

Climate change and forestry in the Asia-Pacific region

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Executive Summary

As a result of the potential impacts of climate change, some forests or forest communities within the Asia-Pacific region may disappear entirely, while others may experience drastic reductions in size. China's boreal forest is predicted to lose 70% of its area and tropical montane cloud forests may be completely lost from the region.

Due to the diversity of the Asia-Pacific region's forests and other ecosystems, there are numerous ecotones or boundaries between forests and other systems across the landscape. Climate change is expected to have drastic impacts at these community margins. However, not all the changes will be negative for forests. Forest migration to increased altitudes and latitudes means that total forest area in the Asia-Pacific region may actually increase. An increase in forest area will help reduce climate change caused by greenhouse gases (GHG) through carbon storage mechanisms. However if the existing forests are lost to other non-forest ecosystems, total forest cover may decline because there will be less viable area to inhabit with increasing altitude on mountains and towards the poles.

In general, any change in climatic variables (moisture, temperature, wind) will alter ecosystems or forests. The magnitude of the climatic change and its variability will largely dictate the changes in forests or ecosystems. Although global climate model outputs are often referred to as "predictions", each output really represents one possible outcome, or 'scenario' of the future. One of the most difficult things to predict is human behaviour, and therefore most climate change modelling exercises include different runs based on various emission scenarios. Whether global GHG levels, represented as 'carbon dioxide equivalents', increase or decrease over the next century, and the magnitude of that increase or decrease, depends on a multitude of political, social, and economic factors that are not easily predicted. In essence it depends on human behaviour and both political and personal will to change. The magnitude, extent and nature of atmospheric emissions, agriculture, energy, transport, human population growth and land-use or land-cover change will all affect the impact climate change has on forests of the AP, as much as forest resilience does.

Although climate change in the Asia-Pacific region presents many challenges for forest managers and users, it also presents opportunity. Forests store carbon efficiently and effectively, while providing additional goods and services – from recreation, to timber and non-timber forest products. While there are numerous potential impacts that climate change may have on forests of the Asia-Pacific region, the fate of the region's forests depends on our ability to reduce GHG emissions, mitigate anthropogenic climate forcing, and adapt our forest management practices. The treatment and design of managed forests to be more like natural forests will increase both carbon storage and resilience, and may provide the forests of the Asia-Pacific region the qualities they need to survive and flourish in an uncertain and changing climate.

1. Introduction

Climate change may be one of the most pressing issues facing humanity in the 21st century. Since the links were first made between rising global temperatures and anthropogenic emissions of greenhouse gases (GHG) in the 1980s, hundreds of thousands of academic papers have been published on the topic. In addition to research, policies have been crafted, and international collaborations forged. For instance, the Intergovernmental Panel on Climate Change (IPCC) was established as a joint endeavour between the United Nations Environment Programme (UNEP) and the World Meteorological Association (WMO), in order to lead the scientific assessment of global climate change. Formed in 1988 and endorsed by the United Nations (UN) General Assembly, the IPCC has since published a number of scientific and technical reports on climate change impacts, mitigation and adaptation measures.

According to the IPCC's 4th Scientific Assessment Report, the Asia-Pacific (AP) region may experience some of the most severe climate change impacts when compared with the global average. For the purposes of this paper, the Asia-Pacific region is defined as the area covered by the Asia Pacific Economic Commission (APEC), thus including countries bordering the eastern Pacific and countries within the Indian subcontinent. This region and its ecosystems are also particularly vulnerable to climate change impacts (Preston et al. 2006) because of the low capacity to adapt to the changes and to cope with environmental disasters. Some of the potential impacts include temperature increases, sea-level rise and alterations in precipitation regimes (Preston et al. 2006; USAID 2010; Vickers et al. 2010). The AP is also unique in that it encompasses both developed and developing nations, and it contains virtually every forest type on Earth, including tropical, sub-tropical, cold and temperate rainforests, boreal forests, montane forests, mangrove forests, temperate mixed forests and coniferous forests. Furthermore, many of the region's inhabitants rely on forests and other natural systems to provide economic, ecological and cultural resources.

Forests are particularly sensitive to climate change. Each tree species has specific temperature, moisture and nutrient ranges within which it grows and reproduces. Even individual trees or understory plants within a species may have specific habitat requirements. Shifts in these regimes, or in a plant's niche, can influence numerous stand- and population-level functions. Climate change also works in concert with other stressors such as air pollutants to further affect forests and alter their functions. For those (human) communities across the AP region that rely on forests and their services, climate change means uncertainty and challenges. Although forests face many threats from climate change, they also present many opportunities. Forests store carbon, balance water and nutrient cycles, filter the air, and provide both timber and non-timber forest products. While it is imperative that we mitigate and reduce the impacts of climate change, we must also adapt our practices and traditions to accommodate climate uncertainty and changes to the forests on which we depend, while exploiting the opportunities that forests provide.

This review focuses on existing research related to forests and climate change in the AP region. In addition to a thorough analysis of potential impacts to forests, we present tools and practices

by which forest managers and users throughout the AP region can learn how to mitigate and manage their forest systems in light of uncertainties and challenges.

