suggested that while dominant species in a community are responsible for much of the ecosystem function, minor species maintain resilience in the community over the long term. This is based on the theory that, while dominant and minor species may be functionally similar and appear redundant, minor species have different responses and tolerances to environmental conditions that are not currently present and they maintain resilience in the community by allowing the maintenance of function under changing conditions. The loss of specialist species may entail lower rates of ecosystem processes and some functions performed by specialists may not be carried out at all – for example, the decomposition of particular substrates or the pollination of certain species.

## 5.3 Landscape patterns

Biodiversity in Queensland is embedded in multi-use landscapes with a variety of ecological, social and economic values. Landscape pattern and form influences many aspects of species population dynamics. In particular, landscape pattern has the potential to significantly affect a species' capacity to respond to climate change, both directly and indirectly. While, at larger scales, species are expected to expand or shift their ranges towards higher latitudes or higher elevations in response to climate change, in reality, fragmented landscapes act as barriers to range shifts for most species. Many natural or semi-natural ecosystems have become fragmented and embedded in a landscape matrix with low permeability to dispersing individuals (Murphy & Lovett-Doust 2004; Vos *et al.* 2008). Climate change may also directly impact landscapes particularly if humans adapt the use of land. For example, climate change will lead to production increases in some places and to abandonment of production in other places with consequent changes in land-use, landscape composition and structure. In a global analysis, land-use change in itself was predicted to have the largest global impact on biodiversity by 2100, while climate change was predicted to be the second most important driver of biodiversity change (Sala *et al.* 2000). The strength of interaction between the two drivers is virtually unknown. More recent scenarios of biodiversity change for the 21st Century (Leadley *et al.* 2010) suggests thresholds, amplifying feedbacks and time-lag effects leading to "tipping points" will be widespread and make the impacts of global change on biodiversity hard to predict, difficult to control once they begin and slow and expensive to reverse once they have occurred.

One way that landscape pattern influences a species response to climate change is through the availability and connectivity of suitable habitat for range expansion or shift. Species may experience crucial bottlenecks in their geographic range, where low connectivity or too few suitable habitat patches inhibit spatial responses to climate change. Changes in climate will potentially increase the amount of suitable habitat in some parts of a species' geographic range in addition to new suitable habitat in places beyond its present range. For example, species that presently occupy relatively connected lowland habitats and hills may need to expand their range into higher altitudes of mountain ranges becoming fragmented and isolated by relatively inhabitable valleys as the climate warms. At a biogeographical scale, a species' range can be conceptualized as a patchwork of regions varying in the amount and density of suitable habitat and spatial cohesion with different capacities to function as an effective source of colonizers for newly available suitable habitat. The colonization of newly suitable habitat requires that distances between patches or between networks of patches can be crossed with enough probability of success to enable establishment of a new viable population. Colonisation success can be thought of as being influenced primarily by three species-specific landscape factors; distances, matrix permeability and physical barriers; which all influence connectivity of the landscape. However, connectivity is not just a function of colonisation capacity. It also incorporates habitat area, quality and arrangement and is thus a complex and dynamic variable. Despite this complexity, increasing connectivity has emerged as an adaptation strategy for conserving biodiversity in the face of climate change (Hodgson *et al.* 2009). Not surprisingly, for species with different life-history characteristics, colonization capacities and habitat requirements, we have as yet no systematic insight into the amount of landscape connectivity required to allow persistence; although new landscape modelling tools allow these facets of species ecology to be parameterised as knowledge accrues (Drielsma & Ferrier 2009; Ferrier & Drielsma 2010). Defining connectivity for multiple species simultaneously adds another layer of complexity.

Regardless of the uncertainties in measuring the degree of connectivity in a landscape, it is widely considered that increasing habitat area is an effective way of increasing connectivity (Hodgson *et al.* 2009). More area generally means more individuals, more resources, more heterogeneity and smaller distances between suitable habitats where new populations can establish. Increasing quality of habitat also increases carrying capacity and generally results in increases in abundance. There are many examples in Queensland of reductions in habitat area or quality leading to reductions in population size and/or density for a range of organisms. For example, in south-east Queensland squirrel gliders (*Petaurus norfolcensis*) occur in a range of habitat patch sizes from 10-20ha through to 1000ha but their density is significantly lower in smaller patches. Their habitat in south-east Queensland is being modified and cleared at a relatively rapid rate, reducing habitat linkages and increasing isolation of smaller habitat patches (Rowston *et al.* 2002). Many studies using patch occupancy models to investigate the impacts of different amounts and patterns of habitat loss on species persistence, have shown that a threshold of habitat availability exists above which a species is able to maintain a high degree of patch occupancy and below which the species rapidly approaches extinction (Travis 2003). The current level of fragmentation in south-east Queensland appears to be above the threshold beyond which the squirrel glider can persist, however, its low capacity for dispersal and its high habitat specialisation along with the pattern of its occupancy in the landscape suggest there will be a threshold below which its population will decline dramatically (Rowston *et al.* 2002). Also using patch occupancy models, Travis (2003) showed a similar threshold for climate change. Below a critical rate of climate change, a species maintains high patch occupancy throughout the period of climate change but, above the critical rate, the species is unable to keep pace and patch occupancy declines rapidly. Importantly, Travis (2003) found that during climate change the habitat threshold occurs sooner and that, similarly, species suffer more from climate change in a fragmented habitat.

Although it is difficult to document examples of this phenomenon empirically, some inference can be drawn from detailed studies of the determinants of abundance of some species. The grey mistletoe (*Amyema quandang*) is a widespread semi-arid species which provides an almost year-round supply of food for nectarivores and frugivores in acacia woodlands where few other sources of fruit and nectar are available. In the Brigalow Belt region of Queensland, it increases in abundance in areas with low connectivity, particularly along narrow linear strips of remnant vegetation. Grey mistletoe thus may be of critical importance for the maintenance of biodiversity in parts of the highly fragmented brigalow landscapes by increasing landscape connectivity for animals dependent on nectar and fruits (Bowen *et al.* 2009). However, grey mistletoe abundance may be susceptible to extended drought periods. An extended drought that significantly reduces mistletoe abundance has the potential to trip a threshold for the many dependant animals that rely on it to increase functional connectivity in these highly fragmented landscapes.

Habitat fragmentation, like climate change, may benefit some native species that occupy the edge between disturbed and intact vegetation. However, an advantage to one native species from habitat fragmentation may have indirect and more pervasive effects on plant and animal communities if the dynamics of inter-specific interactions are altered. At some scales, the effects of climate change and habitat fragmentation may be opposing, with the positive effects of range expansion due to climate change mitigating the negative effects of habitat fragmentation. However, across entire geographic ranges, these phenomena are likely to act in the same direction, such that their combined negative effects can drive a species to extinction (Travis 2003).

Spatial heterogeneity; that is, the spatial patchiness and variability in landscape patterns; is an important component of landscapes which may serve to buffer species against climate change. For example, heterogeneity in habitat quality may dampen the effects of local disturbances. A modelling study of the effect of spatially correlated fires on a metapopulation of Leadbeater’s possums (*Gymnobelideus leadbeateri*) in a fragmented woodland landscape confirmed that the effect of large scale disturbances is mitigated by spatial heterogeneity of habitat patches (McCarthy & Lindenmayer 2000). Heterogeneity provides opportunities for populations to survive different extremes by shifting between different types of vegetation, soils, aspects or elevations (Hodgson *et al.* 2009). The role of heterogeneity as a form of refugia is discussed in more detail in Section 5.4.

## 5.4 Refugia from climate change

### 5.4.1 Introduction

Refugial features in the landscape have been important to the persistence of native species under past natural climate change associated with glaciation/deglaciation periods (Carnaval & Moritz 2008; Mackey *et al.* 2008). Refugia are highly likely to be of continuing importance for the maintenance of biodiversity in the face of current and future anthropogenic climate change. Landscape refuges are areas of the landscape that provide protection for plants and/or animals from unsuitable or threatening conditions or events and allow them to persist. The application of the term refugia is used to represent a wide range of situations (Ashcroft 2010) from large interglacial refugia, through to micro-refugia for a particular population, to individual requirements such as tree-hollows, nests and rocky outcrops. Larger patches of refugial habitats are often recognised for their high levels or unusual types of biodiversity and have been protected as national parks and reserves (DERM 2010a; DERM 2010b; Taylor & Figgis 2007). Relatively intact natural habitat, such as riparian zones, windrows, reserves, National Parks and state forests can all be thought of as different scales of environmental refuges from often highly modified contemporary landscapes.

Low (2011) notes that the survival of some species will increasingly depend on them accessing microhabitats that are unusually cool, wet, humid or protected from fire. Such locations could include the largest rock piles and logs, caves, large hollow trees, gorges and gullies, the deepest accumulations of litter and the shaded southern sides of steep hills. Temperatures can be relatively stable in places that are decoupled from ambient temperatures, such as gorges that pond cool air (Dobrowski 2011), caves, lower water layers in deep pools and deep soil layers. The distribution of these places in the landscape is not well documented for many regions, although the use of them for thermoregulation by many species has been often noted (see Section 4.2).

There are many other ways to view refugia in the landscape. Isolated populations in suitable microclimates near the boundary of a species geographic range may be considered refugia. In most places migration and range expansion at the leading (warming) edge of a species range in response to climate change will not be apparent as a moving front; isolated, low-density populations will initially establish followed by increases in abundance and occupancy (Murphy *et al.* 2010b). An analogy is apparent in recent work which highlighted the importance of low-density and isolated peripheral populations, existing in ‘cryptic refugia’, to range expansions following the last glacial maxima in North America and Europe (Dobrowski 2011; McLachlan *et al.* 2005). These ‘cryptic refugia’ are likely to be very important in enhancing the capacity of species to persist, adapt and migrate in response to climate change and should be a high priority for identification and protection to secure intrinsic adaptive capacity.

Refugia might not even be tangible and identifiable as ‘places’ in the landscape. For example, the particular way that a landscape is managed might actually create the conditions suitable for persistence of a species though change. Climatic gradients, landscape heterogeneity and spatial patterning in the landscape all generate refugia in a number of contexts (Keppel *et al.* 2012). Results of ecological modelling suggest that heterogeneous landscapes, where species are generally patchily distributed, are more likely to experience different micro-environmental conditions to the same climatic perturbations and therefore their population dynamics will fluctuate asynchronously. As a consequence, species with sub-populations in heterogeneous landscapes are less likely to go regionally extinct (Loreau *et al.* 2003). The species diversity in heterogeneous landscapes may also contribute to resilience as described in the section above (Section 5.2). It is predicted that the relatively larger regional species pool in heterogeneous landscapes increases the probability of recruiting key species and maintaining ecosystem functioning during periods of high environmental stress or rapid change (Virah-Sawmy *et al.* 2009). Spatial heterogeneity has been demonstrated to have contributed to enhanced species survival during climatic fluctuation of the past (Virah-Sawmy *et al.* 2009).

As refugia can be a product of both climate and habitat, they are best identified for an individual species. Where multiple species' refugia coincide, "ecosystem hotspots" result (Vos *et al.* 2008). In addition, some refugia will be dynamic as climates change; refugia may move locations, some will disappear and other areas will begin to offer protection under a changing climate. Recognising that the study and discussion of refugia has often been *ad hoc* and descriptive in nature, Keppel *et al.* (2012) set out to provide a habitat-based concept of refugia and evaluate methods for their identification. They define refugia as environmental habitats with space and time dimensions that operate on evolutionary time-scales and that have facilitated the survival of biota under changing environmental conditions for millennia. Their framework aims to allow the identification and description of refugia in any environment.

### 5.4.2 Temperature

Mountainous regions of Queensland are major areas of endemism and are predicted to serve as important refuges for species under climate change (Shoo *et al.* 2011; Williams *et al.* 2003). Shoo *et al.* (2011) used a combination of weather data and spatial modelling to identify 'cool climate refugia' in the Wet Tropics. They constructed a spatial surface of maximum air temperature and took into account climate mediating processes, such as elevation, proximity to coast and foliage cover. Cool climate refugia were defined as those areas falling within the lower quartile (i.e., less then 26°C) of maximum temperature of the warmest period under a rainforest canopy. These areas encompassed 2109 km$^2$ of rainforest. The lowest maximum temperatures occur in major montane areas including the Atherton, Lamb, Carbine and Windsor Uplands, with smaller fragments occurring on Thornton, Finnegan, Malbon-Thompson, Bellenden Ker-Bartle Frere and Elliot Uplands (Shoo *et al.* 2011). The majority of this habitat (85%) is already contained within the established reserve network.

Shoo *et al.* (2011) assessed the temperature niche of 153 species of rainforest vertebrates in the Wet Tropics (26 mammals, 70 birds, 32 reptiles and 25 amphibians and including 62 endemic species), with respect to the location of cool refugia. They found that 30% of all species and 45% of endemics have core habitat contained within the coolest quarter of extant rainforest; a further 39% of endemics have mean temperatures over their range of occupied habitat that lie within 1.5°C of temperatures that delineate cool refugia. A specific example of the potential impact of climate warming on an endemic species restricted to high-altitude cool-climate refugia is demonstrated by modelling research on the golden bower-bird (*Prionodura newtoniana*) (Hilbert *et al.* 2004). With future warming, assuming a 10% decrease in rainfall, potential habitat is reduced from 1564 km$^2$ to 582 km$^2$ (1°C warming), 163 km$^2$ (2°C warming) and 37 km$^2$ (3°C warming). With 3°C of warming, golden bowerbirds could be restricted to the peaks of Bartle Frere, Bellenden Ker, Mt Fisher on the Evelyn tableland and very small patches at the top of peaks in the Lamb Range. If the upper range of predicted warming for this century occurs (5.8°C), no appropriate environments would remain in the Wet Tropics; the closest rainforests to the south are far removed across significant habitat barriers suggesting natural range migration is unlikely (Hilbert *et al.* 2004). Assisted migration may be required for the species to persist beyond the middle of this century if the present trend of warming continues.

Molecular analyses support the theory that isolated tropical montane areas harbour high levels of narrow range species because of their resilience to past climate change (Bell *et al.* 2010). Populations of tropical montane species appear to have persisted in separate isolated refugia resulting in high levels of genetic divergence during past glacial cycles. For example, the montane specialist skinks *Lampropholis robertsi* and *L. coggeri* which are endemic to rainforests of north-east Queensland, exhibit pronounced phylogeographic structuring for mitochondrial and nuclear genes, attesting to low dispersal and high persistence across multiple isolated regions during past climate cycles (Bell *et al.* 2010). So, although narrowly endemic montane species appear especially prone to extinction with climate change, they may be more resilient than expected initially. However, peak temperatures during past interglacial events were approximately 2°C warmer than today’s temperatures (Hilbert *et al.* 2007b). Warming above this level is still likely to remove suitable habitat entirely.

