Climate change impacts on the terrestrial biodiversity and carbon stocks of Oceania

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We review the threats from anthropogenic climate change to the terrestrial biodiversity of Oceania, and quantify decline in carbon stocks. Oceania's rich terrestrial biodiversity is facing unprecedented threats through the interaction of pervasive environmental threats (deforestation and degradation; introduced and invasive species; fragmentation) and the effects of anthropogenic climate change (sea level rise; altered rainfall patterns and increased fire frequency; temperature rises and increased storm severity, extreme weather events and abrupt system changes). All nine of Oceania's terrestrial biomes harbour ecosystems and habitat types that are highly vulnerable under climate change, posing an immense conservation challenge. Current policies and management practices are inadequate and the need for new legislation and economic mechanisms is clear, despite powerful interests committed to limiting progress. Mitigation can be achieved by increasing the effectiveness of the protected area network, by maintaining and effectively managing existing carbon stocks and biodiversity, and by reforestation to sequester atmospheric carbon. A price on carbon emissions may encourage less carbon-intensive energy use while simultaneously encouraging reforestation on long-cleared land, and reducing degradation of native forests. However, realizing these changes will require societal change, and depend on input and collaboration from multiple stakeholders to devise and engage in shared, responsible management.

Key words: Carbon sequestration; endemic species, refugia, deforestation and habitat destruction and degradation, reforestation, resilience, restoration, Pacific island nations, REDD.

INTRODUCTION

CLIMATE is the primary driver of composition, structure and function in terrestrial ecosystems (Schimper 1903; Good 1931), with biogeographic history, edaphic factors and disturbance also important (Bond *et al.* 2005; Mucina and Wardell-Johnson 2011). There is scientific consensus that serious anthropogenic climate change is taking place, and that early projections were highly conservative (Steffen *et al.* 2009; Climate Commission 2011). The major processes by which greenhouse gases influence Earth's climate are well accepted and long established (Crawford 1997). Vastly increased greenhouse gas emissions since the industrial revolution (~1750) are the overwhelming driver of recent global warming (Climate Commission 2011). Altered climates, associated with elevated atmospheric CO_2 concentration, have already been documented in many areas of the world (IPCC 2007) and the speed of anthropogenic climate change is likely to have considerable impacts on terrestrial biodiversity.

The biodiversity, as well as the geological and cultural diversity of Oceania are extraordinary (Keast and Miller 1996; Legra *et al.* 2008). Oceania includes Australia and New Guinea in the west, New Zealand and offshore islands in the south, and the Pacific Islands in Melanesia, Micronesia and Polynesia in the east (*sensu* Kingsford *et al.* 2009; Kingsford and Watson this volume), covering over 80 million km². Because many of Oceania's islands are small and far apart, the combined land area is about 11% of the total, with Australia (84%), New Zealand (3.0%), New Guinea (11%) and New Caledonia (0.21 %) accounting for over 8.4 million km² (98.6%). Australia, parts of New Guinea, New Zealand and New Caledonia are of Gondwanan origin and act as major evolutionary centres and source areas for Oceania's biodiversity (Whiffin and Kikkawa 1992; Keppel *et al.* 2009). Many Oceania islands are geologically recent and include volcanic, fertile islands, elevated limestone islands and, infertile reef atolls or coral cays (Mueller-Dombois and Fosberg 1998).

Based on a substantial increase in recent fossil fuel use, and continuing emissions from land-use change, Earth is on course to warm by 3.5°C by 2100 (IEA 2010), with a range of 1–5°C, depending on projected emission scenarios (IPCC 2007). Climate change models are restricted in scale and dependent on different emissions, and hence warming scenarios (see Yates *et al.* 2010). Furthermore, the nature of future local and regional climates remains uncertain (Araujo and Rahbek 2006). By 2100, climatic models forecast rising temperatures, an increase in extreme events and a rainfall decline for most of Oceania, with the exception of the tropics (CSIRO 2007). This will affect biota and carbon stocks of Oceania, already depleted by millennia of anthropogenic influences.

We conservatively examine climate change impacts on biodiversity and carbon stocks in Oceania for global warming of about 2°C, even though more than 3.5°C rise in global temperature by 2 100 is highly likely (see

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Meinshausen *et al.* 2009; IEA 2010; Climate Commission 2011). Warming above 2°C will strain humanity's capacity to secure sufficient, tractable environments, producing a runaway greenhouse effect through feedback mechanisms (Cox *et al*. 2000). We take the optimistic position that humanity will substantially decarbonize energy requirements and the land-use footprint, to reduce atmospheric greenhouse gas levels and prevent increased impacts. We firstly highlight the importance of Oceania's terrestrial biodiversity and carbon stocks. We then examine direct climate change impacts and indirect interactions with pervasive environmental threats, on biodiversity and carbon flux in terrestrial ecosystems in the nine. Finally we evaluate policy and management approaches to limit biodiversity and carbon losses, under changing climatic conditions.

Terrestrial biodiversity and carbon stocks in Oceania

Terrestrial biodiversity in Oceania is extremely rich and highly threatened on a global scale. As a result of island biogeographic processes, terrestrial biomes in Oceania have high species diversity, lineage diversification and localized endemism (Steadman 2006; Keppel *et al*. 2009; Kier *et al*. 2009). The insular nature of much of the region has probably stabilized local climates, moderating effects of global climatic change (Barnett and Campbell 2010). Nevertheless, the biota and ecosystems are considerably depleted throughout much of Oceania. For example, 50% of the global decline in mammal species over the last 200 years is Australian (Mackey *et al*. 2008), and human settlement of the Pacific Islands has resulted in the extinction of 20% of the global bird fauna (Steadman 1995). The occurrence of six of the world's 34 global biodiversity hotspots (Polynesia and Micronesia, Melanesia, New Zealand, New Caledonia, Wallacea and south-western Australia (Myers *et al*. 2000; Mittermeier *et al*. 2005; Kingsford *et al*. 2009) highlights both the richness and the threats facing the biodiversity of Oceania.

Of the nine global biomes recognized for Oceania (CIESIN 2007) (see Fig 1), "Desert and Xeric Shrubland" (*Arid regions*) and "Tropical, subtropical grasslands, savannas, shrublands" (*Tropical savanna*) occupy more than 60% of terrestrial Oceania (39% and 24%, respectively), mostly within Australia (Table 1). "Mediterranean forest, woodlands, scrub" (*Mediterranean woodlands*), "Temperate grasslands, savannas, shrublands" (*Temperate grasslands*), and "Temperate broadleaf, mixed forest" (*Temperate forests*) all occupy significant areas (particularly of Australia). "Montane grasslands, shrublands" (*Alpine regions*), "Tropical, subtropical dry broadleaf forest" (*Tropical dryforest*), "Mangroves" (*Mangroves*) and "Tropical, subtropical moist broadleaf forest" (*Tropical rainforest*) occupy relatively small areas overall (Table 1), but constitute considerable cover in non-Australian Oceania (Olson *et al.* 2001).

Australia

Australia has all nine biomes (Olson *et al*. 2001) (Table 1, Fig. 1), representing unique environments that have evolved in isolation on an island continent with climatic extremes (Steffen *et al*. 2009). Australia's edaphic and climatic characteristics, particularly ancient and leached nutrient-poor soils, are reflected in specialized biotic adaptations. This is exemplified in the Mediterranean climate-region of south-western Australia (Hopper 2009; Mucina and Wardell-Johnson 2011) with over 7 300 (49% endemic) of Australia's approximately 25 000 vascular plant taxa (Hopper and Gioia

Table 1. Biomes, their land area in Australia and all Oceania; their rangeland component and carbon loss from broad-scale deforestation. Note that the edges of biomes are not distinct over distances of up to 100 km and therefore only major areas can be mapped; e.g., mangroves (Macnae 1965) and small, remnant patches of Tropical dryforest (Fensham 1996) occur in Australia. Methods and data sources are are given in Appendix 1.