2. Changes to forest structure and function

Climate is the most important determinant of plant community composition (Belyazid et al. 2011). Factors such as temperature and moisture influence not only forest plant communities (Lloret et al. 2009), but also nutrient dynamics and soil processes within a forest stand (Belyazid et al. 2011). The following examines existing knowledge on forest processes and their climatic drivers in relation to a changing climate in the AP.

2.1 Temperature and moisture changes

The primary direct impact of a changing climate is an increase in temperature. This factor has led to the catch phrase “global warming” even though it simplifies and even misrepresents the science behind what is actually changing. The AP region will not necessarily get warmer all the time and across the whole region. However, a predicted annual average temperature increase across the AP of 0.5–2°C by 2030 and 1–7°C by 2070 (Preston et al. 2006) will affect the ability of species, individuals, communities, or ecosystems to survive within their current habitat, particularly within habitat edges or regions of furthest extent where climatic shifts are more likely to mean changes in community composition and structure. Furthermore, global circulation models predict that areas of high latitude and high altitude will experience greater warming than the global or regional averages (IPCC 2001a).

With this projected temperature increase comes an increase in weather extremes such as drought and aridity in dry seasons, enhanced precipitation in wet and monsoon seasons, tornadoes, etc. (Preston et al. 2006; Meehl et al. 2007). An increase in temperature will lead to changes in weather patterns, which are controlled by temperature and pressure gradients. Therefore temperature change will indirectly affect moisture regimes. Increases in temperature also lead to increased rates of evapotranspiration, and earlier snowmelt (Chmura et al. 2011), which unless accompanied by increased rainfall will result in overall losses in moisture and potential drought conditions. The combined impacts of changes in temperature and moisture on forest species in the AP are far-reaching, complex and largely unknown. Furthermore the vulnerability of forests to climate change in the AP exhibits a high degree of spatial variability (Preston et al. 2006).

Parts of the AP are already experiencing increased drought frequency, such as coastal Pakistan, the east coasts of India and Indonesia (IPCC 2001b), parts of eastern Australia, Burma, eastern Russian Federation (Walther et al. 2002), Nepal (Rijal 2008), and western North America (Michaelian et al. 2011), particularly Hawai'i and California (Walther et al. 2002). In addition, future drought conditions have been predicted for eastern and central Australia (Mpelasoka et al. 2008), arid and semi-arid Asia (Preston et al. 2006), all the United States mainland (Dale et al. 2001), and with increased risk across the planet's mid-latitudes (Meehl et al. 2007). Despite existing and predicted areas of moisture deficit, other parts of the AP will see increases in precipitation driven by climate change. For instance on a global scale, climate change is expected to increase the intensity of tropical cyclones (Preston et al. 2006; Meehl et al. 2007) and other

precipitation events. The coasts of China, Bangladesh and the Philippines (IPCC 2001b) and much of central Asia (IPCC 2007a) have already experienced increased rainfall, and south Asia is expected to experience further increases in summer (monsoon season) rainfall, while springtime rainfall is expected to increase in northern and central China and the Pacific islands (Preston et al. 2006). Different models predict somewhat different outcomes (IPCC 2007a); however, Meehl et al. (2007) predicted increased precipitation in the tropics near the equator and at high northern latitudes. The global average of these model predictions was both increased precipitation and an increased number of ‘dry days’, implying an increase in the occurrence of climatic extremes.

2.1.1 Species and range shifts

A warming climate is expected to cause shifts in species ranges, niches, or ‘climatic envelopes’. Shifts in the distribution of forests and other plant communities are expected to be the most extreme and rapid at ecotones – the boundaries between ecosystems – especially in semi-arid areas (Allen and Breshears 1998; Foster 2001). In general, in the absence of water-stress, forest ecotones are primarily controlled by temperature (Ohsawa 1995). Assuming overall warming trends, forest ranges or ‘climatic envelopes’ are expected to shift to higher altitudes or towards the poles (Walther et al. 2002). China in particular is expected to experience forest migration into previously unforested areas (Preston et al. 2006). For instance, broad-leaved forests in East China are expected to move northward by up to 3° latitude (Tsunekawa 1996). The AP’s northern reaches may see tundra ecosystems become encroached by shrubland and forest (Karlsson et al. 2011). For instance in British Columbia, Canada, projected climate warming scenarios predict that tree species will gain 100 km of northern range per decade (Hamann and Wang 2006).

Many climate-related range shifts have already occurred throughout the AP. The Russian Federation (Esper and Schweingruber 2004) and Siberia (Devi et al. 2008), for instance, have experienced northern treeline migration into tundra in recent years. In Alaska it has been estimated that over the last roughly 50 years 2% of unforested tundra area has changed to forest (Chapin et al. 2005); while 2600 km² of birch forests on Alaska’s Tanana flats have been replaced by floating mat fens (Hinzman et al. 2005), and the white and black spruce forests of the Mantasta Pass are being replaced by sedge meadows and bogs (Osterkamp et al. 2000). Similarly, in Queensland, Australia, rainforest has been expanding into the former reaches of eucalypt forest and grasslands (Hughes et al. 2003), although some of these changes may be more associated with changes in the anthropogenic fire regime than with climate change.