Landscape heterogeneity that influences thermal patterns may be a critical habitat feature capable of limiting the distribution and abundance of plant and animal species throughout the day, especially in hot, arid environments (Limb *et al.* 2009). For example, lower soil surface temperatures in areas with higher shade may provide critical thermal refugia for many animals on hot summer days (Limb *et al.* 2009). In addition, interactions between air temperature and disturbances, such as livestock grazing, can create thermal heterogeneity by influencing soil moisture, photosynthetic, respiration and decomposition rates and by creating unique habitat conditions that ultimately influence the distribution and abundance of plant and animal populations.

### 5.4.3 Fire

With respect to fire, topographic features, such as mountains and rivers, other moist areas, such as gorges and wetlands, and areas with very low fuel loads can act as barriers to the spread of fire (Banks *et al.* 2011). These produce patches in the landscape where fire frequency and/or severity are reduced. At a fine scale these areas provide immediate protection from fire and provide food and shelter after fire to support animal populations while at a landscape scale they influence the frequency at which areas burn and increase the likely exposure of flora and fauna to fire. Even during extensive, high intensity fires, unburnt patches or patches that are only lightly burnt occur within the fire perimeter (Bradstock *et al.* 2010; Turner *et al.* 2003).

Areas that act as both fire and climatic refugia will likely provide *in situ* habitat for biodiversity as climate change continues. They are important in post-fire regeneration processes in temperate, tropical and arid systems, for a wide range of species – plants, vertebrates and invertebrates (Joseph *et al.* 2011; Mendelsohn *et al.* 2008; Wimberly & Kennedy 2008). However, these climate-fire refugia may be at risk from climate change due its interaction with both fire (Nitschke & Innes 2006) and land use (Mackey *et al.* 2008). Thus, if fire refugia are threatened, the persistence of species that depend on them may be undermined.

The management of fire refugia and habitat complexity is thus an important conservation issue. Indeed, complex components of landscapes, like the dense understoreys in forests, can be thought of as refugia. For example, Lindenmayer *et al.* (2009) argue that, for temperate eucalypt forests, "care should be taken with back-burning during unplanned fires and the spatial and temporal arrangement of prescribed fires to ensure unburned vegetation remains as refugia to facilitate bird persistence." A similar principle is likely to apply in the forest types of Queensland. Conservation managers also need to be aware of shifts in interactions among native species driven by habitat disturbance. For example, current disturbance patterns along the eastern seaboard of Australia have advantaged the native, irruptive bird (the noisy minor), at the expense of other birds. Simplification of woodland understorey is implicated in this process; hence modification of fire regimes and other disturbances such that the complexity and density of the understorey is increased is a desirable conservation management objective (Howes & Maron 2009).

### 5.4.4 Drought

Permanent water bodies, springs and river systems provide water in much of the drier parts of Queensland (see Kroon *et al.* 2012). Permanent water holes are an important resource for many species during the extended dry season in the northern savannas (James *et al.* 1999) when most rivers and streams typically dry up. For example, Ritchie *et al.* (2008) found that the antilopine wallaroo's distribution and abundance was highly dependent on the availability of standing water.

The influence of this water can often be seen in the landscape pattern of trees, where riparian zones support tree populations that quickly diminish with distance from water. Human activity has both destroyed and created critical drought refuges in Queensland. For example, the introduction of domestic livestock across Queensland caused the trampling and destruction of many natural water points utilised by native animals. These losses are compensated by the pastoral sector's watering infrastructure that now support biodiversity in semi-arid Queensland; although not without detrimental effects since these also harbour introduced predators and concentrate grazing pressure (James *et al.* 1999).

Figure 11: Riparian vegetation, Beatrice River, North Queensland (credit: CSIRO science image EM0490).

# 6. REGIONAL VARIATION

## 6.1 Introduction

Queensland provides a wide variety of habitats that arise because of combinations environmental factors (e.g., variation in rainfall, temperature, soil and topography) and ecological processes, previously outlined. These ecosystems support an equal array of plant and animal species, life-forms that have co-evolved or adapted to historical conditions and, depending on the environmental amplitudes of their normal habitats, will differ in their tolerance of present and future change. The continuing effects of climate change will therefore differ between Queensland's regions. In this section, likely broad regional variation in the major types of ecological change described in previous sections, are discussed.

## 6.2 Significant regions and ecosystems

At biogeographic scales, the ecosystems most vulnerable to change are those with restricted geographic distributions, narrow environmental tolerances or systems near the limit of their environmental tolerance and those that are subject to synergistic threat syndromes (Hughes 2011; Laurance *et al.* 2011). The biota of these ecosystems is least likely to have the time or space in which to adapt to rapid climate change.

A number of regions and ecosystems in Queensland are identified as being of particular at risk from climate change. These are summarised below.

- High elevation mountain ecosystems. These systems occur along the east coast of Queensland from Cape York Peninsula through tropical latitudes including the Wet Tropics and sub-tropical latitudes to southern Queensland. Many of the habitats in the ecosystems are elevationally restricted with narrow environmental envelopes and may be near climatic thresholds (Laurance *et al.* 2011). A reduction in cloud stripping (Section 2.4) increased extreme weather events (Section 4.7) and interactions with other threats (Section 4.5) are considered to pose a high risk to the large number of restricted and endemic species sustained by these systems (Hughes 2011; Laurance *et al.* 2011; Williams *et al.* 2003).

- Dry rainforest types. These forests include vine thickets, monsoonal vine thickets and semi-deciduous rainforest types which occur in fire-shadow refugia scattered across much of northern Queensland (Laurance *et al.* 2011). Because of their highly restricted and patchy distributions, their narrow environmental tolerances and their vulnerability to change from threat syndromes (Section 4.5), the drier rainforests have been considered vulnerable to tipping points resulting in significant changes in ecosystem properties.

- Tropical savannas: Tropical savannas extend over much of the northern third of Queensland. Changing fire regimes (Section 2.6 and 6.5), extreme weather events (Section 4.7) and interactions with other threats including invasive species and land-use impacts (Section 4.5), are considered to have the potential to accelerate significant ecosystem change (Laurance *et al.* 2011).

- The Gulf region. Low (2011) postulates a dominant northward movement of biota toward the coast, however, species movements there will be limited by the Gulf of Carpentaria expansion as rising sea levels flood southward into low lying areas (see Bustamante *et al.* 2012). The Cape York and Gulf regions are predicted to experience novel climates (i.e., climates not currently existing) as well as the disappearance of some extant climates (Williams *et al.* 2007) (see also Ferrier *et al.* 2012).

- Arid and semi-arid regions: Shifts in seasonality or intensity of rainfall events, changed fires regimes as a result of warming and drying trends (see Section 2.6 and 6.5) and increased $CO_2$ (see Section 2.2) are likely to result in significant modifications to vegetation composition and structure in these ecosystems (Steffen *et al.* 2009).

- Channel Country of south-west Queensland. Occurring within the semi-arid climate zone, this region supports wildlife living close to their temperature limits during summer. Low (2011) considers catastrophic losses of wildlife could occur during heatwaves and drought, if shelter habitats also decline.

- Areas of high endemism: Areas with concentrations of endemic species are particularly noteworthy as evolutionary arcs for biodiversity. By definition they contain many species with narrow geographic ranges, which often equate with a narrow ecological range and increased extinction vulnerability. Major centres of vascular plant endemism in Queensland occur in the Wet Tropics, Border Ranges and Iron Range/McIlwraith Range area of Cape York (Crisp *et al.* 2001). Depending on the buffering effects of refugia in these heterogeneous mountain regions, climate change may cause a marked increase in species extinction rates (Section 5.4).

## 6.3 Change in rainfall

Changes in rainfall affect water availability. Rainfall has been declining over much of Queensland since 1900 (see Williams & Crimp 2012). The drying trend is evident since 1950, particularly in central, southern and coastal Queensland (DERM 2009d). There is considerable uncertainty in rainfall projections, with the median estimate showing little change in northern Australia and a tendency to increasing decline for the higher emission scenarios (DERM 2009d). Reductions in rainfall due to climate change, although uncertain, have the potential to transform terrestrial ecosystems (Low 2011). A biogeographic summary of the regional influences on rainfall, based on rainfall variability gradients, is presented in Figure 12. The following narrative reflects the schematic patterns presented in the figure.

South-western Queensland currently experiences low, infrequent rainfall. The ecosystems in this region have adapted to this rainfall regime which supports low shrubs and grasses capable of surviving long periods without rainfall. This region also receives flood waters from intense tropical lows originating from the coast. Low (2011) also notes the possible thermal limitation of biota in this region and considers catastrophic losses of wildlife could occur during heatwaves and drought. This biome may not be susceptible to further declines in rainfall within limits. However, if species are dependent on the current limited sources of water and these resources decline during heat events the current population abundances will not be supported.

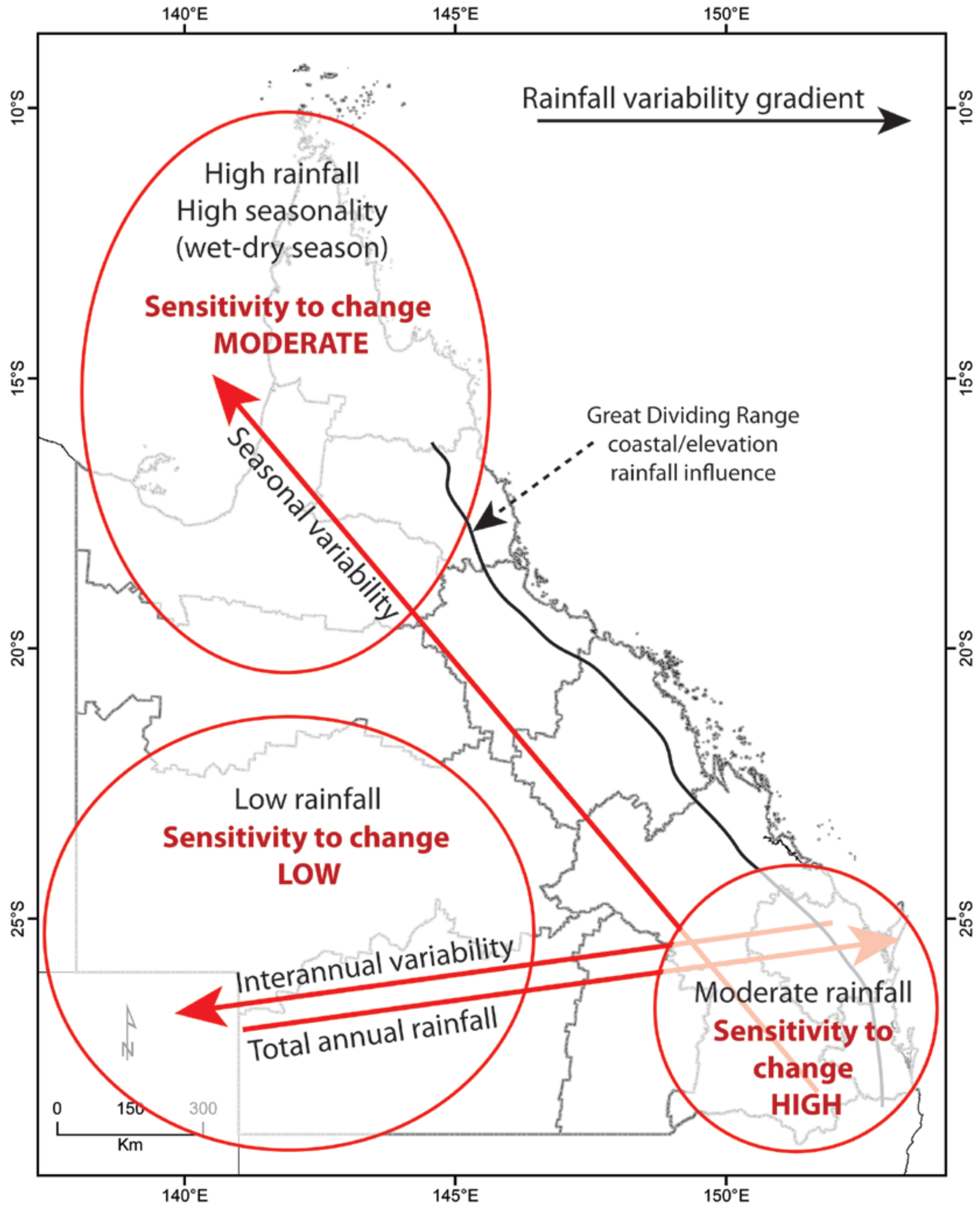


Figure 12: Summary of regional influence of rainfall based on rainfall variability gradients.

Northern Queensland (Cape York, Gulf Region and Far North Qld) is influenced by the tropical monsoon and receives regular, reliable rainfall. However, this rainfall occurs predominantly in the summer months and the very low to no rainfall periods of the winter (dry season) results in a regular, annual drought for trees. This rainfall seasonality is a fundamental driver of the structure and composition of savanna vegetation. While this system may appear insensitive to changes in rainfall, it is highly dependent on the length of the wet and dry seasons. An increase in the dry season may place significant pressure on trees already persisting for the summer rains and the incremental loss of tree cover would cascade into changes in the composition of other plants and dependant animals.

South-eastern Queensland has higher annual rainfall than western Queensland and less seasonal variability than northern Queensland (Figure 12). This availability of water from rainfall supports denser, open forests and woodlands and, in turn, the human population of the south eastern region. Climate change is likely to result in changes in the regional distribution of rainfall (Whitfield *et al.* 2010). We can gain an insight into likely effects of these changes on ecosystem patterning across these regions by considering the end points of the two major rainfall gradients: interannual and seasonal variability (Figure 12). A reduction in rainfall or an increase in seasonality would require the tree populations and ecosystems of south eastern Queensland to adapt to a lower rainfall environment.

Some researchers have noted that the climate change implications of declining rainfall on primary productivity in water-limited regions may be moderated by changes in water and nitrogen use efficiency as a result of $CO_2$ fertilisation (Hughes 2003; McMurtrie *et al.* 2000). However, the outcome for vegetation composition is likely to be more complex due to biotic interactions and individual life history dynamics (Hovenden & Williams 2010; Stokes *et al.* 2005). An increase in water use efficiency and biomass production as a result of $CO_2$ fertilisation will necessarily result in full utilisation of accessible soil water resources within a stand of trees, potentially leading to larger or more individuals (see Box 1). Therefore, the advantage of water use efficiency resulting from $CO_2$ fertilisation is likely to be short-term in a region which is already experiencing declining rainfalls.