Country and Biome	Area(km^2, 1000s)	% of Oceania	Area deforested (km^2, 1000s)	% deforested	C in biomass lost (Pg)	C in Bio mass lost (% of potential)	% area commercial rangeland
Australia (total)	7590	84	950	13	6.2(±1.7)	28(±7.9)	49
Arid regions	3510	39	20.0	0.571	0.0317	0.88	46
Mediterranean woodlands	721	8.0	312	43.2	1.11	50	17
Alpine regions	12.2	0.14	0.519	4.25	0.00531	3.9	0
Temperate grasslands	627	6.8	1789	28.5	1.61	39	57
Temperate forest	485	5.5	234	48.2	2.19	44	0.12
Tropical rainforest	33.2	0.38	8.98	27.0	0.166	26	3.1
Tropical savanna	2130	24	198	9.30	1.12	17	74
New Zealand (total)	279	3.1	154	71	3.1(±0.41)	75(±10)	3.7
New Guinea (total)	971	11	145	14	3.9(±2.6)	15(±10)	0
Hawaii (total)	16.72	0.19	8.28	58	0.12(±0.06)	55(±27)	31
Solomon Islands (total)	28.4	0.31	6.54	23	0.19(±0.070)	23	00
New Caledonia (total)	18.7	0.21	15.6	84	0.37(±0.21)	84	
Tropical rainforest	14.3	0.048	11.3	42	0.058	42	0
Tropical dryforest	4.35	0.16	4.31	90	0.064	90	0

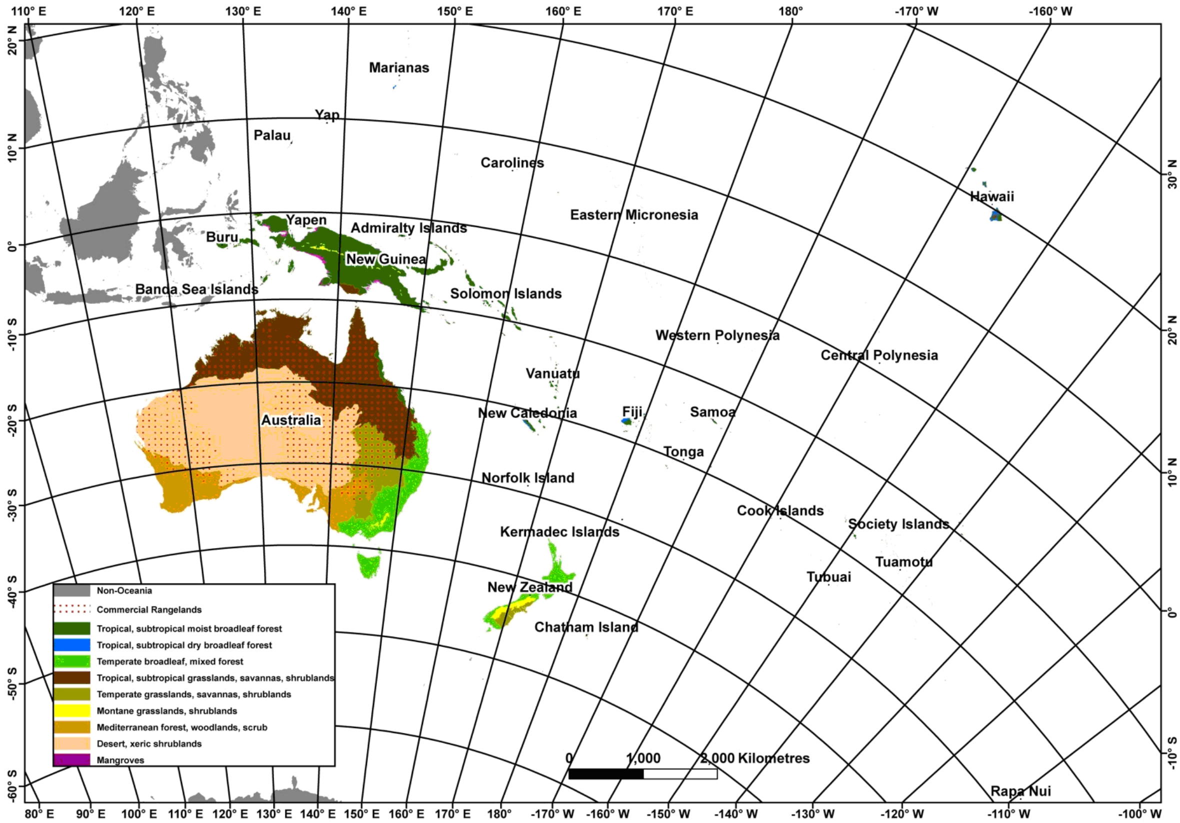

Fig. 1. Terrestrial biomes of Oceania. Map based on biome GIS data from Olson *et al.* (2001) and the commercial rangeland GIS layer from Dean *et al.* (2009). Projection = Lambert Conformal Conic, datum = Australian Geodetic Datum 1984, central meridian = 135° E, standard parallel 1= 18° S, standard parallel 2= 36° S, linear unit = metres. High resolution maps are available in Appendix 1.

2004). High rates of species' richness and endemism also occur in Australia's once more-expansive *Tropical rainforest* (Williams *et al.* 2003); in eastern Australia's *Temperate forests*; and across the vast *Tropical savanna*. The remaining relatively undisturbed south-eastern Australian temperate forests are among the world's most carbon dense forests (Keith *et al.* 2009). These complex, multi-aged and layered forests have fast growth rates, low decomposition rates, and occur in relatively cool and moist environments.

New Zealand

Three of the nine global biomes occur on the two main islands of New Zealand (*Alpine regions, Temperate forest* and *Temperate grasslands,* Fig. 1), where the terrestrial diversity is distinguished by the absence of native land mammals. With the exception of bats, mammals became extinct 26 – 35 million years ago, when most or all of New Zealand's landmass (268,000 km^2) was submerged (Pole 1994; McGlone 2005). This led to a remarkable diversity of endemic (70%) bird species, prior to the arrival of humans (Steadman 1995). Today 80% of the c. 2 300 vascular plant species and all species of bats, amphibians and reptiles are endemic (Allen and Lee 2006; McGlone *et al.* 2010). Prominent examples of evolutionary relicts with Gondwanan ancestry include three tuatara species (*Sphenodon* — Sphenodontidae), the only surviving genus of the reptile order Sphenodontia, restricted to small islands (Lutz 2006). The Gondwanan heritage is also particularly noticeable within the diverse invertebrate fauna with intensive lineage diversification, typified by a substantial proportion of large and flightless insects (Cranston 2010).

New Guinea

Due to its varied topography and complex evolutionary history (Pieters 1982; Heads 2001), New Guinea has the greatest diversity of habitats among islands of comparable size worldwide (Miller *et al.* 1994). New Guinea (including New Britain, New Ireland and other archipelagos) is dominated by *Tropical rainforest* (92% of land area, Fig. 1) but includes *Tropical savanna* (3.0%), *Alpine regions* (1.8%) and *Mangrove* biomes (3.1%). Cold tundra, bogs, montane rainforests, cloud forests and coastal ecosystems occur within these biomes (Johns 1982). New Guinea's terrestrial biota is highly diverse and typified by high endemism, despite its close evolutionary links with Australia (Miller *et al*. 1994). Covering less than 0.5% of the earth's land surface (971,200 km²), New Guinea harbours about 10% (more than 1,550 terrestrial species) of the world's vertebrate fauna and 7% (more than 20,000 species) of the world's described, vascular plant taxa (Allison 1996; Legra *et al*. 1996). Lowland terrestrial environments are dominated by plant taxa of Asian origin while montane and cloud forests include evolutionary, relict plant taxa that evolved in rich, fertile environments (Miller *et al.* 1994; Haberle 2007).