While some changes and shifts in ecosystems occur with latitude or across the landscape, others will depend on altitudinal variation. It is estimated that for every 10 km of latitudinal range shift we can expect a 10 m shift in range with altitude (Jump et al. 2009), and by the time global atmospheric CO₂ concentrations have doubled, tropical montane forests can expect upslope shifts in climate on the order of 200–300 m (Still et al. 1999). Others have estimated the shift of alpine species to higher latitudes at a rate of 11 m per decade, and to higher latitudes at a rate of 16.9 km per decade (Chen et al. 2011). Southern India’s tropical montane forests may see a transition from C3-dominated evergreen forests to C4-dominated grassland systems (Sukumar et al. 1995). Similarly, the Tibetan Plateau is expected to experience the conversion of desert steppe systems,

to those of forest and grassland (Preston et al. 2006). Alpine shifts in forests have already occurred in parts of the AP. Over the last century Siberia's forests have expanded by 20–60m in altitude into previous alpine tundra systems (Devi et al. 2008) due to shifts in climatic variables, and on Australia's Mt. Hotham *Eucalyptus pauciflora* has expanded to replace subalpine grasslands (Wearne and Morgan 2001). Similarly Southern California's dominant alpine plant species gained approximately 65 m in elevation from 1977–2007 (Kelly and Goulden 2008) and the forests of New Zealand began to display climate-induced shifts in altitude in the early 1990s (Wardle and Coleman 1992).



Figure 1. Species such as *Rhododendron veitchianum*, a primarily epiphytic species occurring at altitudes of 1200 – 2400 m in Myanmar, Laos and Thailand are vulnerable to climate change.

2.1.2 Drought and water stress

Drought and water stress induced by climate change in the AP (Allen et al. 2006; Preston et al. 2006) could also induce shifts in forest ecosystems (Allen et al 1998) and could produce significant changes in plant community composition. For instance drought-induced reductions in

seedling germination and increases in seedling mortality (Lloret et al. 2009) effectively reduce regeneration success (Chmura et al. 2011) that can lead to population decline. Drought may also directly impact adult trees by limiting growth or causing drought-related physiological injuries (Allen et al. 2010; Chmura et al. 2011), and drought has been found to reduce shade tolerance in some woody species (Battaglia et al. 2000, Sánchez-Gómez et al. 2006). Drought also alters gas exchange and vapour fluxes between forests, the atmosphere, and soils (Chmura et al. 2011), and could significantly reduce photosynthesis while increasing respiration, thereby reducing overall forest productivity (Goldammer and Price 1998). Drought is also positively associated with a number of plant diseases, both complementarily and synergistically (Desprez-Loustau 2006).

Climatic trends such as warming and drought are already increasing mortality and dieback in the Asia Pacific (Allen et al. 2010). Increases in temperature and related moisture stress are being blamed for increased tree mortality throughout the forests of western North America (van Mantgem et al 2009). For instance, in recent years California has experienced dieback in the old-growth mixed conifer forests of Yosemite National Park – dieback that has been linked to drought conditions and reduced spring snowpack volume (Guarín and Taylor 2005). Drought-induced direct forest mortality has already been witnessed across the AP, including water-stress related *Abies koreana* decline in South Korea (Lim et al. 2008); drought-induced aspen (*Populus tremuloides*) mortality across Western Canada’s provinces of Alberta and Saskatchewan (Hogg et al., 2008; Michaelian et al. 2011); birch (*Betula spp.*) forest decline in the southern Russian Federation (Ranson et al. 2011), and eucalypt mortality induced by water stress, documented in north-eastern Australia (Fensham et al. 2003; Fensham and Fairfax 2007). The largest changes in moisture balance associated with climate change are expected to occur in parts of the tropics, (Goldammer and Price 1998; Meehl et al. 2007; IPCC 2007a) and mid-latitudes (Meehl et al. 2007; IPCC 2007a).

Drought may also indirectly impact forest composition and increase adult mortality via increases in insect pests. Such climate-induced infestations have so far been observed throughout the AP, such as in east-central China in the late 1990s when a drought-induced red turpentine beetle (*Dendroctonus valens*) outbreak affected 0.5 million ha of Chinese Red Pine (*Pinus tabulaeformis*) forest (Wang et al. 2007a). The northwestern region of the Russian Federation has experienced similar drought-induced beetle outbreaks, resulting in spruce forest dieback (Kauhanen et al. 2008). Similarly, warmer temperatures and particularly mild winters have caused a mountain pine beetle (*Dendroctonus ponderosae*) infestation in Western Canada (and parts of the USA), which has been devastating the region’s pine forests since the 1990s (Smith et al. 2012), particularly in British Columbia. The infestation has now spread eastward over the Rocky Mountains into Alberta, with the range expansion being enabled by milder climatic conditions. Additional insect, particularly beetle, outbreaks in the AP that are being blamed on a warmer and/or drier climate include the spruce beetle (*Dendroctonus rufipennis*) in the southwest Yukon territory, Canada, and southwest Alaska, USA (Berg et al. 2006), and bark beetle outbreaks in California (Miller and McBride 1999). More insect outbreaks are expected throughout the AP’s forests in coming decades. Milder winters are also thought to contribute to the enhanced survival and proliferation of bacterial and fungal plant pathogens (Harvell et al. 2002).