## 6.4 Changes in temperature

Increases in maximum summer temperatures are more likely to influence species than increases in mean temperatures (Low 2011). Summer maximum temperatures increase on an east to west gradient over much of Queensland along the same decreasing rainfall gradient. While species living in western Queensland are able to tolerate high temperatures, extended heat waves may be critical resulting in large numbers of deaths as these species may be living closer to thermal limits under normal conditions. This is also true for coastal areas that are currently buffered from extreme heat events. In these situations, animals may be susceptible to even modest increases in the number of consecutive days with extreme temperatures.

## 6.5 Change in fire regimes

As outlined in Section 3.2, regional variation in fire regimes occurs because of landscape scale variation in moisture, temperature and ignitions. Figure 13 presents a summary of the regional biogeographic processes defining the current fire regimes in Queensland and how these may change as a consequence of the impacts of climate change on key weather and fuel characteristics. This regional variation broadly corresponds with patterns in rainfall variability and seasonality (Figure 12). People occupy all regions of Queensland and because different groups of people have different approaches to the seasonality and frequency of ignition there are substantial differences in regional patterns of ignition events leading to fire.

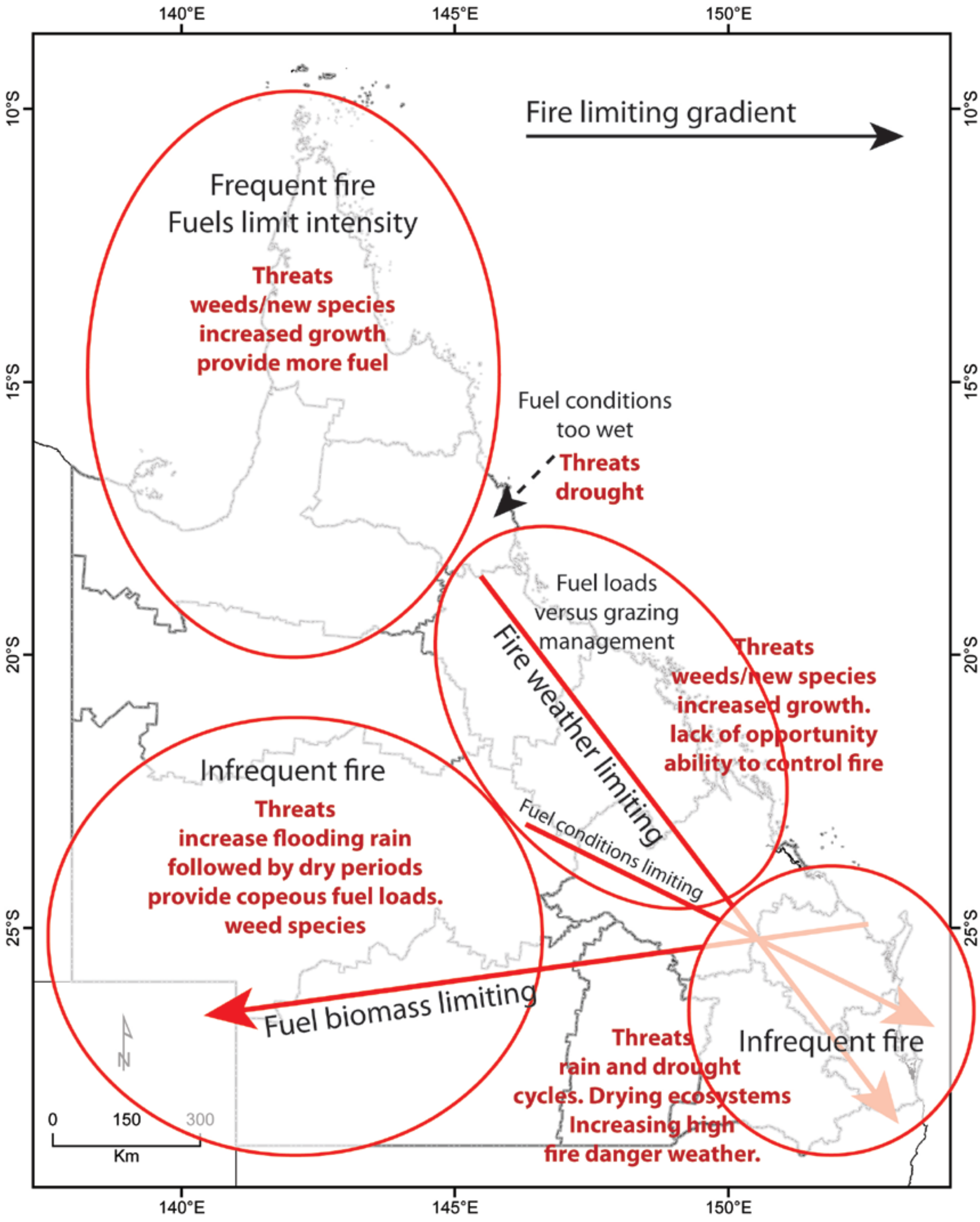


Figure 13: Summary of regional limitations to the occurrence of fire and future threats resulting in increases in frequency or intensity of fire.

The four switch model previously discussed (section 3.2) could be used to explore the potential impacts of climate change on fire regimes and biodiversity in different regions of Queensland. Rainforests and brigalow ecosystems, despite vastly different bioclimatic settings, are both characterised by potentially long intervals between fires (Russell-Smith & Stanton 2002). Both have relatively closed canopies in the mature state and fire regimes are essentially limited by the recurrence of drought. Climate change induced changes to fire regimes in both biomes are likely to be highly dependent on the recurrence pattern of other disturbance agents, such as drought and cyclones in the case of the rainforests and drought and invasion by exotic grasses in the case of the brigalow.

In the savannas, fire regimes will continue to be driven by the annual wet-dry tropical climate. The reliable wet season, followed by a prolonged dry season where grasses cure and weather conditions support fire spread, ensures that fires can potentially burn annually in this region. Moreover, ongoing active management of landscapes through the use of prescribed burning is, and will remain, a key component of most natural resource management objectives. Remoteness is of itself no barrier to the evolution of inappropriate fire regimes (Woinarski *et al.* 2007). There are three main, interrelated landscape issues that underpin the application of prescribed burning for biodiversity conservation in savanna landscapes – the use of early dry-season prescribed fires to: (i) mitigate the impact of more intense and extensive late dry-season fires (Dyer *et al.* 2001; Williams *et al.* 2009); thereby (ii) decreasing fire frequency (Andersen *et al.* 2005); and (iii) maintaining landscape heterogeneity (Russell-Smith *et al.* 2003; Whitehead *et al.* 2005; Woinarski *et al.* 2007); and (iv) the use of prescribed fire to manage the abundance of invasive plant species (Radford *et al.* 2008). These broad fire and conservation management objectives and the use of prescribed fire in landscape management are unlikely to change as a consequence of climate change. In pastoral country within the savannas, it is likely that key features of the pastoral landscape in north Queensland – relatively small property sizes (on a savanna-wide scale), relatively high utilization rates and relatively low rates of ignition – will continue to be key determinants of available fuel and hence regional fire regimes.

Figure 14: Naturally occurring fire. In north Australian savannas, the high incidence of lightning strikes during the build up to the tropical Wet Season (October to November) ensures fire is a regular event (credit: CSIRO Ecosystem Sciences, science image DA0218).

The savannas present a nationally important global change-fire regime problem; that of the spread of exotic species, particularly grasses. Such species may alter fire regimes more than climate change alone; as in these systems the regimes are limited by fuel and ignition more so than climate (Bradstock 2010; Williams *et al.* 2009). For example, gamba grass (*Andropogon gayanus*) has an extensive potential range across the savannas and can produce up to 20 tonnes per hectare of biomass (Douglas & Setterfield 2005). It thus influences the fuel mass 'switch' (see section 3.2). Gamba grass also cures later in the dry season than native grasses, resulting in changes to fire season (the availability to burn switch). Increases in fire intensity can increase tree mortality compared with savanna fires burning in native grass fuel loads (Setterfield *et al.* 2010) and result in rapid ecosystem transformation to exotic grass-dominated structure (Rossiter *et al.* 2003). Using the FLAMES fire simulation model (Liedloff & Cook 2007), Williams *et al.* (2009) undertook a 50-year simulation of fire regimes where native perennial/annual grass understorey was replaced by gamba grass. Repeated, intense fires reduced the tree basal area of the model savanna stand. This will have consequences for habitat structure and complexity and the management of biodiversity of the savannas if the range of gamba grass continues to expand. Most importantly, it was not possible to produce a similar result using the FLAMES model based on native fuel loads combined with estimates of temperature change from climate change scenarios (Williams *et al.* 2009). In the arid zone, the potential influence of the introduced buffel grass, *Pennisetum ciliare* (syn. *Cenchrus ciliata*), (Clarke *et al.* 2005) on fire regimes is analogous to that of gamba grass in the savannas. This species has high drought tolerance, thus providing levels of biomass and spatial connectivity for the promulgation of fire that may exceed that contributed by native grasses and herbage (Clarke *et al.* 2005). Thus, the future fire regimes in the savannas are more likely to be affected by invasion of exotic grasses than by climate change (Bradstock 2010; Williams *et al.* 2009). Management efforts should be directed now towards control of these grasses.

The subtropical sclerophyll forests of Queensland's south-east region has experienced a relatively low period of fire activity in recent years as a result of drought but rainfall from recent cyclonic depressions or wet summers allows fuels to accumulate. This, in combination with climate change, tends to produce weather conditions (high temperatures, low humidity and winds) that will lead to fires that are difficult to manage. Introducing new and novel fire regimes (fire intensity and frequency) has the ability to modify habitat structure by removing fire sensitive species and promoting fire tolerant and often flammable plant species. Such regimes can also open up tree canopies and promote the growth of grasses. Increased fire frequency is likely to increase the proportion of grass in the fuel bed and decrease the proportion of shrubs. These forests present an interesting climate-change-fire-biodiversity management problem. The biome area is important for production forestry (Guinto *et al.* 1999) and is also recognized as having a high proportion of Australia's threatened or near-threatened bird species, particularly hollow nesters (Garnett & Crowley 2000). A key fire management issue is the use of prescribed fire to mitigate risk to life and property posed by unplanned fire, while at the same time achieving appropriate spatial and temporal distribution of regime components to minimise risk to biodiversity (Williams *et al.* 2009). In discussing the conservation management of birds in this forest type, Smyth *et al.* (2002) argue that management should be aimed at avoiding extinctions of populations rather than maximizing species abundances. This is a substantial task in conservation management globally (Keith *et al.* 2002) and requires investment in monitoring schema that focus on population viability parameters, as opposed to more traditional foci, such as counts of presence/absence or simple abundance. These techniques may be rapid and easy to apply (e.g., Mac Nally *et al.* 2010).

Given the matrix of disturbance types and that the relationship between fire regimes, climate change and fauna in general is still poorly understood (Driscoll *et al.* 2010), the southeast Queensland region represents an opportunity to invest in this area of conservation biology.

The availability of fuel is the limiting factor for fires in the arid zone of western Queensland. This region requires infrequent, large rainfall or flooding events to produce fuel loads capable of carrying landscape fires. Fire frequency may increase with more rain, floods or the spread of grassy weeds.

Central Queensland regions lie somewhere on the continuum between the other three areas discussed where there is some influence from the tropical wet/dry season, aridity west of the Great Dividing Range and periods with when fire weather conditions may limit fire activity (Figure 13). This region is also dominated by pastoral land management where there is a competing relationship between fire and livestock for the use of fuels/fodder. Grazing reduces grass biomass to levels that limit fire spread; effectively excluding fire from this region (Dyer *et al.* 2001). This effect is clear when fire frequency maps of northern Australia are viewed (e.g., Figure 15), with less fire than would be expected in much of the grazing country of Queensland. Climate change could result in changes in fire regime in this region. $CO_2$ fertilisation, plus intense rainfall from cyclonic depressions can result in more pasture production while increasing severe fire weather conditions may limit the opportunity to undertake prescribed burning and, these confounding factors could provide conditions that result in the propagation of unplanned fires.

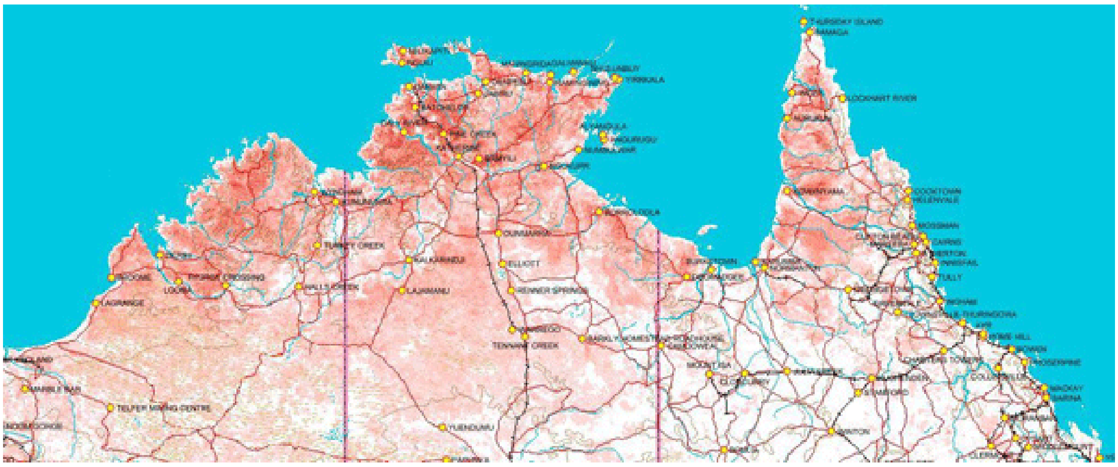


Figure 15. Fire frequencies in northern Australia 1997-2005 as detected by NOAA satellites (Landgate). The darker red areas show areas burnt in most of the 9 years, the lighter pink areas are burnt less frequently and the white areas unburnt by large fires. Image source: Savanna Explorer, Fire in Australia's Tropical Savannas, http://www.savanna.org.au/all/fire.html. Data source: North Australia Fire Information (NAFI), http://138.80.128.152/nafi2/.