New Caledonia

New Caledonia (19 100 km²) includes two global biomes (*Tropical rainforest* and *Tropical dryforest*), associated with more than 3,270 species of vascular plants and the highest biodiversity per unit area of any region in the world (Kier *et al.* 2009). More than 75% of species and 14% of genera of native angiosperms and gymnosperms are endemic to New Caledonia (Jaffre 1993). Although of continental origin, the island was mostly or entirely submerged in the Oligocene, implying subsequent colonization and divergence of biodiversity (Grandcolas *et al.* 2008). Geographic isolation, dispersal, ecological diversification and hybridization produced the present biota (Bartish *et al*. 2005; Keppel *et al*. 2011a).

Other Pacific Islands

The other, smaller islands of Oceania harbour a diverse biota with numerous endemic species (Mueller-Dombois and Fosberg 1998; Keppel *et al.* 2009). For example, the combined East Melanesian Islands and Polynesia-Micronesia hotspots have more than 12 000 vascular plant, and more than 500 bird species (Mittermeier *et al*. 2004). Pacific islands are mostly of volcanic origin, harbouring most of the locally endemic biota, but also include uplifted limestone, and coral atoll islands (Mueller-Dombois and Fosberg 1998). The oldest and still emergent islands date to less than 40 million years ago (Yan and Kroenke 1993). Generally, greater age, size and isolation of islands equates to greater diversity and endemism (Keppel *et al.* 2010), reinforcing the importance of biogeography.

Impacts of climate change on Oceania's biodiversity and carbon stocks

Compared to the Northern Hemisphere, there is limited understanding of climate-change related trends derived from long-term datasets and phenological monitoring in Oceania (Hughes 2003). However, recent observations in Oceania (e.g. Welbergen *et al*. 2007; Chambers 2008; Pickering *et al*. 2008) reveal changing patterns consistent with a warming Earth. Past climate change shifted distributions of taxa and biomes (Stevenson and Hope 2005; Byrne 2008) and anthropogenic induced climate change will probably have similar consequences. Sea level rise and associated inundation, rainfall and hydrological change, temperature rises, more extreme weather events, and an increased likelihood of abrupt system changes (Leslie *et al.* 2007; Richards and Timmermann 2008; Climate Commission 2011) will considerably impact on terrestrial biodiversity and carbon stocks (Table 2), interacting with one another and existing pervasive threats (deforestation and degradation, introduced and invasive species, and fragmentation). All nine biomes include ecosystems, ecological communities and species threatened by <2°C of anthropogenic climate change (Table 2) and many species and ecological communities are already threatened, regardless of climate change.

Sea level rise

Sea-level is projected to rise (<20cm-2m) to 2100 and later, regardless of future greenhouse gas emissions (Solomon *et al*. 2009). Rising sea levels are already reducing the small landmass of islands, and further aggravating existing environmental stressors in coastal zones throughout Oceania, such as erosion and salinization of soil and groundwater (McLeod and Salm 2006). A sea-level rise of 1 m would cause high inundation of substantial land masses of islands during spring tides and intensified storms, rendering small, low-lying coral atoll islands (e.g., Tuvalu) uninhabitable (Barnett and Campbell 2010). Low-lying coastal vegetation will experience severe reduction in area (e.g., Legra *et al*. 2008), with impacts inversely related to shoreline gradient (Table 2). Coastal ecosystems (e.g., mangroves and salt marshes) will decrease in extent because there is little space for retreat. This will considerably reduce carbon stocks because mangroves are highly carbon-dense ecosystems (Kauffman *et al.* 2011). Mangroves are also adversely affected by high temperatures (> 35°C) and high salinity from reduced rainfall (McLeod and Salm 2006). Rising sea levels will exacerbate existing problems of high exploitation and urbanisation of habitat, with considerable uncertainty of cascading effects on biodiversity and carbon stocks (Mack 2008).

Temperature rises and increased storm severity

Average air temperature has increased at a rate of 0.17 per decade over the past three decades (Climate Commission 2011), noticeably affecting montane and alpine regions (Table 2). Ranges of tropical threatened species have contracted due to losses of bioclimatic envelop,

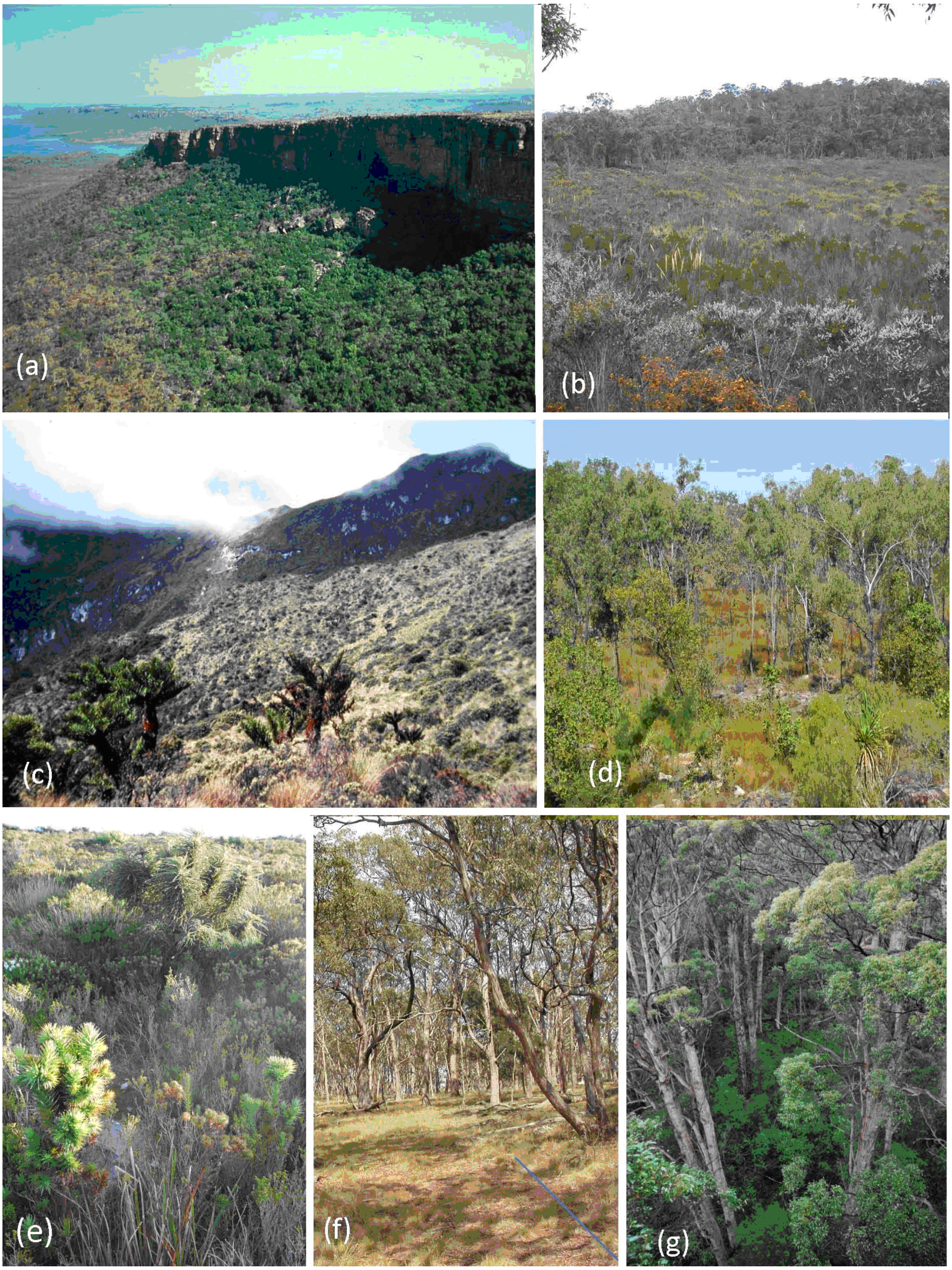

Fig. 2. Examples of Oceanian environments and habitats vulnerable to < 2 °C warming. These include areas harbouring ecological communities that are vulnerable to cattle and fire such as isolated tropical dry forest fragments in the Kimberley, Western Australia (1a); very restricted habitats such as moist, organic-rich swamp habitat in Mediterranean-climate south-western Australia (1b); upper slope habitat of montane environments such as alpine treefern and grassland communities in the Sarawaget Highlands, Papua New Guinea (1c); species-rich sandstone gully headwater habitat in the Tropical Savanna Biome, Northern Territory, Australia (1d); species-rich heathlands above 1000 m in the otherwise flat landscapes of south-western Australia (1e); high altitude grassy eucalypt forest in the Temperate Forest Biome of south-eastern Australia (1f); and high rainfall environments in areas of projected rainfall decline such as the tall open-forests of Mediterranean-climate south-western Australia (1g). Not all of these environments are rare at present. Photograph credits 1a, b, c, d, e, f. g: G. Wardell-Johnson.