Figure 2. Korean fir (*Abies koreana*) in some areas has been reported to be suffering from water stress in its native habitat (Lim et al. 2008).

2.1.3 Invasive species

In addition to increasing the occurrence of invasive insect outbreaks, invasive alien plant species may also become more troublesome in a changed climate, especially in cases where their climatic envelope is expanded or shifted (Smith et al. 2012). These invaders may compete with existing forest plants for light, water or nutrients; and like the introduced insects that feed on native species, these plants may have no natural predators. As such, the newly found habitat of an invader is considered its ‘fundamental niche’ being that it is potentially relieved of pressure from predation and competition, whereas its native range would be considered an organism’s ‘realised niche’ (Taylor et al. 2012). Numerous invasive species already occur throughout the Asia-Pacific and climate change may allow them to proliferate even further by adapting their habitat accordingly. For instance the invasive tree *Robinia pseudoacacia* L. of European origin is expected to expand its range in Australia, particularly in montane and low-lying forest areas, as the climate warms (Klabinbauer et al. 2010). One of the most insidious invasive plant species is the woody shrub *Lantana camara* L. that has already led to environmental and economic damage throughout tropical and subtropical Asia and Australia. While future climate scenarios have predicted that *L. camara*’s suitable range will contract on a global scale, parts of Australia and inland China are predicted to experience an expansion in the distribution of this nuisance species (Taylor et al. 2012) – known for reducing species diversity and soil fertility, and for altering both

soil properties and ecosystems processes (Day et al. 2003). Through these means, invasive species such as *L. camara* are able to create entirely novel disturbance regimes, alter existing disturbance regimes (Mack and D'Antonio 1998) and disrupt (Preston et al. 2006) or completely replace (Allen et al. 2010) existing forest ecosystems.

2.1.4 Wildfires

In addition to drought- and heat-initiated insect outbreaks, or climatic shifts that favour invasive species, it is predicted that climate change will alter the patterns and frequencies of other forest threats such as wildfires and diseases (Dale et al. 2001; Preston et al. 2006; Chmura et al. 2011). Wildfires for instance are expected to increase in parts of the tropics and in the boreal forests of Canada and Alaska (Preston et al. 2006). In western North America forest fires have already increased and the total area burned doubled in the last twenty years of the 20th century (Stocks et al. 2000). In British Columbia, Canada, the occurrence of forest fires is positively correlated with Pacific Ocean surface temperatures (Wang et al. 2010), as well as drought and record level terrestrial air temperatures (Hamann and Wang 2006). Increased warming and drying of forests increases susceptibility to forest fires, as do the large numbers of dead trees associated with drought or insect dieback. This susceptibility is further enhanced when fire suppression has allowed the build-up of fuel on the forest floor and in the forest understorey. Fire suppression and some forest management techniques lead to the development of particularly dense stands that are especially prone to forest fires (Allen et al. 2010). Modelling has predicted that 'fire weather severity' – weather that promotes the occurrence of fire ignition and propagation – will increase by 95% in south-central British Columbia, Canada, and that fire season length will increase by 30% over the same period (Nitschke and Innes 2008). Similarly, the severity of fire hazards is expected to increase by 30% in Alaska over the next century (National Assessment Synthesis Team 2000) and both the length of the fire season and the severity of wild fires are anticipated to increase across southern Canada and northern Russian Federation (Stocks et al. 1998). An increase in biomass, as anticipated from increased precipitation, nitrogen deposition, CO₂ fertilisation, and increased temperature, will contribute to larger and more frequent fires across western North America (National Assessment Synthesis Team 2000). The frequency of thunderstorms and associated lightning induced wildfires are also expected to increase in boreal regions (Juday et al. 2005; Krawchuk et al. 2009), the tropics (Goldammer and Price 1998; Cochrane 2003), and throughout the United States (Price and Rind 1994).

The frequency, intensity and duration of forest fires have already increased in parts of the AP with devastating fire events having occurred in British Columbia, Canada (Cohen 2011); Australia (Fensham et al. 2003; Cohen 2011); the Russian Federation (Sukhinin et al. 2011); South Korea (Park et al. 2008); and California, USA (Bytnerowicz et al. 2007) in the past decade. Furthermore, drought and forest fires are being blamed for a loss of forest biodiversity in Indonesia (Cruz et al. 2007). Although many forest ecosystems, such as those along the west coast of North America, were once dominated by fires (Chmura et al. 2011) that aided in nutrient cycling, seed germination and the creation of canopy gaps (Boerner 1982), increases in the intensity and duration of wild fires, means they may rapidly alter forest ecosystems and lead to complete shifts in community structure. In some places, such as in the AP's tropical areas, increases in forest fire activity may promote relatively fire-tolerant tree species over grass-dominated communities (Goldammer and Price 1998), thereby increasing total forest area.

Forest fires may also select for forest species that regenerate (resprout) from coppice or rhizomes (Goto et al. 1996; Fensham et al 2003) rather than those that require seed germination, since seed banks are often destroyed in forest fires.



Figure 3. Fire damage to mixed eucalypt forest in the Murrindindi Valley, Victoria, Australia. This photograph was taken about 9 months after the fire (in February 2009) occurred and re-sprouting on some trees is already evident.