## 6.6 Storms and cyclones

All coastal regions of Queensland are at risk of cyclonic activity, with greater risk in the tropical north (BOM 2011). A number of large category 5 cyclones have been experienced in recent years (Tropical Cyclones Yasi in 2011, Monica in 2006 and Larry in 2006). Inland regions are also susceptible to these large systems experiencing both wind and flooding rain. The effects of tropical storms and cyclones are likely to interact with rising sea levels and tidal processes to precipitate change in unprotected and erosion prone regions. Regional variation in these effects is discussed in the context of coastal and marine ecosystems (Bustamante *et al.* 2012).

# 7. ECOSYSTEM SERVICES

## 7.1 Introduction

Healthy ecosystems have intact ecological processes and functions that support the efficient delivery of ecosystem services that all Queenslanders benefit from, such as clean water and air, food and materials, as well as less obvious services, such as provision of recreational areas and even aspects of the environment that determine Queensland’s identity including sandy beaches and iconic species (see Williams *et al.* 2012b). Ecosystem function also includes ecosystem processes that don’t directly benefit humans but without which many ecosystem services would likely collapse. These include the facilitation of energy transfer along food webs and the major processes of carbon, oxygen and nitrogen cycling. These are in turn influenced by the numerous interactions that occur within and among species, as well as between species and environmental gradients, such as rainfall, temperature and geology (Traill *et al.* 2010). Changes in the function of natural systems as a result of climate change will directly affect the ecosystem services provided, resulting in potential economic and amenity losses. Some empirical evidence suggests changes in ecosystem processes have already occurred in response to climate change (Traill *et al.* 2010) and we draw on these examples in the discussions below.

Predicting change to ecosystem services due to changes in biodiversity has proven to be challenging, partly because the same kind and level of ecosystem service can be potentially derived from different combinations of species. For example, some services, like soil erosion prevention or flood risk alleviation or water purification, may be largely independent of the composition of the biodiversity: any plant may do (Montoya & Raffaelli 2010). Other services, such as those for pollination and seed dispersal, however, may be closely coupled to the presence of particular species (see Figure 16). Regardless, if ecosystems are altered dramatically, as is already the case with widespread losses of biodiversity due to existing threat-syndromes (see Section 4.5), the supply of ecosystem services from those systems is unlikely to be sustained.

The spectrum of ecosystem services is often divided into broad groups representing those that are provisioning, regulating, cultural and supporting, following the Millennium Ecosystem Assessment framework (Alcamo *et al.* 2003) (see details in Williams *et al.* 2012b). Some of these relate to individual species, such as the discovery of drugs and insecticides, whereby the extinction of a species through climate change will result in the loss of this future discovery (Wildman 1999). Other services, such as pollination, rely on the presence of pollinators as well as healthy, diverse ecosystems that provide suitable habitat for vectors of pollination services to nearby commercial crops and even urban market gardens (Jackson *et al.* 2007). At even larger scales, healthy landscapes that have tree populations and adequate ground cover maintain hydrological cycles and provide clean freshwater to catchments that can be used for urban water supplies (Brauman *et al.* 2007).

Most ecosystem services can be broadly classified as operating on local, regional, global or multiple scales (Kremen 2005). For example, native species that provide pollination services or pest control on crops generally operate at a local scale, while tree stands contribute to climate regulation at local, regional and global scales (local: shading; regional: rainfall patterns; global: carbon sequestration). Understanding the spatial scales at which ecosystem functions and services operate and the types of changes that may be expected as natural systems change will help in developing appropriate adaptation options.

The terrestrial biomes of Queensland are contiguous with marine and aquatic ecosystems across the coastal and riparian zones. Runoff from terrestrial landscapes enters rivers and coastal marine habitats (for details, see Kroon *et al.* 2012). A healthy terrestrial system, with intact vegetation cover and functioning riparian habitat, provides an ecosystem service by slowing overland water flows and acting as a filter and trap for sediments resulting in water quality benefits (Brauman *et al.* 2007). Extensive clearing of lowland native vegetation for agriculture and ground cover disturbance in northern river catchments has resulted in sediment and nutrient loads and agricultural pollutants being transported to the Great Barrier Reef. This has caused declines in fish and coral populations and increased incidence of algal blooms (Fabricius *et al.* 2005; Fabricius 2005; McCulloch *et al.* 2003). This link demonstrates that changes in terrestrial systems can influence a wide range of other systems providing their own suite of ecosystem services.

This section presents some of the terrestrial ecosystem services provided by Queensland's natural ecosystems and biodiversity, highlighting how climate change is likely to alter these services. A summary of how climate change influences different categories of ecosystem services through its effects on species, ecosystems and landscapes and the likely flow on of effects to sectors that benefit from the provision of each service is given in Table 4 (Appendix 1). For further details about ecosystem services and sectors in Queensland, see Williams *et al.* (2012b).

## 7.2 Biodiversity, ecosystem function and services

Biodiversity—including the number, abundance and composition of genotypes, populations, species, functional types, communities and landscape units—strongly influences the provision of ecosystem services and therefore human well-being (Díaz *et al.* 2005). Consideration of all levels of biodiversity is essential in order to understand its role in ecosystem function and in the provision of ecosystem services. A large body of work has demonstrated that biodiversity generally enhances many ecosystem process rates which provide ecosystem services, such as resource use, nutrient cycling and biomass production, across a wide spectrum of organisms and systems (Balvanera *et al.* 2006). In other contexts, however, the link between biodiversity and ecosystem function is less obvious.

There is increasing consensus among scientists that maintaining biodiversity confers resilience on ecosystems undergoing change (Elmqvist *et al.* 2003; Isbell *et al.* 2011). Resilience itself is a complicated issue and is difficult to quantify and value but is usually defined as the capacity of an ecosystem undergoing disturbance to withstand the pressure without the loss of ecosystem function. We discuss the concept of resilience in the context of climate change in Section 5.2. While ecosystem process rates have been shown to vary across different levels of biodiversity, researchers have shown that the rate of ecosystem process change is much less pronounced across high levels of biodiversity. This implies that high levels of biodiversity provide resilience to ecosystems which may continue to deliver ecosystem services even if the underlying composition and structure of species changes (Montoya & Raffaelli 2010).

Empirical work also demonstrates the role of species diversity in the maintenance of ecosystem services (Isbell *et al.* 2011). For example, recent research by Steudel *et al.* (2011) highlights the role of biodiversity in maintenance of ecosystem function and services, such as productivity and water quality. They found that the productivity of diverse marsh plant communities was more consistent across a range of environmental conditions than that of less diverse communities. The authors suggested that this was a result of 'complementarity', whereby production is maximized by differences among species in their use of resources and environmental needs (see also Mora *et al.* 2011). Similarly, much research has shown that more diverse systems with complementary species are more efficient at removing nutrients from soil and water than are systems with fewer species (Cardinale 2011). Diverse ecosystems are also disproportionally affected by human resource appropriation than less diverse ones (Mora *et al.* 2011).

Biodiverse communities also provide a vital ecosystem service in the form of conferring resistance to invasion (Levine & D'Antonio 1999; Maron & Marler 2007). All else being equal, more diverse communities are more likely to contain highly competitive species and fewer vacant niches and therefore are more resistant to invasion (Díaz *et al.* 2005). For example, it is generally considered that because of the very high species diversity in rainforest, all the available niche space is fully utilised making it more difficult for invading species to establish successfully. This is why disturbance is considered such a critical factor in promoting invasions in rainforests: disturbance creates opportunities for exotic species to claim previously utilized space and resources. Climate change effects that reduce diversity or increase disturbance have the potential to reduce the resistance of native species communities to invasion from weeds and pests.

## 7.3 Species interactions and biodiversity services

Preserving interactions among species is critical for maintaining long term production of food and materials in terrestrial ecosystems. The production of food and fibre depends on the ability of the organisms involved to successfully complete their life cycles. For most plant species, this requires interactions with pollinators, seed dispersal agents, herbivores or symbionts (Díaz *et al.* 2005).

Biodiversity provides important pollinators, seed dispersers and pest control agents on which agriculture and forestry depend. Pollination is a key ecosystem function: an estimated 60-80% of wildplants and 35% of global crop production depends on animal pollination (Klein *et al.* 2007). It has been estimated that in north Queensland rainforest communities, more than 80% of plant species are partly, or entirely, pollinated by insects (Armstrong & Irvine 1989). Crop pollination by wild pollinators is an ecosystem service of immense value that is under-researched and under-valued in the tropics (Martins & Johnson 2009). For example, the native lycid beetle, *Metriorrhynchus rhipidius* (Macleay), the flower wasp, *Campsomeris tasmaniensis* Saussure and the halcid bee, *Homalictus* sp. all make important contributions to the pollination of macadamia in Queensland (Blanche *et al.* 2002; Vithanage & Ironside 1986). Stingless bees, especially species of the *Melipona* and *Trigona* genera are known to be effective pollinators of a least nine crops and contribute to the pollination of about 60 other crop species (Heard 1999). Tropical rainforest beetles in north Queensland are important native pollinators of custard apples in commercial orchards (Blanche & Cunningham 2005) and hawk months pollinate papaya (DPIF 2010; Morrisen 1995).

The interactions between plants and symbiotic microorganisms, such as mycorrhizal fungi, endophytic fungi and nitrogen-fixing microorganisms, can greatly influence ecosystem processes and have considerable impacts in the provision of ecosystem services by natural and agricultural ecosystems (Díaz *et al.* 2005). Ecosystem productivity and carbon accumulation may be enhanced by nitrogen-fixing microorganisms. These include both nitrogen-fixing bacteria and mycorrhizal fungi in symbiotic relationships with plants (especially but not exclusively, legumes) and free-living microorganisms (Thrall *et al.* 2000). For example, in a review of what is known about the role of arbuscular mycorrhiza in plant performance, Smith *et al.* (2010) suggest that a wider range of plants will potentially benefit from the symbiosis than previously believed. Arbuscular mycorrhiza symbioses are formed by approximately 80 percent of vascular plant species in all major terrestrial biomes and may ameliorate abiotic stresses, such as nutrient deficiency and water stress (Smith *et al.* 2010), supporting some level of *in situ* adaptation to climate change.

## 7.4 Individual species' services

The loss of individual species may have great impacts on the wellbeing of local people and their sense of place. Iconic species, such as cassowaries (Figure 16) serve a number of functions. As well as playing a vital role in ecosystem services through dispersal of large fleshy fruits (Bradford *et al.* 2008; Crome & Moore 1990; Westcott *et al.* 2005), they culturally define local regions (Hill *et al.* 2010a; Hill *et al.* 2010b). The loss of such species through habitat changes related to climate change, for example through limited food resources following destructive cyclones (Section 4.7), will have impacts on natural amenity and recreational values as well as on ecosystem function (Latch 2007; Turton 2008).

The iconic Antarctic beech, *Nothofagus moorei* (Nothofagaceae), is at the northern limit of its geographic range in the Gondwana Rainforests World Heritage Area. The species only occurs in areas with high (1158 – 3089 mm/year) and consistent (>45 mm/month) rainfall (Laidlaw *et al.* 2011). With predicted changes to rainfall and reduced water inputs from cloud interception in the Gondwanan Rainforests it is possible that this species will struggle to persist in Queensland (Laidlaw *et al.* 2011).

A slight change in breeding success or competition with other species may cause even common species to disappear from an area, as in the case of an Australian icon, the magpie (Gibbs 2007) (see Section 4.3). Even species that form part of the Australian or Queensland character and considered common, such as koalas, kookaburras or platypus, may find that climate change is the final additional pressure that causes these species to decline or become extinct locally (e.g., Johnson *et al.* 2009). It is also very difficult to understand the full suite of interactions that these species rely on and are impacted by. It may be changes in pollinator species, food sources or competition with invading species, or compounding existing threats, such as habitat loss, that ultimately cause their decline.

Ecosystem services can be modified through ecosystem functions as a result of the arrival of new species or with a change in abundance of species already present (Balvanera *et al.* 2006). Long-lived, woody plant species are likely to dominate the processes of modification through structural habitat changes that also influence fire regimes and affect the nature of supporting habitat for fauna. This process is currently evident in Queensland through weed invasion (e.g., Grice & Setter 2003) and woody vegetation thickening (Witt *et al.* 2009). Increases in weed or feral animal abundance or woody thickening can affect the value of ecosystem services, for example, by causing loss of pasture production though competition or by increasing the cost of cattle mustering in pastoral properties (Lankester 2006). In a review of landholder opinions, Lankester (2006) reports that fire was seen by many graziers as the only economical and practical way to manage woody vegetation thickening. Crowley *et al.* (2009) examined the invasion of native grasslands on Cape York Peninsula by *Melaleuca viridiflora* and other woody species, a process that has been considered an inevitable consequence of climate change. They conclude that a fire regime with regular storm-burning is an effective way to control recruitment of *M. viridiflora* and maintain a stable open vegetation structure in these ecosystems.

Figure 16: The Southern Cassowary (*Casuarius casuarius*) is an iconic species in the Wet Tropics region (credit: Adam McKeown, CSIRO Ecosystem Sciences); it is featured on the corporate logo of the Wet Tropics Management Authority. Cassowaries provide a vital ecosystem service in that they are the only animals capable of dispersing the very large seeds of more than 70 species of native rainforest trees. They also assist in the long-distance dispersal of at least another 80 species (WTMA 2011).

## 7.5 Ecological communities and ecosystem services

Climate change has the potential to radically alter the composition of natural communities in Queensland through individual species abundance changes. For example, as discussed in Section 7.4, changes in individual tree species abundance can alter habitat structure and modify many of the ecosystem services provided by forest and woodland vegetation. Many of the disturbance processes affecting tree stands are already known and cause some of the biggest land management problems seen today. These include woody thickening (Gifford & Howden

2001; Lankester 2006), weed invasion (Grice & Setter 2003; Wilson & Scalan 2008), wildfire, drought dieback (Fensham *et al.* 2003; Fensham *et al.* 2009), storm and cyclone damage (Turton 2008; Turton & Dale 2007), erosion from over-grazing (e.g., Prosser *et al.* 2002) and tree clearing (McGrath 2007). Many of these processes are closely related to climate and rainfall (Woodward & Lomas 2004), suggesting that any change in future climate has the potential to exacerbate changes to tree stands that are already under pressure (See Box 1). Sustainable grazing management, for example, has been shown to improve pasture condition and economic profitability (see Figure 17).