Table 2. Examples of documented climate-related threats and their impacts on biodiversity in different regions of Oceania.

Climate – related threat	Biome	Biodiversity impact (ecosystem, species	Region most affected	Source
Sea level rise	Mangrove (also coastal and lowland ecosystems)	Ecosystem squeeze, loss of swamp forests	Pacific Islands, New Guinea, coastal areas of low shoreline gradient	Barnett and Campbell 2010; Laurance *et al*. 2011; Legra *et al*. 2008; Low 2011; McLeod and Salm 2006
Temperature rises and increased storm severity	Alpine regions	Encroachment of tree taxa into subalpine meadows, retreat of glaciers and snowlines, invasion of exotic species	Australia, New Zealand, New Guinea	Allen and Lee 2006; Hope 2008; McDougall *et al*. 2008; Mullan *et al*. 2008; Nicholls 2005
	Tropical rainforest (Montane)	Contraction of vertebrate species ranges, lifting of cloud base leading to loss of habitat and endemic species, invasion of exotic species	Montane areas of Australia's wet tropics and Pacific islands	Hilbert *et al*. 2001; Schneider and Moritz 1999; Shoo and Williams 2004; Williams *et al*. 1996; 2003
	All biomes	Invasion by exotic species, increased likelihood and increased frequency and intensity of fire, changed species composition, loss of carbon	Pacific islands, Wet tropics, savanna and arid regions of Australia	Curran *et al*. 2008; Elmqvist *et al*. 1994; Keppel *et al*. 2010; Webb *et al*. 2011
	Temperate forest	Bell miner associated dieback (BMAD)	Temperate south-eastern Australian forests	Wardell-Johnson and Lynch 2005; Wardell-Johnson *et al*. 2005 Saunders et al. 2011
	Mediterranean woodland	Reduction in species range and/or extinction	South-western Australian woodlands	
Altered rainfall patterns and increased fire frequency	Mediterranean woodland	New hydraulic regime, forest density loss, reduction in open-forest, loss of soil carbon, reduction in species ranges and/or extinctions	South-western Australian forests, woodlands and shrublands	Dean and Wardell-Johnson 2010; Judd *et al*. 2008; Petrone *et al*. 2010; Wallace *et al*. 2009; Gibson et al. 2010; Yates et al. 2010; Wardell-Johnson 2000
	Tropical savanna	Increased prevalence of exotic species, increased fire intensity and frequency, loss of carbon stocks	Australia	Butler and Fairfax 2003; D'Antonio *et al*. 2011; Jackson 2005; McDonald and McPherson 2011
	Tropical rainforest and Tropical dryforest	Loss of species and forest cover, increased forest mortality, changes in species composition, carbon loss	Australian wet tropics, New Guinea, New Caledonia, Pacific islands	Edwards and Krockenberger 2006; Fensham 1996; Gillespie *et al*. 2011; Keppel and Tuiwawa 2007; Mack 2009; Parmesan 2006;Shoo and Williams 2004; Shoo *et al*. 2011
	All biomes	Loss of soil organic carbon	Australia and elsewhere	Dean *et al*. 2009; McKeon *et al*. 1998

particularly temperature rise (Table 2). Tropical montane cloud forests are already restricted to the highest peaks in Australia and on many Pacific islands. However, they continue to support a considerable proportion of the endemic biodiversity (Meyer 2010), and provide refuges for taxa from highly degraded lowlands (Benning *et al*. 2002). The forecast lifting of the cloud base (Still *et al*. 1999; Pounds and Puschendorf 2004) is predicted to destroy much of the ecosystem and its endemic species (Williams *et al*. 1996; Loope and Giambelluca 1998; Table 2). Cloud forests and other highland rainforest types in northern Queensland are projected to halve with a temperature increase of 1°C and a small reduction in rainfall (Hilbert *et al*. 2001).

Most alpine environments in Oceania, apart from in New Zealand, are small and isolated (Wardle 1988; Mueller-Dombois and Fosberg 1998), with high levels of endemism (Smith 1982; Halloy and Mark 2003). Global warming poses a severe threat, especially for Australia, which has limited capacity for altitudinal migration because of limited elevational range (Smith 1982; Green and Osborne 1994). Increasing land temperatures in New Zealand have contributed to reduced frost frequency and alpine snow mass and retreating glaciers and snowlines (Mullan *et al*. 2008). While this will increase short-term carbon biomass, increasing fire frequency and intensity with elevated temperatures is likely to subsequently reduce soil organic carbon. Furthermore, high-intensity storm events, prolonged periods of drought and frequent high-intensity fires will assist invasion of exotic taxa into high-altitude habitat, reducing native vegetation (Pickering *et al.* 2008; Table 2). Increased sea temperatures also lead to extreme, high-intensity storms and cyclones (Leslie *et al.* 2007), increasing flood frequency and tree mortality (Elmqvist *et al*. 1994), leading to loss of carbon in all biomes. Selective mortality of species with vulnerable traits will change species' composition and structure of forests (Curran *et al*. 2008; Keppel *et al*. 2010; Webb *et al*. 2011).

Altered rainfall patterns and increased fire frequency

Australia's total annual rainfall has slightly increased, over the past century (Climate Commission 2011) but with substantial regional variation (CSIRO and BOM 2007), increasing in north-western Australia and with a substantial reduction along the central east coast and in south-eastern and south-western Australia (Suppiah *et al*. 2007). Reductions in winter rainfall in Mediterranean south-western Australia since the mid-1970s are attributed to increased greenhouse gases, natural climate variability, and land-use change (CSIRO 2007, 2009; Bates *et al*. 2008). Water availability is similarly likely to decline in south-western and south-eastern Australia. In Australia's wet tropics, severe intensification of the dry season and changes to seasonal precipitation patterns are forecast to alter ecosystem dynamics (Schloss *et al*. 1999), and impact severely on taxa restricted to moist environments (e.g., microhylid frogs, Table 2).

Reduced rainfall and long droughts will increase the frequency of fires, potentially affecting tropical dryforests, probably the most threatened biome in Oceania (Gillespie *et al*. 2011). Introduced grasses may also increase fire intensity and frequency in savanna and arid landscapes, increasing carbon emissions and biodiversity loss (D'Antonio *et al*. 2011; McDonald and McPherson 2011). Furthermore, modest changes in climate may disproportionately increase frequency and intensity of extreme events (e.g., high-intensity fires, extensive droughts, heat waves, high-intensity storms, Alexander *et al*. 2007; Climate Commission 2011), detrimentally affecting native biodiversity in Oceania (Low 2011). For example, extreme droughts and fires associated with El Niño events have produced considerable mortality in rainforests (Edwards and Krockenberger 2006; Mack 2008). This may change thresholds, potentially rapidly changing or reorganizing ecosystems to alternative states (Climate Commission 2011), with natural recovery unlikely (D'Antonio *et al*. 2011). For example, the rainfall decline in south-western Australia has shifted stream flow regimes from perennial to ephemeral, reduced water-tables and runoff coefficients, and led to the development of a new hydrological regime (Petrone *et al*. 2010). Widespread forest density loss in the lower rainfall eastern areas of the jarrah (*Eucalyptus marginata*) forest may be from altered hydrology (Wallace *et al*. 2009).