2.1.5 Shift in seasons

Another anticipated climatic change that will impact forests is a shift in seasons. For instance the earlier onset of spring and the postponed arrival of autumn are expressed as earlier leaf unfurling and delayed senescence, creating a lengthened growing season (Walther et al. 2000; Chen et al. 2005). A lengthened growing season means increased biomass accumulation, carbon storage and potentially the propagation of previously foreign species. Phenological evidence of an earlier spring has been observed in the deciduous woody plants of western Canada (Beaubien and Freeland 2000; Beaubien and Hall-Beyer 2003), the western United States (Cayan et al. 2000), eastern (Chen et al. 2005) and temperate (Chen and Xu 2011) China; and as early spring flowering in Australia (Gallagher et al. 2009), and early winter flowering in the Japanese apricot

Prunus mume (Doi et al. 2007). However others suggest that in temperate systems, a warmer winter can mean inadequate chilling, resulting in delayed bud burst and adverse impacts on flowering or seed germination in species that rely on cold winters (Chmura et al. 2011).

Photosynthesis in plants actively converts and stores CO₂ as carbohydrate. However, photosynthesis and its related carbon storage are not constant throughout the growing season, across climates or between species. For instance, photosynthesis rates are reduced in the autumn compared with the spring and summer; however a later and warmer autumn, as expected with climate change, will increase the rate of soil microbial decomposition and therefore increase soil emissions of CO₂, faster than autumnal rates of photosynthesis can consume it. The consequence is increased forest emissions of CO₂ in the autumn under a changed climate (Hinzman et al. 2005). Delayed autumn colouration in temperate deciduous vegetation has been reported in eastern China (Chen et al. 2005), across temperate China (Chen and Xu 2011), and in Japan (Matsumoto et al. 2003). Conversely, Kozlov and Berlina (2002) describe a delayed spring and an advanced autumn on the Kola Peninsula, Russian Federation, leading to a decline in the length of the frost-/snow-free period, once again illustrating the nonlinear and heterogeneous nature of a changing climate. It has been suggested that changes in seasonal plant phenology may present a simple and effective indicator of the local effects of climate change on forest ecosystems (Kozlov and Berlina 2002; Beaubien and Hall-Beyer 2003; Chen et al. 2005; Gallagher et al. 2009).

Another seasonal impact of climate change is the timing of spring thaw. An earlier ground thaw in temperate and higher altitude climates further acts to lengthen the growing season. The associated increases in evapotranspiration flux may act to reduce water levels (Derome et al. 2006) in colder parts of the AP – such as the northern reaches of Canada, China and the Russian Federation. An extended frost-free period will also enhance bacterial decomposition and increase greenhouse gases released from northern soils (Derome et al. 2006), as well as increasing mineral weathering and the associated release of nutrients (Wrona et al. 2005). Moreover, widespread thaw is expected to occur throughout extensive permafrost regions by the end of this century (Meehl et al. 2007), which will help facilitate northward treeline migration and promote the development of thermokarst features (Karlsson et al. 2011), as has occurred in Alaska (Osterkamp et al. 2000). At the same time, some areas of forest may be disturbed, with melting permafrost resulting in the development of “drunken” forests as the trees lose their support (Bunnell and Kremsater 2012).



Figure 4. “Drunken forest” at Mayo, Yukon Territory, Canada, caused by melting permafrost. The unstable ground results in the trees losing their stability and falling over in different directions.

The timing of snowmelt and associated runoff may also be of importance to forests. Snowmelt provides crucial springtime moisture in temperate forest systems, and can further promote earlier plant growth and a lengthened growing season (Hinzman et al. 2005). Snow also provides a very reflective surface, with a high albedo. A lengthened snow-free period, brought on by an early spring melt, decreases surface albedo. The land surface consequently absorbs more solar radiation and gets warmer, thereby creating a positive warming feedback loop (Hinzman et al. 2005). Peterson et al. (2001) found that spring snowpack depth and the timing of snowmelt, were the best predictors of the growth of mountain hemlock (*Tsuga mertensiana*) in North America’s Pacific Northwest. Growth and productivity were strongly limited by a deep and late winter snowpack in both high elevation and southern ranges, in addition to the growth limitations imposed by winter precipitation and spring temperatures – two variables strongly related to snowpack formation and persistence. The growth of high elevation *T. mertensiana* forests was also strongly related with the occurrence of the Pacific Decadal Oscillation index (PDO), which shifts between warm and cool phases at 20–30 year cycles with cool phases being associated with increased winter snowpack, higher overall precipitation and cooler winter temperature, and reduced growth of higher elevation mountain hemlock. Conversely the growth of low elevation *T. mertensiana* forests in the Pacific Northwest, were positively influenced by the PDO cool phase. These findings not only illustrate the potential sensitivity of forests to changes in snow

cover and time timing of snowmelt, but show that a single species' response to seasonal changes can vary throughout a species' range, and how a changing climate may affect the growth and productivity of forest species. Furthermore, because climate change is expected to cause reduced spring snowpacks and potentially drier and warmer summers, the growth and productivity of mountain hemlock are expected to increase across its higher elevation northern ranges, while some of the lower elevation or southern populations may experience productivity and growth declines (Peterson et al. 2001).