Native tree stands in Queensland provide a wide range of ecosystem services. These vary from provisioning services, such timber production and water capture and storage, honey production, fruit production for fauna, to regulating services, such as water table regulation, climate regulation, wind protection and stream-bank erosion protection. Supporting services ensure the maintenance of catchment hydrological processes and habitats. Humans benefit from interacting with nature through recreation areas and conservation areas, in such iconic settings as the Scenic Rim, Fraser Island and the rainforests of the Wet Tropics.

Climate change may directly or indirectly result in areas that are currently devoid of trees increasing in cover resulting in loss of grass species pastures and open ground (Henry *et al.* 2002; Witt *et al.* 2006; Witt *et al.* 2009). Areas with current stands of trees may experience decreases as a result of climate change, for example through disease (Section 4.5), changes in fires regimes (Section 2.6) or through physiological changes at the individual level (Box 1). Trees are heavily dependent upon soil characteristics, such as depth and texture and the amount and distribution of rainfall including daily rate of rainfall, seasonality and inter-annual variability (Liedloff & Cook 2007). The variability in these factors across Queensland is responsible for the current pattern of tree cover and structure with additional changes driven by anthropogenic factors, such as clearing, grazing and selected timber extraction (Accad *et al.* 2010; Accad *et al.* 2008; Eyre *et al.* 2011). While climates currently experienced by tree stands in a given location will change, the spatial distribution of soil types which is a critical determinants of the distribution of some species, may limit adaptive response to existing climatic envelopes (Aitken *et al.* 2008).

Slight changes in the structure of tree stands can have flow-on effects to modify ecosystem services. Thinning of tree stands; for example, through increases in pathogens (Section 4.5) allows more light to reach the ground resulting in more production of grasses, increasing heat levels and reducing shade. Increases in dry periods, such as during El Nino years, or a slight decrease in winter rainfall, can cause understorey grasses to cure and provide fuel for fires (Section 2.6). This cycle can potentially alter forests and woodlands of southern Queensland so that they more closely resemble tropical savannas and increase the frequency of fire experienced. Fire in the landscape influences the outcome of many ecosystem services, such as conservation of biodiversity, catchment water quality, erosion, damage to infrastructure and loss of domestic livestock.

Modification to vegetation communities in foreshore and riparian zones; for example, due to cyclone damage (Section 4.7); has the potential to severely alter the ecosystem services they provide. For example, littoral rainforest communities in Queensland provide a range of ecosystem services (DEWHA 2009b; Gallagher *et al.* 2010); they are an important buffer to coastal erosion and wind damage (Meier & Figgis 1985) and provide natural refugia, suitable nest sites and food resources for resident and seasonally migratory species (Williams 1993). Moreover, the nationally listed species *Ducula bicolor* (Pied Imperial Pigeon), a migratory species from north of New Guinea, feeds on fruit associated with mainland littoral rainforests and disperses the seeds on offshore islands where it roosts (Threatened Species Scientific Committee 2007).

Figure 17: Brahman cattle grazing with good ground cover in the Burdekin River Catchment area. Sustainable grazing, which includes practices such as wet season spelling and conservative stocking, has been shown to improve pasture condition and economic profitability. It also shows promise as a key strategy to reduce the loss of sediments and nutrients in run-off to the Great Barrier Reef (credit: CSIRO Land and Water, science image BU7022).

## 7.6 Landscapes and ecosystem services

Many ecosystem services are influenced by patterns and processes operating at the landscape scale. These ecosystem processes and services are inherently variable in space and time in part because they occur on spatially heterogeneous landscapes (Loreau *et al.* 2003). Intact, healthy landscapes supply the ecosystem services Queenslanders rely on, providing: clean catchments for the collection of unpolluted freshwater; refuges for flora and fauna; influencing regional weather and climate patterns; providing treasured landscapes for recreation; and providing resources to surrounding areas, such as pollen and surface, stream and ground water. Intact landscapes are also vital in providing the space and connectivity of habitat to allow species to respond to climate change (Section 5.3). The long-term sustainability of ecosystems and the services they generate requires landscape-scale and ecosystem-based conservation measures. In human-dominated landscapes undergoing climate change, diverse ecosystems support minor species that may become important as soon as others disappear (Section 5.2).

## 7.7 Regional changes in ecosystem services

Regional changes to ecosystem services will depend on the predicted regional climate changes and the services that human value from the environment in different places. For example the provision of erosion regulation by native vegetation occupying coastal dunes will be important in mitigating the impact of sea level rise on nearby settlements (e.g., see Figure 18). Eroded coastal dune systems and saltwater intrusion in freshwater wetlands will negatively affect public uses, such as tourism and recreation (Environment Planning 2011). In some regions, coastal erosion processes will lead to the loss of public assets, such as beaches and protective dune systems (for details, see Bustamante *et al.* 2012; Williams *et al.* 2012b).

Figure 18: Intact littoral vegetation helps to protect coasts from erosion. Casuarina on beach, North Queensland (credit: CSIRO science image, EM0484).

# 8. MANAGING ADAPTIVELY FOR CLIMATE CHANGE

> “*Anthropogenic climate change will probably require anthropogenic intervention in species conservation. We make the problem - we fix the problem (or try).*” David Bush, CSIRO, July 2007

## 8.1 Introduction – current programs of work

Adaptation in the context of managing for climate change is the ability of a system (where the system includes both the environment and human management) to adjust to climate change (including climate variability and extremes), to moderate potential damages, to take advantage of opportunities or to cope with the consequences (Parry *et al.* 2007). A detailed discussion of adaptation frameworks for biodiversity and ecosystem management and adaptation options applicable across terrestrial, freshwater aquatic and marine ecosystems is given in Dunlop *et al.* (2012).

Queensland’s draft biodiversity strategy outlines the direct and indirect initiatives for conserving biodiversity that currently apply at State, National and International levels (DERM 2010a). These initiatives are relevant to different values or aspects of biodiversity and their management (Table 5 in Appendix 2). Recognising the holistic definition of biodiversity[1] and variability at different scales (landscapes, ecosystems, species, genes)[2], the draft biodiversity strategy aims to: i) reverse the decline in biodiversity; and ii) increase the resilience of species, ecosystems and ecological processes. The clustering of current initiatives in Table 5 (Appendix 2) aims to show which aspects of biodiversity are the principle targets of concern for management. Most of these initiatives reflect a static view of nature and represent programs of work that were established before the effects of climate change on the environment and biodiversity were understood. Climate change is now recognised as a major threat to terrestrial and marine biodiversity and ecosystem function in Queensland (DERM 2010a). In recent years, an increasing number of international, national and State level conventions, agreements and policies have been developed.

Building climate change resilience into Queensland’s biodiversity is a key underpinning of the State’s draft strategy and an important step forward in the way biodiversity is managed in Australia (DERM 2010a). This follows on from the 2001 agreement of government’s that resulted in an action plan to help coordinate the activities in addressing the impacts of climate change on species, communities and ecosystems in Australia (Australian and New Zealand Environment and Conservation Council 2001; NRMMC 2004). These actions were precipitated by concerns about Australia’s vulnerability to changes in temperature and rainfall projected to occur throughout the 21st century that had been raised in the third assessment report of the intergovernmental panel on climate change (Houghton *et al.* 2001). A further series of State and National documents and discussions have increasingly raised awareness about the potential impacts of climate change on biodiversity (Australian Centre for Biodiversity 2008; Australian National University 2009; Council of Australian Governments 2007; DCC 2009; Department of

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[1] ‘the natural diversity of wildlife (including plants and animals), together with the environmental conditions for their survival’ Queensland’s *Nature Conservation Act 1992*

[2] Regional diversity—the different kinds of landscape; Ecosystem diversity—the different communities of plants and animals; Species diversity—the number of different species in an area; Genetic diversity—diversity in the genetic make-up of individuals and populations. Queensland’s *Nature Conservation Act 1992* as defined in DERM (2010a)

the Environment Water Heritage and the Arts 2009; DEWHA 2009a; Dunlop & Brown 2008; Garnaut 2008; Gilmore *et al.* 2008; Hennessy *et al.* 2007; Hilbert *et al.* 2007a; Howden *et al.* 2003; Hughes *et al.* 2010; Krockenberger *et al.* 2003; National Reserve System Task Group 2009; NRMMC 2010; Steffen *et al.* 2009; Taylor & Figgis 2007; Whitfield *et al.* 2010; Williams & Price 2008; Williams *et al.* 2009).

This section reviews the types of ecological changes described for terrestrial biodiversity in this report and defines a set of “ecological change phenomena” that represent the different types of terrestrial biodiversity responses or outcomes arising from, or driven by, climate change. This provides a structure within which to consider possible adaptation responses to the likely effects of climate change on biodiversity. Management responses that take into account the dynamics of climate change can be linked to some of these ecological changes and related to adaptive pathways for the management of biodiversity.

## 8.2 Ecological change phenomena

A number of terrestrial ecological change types or phenomena can be related to climate change. These arise as species respond to their changing environmental conditions. Changes in the seasonal timing of climatic events affect the phenology of species through their life cycles and result in changes in species interactions and changes in the match between species and their habitat in their current locations. These changes and responses occur at the individual species level, cascade through species interactions to alter the composition of ecosystems and, at larger scales, are reflected in changes to the structure, function and productivity of ecosystems. The following broad groups of ecological change; driven by the interacting effects of climate and environmental changes; can be summarised as follows:

- Changes in species’ phenology, behaviour and interactions (food webs, competition, facilitation) (Sections 4.2, 4.4).

- Changes in species’ abundance, distribution and resilience to climate variability (Sections 0, 4.6).

- Changes in species resistance and exposure to disturbance, pathogens and disease (Sections 4.5, 4.7).

- Changes in overall ecosystem productivity and nutrient status (Sections 2.2, 4.1).

- Geographic changes in ecosystem types through cumulative change in structure, function and composition (Section 7.5).

- Changes in landscape function and ecosystem services (Section 7.6).

These changes reflect the ability of an individual of a species at a site to survive, adapt or move to another site where it may be better suited to local environmental conditions. These abilities often depend on where the species in its current location is actually positioned within its environmental envelope. Species on the cooler margin of their ecological range may, in the near future, find themselves in more suitable habitat or, conversely, species on the warmer margins of their distribution range may be under increasing environmental stress.

At the ecosystem level, the aggregate changes at the species level combine with changes in productivity responses due to enhanced photosynthesis rates with increasing atmospheric carbon dioxide concentrations. This response of primary productivity may be offset by changes in water and nutrient use efficiencies that cascade through the food web to affect the nutritional status of inter-dependent fauna species. As a result of these interactions, the structure, function and composition of the ecosystem may change. The availability of water in the landscape will be a key determinant of the ability of species to tolerate or adapt to the other changes arising from climate change, particularly when combined with climatic extremes that result in storms, floods and fire. The cascade of ecological change combined with climatic extremes and variability may result in rapid episodic state changes in ecosystem structure and function and erode the value of goods and services provided through natural systems.

As a result of climate change altering the geography of environmental conditions, ecological changes will cascade through biological systems and become detectable phenomena at all levels of organisation – species and populations, ecological communities and ecosystems and across landscapes and through a wide range of ecosystem services. These change phenomena may be monitored through observations of species diversity within a site (e.g., richness, abundance, dominance and demography) and accumulate as differences in diversity between sites (e.g., compositional dissimilarity in the number, abundance and phylogenetic or functional make up of species). Across whole landscapes or regions, these changes aggregate into temporal and geographic patterns of emergent properties of biodiversity, such as land cover, vegetation structure, composition and, ultimately, changes in the function and productivity of ecosystems.

The adaptive management of biodiversity under climate change will therefore need to be sensitive to a wide range of natural responses and changes that will occur. Managers will need to consider whether to observe and passively facilitate this change or whether it is appropriate and necessary to actively promote or manage the change. In this context, the traditional objectives of biodiversity management may need to be adjusted in order to recognise and manage these ecological phenomena (Prober & Dunlop 2011). New concepts of ecological change and biodiversity management objectives necessarily underpin adaptive management under climate change and are discussed in more detail in Dunlop *et al.* (2012).

## 8.3 Biodiversity management objectives

Terrestrial biodiversity in Queensland is currently managed under three primary objectives, as outlined in the draft biodiversity strategy (DERM 2010a):

- Building protected areas on public and private land as a foundation for landscape resilience and to connect people with nature.

- Conserving species and their habitats as a safeguard against extinction.

- Managing the extent, condition and connectivity of natural environments to build ecological integrity and landscape resilience in the presence of climate change and other stressors.

Actions underpinning these three primary objectives are supported by three further objectives (DERM 2010a):

- Building the necessary knowledge base in data, science and management, to support implementation of the primary objectives, including benchmarks and monitoring to enable strategic responses.
- Shared responsibility and coordination across government in the management of biodiversity.
- Increased participation by the community and industry in caring for biodiversity.

These objectives are designed to be responsive to new information about how climate change is likely to affect biodiversity and be flexible, so that adaptation strategies can be developed, tested and adjusted as land managers and the community learn what to expect and which approaches work best (Lawler 2009; Mawdsley *et al.* 2009). Anticipating how biodiversity will respond to climate change and considering options for managing biodiversity in this context, the following section provides an overview of possible management options linked to climate adaptation.

## 8.4 Managing biodiversity and ecosystems under climate change

An important concern about the impact of climate change on biodiversity and ecosystems is the existing legacy of degradation from unsustainable land management practices and their ensuing interaction that results in “threat syndromes” (Section 4.5). Many ecosystems have already experienced large-scale perturbations resulting in species extinctions, abrupt species shifts through introductions and changing disturbance regimes as a result of human activities. This concern implies that heavily exploited ecosystems may be irreparable and require expensive intervention in order to stabilise and buffer landscapes from the disruptive effects of climate change. In a recent review of global environment evidence, however, Jones and Schmitz (2009) found that ecosystem recovery times can be measured on timescales of decades to half-centuries, suggesting the potential for a rapid adaptive transition by humankind to more sustainable use of ecosystems. Recovery times for functioning terrestrial systems were found to be the longest, largely because turnover times for the longest living species are greater than in aquatic and marine systems. Furthermore, terrestrial ecosystems have experienced massive disturbances, such as agriculture, deforestation and logging, from which recovery is relatively slow.