Pervasive threats to terrestrial biodiversity and carbon stocks

Existing drivers of biodiversity and carbon loss (Dean and Wardell-Johnson 2010) continue to impact terrestrial biomes in Oceania (Frazer 1997; Lees 2007; Lindenmayer 2007; Kingsford *et al.* 2009; Bryan *et al*. 2010; Woinarski 2010). Deforestation and degradation of terrestrial biomes for urban development, agriculture and logging remains the predominant threatening process in Oceania (Lindenmayer 2007; Kingsford *et al.* 2009; Woinarski 2010), and is the major cause of extinctions (Sodhi *et al*. 2009) and carbon loss. Thus more than half of all IUCN red-listed species in Pacific island nations are threatened by habitat loss (IUCN 2011), and Papua New Guinea lost almost 20% of its carbon stocks through deforestation and degradation from 1972-2002 (Bryan *et al*. 2010). The carbon stock loss in New Guinea is approximately 50% from logging and 50% from subsistence agriculture (Shearman and Bryan 2011), with area of each increasing. Logging in the Solomon Islands and Papua New Guinea is progressing at a rate that will leave little mature and undisturbed forest within a decade (Frazer 1997; Keppel 2006; Shearman and Bryan 2011).

Depletion and fragmentation by agriculture and logging in Australia and New Zealand over the past 200 years (Table 1) has sometimes led to novel ecosystem dynamics, including those involving the bell miner (*Manorina melanophrys*, Table 2). In New Zealand, ecosystem loss and fragmentation has occurred throughout the country (Hennessy *et al.* 2007), leading to the indirect loss of forest biomass in adjacent areas, through salinity and increased fire risk from introduced grasses. Deforestation continues in Australia and New Zealand, for agriculture, urbanization and mining, although rates of clearing native forests for agriculture and plantation establishment have substantially declined since 2000.

There are several other important and continuing threats to ecosystems in Oceania, including invasive species, which impact native biodiversity in all ecosystems (Lonsdale 1994; Craig *et al.* 2000; Olson *et al.* 2006). For example, there are many introduced species in New Zealand; at least 30 mammals, 34 birds, 2000 invertebrates, and 2200 plants (Allen and Lee 2006; Norton 2009). In addition invasive species may cause the collapse of native species and ecosystem processes in the Pacific islands (Walker and Vitousek 1991; Meyer and Florence 1996). Mining operations, while occupying relatively small areas overall, often occur in areas of high endemism, and can contribute substantially to habitat degradation (e.g., Paull *et al*. 2006, Fig. 2). Commercial livestock grazing occupies half of Australia's, and about 4% of New Zealand's area (Table 1), making it the largest land use in Oceania. The impacts on rangelands of deforestation, overgrazing, introduced fauna and flora, and altered fire regimes result in a net carbon emission (e.g., Walker and Steffen 1993; Radford *et al*. 2007) and decreased biodiversity (e.g., Williams and Price 2010).

Overall, many existing stressors interact to exacerbate climate-change impacts (Williams *et al.* 2008; Kier *et al.* 2009; Shoo *et al.* 2011). These include deforestation, degradation, habitat fragmentation, invasive weed species, introduced predators and the spread of pathogens (e.g. *Phytophthera cinnamomi* and related taxa, Wardell-Johnson and Nichols 1991; Shearer *et al*. 2004). Understanding the synergy between climate change scenarios and the threats posed by existing pervasive threats is urgently required for incorporation into new approaches to land management (e.g., Hobbs et al. 2009; Hopper 2009).

Management and policy responses under a changing climate

Current policy responses to increased greenhouse gas emissions vary with region and country, but are inadequate to slow or arrest global warming. There is a need to establish effective financial and legislative mechanisms to safeguard current carbon stocks, and re-sequester carbon, while also providing positive biodiversity benefits. Furthermore, protected areas need to be increased and off-reserve conservation improved, including increasing the area of reforestation and improving the management of existing carbon stocks. A price on carbon emissions throughout Oceania can redirect energy use to less polluting forms. New Zealand and recently Australia have shown leadership in reducing greenhouse gas emissions. Broad-scale logging and deforestation of natural forests in New Zealand declined with the introduction of an emissions trading scheme (NZ Ministry for the Environment 2011).

Protected areas, connectivity and refugia

Protected areas may fail to adequately conserve biodiversity with future projected changes in climate (Steffen *et al*. 2009) because they do not provide sufficient area for alterations of species' ranges and distributional patterns in terrestrial biomes. Furthermore, most forest types in Australia will experience future climatic conditions that are currently associated with other forest types (Hughes 2003). In many Pacific islands, the protected-area systems are not only inadequate and unrepresentative (Jaffré *et al*. 1998; Lees 2007; Shearman and Bryan 2011), but are also poorly enforced, resulting in considerable anthropogenic degradation (Shearman *et al*. 2008). Furthermore, conservation processes established in developed countries are often ineffective in Pacific islands because of different social, economic, political and cultural complexities (Hunnam 2002; Hviding 2006). This is exacerbated by large gaps in the knowledge of the distribution of biodiversity (James 2008; Kool *et al.* 2010).

Increasing connectivity along environmental gradients will enhance migration opportunities for many species (Hilbert *et al.* 2001; Heller and Zavaleta 2009; McGlone *et al.* 2010), and allow for changes to species' habitat ranges (Foran and Plody 2002). Hence increases to national reserve systems in Oceania need to focus on meta-population and source-sink population dynamics, species core habitat, migration corridors and stepping stones, refugial habitats