2.1.6 Precipitation increases

Changes in the timing and intensity of snowpack melt will in some areas also be accompanied by increased rainfall – such as in northeast China (Preston et al. 2006). Excessive snowmelt and/or rainfall can cause flooding and water-logging, which can lead to enhanced erosion, tree upheaval and anoxic/anaerobic soil conditions. Anaerobic soil respiration releases methane rather than CO₂, and since methane has approximately twenty times the atmospheric warming potential of CO₂ (Brakkee et al. 2008), water-logging creates a (sometimes temporary) positive climate change feedback.

Rapid snowmelt, large snowpack volumes and increased rainfall intensity also increase the likelihood of landslides and slope instability (Dale et al. 2001). Both rainfall intensity and landslides are predicted to increase in frequency along the southwest coast of British Columbia over the next century (Jakob and Lambert 2009). Even in areas where total precipitation is expected to decline, precipitation intensity is expected to increase (Meehl et al. 2007). Field experiments in China have drawn positive relationships between rainfall intensity and soil erosion which can destabilise forest slopes and lead to an overall loss of topsoil, organic matter and nutrients from a watershed (Wang et al. 2011).

Anywhere that experiences increases in rainfall will also experience increases in landslide risk (Dale et al. 2001; Jakob and Lambert 2009), and climate change may therefore be associated with an elevated frequency of landslides (Santoso et al. 2008; USAID 2010) in areas of complex terrain. China has witnessed numerous flooding and landslide events in the past decade (Wang et al. 2011), and increases in the frequency of landslides have also been reported in South Korea (Park et al. 2008) and Indonesia (Santoso et al. 2008). Although landslides can be debilitating to forests at their origin and destination, forest vegetation helps prevent landslides by maintaining slope stability (Dale et al. 2001; Santoso et al. 2008).

Avalanches, sometimes associated with rapid warming or melting of alpine glaciers, will also increase in frequency, but at higher altitudes. Enhanced rainfall intensity and magnitude, in combination with increased snow and glacier melt (Meehl et al. 2007), will act to further increase the frequency and intensity of flood events throughout the AP (Preston et al. 2006). Glacial destabilisation and outburst floods, for instance, are expected throughout the eastern and central Himalayas (USAID 2010) and in the ranges of western North America (Moore et al. 2009). Landslides, avalanches and flooding are all forest disturbances that will increase in frequency in a changing climate. Flooding can expose and damage plant roots, erode and wash away soils, destabilise slopes, uproot trees, saturate soils, and kill forest vegetation. The magnitude, duration and frequency of flooding, along with the quality of floodwaters, determine

the nature and intensity of forest injury (Glenz et al. 2006). Most forest plants may quickly recuperate following a single flash flood that does not uproot vegetation, but repeated events will prevent that recovery. However, the flooding caused by sea level rise, as discussed in the following section, is more permanent and not as easily recovered from. In the northwest United States future flooding in forests is associated with changes in watershed hydrology brought on by milder winters (Hamlet and Lettenmaier 2007) rather than through an influx of excess moisture from rainfall or glacial melt.



Figure 5. Flood damage to riverine forest, Bellinger River, New South Wales, Australia.

Increased moisture from enhanced precipitation, snow and glacial and melt will also affect forests even if it is below the magnitude required to cause flooding. Increased moisture can ‘over-water’ forests, saturate soils and create water-logged anoxic conditions for roots, a situation few tree species can tolerate (Kozlowski 1997; Niinemets and Valladares 2006). These soil conditions cause roots to switch from respiration to alcoholic fermentation and unless the produced ethanol can be re-used as C in foliage, as seen in flood-tolerant trees, leaves will assimilate less CO₂ and growth will be reduced (Kreuzwieser et al. 2004). Increases in forest moisture can also increase forest susceptibility to bacterial and fungal diseases (Agrios 1997). For instance increased summertime precipitation in British Columbia, Western Canada, has been linked to a *Dothistroma* needle blight (*Mycosphaella pini*) epidemic in *Pinus contorta* forests (Woods et al. 2005).



Figure 6. Dothistroma needle blight damage to lodgepole pine (*Pinus contorta*). Bell II, British Columbia, Canada. The widespread damage caused by this disease in central British Columbia, Canada, is believed to be related to changing climatic conditions.

Increased moisture can however have some benefits for forests, such as increased success and rates of seedling germination (Khurana and Singh 2001) and increased tree growth (National Assessment Synthesis Team 2000; Chumara et al. 2011). More overall rainfall is predicted in north and southeast Asia (Stern 2007), the Solomon Islands (USAID 2010), and Southeast China (Preston et al. 2006), while in South and Southeast Asia, more rainfall is predicted to occur in the wet (monsoon) seasons (Stern 2007). However in some areas such as Indonesia, and temperate-

west arid- and semi-arid Asia, even wet season rainfall is expected to decline (Preston et al 2006). In general, dry areas (and times) may become drier (Stern 2007). Despite any potential impacts of flooding and saturation, the consequences of droughts induced by climate change are expected to be more important for forests (Chmura et al. 2011).