Many ecosystem restoration projects use contemporary reference systems or benchmarks as restoration targets, having dismissed the idea of restoring ecosystems back to ‘natural’ or pre-human states (Jones & Schmitz 2009). This is an important biodiversity management concept for climate change adaptation also (Thomas 2011). In many instances, climate adaptation will require management actions that are sensitive or able to recognise the changes that are occurring and use this knowledge in making decisions about whether changes are advantageous to the system and can be allowed or have adverse consequences and should be resisted (e.g., revised definition of invasive species; Webber & Scott 2012; Webber *et al.* 2011a). For example, highly-mobile range-shifting native species may exhibit functional behaviour more similar to invasive species in their recipient communities. How such “invasive natives” are dealt with is important to consider, especially if they threaten to impact on the conservation status of non-mobile species with low resilience to climate change. The purpose of adaptation is to allow the system to adjust to climate change by moderating potential damages, taking advantage of opportunities or coping with the consequences. In this section, we identify possible adaptation options for managing biodiversity on the basis of the potential ecological responses available to biological organisms under new climates (see Section 4).

The rapid nature of current climate change means that evolution as a mechanism for individual species to adapt to changing conditions is only likely for species with relatively short life cycles (see Section 4.2.4). The remaining response options available to species are persistence, obligatory dispersal, range expansion or shift and extinction (e.g., see Bashar 2011; Lawler 2009):

- Persistence. The new climate is within the current envelope of environmental conditions tolerated, or adapted to, by the species by virtue of its genetic variation, phenotypic plasticity or behavioural responses. For example, nocturnal and burrowing behaviours may enable a species to avoid damaging conditions or retreat in viable populations to thermal refugia in a heterogeneous landscape where the climate is constantly moderated.

- Obligatory dispersal. The species cannot persist in the new climate and must move to new locations with environmental conditions within its envelopes of tolerance. Dispersers must have the capacity to physically move with the changing climate at a sufficient rate to track areas with suitable climates or alternately will have to be moved to new areas with suitable climates if they are unable to move on their own (i.e., managed relocation).

- Range expansion or shift. Some species may be able to expand into new habitat and new climates that were not previously available. This may be the case for introduced species that currently occupy a small proportion of their potential environmental range or for species that have been suppressed by a superior competitor that becomes disadvantage in the new climate.

- Extinction. Species that are unable to tolerate the new climate will become extinct if they are unable to move at all or far enough or fast enough without assistance or are disadvantaged and unable to compete with co-occurring species. Some species in this situation may have some capacity to move but may be faced with barriers to movement due to natural and artificial habitat disjunctions due to land clearing and urban development.

The mechanism used by a particular species in adapting to changing conditions may not always be obvious. For example, the distinction between expanders and obligatory dispersers may in large part depend on whether the species is limited physiologically (e.g., genetic constraints on metabolic temperature thresholds) or ecologically constrained (e.g., due to availability of pollinators or a superior resource competitor). Information about the processes that limit species abundances near the leading edge of their range boundary, from where they might potentially expand or retract, will be essential in developing management strategies that provide species with the best possible chance of tracking suitable climatic space (Murphy *et al.* 2010b).

Possible management responses or adaptation options fall into five groups on the basis of species' responsive options (after Bashar 2011; Lawler 2009): maintain the current conservation strategy, expanding the reserve system, matrix management, translocation or facilitate dispersal and *ex situ* conservation. Each of these options are discussed below in consideration of current and proposed management strategies in Queensland.

### 8.4.1 Maintain the current conservation strategy

This includes the current reserve networks and current conservation activities but any gains are likely to be offset by loss of services derived from biodiversity. This strategy will result in the protection of some but not all biodiversity. Although a management option, this strategy is not climate change adaptive because avoidable impacts are not addressed. The new draft biodiversity strategy for Queensland (DERM 2010a) already foreshadows the development of management objectives that are flexible and responsive to new information. Ongoing management strategies that address the non-climate stressors on biodiversity (e.g., fire management, introduced weeds and feral animal management) and that also take account of the interacting effects of climate change continue to be important adaptation options (see Sections 4.5 and 6.5). It is also clear that new criteria for deciding on priority species for conservation management will be needed in the context of climate change. Species that are important ecosystem engineers or that provide functional redundancies that confer resilience to climate change may need to be considered as additional targets of biodiversity management. In some extreme cases, the potential role of introduced species as functional substitutes for extinct taxa may need to be considered (Schlaepfer *et al.* 2011), although careful consideration of unintended negative impacts is essential.

### 8.4.2 Expand the reserve system

This requires the identification and strategic acquisition of additional reserves to minimise the impacts of climate change on biodiversity. The Queensland government is committed to expanding the protected area estate (DERM 2009d; DERM 2010b) in ways that facilitate adaptive responses and build resilience in biodiversity and ecosystems. This activity is to be implemented in part through climate change corridors, which includes restoring vegetation and reconnecting fragmented ecosystems to enhance ecological viability and resilience (DERM 2009a). This action could be strategically implemented in the context of expanding the habitat for native plants and animals, depending on the previous land use of the corridors and how they are managed. Coupled with this initiative is a renewed focus on fire management in national parks that take climate change forecasts into account (DERM 2009c). The current strategy for identifying priorities for reserve acquisition – comprehensive, adequate, representative and replicated (NRMMC 2005) – will be supplemented by a targeted strategy that includes the identification of three types of refugia: evolutionary hotspots, ecological refugia and habitat remnants (DERM 2010b).

### 8.4.3 Manage the matrix

This requires a whole of landscape view of biodiversity. Most biodiversity is found outside reserves and its existence depends on the type and configuration of land use activities that are taking place. Matrix management aims to ensure permeability to movement so that species can both persist within the matrix and, those with the capacity, can track changing climates. A range of management strategies and incentives will be needed to ensure understanding, cooperation and coordination among land owners and stakeholders who have diverging and often competing interests. Biosphere reserves like the Noosa Biosphere[3] (Noosa Biosphere 2010) and World Heritage Areas like the Wet Tropics[4] (WTMA 2008) represent examples of, or experiments in, coordinated matrix management. Clear objectives are necessary, such as the *Climate Change Corridors for Biodiversity* initiative (DERM 2009a) which addresses landscape-level conservation by softening the matrix at a range of scales to facilitate species movement and gene flow. As noted in Section 5.3, increasing connectivity has emerged as an adaptation strategy for conserving biodiversity in the face of climate change (Hodgson *et al.* 2009; Mawdsley *et al.* 2009). Other initiatives in Queensland are based around cooperative agreements, such as Nature Refuges, which are recognised contributions to Australia's National Reserve System but remain under landholder control through specified partnership management plans (DERM 2010b).

Matrix management will be an increasingly important target for building resilience and supporting biodiversity adaptation under climate change. In this context, carbon mitigation measures through the establishment of carbon markets, such as the *Carbon Farming Initiative* (Garnaut 2011), are likely to provide a significant future source of investment for reforestation, which if sensitively implemented and managed can also provide biodiversity co-benefits (Commonwealth of Australia 2011). Queensland is encouraging biodiversity co-benefits in reforestation projects by providing information about the carbon storage potential and growth rates of native regrowth vegetation, as a cost-effective option for the carbon market (DERM 2009b; Fensham & Guymer 2009). As these initiatives expand, additional information, standards and tools will be required to support clarity about decisions that will effectively "lock-in" and transform landscapes in ways that may not be fully compatible with the balance of uses considered sustainable and adaptive under climate change (Bekessy & Wintle 2008; Bekessy *et al.* 2010; O'Connor & Patterson 2011). Other examples of matrix management include land management agreements under the Delbessie Agreement which establishes sustainability objectives and protects areas of significance to biodiversity on rural leasehold land (Eyre *et al.* 2011; Hassett *et al.* 2010). There is considerable potential to build community support for land management objectives that are sensitive to the needs of biodiversity and natural ecosystems under climate change. In this context it will be important to include biodiversity conservation into the broader societal adaptation process, which is happening through Queensland's adaptation planning (Queensland Government 2007) and draft biodiversity strategy (DERM 2010a).

### 8.4.4 Translocation

Translocation (sometimes referred to as facilitated dispersal, assisted colonisation or assisted migration) is the process of physically moving biodiversity from one location where the climate has become unsuitable to a new location which is now climatically suitable for the species. Translocation is a tool for managing the continuing existence of species that may be threatened with extinction and cannot be maintained in their current habitat (e.g., Brownlie *et al.* 2009). In other cases a population may have become locally extinct and is re-established with additional protection measures (e.g., the bilby in western Queensland DERM 2010a). Translocation is a tool of "last resort" under climate change and is more likely to be considered in highly fragmented situations where an inhospitable matrix precludes species movements and

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[3] Noosa Biosphere Reserve, http://www.noosabiosphere.org.au/

[4] Wet Tropics World Heritage Area, http://www.wettropics.gov.au/index.html

intervention is necessary (Minteer & Collins 2010; Opdam & Wascher 2004). McIntyre (2011) describes relevant ecosystem attributes that make translocation a sensible and low-risk strategy in fragmented habitats, based on an analysis of Australian temperate grassy woodlands. These include (1) a biota adapted to resource conservation, (2) an historically naturally connected landscape with component species having wide distributions over a large climatic gradient and (3) current land use unrelated to endogenous disturbance regimes under which the component species evolved. Iconic species may also be candidates for translocation but this is unlikely to be a cost-effective management strategy for the majority of species threatened by climate change.

In other situations, translocations may be considered to facilitate functional stability in an emerging ecosystem or to moderate the dominance of invasive species in highly perturbed systems. For example, biodiversity co-benefits in planted systems under the carbon farming initiative will effectively be translocation experiments for native species and ecosystems. This will require new objectives for biodiversity management (Prober & Dunlop 2011) to enable proactive selection of species that are pre-adapted as ecosystem engineers, or are functionally compatible (e.g., ectomycorrhizae; Courty *et al.* 2010), to facilitate restoration and provide supporting habitat for the species that will follow. Careful choices need to be made about the geographical and temporal components of facilitated movement of species to ensure invasion mistakes of the past are not repeated (Thomas 2011; Webber *et al.* 2011a). Translocation involves introduction of species that could become invasive; these introductions require careful screening and clear objectives before such actions are approved (Minteer & Collins 2010; Webber & Scott 2012). However, one could imagine a future ecosystem at risk; for example, following repeated cyclone disturbance closely followed by drought, fire, invasive weeds and the loss of keystone dispersal agents, such as cassowary and flying fox, into which functional species may be reintroduced to enhance ecosystem resilience and buffer against, or support, transformational change.

### 8.4.5 *Ex situ* conservation

Species may become extinct in the wild but preserved in zoos through captive breeding programs, in botanical gardens, wildlife parks, aquaria, seed-banks or cryopreservation units. This is an important aspect of conservation but rarely enough to save a species from extinction and techniques are often experimental or controversial.

### 8.4.6 Adaptive management

In the face of considerable scientific uncertainty surrounding how different functional units of biodiversity will respond to climate change, management needs to be adaptive, flexible and responsive to new information. This requires some new objectives along with a proactive and coordinated approach to monitoring change that enables participation at all levels of government and the community (DERM 2010a). For example, significant gains in knowledge can be achieved through emerging citizen science initiatives, such as the Australian phenology network[5] (Donnelly *et al.* 2010), the National Ecological Meta Database[6] (Chambers *et al.* 2007) and the Atlas of Living Australia[7] (Sealie & Lawrence 2010). The combined efforts of

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[5] See, http://www.climatewatch.org.au/about/climate-change

[6] The National Ecological Meta Database project, http://www.bom.gov.au/nemd/About.shtml

[7] http://www.ala.org.au/

systematic research through established institutions and community volunteers in natural history will be needed to track and monitor ecological changes in order to be responsive and adaptive to the needs of biodiversity across a wide range of geographic domains as climate change continues. For example, existing international agreements on migratory birds[8] (i.e., JAMBA, CAMBA, ROKAMBA) represent important climate adaptation responses at global levels but these need to be coupled to local monitoring of shifting phenology patterns as flight paths and the location of supporting habitats potentially change. Monitoring and meta-analysis of available data combined with spatial forecasting techniques (e.g., see Ferrier & Drielsma 2010) will help identify the timelines over which ecological changes may be expected and therefore the timeframe for decisions to be made about changing management strategies. Further discussion of these topics can be found in Dunlop *et al.* (2012).

## 8.5 Climate change, fire and biodiversity management

In their national review of the implications of climate change for fire and biodiversity management, Williams *et al.* (2009) suggested that fire management in areas managed for biodiversity will become more complex in the coming decades. This will be due to:

- a poor initial understanding of endogenous fire regimes for many communities;
- uncertainties associated with the direction and magnitude of the effects of climate change on fire regimes;
- uncertainties concerning the interactive effects of climate change and changing fire regimes on biodiversity;
- potential trade-offs that will be required to manage biodiversity values in the face of potentially more frequent and/or intense fires; and the
- need to manage fire regimes to account for multiple landscape values, such as biodiversity conservation, greenhouse gas abatement, carbon sequestration, smoke management and water yield.

A key message for fire and conservation managers is to expect and prepare for, change. Some change is inevitable as a consequence of climate change, as stressed by Dunlop and Brown (2008). Heller and Zavaleta (2009) reviewed climate change–biodiversity management recommendations from the past 22 years. Adaptation requires greater regional, institutional coordination; incorporation of climate change into all planning and action; and greater efforts to address multiple threats. They discussed several generic, practical actions that could be undertaken (such as regional planning, more protected areas, enhancing landscape connectivity and building resilience) but there was no consideration of how to manage changing fire regimes. In Queensland, however, the need to review fire management strategies under climate change is a current initiative of park management. These considerations need to be expanded throughout the matrix to ensure compatibility in fire management and a joint understanding of the risks of climate change to biological as well as human assets.

As noted in Section 4.5.1, because of considerable uncertainty, there is also a need for more detailed region-specific research into the potential impacts of climate change on fuel dynamics

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[8] Bilateral migratory bird agreements: http://www.environment.gov.au/biodiversity/migratory/waterbirds/bilateral.html

and fire regimes in Queensland. The four switch model discussed in section 3.2 can be used to explore the potential impacts of climate change on fire regimes and biodiversity in different regions of Queensland. Altered fire regimes may affect life history dynamics resulting in potential shifts in species composition or abundances that reinforce that regime.

Because fire, when combined with other disturbances such as drought and storm damage, has the potential to transform ecosystems over relatively short time frames, proactive management under climate change is an imperative. For example, management efforts in the tropical savannas should be directed now towards control of the fire promoting grasses that are otherwise likely to precipitate transformational changes in the ecosystem as a result of climate change and fire regime interaction. In all situations, the contemporary and projected compositions of the natural community under climate change and their lifecycle responses to fire need to guide prescribed burning regimes and fire suppression management responses.