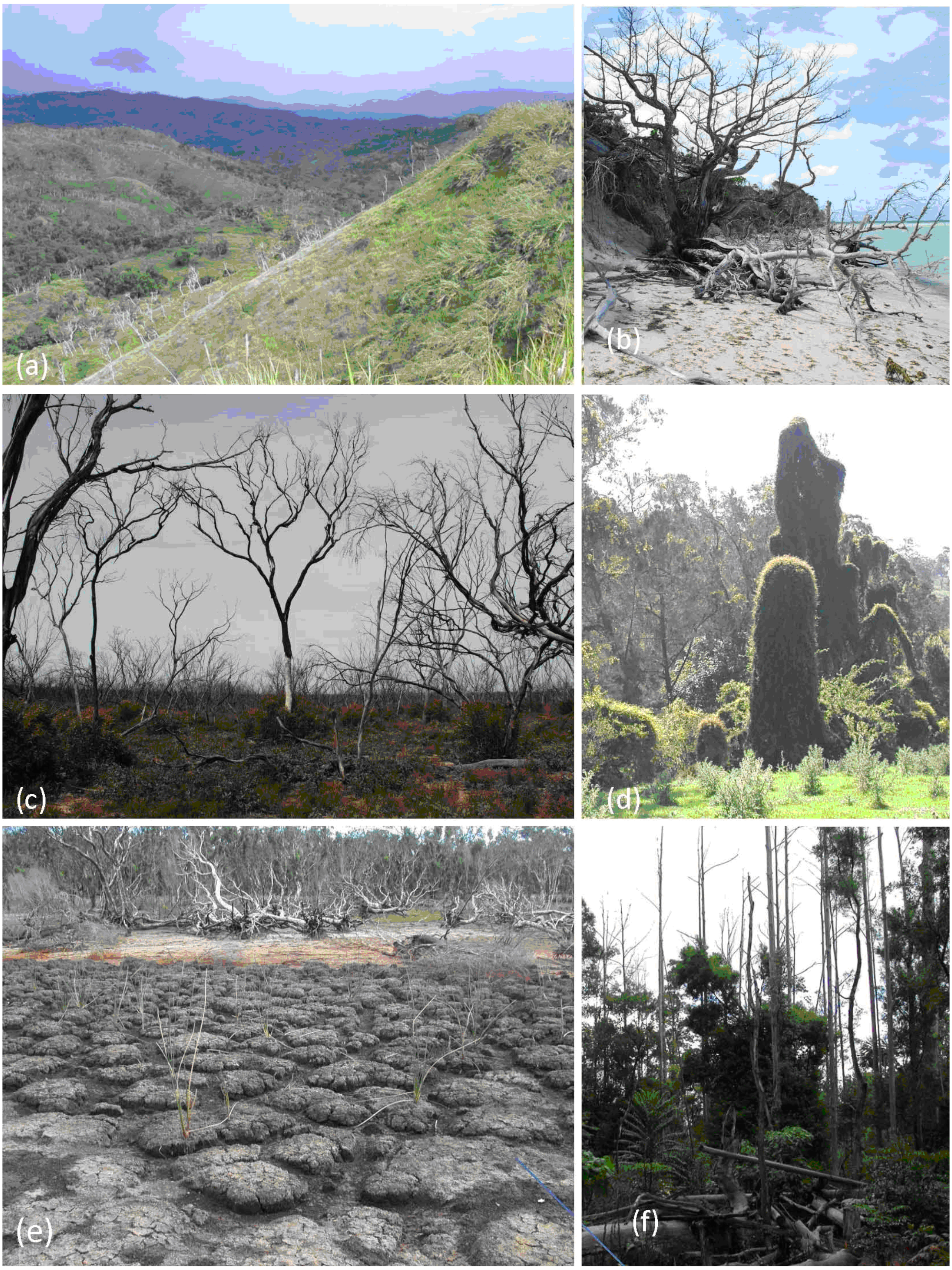

Fig. 3. Examples of interacting threatening processes that exacerbate the impacts of anthropogenic climate change. These include deforestation, weed invasion and frequent fire in former Tropical dryforest habitat of New Caledonia (2a); sea-level rise, increased wind intensity and storm surges on pacific Islands such as Ouvea, New Caledonia (2b); increasing fire frequency associated with higher extremes of wind and temperature such as in woodland habitat in the Great Western Woodlands, south-western Australia (2c); increasing impacts of weed invasion such as by cats-claw creeper (*Macfadyena unguis-cati*) in areas of disturbed Temperate forest in Queensland, Australia (2d); an increasing incidence of peat fires associated with drying wetlands in south-western Australia (2e); and the interacting impacts of insects, birds and disturbance leading to Bell-minor Associated Dieback or BMAD, in south-eastern Australian forests (2f). Photograph credits 2a, b, d, e, f. g: G. Wardell-Johnson: 2c: C. Dean.

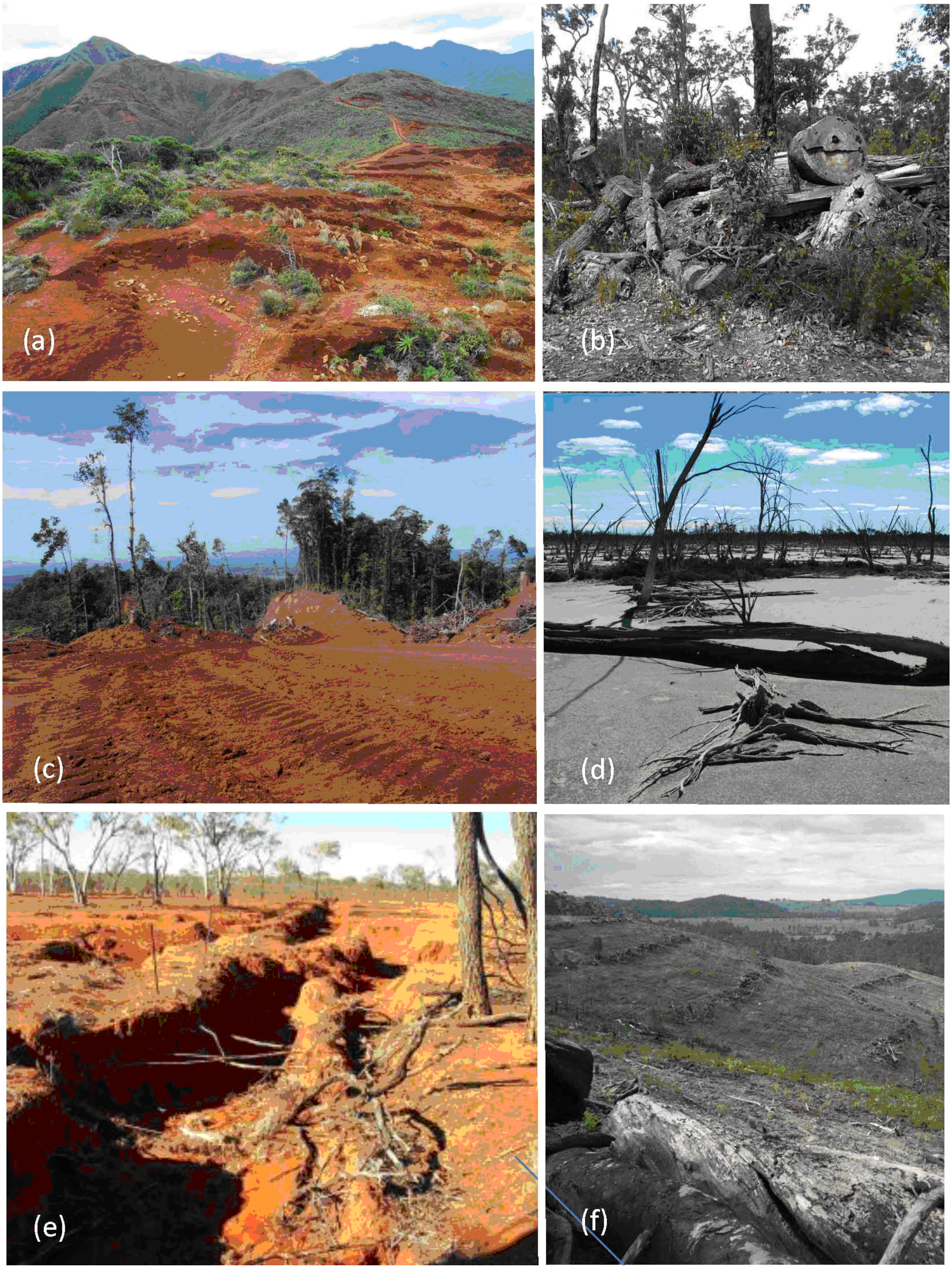

Fig. 4. Examples of land management activities that exacerbate the impacts of anthropogenic climate change. Disturbance of fragile, ancient soils leading to erosion, biodiversity loss and long-term carbon emissions such as in old ultramafic soils in New Caledonia (3a); Logging of mature native forests such as in jarrah (*Eucalyptus marginata*) forest in south-western Australia emits carbon and changes forest structure, leading to the establishment of a higher-water-demanding regrowth stand – exacerbating further drying of the landscape (3b); Logging in Pacific island nations such as in the Solomon Islands represents a significant carbon emission, a serious biodiversity loss, and predisposes terrestrial biota to climate-change-induced stresses (3c); Deforestation for agriculture in old-stable landscapes such as in south-western Australia has led to water table rises and concomitant landscape salinity, rendering restoration problematic (3d); Continued overgrazing of Australia's rangelands has led to widespread erosion and loss of soil organic carbon and concomitant biodiversity loss (3e); Conversion of old-growth forests to plantations such as in this site in Tasmania, Australia, provides continuing carbon emissions and biodiversity decline (3f). Photograph credits 3a: G. Wardell-Johnson, 3c: Patrick Pikacha, 3b, d, e, f: C. Dean.

and adaptation pathways (Parmesan 2006; Williams *et al*. 2010).