2.1.7 Cloud cover changes

Because of the uncertainty surrounding clouds and how they will respond to and influence a changing climate, they are discussed here only briefly. Cloud cover may increase due to increased evapotranspiration and therefore act to reflect incoming solar radiation back to space and act to cool Earth's surface, which reduces the impacts of climate change and reduces evapotranspiration, creating a negative feedback. However, increased cloud cover also traps (absorbs) heat near the surface and may assist in both warming and further evaporation and cloud formation – a positive feedback. Whether radiation is reflected or absorbed depends on cloud properties such as droplet size, and liquid water path, which are yet to be completely resolved or agreed upon by models (Quaas et al. 2009). However, increased or decreased cloud cover can affect forests beyond either warming or cooling feedback loops. Cloud cover reduces incoming solar radiation leading to growth reductions in forest plants (Niinemets and Valladares 2006). Research has shown that seedlings are generally more tolerant to shade than adults (i.e. can grow in shaded conditions) (Niinemets and Valladares 2006) and that this tolerance decreases with plant age in some species (Lusk 2004).

Cloud cover influences the amount of evapotranspiration and therefore water availability in forest soils. Cloud cover also influences stomatal opening and closing, and vapour losses from leaves or needles. Increased cloud cover, even when it is not associated with increased precipitation, may still provide some protection against drought and water stress in forest plants. Cloud cover and shade can also influence plant susceptibility to disease (Desprez-Loustau et al. 2006). However, because of the uncertainties involving cloud cover in global climate models, we do not know which areas may experience more or less cloud cover and when. Tropical cloud forests rely on cloud as the main source of moisture (Foster 2001) and as mentioned previously, cloud forests are particularly at risk from climate change.



Figure 7. Coast redwoods (*Sequoia sempervirens*), such as these at the Hunnewell - Donald Memorial Grove in California, USA, may be very vulnerable to climate change because of their reliance on coastal fogs to supply moisture.

2.1.8 Wind patterns and magnitude

Wind is the result of differential heating and cooling across the land surface and the resultant development of surface pressure gradients. At the surface, air moves from high (sinking) to low (rising) pressure regions. Because climate change threatens to modify the magnitude of incoming solar radiation, cloud cover, moisture fluxes and temperature, wind will also ultimately be affected. Wind is associated with seasonal storms such as the northeast winter monsoon and the southwest- and east-Asian summer monsoons that affect much of southern Asia (Preston et al. 2006). The interannual variability of these monsoons is strongly linked with the El Niño Southern Oscillation (Kawamura et al. 2004) – a meteorological phenomenon that results from unusually warm ocean temperatures in the eastern Pacific near the equator. El Niño contributes to periodic drought (Asia Development Bank 2005; Kumar et al. 2006) and sea-level changes (Preston et al 2006), and its frequency is expected to increase with climate change (Asia Development Bank 2005). If El Niño events become stronger and more frequent, tropical Asian Monsoons are expected to become increasingly variable (Adger et al. 2001) and tropical cyclones more intense (Asian Development Bank 2006). Mesoscale meteorological events such as El Niño are the source of small-scale local wind events; however because the grid-sizes of

most global climate change models (GCM) are larger than the scale of these events, changes in local winds cannot be readily predicted (Dale et al. 2001).

Overall increases in peak winds are likely to occur with climate change throughout the AP, and around the globe (Meehl et al. 2007). Windstorms are already the most common natural disaster reported in the Pacific Islands (Preston et al. 2006), and the return frequency of extreme wind events on the islands has been shortening in recent years, making high winds even more frequent (Asia Development Bank 2005). The number of high wind events has also been increasing over the past few decades across Northern Alaska and other arctic regions (Hinzman et al. 2005). High surface winds are associated with increased lightning strikes and forest fire intensity in northern boreal regions (Juday et al. 2005). Wind-speeds associated with tropical cyclones are also predicted to increase (Meehl et al. 2007), and there is evidence that the annual number of Category 4 and 5 hurricanes has been increasing globally over the past 30 years (Meehl et al. 2007) and that hurricanes may continue to increase in frequency (Foster 2001). Typhoons are the Pacific version of a hurricane, both being severe tropical cyclones with potentially disastrous consequences for forests. Furthermore, the frequency of thunderstorms capable of contributing to tornado formation, appears to have increased in recent decades (Dale et al. 2001), and positive relationships have been found between tornado frequency and mean monthly temperature in western Canada (Etkin et al. 1995).

Changes in the usual strength or direction of winds can alter forests in a number of ways. Strong winds are capable of damaging and uprooting forest trees and other vegetation over large areas. Individual tree and stand traits influence the type and magnitude of wind damage. For instance dead or tall trees (Veblen et al. 2001), and thin stands or those with shallow-rooted individuals (Dale et al. 2001), are the most susceptible to severe wind damage. Windstorms that damage forests have occurred in the AP and are expected to increase in frequency and magnitude. A severe windstorm in northern Colorado in the western United States in 1997, damaged over 10,000 ha of sub-alpine forest (Veblen et al. 2001), while in Hokkaido Japan, a windstorm that occurred in 1954 is credited for the shift from boreal- to mixed-forest (Osawa 1992; Ishizuka et al. 1997).