Heterogeneous landscapes that offer refuges from fire are also important places for the retreat and persistence of species (see Section 5.4). For example, Low (2011) recommends managing fire regimes to ensure a mosaic of shelter habitats at fine scales are maintained as temporal, cool refugia to facilitate fauna persistence during periods of extreme heat. This includes litter and dense understorey in forests. Knowledge of fire refugia at multiple scales and management requirements to ensure vulnerable habitats are protected from repeated disturbance is a key principle for biodiversity adaptation.

## 8.6 Climate change refugia, vulnerable species and regions

The identification and protection of local refugia likely to support the persistence of biodiversity at multiple scales is a common recommendation for biodiversity adaptation to climate change (Dobrowski 2011; Howden *et al.* 2003; Keppel *et al.* 2012; Low 2011; Mackey *et al.* 2008; Prober *et al.* 2011; Shoo *et al.* 2011; Vos *et al.* 2008). Intact heterogeneous landscapes are likely to contain a wide range of microhabitats that buffer species from climatic extremes – the relatively larger regional species pools in heterogeneous landscapes increases the probability of recruiting key species and maintaining ecosystem functioning during periods of high environmental stress or rapid change.

Refugia represent intrinsic adaptive capacity in the landscape. It is likely that different types of refugia will become important as climate change continues. Species populations presently located at the leading (warming) edge of their ranges are likely to occur in low density in cryptic refugia. The identification and protection of cryptic refugia will be important in enhancing the capacity of species to persist, adapt and migrate in response to climate change. Some refugia may provide only temporary safe havens for species but represent opportunities for future management given that decisions are initially likely to be overwhelmed by the anticipated cascade of change requiring strategic or triage responses.

As climate change continues, species are expected to expand or shift their ranges towards higher latitudes or higher elevations (Section 5.3) but, in actuality, fragmented landscapes pose barriers to many range-shifting species. Using spatial assessment and forecasting tools, it may be possible to identify vulnerable regions and ecosystems where significant barriers may exist given the anticipated velocity and direction of climate change (e.g., see Ferrier *et al.* 2012). For example, taking into account existing areas of remnant vegetation, those parts of the landscape that may benefit in the future from adaptation management actions, such as enhancing vegetation (e.g., regrowth management and ecosystem restoration) or assisting long-distance migration / colonisation, could be evaluated using ecological models coupled with climate change projections. Trial adaptation planning assessments using spatial tools conducted in partnership with land managers and conservation planners need to be guided by specific objectives and criteria.

## 8.7 Climate change and invasive species

Introduced plants and animals that become naturalised and aggressively invade natural ecosystems have been identified as a significant threat to biodiversity, even in the absence of climate change. In combination with climate change, invasive species are expected to contribute to interacting processes or ‘threat syndromes’ that could precipitate major environmental change (Sections 4.5 and 6.5). In addition to known weeds, sleeper and alert weed species are expected to become more active as climate change continues (Scott *et al.* 2008). Moreover, range-shifting native species may become invasive in their recipient ecosystems and require similar management to invasive aliens. For example, sleeper and alert weed species from the far north of Australia can be expected to be displaced southwards by over 1000km (Scott *et al.* 2008; van Klinken *et al.* 2009). Recommendations, such as those outlined for South Australia (Kriticos *et al.* 2010), also apply to biodiversity management under climate change in Queensland, for example:

- Maintain or establish long term monitoring plots or procedures in key environmental and agricultural areas so that changes in distribution and abundance of weed flora can be detected early enough to apply appropriate management strategies.
- Introduce climate change considerations into weed risk assessments and develop alert species and sleeper weeds lists.
- Use climate change considerations by planning region to identify the future risks posed by weed species that may either not be present or may be currently present as benign (sleeper) populations.
- Use climate analogue analyses to identify potential weed species that presently occupy climates that are similar to Queensland’s recent historical or expected future climates. The list of potential weeds can then be used to investigate the presence of these species in Australia (e.g., in horticulture) and to undertake weed risk assessments.
- Develop strategic research priorities in the area of climate change and invasive plants.

## 8.8 Local habitat management

The adaptive capacity of biodiversity under climate change will depend in large part on the perceptions and will of local people sharing responsibility for the protection and management of biodiversity in their own backyards and commons. Existing community networks and non-government organisations will likely need more information to develop a collective understanding of the range of management actions likely to be adaptive for biodiversity. Local habitat management plans developed through community, government and research partnerships, such as that developed for Mission Beach, will be needed (Hill *et al.* 2010a; Hill *et al.* 2010b). Tools and information to facilitate the inclusion of climate change concerns in community-based plans, linked to adaptive strategies in broader regional plans, will be needed. For example, litter retention to provide a range of micro-thermal habitats for species to shelter during hot periods is an action that could be incorporated in local community-based plans. Habitat heterogeneity at fine scales will help species with behavioural adaptive mechanisms, such as reptiles that seek shelter to avoid hot conditions (Low 2011).

With greater awareness of the values, local communities can help build biodiversity resilience to climate change. For example, the maintenance of a matrix of natural ecosystems, interspersed and adjacent to human settlements and agricultural fields, can provide significant insurance that pollination services remain intact. While people can sometimes relate better to the services provided by natural ecosystems, than to the intrinsic values of biodiversity, as a reason to protect habitats, it is important to recognise that many services are substitutable while extinction is forever. The services that biodiversity provides in the form of mitigation, protection, buffering and options, cannot be measured or valued in the short-term but the value of these services will be increasingly realised over the long-term (see Williams *et al.* 2012b). Understanding the spatial scales at which ecosystem functions and services operate will help in developing appropriate adaptation options at different scales – local, regional and global (see Section 7).

## 8.9 New ways to manage ecosystems

Existing strategies for managing biodiversity evolved during a period of relative climate stability and in response to the increasing acquisition of habitats for human uses. Community concerns about declining populations and extinctions were put into effect through legislation, such as the *Nature Conservation Act 1992* and the *Vegetation Management Act 1999*, to protect threatened species and ecosystems and manage remnants habitats. However, the values and criteria used to identify species and ecosystems “at risk” may need to be expanded to take into account species that are transformative under climate change, in positive ways, in maintaining or supporting functioning ecosystems and habitats. Research is needed to define what these new values are that facilitate climate adaptation and how particular species and ecosystems functions and processes would be identified and pro-actively managed under climate change (Bennett *et al.* 2009). Such research could be used to inform ecosystem reconstruction efforts coupled with initiatives, such as carbon farming. Robust and effective planning for the ecological future is needed before options are foreclosed (Bennett *et al.* 2009).

Traditional conservation management will need to adjust with new information about the types of changes that can be expected under climate change. For example, land managers will need to prepare for the appearance of new and vagrant species and foster these populations or new species interactions, if this seems adaptive, sometimes at the expense of other species in their care. Some changes in interactions among native species may be driven by changes in habitat disturbance regimes. Land managers will become landscape architects, fostering ecological changes through an evolutionary management strategy.

The definition of a native species for management purposes will also require rethinking under climate change (Webber & Scott 2012). For example in permanently invaded ecosystems, Carroll (2011) suggests “conciliatory strategies” that incorporate the benefits of non-native species in addressing the practical needs of ecosystem change. This includes slowing rates of evolution to resistance, promoting evolution of indigenous biological controls, cultivating replacement services and novel functions and managing native–non-native co-evolution. This management strategy utilises the predictive power of evolutionary theory to offer diverse and flexible pathways to more sustainable outcomes (Carroll 2011). Webber & Scott (2012) suggest a new terminology to ascribe native or alien status to species in rapidly changing climates. Their approach is based on the concept of a 'projected dispersal envelope' which describes the area in which a population of a species could be found based on natural dispersal or migratory traits. They recommend using this envelope to distinguish between species that are legitimately moving (perhaps in response to climate change) as opposed to populations that have undergone inappropriate long-distance range shifts.

A wider range of mechanisms (legislation, incentive schemes, market-based instruments, regulations and institutional arrangements) are needed to incentivise the protection and management of biodiversity under climate change. For example, policy tools or market-based instruments (e.g., incentive schemes and biodiversity credits) need to be identified that are capable of promoting synergies between climate change mitigation, adaptation and biodiversity conservation (Hughes *et al.* 2010). This requires clarity in conservation goals and in conceptual thinking around the types and timing of ecological change and management actions taking into account the different types of uncertainty (Hallegatte 2009; Stafford Smith *et al.* 2011). For example, Ferrier *et al.* (2012) describes a hypothetical situation of ecological change over more than 100 years from the stable climates in the middle of last century to beyond the middle of the current century. More specific hypothetical examples are needed to put ecological understanding about the specifics of ecosystem change into practice and, in so doing, identify some of the practical requirements of implementing conservation management under climate change (Morton *et al.* 2009). This could include consideration of the different tactics for reducing decision-making risk (e.g., Hallegatte 2009), examples of which are mainly given in the context of infrastructure type projects; but some of these tactics and strategies are relevant to conservation management also (Stafford Smith *et al.* 2011) (Further details are given in Dunlop *et al.* 2012).

In summing up the missing pieces of information for climate change adaptation, Lawler (2009) observes that "to successfully manage for climate change, a better understanding will be needed of which species and systems will likely be most affected by climate change, how to preserve and enhance the evolutionary capacity of species, how to implement effective adaptive management in new systems and, perhaps most importantly, in which situations and systems will the general adaptation strategies that have been proposed work and how can they be effectively applied." In a subsequent article, Lawler *et al.* (2010) suggests that the most important tool for managing ecological systems in the face of climate change is active adaptive management, in which systems are closely monitored and management strategies are altered to address expected and ongoing changes. Adaptive management is central to Queensland's draft biodiversity strategy. This perspective may only challenged by the cost of implementation and uncertainties inherent in projected climate impacts and therefore in the outcome of management actions. It will be important to develop dynamic landscape conservation plans and identify management strategies that are robust to different future climates that can be implemented without foreclosing options (as discussed in more detail in Dunlop *et al.* 2012).

# 9. CONCLUSIONS

Queensland’s terrestrial biodiversity provides enormous value through the provision of ecosystem services that sustain and support natural, urban and production systems. Some of these ecosystem services are obvious and valued but others are more subtle and may be less obvious until they are no longer being provided. Ecosystem services are tightly coupled to biodiversity and any change in ecosystem structure function or composition will affect the quality and quantity of ecosystem services available to people. Some apparently subtle changes in ecosystems may become magnified through interaction with other stressors and result in a cascade of change that ultimately transforms ecosystems.

The current scientific consensus is that there will be change to natural and human-altered ecosystems in Queensland and this change is likely to be substantial. There will likely be a cascade of effects from changing climatic regimes, such as temperatures and rainfall patterns, on individual species biology and behaviour, to species and populations and to ecosystems and landscapes. These changes will result in shifts in plant and animal species distributions and changes in interactions between species. This will be compounded by regional changes in other threats such as the frequency and intensity of fires and the distribution of introduced weeds that further modify ecosystems. Future natural landscapes are thus likely to look and behave quite differently from those we see today.

Intact and biodiverse ecosystems have the greatest resilience to changing conditions. They provide ‘insurance’ in the provision of ecosystem services in the face of climate change. However, many regions of Queensland have a long history of disturbance and modification. Adapting to the effects of climate change will depend on protecting and restoring biodiversity and appropriately connecting intact landscapes across a range of Queensland environments so that natural systems continue to provide vital ecosystem services into the future.

This report presents the likely changes to terrestrial biodiversity as a result of climate change and discusses the capacity of native biodiversity to adapt to change. We emphasise that climate change is one of a number of pressures being placed on Queensland’s natural assets and that the changes we identify must be considered in the context of ‘threat syndromes’ which take into account other major drivers of change, such as land-use change, invasive species, fire and disease. In a triage based approach, reducing the current threats to Queensland environments is the first step in ensuring these systems are intact and diverse enough to ensure resilience and the maintenance of ecosystem function as the climate continues to change.

# APPENDIX 1

Table 4: General summary of how species, ecosystems and landscape pattern influence the ecosystem services offered by natural terrestrial systems in Queensland and likely influences on sectors benefiting from each service.

| Type | Category | CC influences | Species existence | Ecosystem health | Landscape pattern | Threat syndromes | Sector influenced |
|---|---|---|---|---|---|---|---|
| Provisioning | Freshwater | Relative humidity changes, temperature | Plant species in high altitudes precipitate water from clouds | Moist montane environments support specialised plants and water dependent species (e.g., frogs) | Requires altitudes higher than cloud formation layer, especially during the dry season | Loss of specialised plants and animals in the warming and drying environment, due to rising altitude of cloud layer | Conservation of "at risk" species and unique ecosystems; reduction in montane water provision during the dry season |
| | Water quality | Rainfall, extreme events, changes in fire regime | Tree species suppress watertable and reduce the effects of dry land salinity in sensitive landscapes | Sufficient plant and litter cover to protect soil from erosion. Healthy plant-soil water processes and filtering | Continuous vegetation cover from upslope to water courses | Landscape degradation due to intensive use, such as unsustainable grazing and reduced vegetation cover may follow fire and drought | Water supply quality and quantity, conservation of "at risk" species, agriculture productivity |
| | Genetic resources | Temperature, season, rainfall, extreme events | A diverse mix of native flora and fauna required | Undisturbed, intact ecosystems able to support native communities, with a range of ecological provenance types | Representative, replicated and viable natural habitats across a wide range of landform types and soil-climate gradients | Habitat for extant species becoming less suitable as the climate warms, competition from introduced or migrating species and new fire regimes additional pressures | Reduced future options for the acquisition and development of commercial, medical and food security products; reduced potential for technological innovation from biological analogues |
| | Grass biomass | Rainfall, temperature, $CO_2$, changes in fire regime | Perennial and annual grass species | Diverse mix of native grasses | Grass and tree stands in dynamic mosaics and balance maintained through fire management, grazing pressure and climate interactions | Introduced fire-promoting grasses can shift the balance to largely monoculture grasslands, exacerbated by drought; | Reduced landscape productivity and loss of resilience in grass mixtures, increasing sector vulnerability to climate change; conservation of "at risk" species |