The maintenance of refugia minimises environmental stress in natural areas (Wardell-Johnson and Coates 1996; Keppel *et al*. 2011b). While the localized evolution of many narrow endemic species (Hopper and Gioia 2004; Williams *et al.* 2010) may limit the adaptive capacity of biodiversity, refugia and phenotypic plasticity in species may maintain vulnerable biodiversity under climate change (Horwitz *et al.* 2008; Keppel *et al.* 2011b). Examples of refugia include rock-boulder fields associated with sheltered and cooler climatic conditions in Australian tropical montane forests (Shoo *et al*. 2010; 2011), and granite outcrops and wetland ecosystems in south-western Australia (Horwitz *et al*. 2008). Old-growth, mature, and natural forests are also important refugial habitat because they are structurally diverse, harbour phylogenetic relicts (Wardell-Johnson and Coates 1996; Sander and Wardell-Johnson 2011), retain moisture, have relatively low water demands, have a very high carbon store (Dean and Roxburgh 2006; Dean and Wardell-Johnson 2010), and are stable thermodynamically (Keppel *et al*. 2011b).

Detailed knowledge of ecosystem functioning is required for landscape-level conservation under climate change. Understanding the buffering capacity of refugia is necessary for translocation and *ex-situ* strategies, and to address risks of extinction (Shoo *et al*. 2010; Keppel *et al*. 2011b). For this, fine-scale data are required to resolve the important interactions between climatic and landscape factors (Austin and Van Niel 2011). Addressing knowledge gaps in species' resource use and their ability to adapt to climate change is also crucial (Halloy and Mark 2003; Williams *et al.* 2008; Shoo *et al.* 2011). This requires an enhanced understanding of dispersal mechanisms and behaviour, flow-on effects on species interactions, genetic structure within and among species populations (Parmesan 2006), and the extent of phenotypic plasticity. On a broad level, research into the rules of ecosystem functioning is required to understand the dynamics of ecological cascades, and to forecast susceptibility to climate change (Heller and Zavaleta 2009).

Reforestation

Widespread reforestation provides an opportunity to reverse many of the threats associated with near-future climate change in terrestrial ecosystems (see Climate Commission 2011) but it needs to begin soon. Plantations on long-cleared land can increase biodiversity and carbon stocks (e.g., Kanowski *et al*. 2005) but may also depend on strategic land purchase, good land management, and corridor establishment. Many degraded natural ecosystems still retain important conservation value and carbon stocks (see Bishop *et al*. 2010), although others are so badly degraded that restoration or recovery is prohibitive (Hobbs *et al*. 2009). Most restoration projects in Oceania are currently publicly or volunteer funded because they have a relatively low economic return with no current economic value in emission reduction. Current "carbon offsets" are small in scope and effect, but economic value on carbon stocks may profoundly alter the scale of available land use options. Some in the private sector are already responding to changed societal signals (e.g., Saffitz 2010).

Management of carbon stocks

Mature forests have higher carbon stocks than regrowth forests and a more complex insulating structure (Dean and Wardell-Johnson 2010; Hatanaka *et al*. 2011). Continued logging of mature and primary forests increases carbon emissions, which are not offset by storage in wood products (e.g., Dean and Wardell-Johnson 2010; Sathre and O'Connor 2010). Furthermore, substantial amounts of course woody debris from forests are presently used as fuel to offset fossil fuel use (Brown *et al*. 2009) with greenhouse gases from this substitution remaining a net anthropogenic emission. Short commercial logging cycles in remnant forests have produced a mosaic of young stands, greatly reducing representation of mature and old-growth stands and carbon stocks (e.g. Calver and Wardell-Johnson 2004). Degradation of remnant forests also represents considerable carbon loss in temperate and tropical ecosystems (e.g., Roxburgh *et al*. 2006; Shearman and Bryan 2011).

Soil organic carbon is influenced by over-grazing, land rehabilitation, deforestation, regrowth, proliferation of woody cover (thickening), fire and weed interactions, land-use history and climate change (Dean *et al*. 2009). Australian rangelands remain a major emitter of greenhouse gases in Oceania (Dean et al. 2009), requiring a change in management, and improved understanding of the magnitude and importance of rangeland carbon. Avenues which reduce carbon emissions from rangelands, sequester carbon and enhance biodiversity include; reduced deforestation, protection and enhancement of soil organic carbon, reforestation, protection of riparian areas, and use of alternative protein sources such as kangaroos or *in vitro* production (Wilson and Edwards 2008; Dean *et al*. 2009; Taylor and Dean 2009). None of these recommendations are currently being enacted at large scales.

Numerous administrative processes, and intergovernmental and conservation bodies are attempting to reduce the rate of both legitimate and illegal deforestation, through carbon trading with conservation initiatives (e.g., REDD and REDD+ projects — Reduced Emissions from Deforestation and Forest Degradation), and increased village-level forest management (e.g., Seppälä *et al.* 2009). While national planning schemes have been developed for most countries in Oceania (e.g., Schuster and Butler 2001; NBSRTG 2009), implementation and co-ordination is often difficult. Increasing institutional capabilities may enable more effective conservation (Steffen *et al*. 2009), particularly in many Pacific island nations and New Guinea, where governments have little funding and few personnel (Lees and Siwatibau 2009; Shearman *et al*. 2009). There are different mechanisms to make REDD schemes more robust, including ensuring viability of forests under climate change (Fry 2008; Seppälä *et al.* 2009, Hoisington 2010).

Illegal logging undermines carbon conservation including REDD projects and may outweigh legal logging in area (Hoisington 2010). For example, Australian wood product currently includes 9% illegally sourced timber (Hoisington 2010). Banning imported illegal logged wood-products may provide a temporary solution but alternative sources of income will be required to reduce pressure on natural resources (Hunnam 2002; Ningal *et al*. 2008). Carbon trading may provide an avenue, with increasing regional interest (Bond 2006; Kintisch 2009). However, according to trends emerging from western Indonesia, deforestation for oil palm plantations is expected to increase in Oceania. Use of biofuels perversely reduces carbon stocks where mature forests are cleared for palm oil so that the greenhouse benefit of using biofuel rather than fossil fuel becomes questionable (Fargione *et al*. 2010).

Anthropogenic climate change provides the greatest challenge of our time because of the magnitude of its effect, its all-encompassing nature and its interactions across multiple sectors of society (e.g., Hsiang *et al.* 2011). Unfortunately, powerful interests remain committed to limiting the effectiveness of responses (see reviews by Hamilton 2010; Oreskes and Conway 2010). Therefore management and policy responses to climate change will need to encompass a broad consideration of biodiversity which involves social justice (Wardell-Johnson *et al.* 2011). This will require appropriate combinations of local, scientific and indigenous knowledge (see Wardell-Johnson 2007; WinklerPrins and Sander 2003) to improve existing management and policy responses, and develop and implement new approaches that match the magnitude of the challenge.

CONCLUSION

The terrestrial environments of Oceania are highly diverse, harbouring considerable biodiversity. Anthropogenic climate change, whether directly or through synergies with other environmental stressors, poses a serious threat to terrestrial biota and carbon stocks. Urgent, comprehensive and integrated strategies are required to improve conservation of biodiversity and carbon stocks, accounting for differences in culture, social structure and economics throughout Oceania. Many ecosystems are so fundamentally transformed that restoration becomes prohibitive. Even with a rapid turnaround, there will be substantial losses of biodiversity and carbon, with different ecosystems in the future. Humanity must act urgently to stabilize and reduce greenhouse gas emissions, and implement many other priority actions to meet the most important conservation challenge in Oceania's history. All actions and approaches will require significant resources or subsidies for climate-change adaptation — orders of magnitude greater than currently provided.

Past management has resulted in a continuing loss of biodiversity and carbon. Current management is inadequate and will limit our capacity to respond to the all-encompassing threat of anthropogenic climate change. Successes have been small and limited in scope, with a few patchy conservation outcomes, despite the dedication of many. In an environment where it is now widely reported that global warming can no longer be kept below a rise of at least a further 2°C (the edge of dangerous climate change), serious resourcing changes are needed to provide the best chance for biodiversity and society. Closure of the many knowledge gaps associated with terrestrial biodiversity and climate change will not stop unsustainable land use, and further attrition of biodiversity and carbon stocks. Use of combined scientific, local and Indigenous knowledge is essential. There is a clear disjunct between what is understood to be necessary scientifically, and what is potentially achievable socially, economically and politically. Unfortunately these impediments are compounded by powerful interests committed to limiting the effectiveness of urgent responses to climate change.