| Type | Category | CC influences | Species existence | Ecosystem health | Landscape pattern | Threat syndromes | Sector influenced |
|---|---|---|---|---|---|---|---|
|  | Carbon storage | Rainfall, extreme events, $CO_2$, fire | Presence of plant species capable of storing carbon (e.g., long-lived, large trees and extensive root systems support diverse soil biota) | A system with healthy soils, hydrological processes and free of disturbance (e.g., fire, weeds, etc.) will ensure productive tree stands | Healthy landscapes with functional, dynamic hydrological processes (vegetation-soil-atmosphere interactions) | Groundwater changes, storms, cyclones, wildfire, weeds | Carbon-mitigation industry, conservation of “at risk” species, pastoral and agricultural productivity |
|  | Soil carbon storage | Temperature, rainfall | Soil fauna (e.g., microbes, nematodes, earthworms and termites) critical for engineering soil processes and storing carbon | Vegetation cover protects and maintains healthy soils to facilitate maximum carbon storage | Productive landscapes with diverse patterns of vegetation cover and healthy, fertile organic soils | Higher temperatures and drought may reduce vegetation cover and expose soils to drying, degradation and erosion processes, exacerbated by increased fire incidence | Carbon-mitigation industry, conservation of “at risk” species (including loss of soil biota unknown to science), pastoral and agricultural productivity |
| Cultural | Recreation and aesthetic values | Rainfall, extreme temperatures, storms/cyclones, drought, fire | Iconic species, unusual examples of species (e.g., large, old trees) or congregations | Healthy systems with native flora and fauna with seasonal events, such as flowering, breeding and migratory congregations | Vast landscapes (wilderness) without disturbance | Weeds, physical disturbance, fire | Tourism, urban populations, recreation, health |
|  | Spiritual, religious | General consequences of climate change | Species of personal or spiritual value | The functioning of a healthy system | Important aspects of the landscape should maintain current function | Species distribution shifts, weeds and feral animals, changes in fire regime | Health, indigenous people |
|  | Knowledge systems, education and inspiration | General consequences of climate change | Continuing observations of presence of rare species and species behaviours | Monitoring of species interactions and natural system function | Balance of sustainable uses and natural environments in the landscape | Changes in landscape patterns and the balance of uses no longer sustainable | Education, government |
|  | Sense of place | General consequences of climate change | Iconic species or individuals associated with place | Healthy systems or belief systems persist | Iconic landscapes including mountains and water bodies | Weeds, loss of iconic (even common) species from a region | Health, less recreation due to reduced aesthetic appeal |
| Supporting | Nutrient cycling | Temperature, rainfall, extreme events | Critical microorganisms and invertebrates | Productive systems creating biomass | Closed systems not losing nutrients through wind and water to other areas | Disturbance, drought, fire | Conservation of “at risk” species, reduced agriculture and pastoral productivity |

| Type | Category | CC influences | Species existence | Ecosystem health | Landscape pattern | Threat syndromes | Sector influenced |
|---|---|---|---|---|---|---|---|
| | Soil formation and retention | Temperature, rainfall, extreme events | Soil fauna (microbes, earthworms and termites), ground cover, tree roots | Healthy ecosystems ensure water moves slowly across landscapes and into soils | Landscapes with cover, healthy waterways and litter | Disturbance, changes in fire regime | Conservation, agricultural, water provisioning |
| | Primary production | rainfall, extreme events, $CO_2$, fire | Species available, genetics present to handle changing environments | Pollinators present, all other services, such as nutrient and water cycling functioning | Mosaic of habitat patches and land uses for effective plant-animal interactions | Changing rainfall, extreme temperatures, fire, storms/cyclones, overall reduction in primary productivity | Agriculture, forestry, pastoral, conservation |
| | Provision of habitat | General consequences of climate change | Mature trees, shrubs and grasses | A range of ages of trees and shrubs providing hollows and other soil and arboreal support structures | Natural landscapes with a range of substrates, soils, vegetation, water-bodies, etc. | Disturbance, changes in rainfall, changes in fire regime | Conservation of “at risk” species and changes in ecosystem character and appearance |
| | Production of oxygen | Rainfall, storms/cyclones | Plant species photosynthesis – $CO_2$ substrate, oxygen bi-products | Recycling of carbon and oxygen in the atmosphere | Landscapes with intact forests | Loss of native forests | Reduced air quality affects all sectors |
| | Local climate | General consequences of climate change | tree species canopies and associated transpiration influences atmospheric water and weather patterns | Effective ecosystem functioning with carbon, water and nutrient cycling | Areas covered by trees have different albedo to bare/rocky areas and this influences temperatures and radiation | Loss of tree cover, loss of grass and litter exposing bare soils, reduced transpiration, less cloud and local rain | Water provision, health, conservation, tourism |
| | Water cycling | Rainfall, temperature (evaporation), humidity | Soil macropore producing organisms (burrowing animals, soil biota and tree roots), plant canopy and transpiration | Effective ecosystem functioning with carbon, water and nutrient cycling | Intact landscapes with healthy soils, vegetation cover, trees and vegetated water courses | Changes to fire. Loss of trees | Water provision, conservation, agriculture, pastoral |
| Regulating | Invasion resistance | Changes in local climate | Healthy native species | Intact healthy ecosystems minimise the ability of new species to invade | Intact landscapes do not provide corridors for dispersal | Loss of critical species | Conservation, pastoral |
| | Herbivory | Changes in local climate | Native species with natural defences | A mix of plant species and predators | Areas large enough to support populations of herbivores | Fragmentation, distribution shifts in herbivores, monospecific | Conservation, agriculture, forestry |

| Type | Category | CC influences | Species existence | Ecosystem health | Landscape pattern | Threat syndromes | Sector influenced |
|---|---|---|---|---|---|---|---|
| | | | | | | stands | |
| | Pollination | Temperature, season | Continued presence of pollinator populations (e.g., bees) | Diversity of species and flowering times to ensure uninterrupted supply of pollen | Landscape is sufficiently connected to ensure access to flowering plants. Natural habitats may become critical pollinator sources | Extreme temperatures, diseases, pests, shifts in plant phenology, loss of critical flowering species, decoupling plant-animal interactions | Horticulture, market gardeners, agriculture |
| | Seed dispersal | General consequences of climate change | Critical dispersal species | Food supplies to allow persistence of dispersal species | Landscapes large enough and continuous to support large dispersers (e.g., cassowaries) | Fragmentation, weeds, shifts in flowering and seeding patterns, decoupling plant-animal interactions | Conservation, weed management |
| | Pest regulation | Temperature, rainfall | Species native to a region may be undesired in new regions | Disturbed ecosystems assist dispersal of current pest species | Transport networks across landscapes may facilitate spread of pest species | Stress to current systems, climate change enables pest introductions or expanded distribution ranges | Conservation, health, pastoral, agriculture, urban systems |
| | Disease regulation | General consequences of climate change | Stressed species may carry disease, migrating species may transport vectors of disease to new areas and across species boundaries | Disturbed ecosystems may facilitate disease vector dispersal | Transport networks across landscapes may facilitate spread of disease vectors | Stress to current systems, climate change leads to new disease introductions or distribution ranges | Health, pastoral, agriculture, forestry conservation |
| | Natural hazard protection | Rainfall, extreme events | Mature vegetation species provide protection from storms/cyclones, floods and landslip | Ecosystem recovery after disturbance is important to maintain this service | Intact landscapes with vegetation cover in critical disturbance-prone situations | Frequency, type, duration and intensity of disturbance | Urban settlements, coastal environments, mountain environments |
| | Erosion regulation | Extreme rainfall and storm events, sea level rise | Ground cover species maintain costal land and dune integrity during storms | Intact dunes and littoral vegetation in coastal systems protect adjacent freshwater systems | Intact vegetation cover on dunes help maintain coastal pattern, shape and currents (e.g., N / S Stradbroke Island) | Fire regime, trampling and bank disturbance can reduce vegetation cover and expose sand/soil surfaces to erosion processes | Reserves, water provision, urban, recreation |

| Type | Category | CC influences | Species existence | Ecosystem health | Landscape pattern | Threat syndromes | Sector influenced |
|---|---|---|---|---|---|---|---|
| | | Changing fire regimes | Native and introduced grass cover | Intact cover over a range of scales filters water moving across landscapes and helps infiltration | Intact cover across broad landscapes limits erosion into catchments | Grazing, floods, clearing, mechanical activities | Water provision, forestry, pastoral |
| | | Drought, fire, extreme events | Native riparian vegetation protects stream-banks | Healthy riparian zones improve water quality and prevent gully erosion and loss of soil | Intact landscapes with stable river and drainage systems fringed by native vegetation, filtering water and trapping sediments | Grazing/trampling, weed invasion, fire, exacerbate erosion gullies leading to increased sediment in watercourses | Water provision, forestry, pastoral |
| | Salinity | drought, fire, extreme events | Deep rooted tree species use soil water and maintain water tables below the soil surface | Healthy soil water processes and tree transpiration prevent rising water tables and salt build up in soils | Low lying and coastal landscapes are more prone to natural salinity build-up | Drought and loss of tree species can lead to rising water tables and increased salinisation | Salinisation affects all sectors |

# APPENDIX 2

Table 5: Current initiatives (legislation, agreements, policies, programs of work) for conserving biodiversity values: based on information sourced from Tables 1 and 2 in "Building Nature's Resilience: A Draft Biodiversity Strategy for Queensland" (DERM 2010a).

| Biodiversity values (reverse declines and build resilience) | Biodiversity management targets (understand, conserve, manage, extent, condition and connectivity) | | | |
|---|---|---|---|---|
| | Species | Habitats | Viability and ecological integrity | People and environment |
| Species (individuals, populations and meta-populations )<br>*Nature Conservation Act 1992; Biodiscovery Act 2004; Land Protection (Pest and Stock Route Management) Act 2002; Environment Protection Biodiversity Conservation Act 1999 (C'wlth)* | Species Conservation and Management Plans for Dugong, Estuarine Crocodile, Koala, Macropod, Whales and Dolphins, Protected Plants; ONE Plan; WildNET and Wildlife Online; HERBRECS database;; National Biodiversity and Climate Change Action Plan 2004-2007; Convention on International Trade in Endangered Species of Wild Fauna and Flora (1973); Convention on the Conservation of Nature in the South Pacific (Apia Convention, 1986) | State Planning Policy and State Planning Regulatory provisions for koala conservation; Queensland Government Environmental Offsets Policy (2008); Conservation Management Profiles; Essential Habitat Mapping; Caring for Country Outcomes 2008–2013 | Natural Resource Management Plans (back on track biodiversity action plans); Park Profiles; Recovery Action Database; National Framework for NRM Standards and Targets (2002); National Climate Change Adaptation Framework (2007); Australia's Biodiversity Conservation Strategy 2010–2020; International Convention on Biological Diversity (1992); International Convention on the Conservation of Migratory Species of Wild Animals (Bon Convention, 1979); Japan-Australia Migratory Bird Agreement (JAMBA, 1981); China-Australia Migratory Bird Agreement (CAMBA, 1988); Republic of Korea-Australia Migratory Bird Agreement (ROKAMBA, 2007); Convention on the Conservation of Nature in the South Pacific (Apia Convention, 1986) | *Biodiscovery Act 2004; Land Protection (Pest and Stock Route Management) Act 2002; Biosecurity Strategy 2009–2014;* Blueprint for the Bush; Australian Weeds Strategy (2005); Australian Pest Animals Strategy (2005); National Forest Policy Statement (1992) |
| Ecosystems (communities and assemblages of species, meta-communities)<br>*Vegetation Management Act 1999; Regional Forest Agreements Act 2002 (C'wlth);* | Enhancing Biodiversity Hotspots along Western Queensland Stock Routes | Nature Refuges Program; Nature Assist; Trust for Nature; Terrestrial biodiversity and wetland conservation State interests (Areas of Ecological Significance); Conservation Management Profiles; Regional Ecosystem Mapping; National Biodiversity and Climate Change Action Plan 2004-2007; Convention on the Conservation of Nature in the South | Biodiversity Planning and Aquatic Conservation Assessments; Park Profiles; Blueprint for the Bush; Strategy for Australia's National Reserve System 2009–2030; Commonwealth funding for National Reserve System acquisitions; National Climate Change Adaptation Framework (2007); National Framework for the Management and Monitoring of Australia's Native Vegetation (2001); | Blueprint for the Bush; South East Queensland Forests Agreement (1999) and forest transfers process; National Forest Policy Statement (1992) |

| Biodiversity values (reverse declines and build resilience) | Biodiversity management targets (understand, conserve, manage, extent, condition and connectivity) | | | |
|---|---|---|---|---|
| | Species | Habitats | Viability and ecological integrity | People and environment |
| | | Pacific (Apia Convention, 1986) | Nationally Agreed Criteria for the Establishment of a Comprehensive, Adequate and Representative Reserve System for Forests in Australia (JANIS criteria) (1997); National Framework for NRM Standards and Targets (2002); Australia's Biodiversity Conservation Strategy 2010–2020; Caring for Country Outcomes 2008–2013 | |
| Ecological processes (natural systems, landscapes and regions)<br>*Sustainable Planning Act 2009; Coastal Protection and Management Act 1995; Wet Tropics World Heritage Protection and Management Act 1993; Wet Tropics of Queensland World Heritage Area Conservation Act 1994 (C'wlth); Recreation Areas Management Act 2006; Land Act 1994; Cape York Peninsula Heritage Act 2007; Environmental Protection Act 1994; Native Title Act 1993 (C'wlth)* | | Biodiversity Planning and Aquatic Conservation Assessments; Park Profiles | Protected Areas Management Plans; Rapid Assessment Program for protected areas (management effectiveness); World Heritage Areas—Wet Tropics, Fraser Island, Gondwana Rainforests of Australia, Riversleigh and the Great Barrier Reef; Blueprint for the Bush; National Framework for NRM Standards and Targets (2002); Natural resource management plans; Intergovernmental Agreement on the Environment (1992); National Climate Change Adaptation Framework (2007); Australia's Biodiversity Conservation Strategy 2010–2020; Caring for Country Outcomes 2008–2013 | Blueprint for the Bush; Delbessie Agreement; Indigenous Land Use Agreements; Indigenous Management Agreements; Master Plan for Queensland's Parks System (2001); Draft Queensland Coastal Plan (2010); Regional Coastal Management Plans; State Regional Plans (under the *Sustainable Planning Act 2009*); Toward Q2–Tomorrow's Queensland: Green— protecting our lifestyle and environment; ClimateQ: towards a greener Queensland (2009); ClimateSmart 2050; SoE Queensland reporting; Framework and Implementation Plan for a National Cooperative Approach to Integrated Coastal Zone Management (2006); National Principles and Guidelines for Rangeland Management (1999); International World Heritage Convention (1972, 1975); Rio Declaration on Environment and Development (1992); United Nations Framework Convention on Climate Change (UNFCCC, 1992); Kyoto Protocol to the UNFCCC (1998); Noosa Biosphere Reserve (2007) |

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