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APPENDIX 1

Table 3. Selected biome areas of larger islands in non-Australian Oceania. Nomenclature as per Table 1.

Biome	Area(km^2 , 1000s)	% of Oceania
Tropical rainforest	895	9.9
Tropical savanna	29	0.32
Mangroves	30	0.34
Alpine regions	17	0.19
New Guinea total	971	11
Temperate forest	179	2.0
Temperate grasslands	57.8	0.64
Tropical rainforest	0.0330	0.00037
Alpine regions	42.7	0.47
New Zealand total	279	3.1
Solomon Islands	28.4	0.31
Tropical rainforest	14.3	0.048
Tropical dryforest	4.35	0.16
New Caledonia total	18.7	0.21
Tropical rainforest	11.6	0.13
Tropical dryforest	6.92	0.077
Fiji total	18.5	0.21
Tropical rainforest	6.73	0.073
Tropical dryforest	6.62	0.075
Tropical savanna/tundra	3.37	0.037
Hawaii total	16.7	0.19
Alpine regions	60.2	0.667
Temperate grasslands	57.8	0.641
Temperate forest	179	1.980
Tropical dryforest	22.6	0.251
Tropical rainforest	1050	11.6
Tropical savanna	35.1	0.389
Mangroves	30.1	0.334
Oceania, non-Australia total	1440	16
Oceania total	9024	100

BIOME GIS AND CARBON CALCULATIONS

In biological systems carbon (C) forms the backbone of organic molecules, and carbon flux (mass movement per unit area per unit time) is inherently linked to that of other constituent elements (e.g. O, H, N, P and S). Carbon alone is considered here because it forms the predominant greenhouse gas, CO_2. In discussing carbon stocks and fluxes we consider the major organic carbon pools of botanical biomass (above and belowground), soil organic carbon (including micro-organisms), debris (e.g. coarse woody debris) and wood products. "Unit-area carbon stock", (Mg ha^{-1}) indicates the wood-products and emissions (associated with extraction, decomposition and long-term altered returns to soil organic carbon) attributed to the forest area of origin, when quantifying carbon fluxes. When considering soil organic carbon flux, sequestration and emission are slower than those of biomass, due to usually longer half-lives; however the soil organic pool is at least of comparable size to the carbon in biomass (MPIGA 2008). The course woody debris pool and wood-product pools are generally ~20% and ~5% (respectively) of the above ground carbon in [live] forest biomass (e.g., MBAC 2007; Woldendorp and Keenan 2005). The most recent carbon assessment of Oceania was from Khanna *et al*. (1999), where some mechanisms of carbon flux were discussed (e.g., logging, plantations and wood products). However, Khanna *et al*. (1999) did not include some areas of Oceania (e.g. West Papua), and the reported carbon stocks were unusually low, being without the benefit of more recent sampling, mapping and modelling.

Biome GIS data were from Olson *et al*. (2001), the World Wildlife Fund Terrestrial Ecosystems of the World Dataset, available at: http://www.worldwildlife.org/science/data/item6373.html.

Maps of Oceania and Australia (showing state outlines), are avaialbe online so they can be enlarged for viewing, on screen (Supplementary Figure 1 and Supplementary Figure 2: http://pcb.murdoch.edu.au/supp_material.html).

For Australia the deforestation GIS layer was from DEWR (2007) with 0.1 ° x0.1 °, i.e. ~1km pixels. Australian carbon in biomass was derived from potential aboveground biomass (Richards and Brack 2004; with 0.0025 × 0.0025°°, i.e. ~250 m pixels) plus root biomass from biome-specific root:shoot ratios from Mokany *et al*. (2006).

New Zealand carbon data were from Tate *et al*. (2001), areas of deforestation from Ewers *et al*. (2006) and commercial rangeland area was from Greer (2004).

New Guinea carbon data were from the nationwide, calibrated modelling of Brown *et al*. (2001) and deforestation area was from the 1970—2002 remote sensing work of Shearman *et al*. (2008).

For the Solomon Islands and New Caledonia, carbon values for Tropical rainforest were based on diameter at breast height (1.3m, DBH) measurements from thirteen 1 hectare plots of primary forest measured in the work of Keppel *et al*. (2011) in Papua New Guinea, Samoa, Fiji and the Solomon Islands. DBH values ranged from 0.1 m to 1.48 m. Individual tree aboveground biomass was calculated using formula for moist forest (2000–4000 mm yr^{-1} precipitation) from Table 4.A.1 in Schlamadinger *et al*. (2003) and below ground biomass was calculated from the formula for generic Tropical rainforest from Table 4.A.4 in Schlamadinger *et al*. (2003). From these allometrics the mean root:shoot ratio was 0.165 — in agreement with other values for tropical rainforest. Mean carbon in biomass for the thirteen plots was 276(60) Mg ha^{-1}. Carbon per tree was checked by regression against values derived using a second allometric — the allometric reported as the "best" overall fit in Chave *et al*. (2005) using DBH and basic density: the difference in mean carbon density for all thirteen plots, compared with that calculated from Schlamadinger *et al*. (2003), was only -1.7%, being slightly higher than Schlamadinger *et al*. (2003) for trees of low DBH, and slightly higher for trees of high DBH.

For the Tropical Dryforests of New Caledonia and Hawaii, the carbon density used was the global average for tropical dry forests from Brown (1997) plus a below-ground portion from the root:shoot ratio given in Mokany *et al*. (2006) for Tropical dryforest: totalling 122.8 Mg ha^{-1}.

For the Tropical rainforest in Hawaii, the carbon density used was the mean value from a site-specific Hawaiian study in Asner *et al*. (2009), plus a belowground portion from the root:shoot ratio given in Mokany *et al*. (2006) for Tropical rainforest: totalling 211.0 Mg ha^{-1}.

For the Tropical savanna in Hawaii, the carbon density used was that from Richards and Brack (2004) for the Cape York IBRA (International Biogeographic Region of Australia) of 124 443 km^2, on the northern-most tip of Australia, plus a below-ground portion from the root:shoot ratio given in Mokany *et al*. (2006) for Tropical savanna: totalling 81(47) Mg ha^{-1}.

Areas of original forest for each biome for Hawaii were determined from the biome GIS data from Olson *et al*. (2001), with subtraction of areas that had never been forested, such as glaciers, beaches, bare rock etc. The latter areas were derived from the land use/land cover GIS layer from Hawaii Government (2011), which was based on 1997 data. The subtraction indicated that ~85% of Hawaii had vegetation cover prior to human arrival. The rangeland areas for Hawaii were also derived from the land use/land cover data set.

Percentages of deforestation for Hawaiian Tropical Rainforest and Tropical dryforest were from Bruegmann (1996). Area deforested of Hawaiian Tropical Savanna was determined by overlaying anthropogenic land covers from the land use/land cover layer on the Tropical savanna Biome.

Areas of deforestation for the Solomon Islands were from FAO (2010) and for New Caledonia were from Petit and Prudent (2008).

Throughout, biomass was assumed to be 50 wt% carbon. Values for carbon and area were rounded only after summation, for display purposes.

Areas deforested and regrown to plantation or orchards etc were not tallied as forest, because the carbon stock (including wood products) is generally much lower than for the primary forest, unless fertilizer or other management is used, in which case some input is derived externally to the system, and therefore the carbon accounting must cover that area too. Similarly the carbon stock of infrastructure, where it replaced forest, was not included. Secondary forests were counted as forests and not as deforestation, and their change in carbon comes under the heading of forest degradation, which was discussed in the main text.

APPENDIX 1 REFERENCES

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