Strengthened winds of a lower intensity can also cause leaf or flower losses, negatively affecting photosynthesis and seed production, respectively. High winds are also responsible for the stunted and sculpted “krumholtz” trees found in arctic and alpine areas; these may become more common on exposed forest edges. Wind also transports seeds, and stronger winds can transport seeds further distances thereby increasing dispersion and the colonisation of new habitat. Strong winds may also increase the dispersion of infectious fungal spores (Desprez-Loustau et al. 2006) or insect pests, thereby increasing the spread of pathogens among and between forests. However, some insects such as leaf-mining moths of the *Eriocrania* genus, tend to aggregate together at higher windspeeds because they walk rather than fly between host sites (Kozlov 2002). Wind may also enhance the hybridisation of wind pollinated tree species such as spruce (*Picea* spp.) and pine (*Pinus* spp.); however because hybrids rarely inherit a full suite of defence traits, they are often more susceptible to insect attack (Juday et al. 2005). Increased pollen dispersion distances resulting from strong winds can also increase within species breeding and benefit the genetic diversity of forest stands.

Wind also influences vapour losses from forests and high winds may help to further the damaging effects of drought by increasing evapotranspirative losses. However increased wind-speeds over water bodies can increase the atmospheric moisture available for precipitation (Hubbard 1995) and help alleviate drought in forest areas near open water. The impact that increases in wind will have on the forests of the AP depends largely on the characteristics of a particular forest stand, its species, its moisture characteristics and the surrounding landscape and location (open water, exposed edges, etc.).



Figure 8. Hurricane damage to tropical rainforest, Atherton Tablelands, Queensland, Australia. The incidence of hurricanes may increase as a result of climate change, altering the disturbance dynamics of many forests.

2.2 Sea-level rise

The melting of alpine glaciers and sea-ice, along with the thermal expansion of sea-water, are contributing to a larger ocean volume and increases in sea-level (Meehl et al. 2007). Even a minute warming of the planet's average temperatures results in the melting of sea ice and subsequent sea level rise; such changes put all coastal (forest) communities of the AP at risk of inundation, erosion and complete loss (Preston et al. 2006; IPCC 2007a; USAID 2010). Asia is considered to be the most vulnerable region in the world to a 1 m rise in sea-level (USAID 2010). The low-lying deltas of China, India, Vietnam and Bangladesh, as well as the Pacific Islands are particularly at risk of inundation (Preston et al. 2006). Adger et al. (2001) name the

low-lying deltas of Ganges-Brahmaputra, Bangladesh; Irrawaddy, Myanmar; Choo Phraya, Thailand; and Mekong and Song Hong, Vietnam, to be among those most at risk in tropical Asia.

Sea level is anticipated to rise by around 40 cm along the south Asian coasts of Sri Lanka, India, Burma and Bangladesh (Cruz et al. 2007). Some models predict a sea-level rise of up to 85 cm in the coastal AP by the year 2100 (Preston et al. 2006), and global models generally show increases on the order of 18–59 cm by 2099. The mid-range central global estimate is a rate of 15 $\text{cm}\cdot\text{yr}^{-1}$ by the end of the century (Meehl et al. 2007). Coastal islands of the AP are already experiencing sea level increases of the order of 2.5–21.4 $\text{mm}\cdot\text{yr}^{-1}$ (Australian Bureau of Meteorology 2006). Conversely, the United States' National Oceanic and Atmospheric Association (NOAA 2012) reports slight sea-level declines due to vertical land motions along the west coast of North America – especially along the coasts of northern Alaska and the western Yukon Territory, thought to be due to the melting of mountain glaciers. Slight increases in sea level (with averages of up to 6 $\text{mm}\cdot\text{yr}^{-1}$) are reported further south along the coasts of California and Oregon, where some isolated sea-level declines are also reported. Slight declines in sea levels on the coasts of California, Oregon and Washington States are owed to the effects of tectonic subduction beneath the North American continent. Meanwhile, Hawai'i is experiencing increases in sea level of 1–6 $\text{mm}\cdot\text{yr}^{-1}$ due to relative sinking.

Sea level rise is an important consequence of climate change in the AP and will affect coastal forests throughout the region. Complete inundation and increased shoreline erosion (Preston et al. 2006) are two of the most important impacts on coastal forests. Despite some disagreement between models on the actual magnitude of sea-level rise, most agree that the rise will be of greater magnitude in northern and arctic areas, and at lower magnitude in the southern Pacific. However, a band of heightened sea-level rise is anticipated across the Indian Ocean (Meehl et al. 2007). Of the region's coastal forests, those of Bangladesh are thought to be most at risk, since large areas along the Bay of Bengal are at or below 1 m.a.s.l. (Preston et al. 2006).

In addition to sea-level rise from increased ocean volume, climate-change induced increases in the intensity or frequency of systems such as the El Niño – Southern Oscillation, can lead to temporary changes in sea-level in parts of the AP. For instance the Pacific Islands experience sea-level declines in El Niño years (Preston et al. 2006). Other local factors, such as crustal uplift or sinking, and variability in wind speed or air pressure, may also lead to more temporary variations in sea level. However, by the end of the century, all coastal areas are expected to have experienced permanent sea-level rise (Asia Development Bank 2006).

Coastal forests will also see an increase in soil salinity (Adger et al. 2001; USAID 2010), which may hamper forest reclamation, management and adaptation in coastal zones. Most trees are not salt tolerant and case studies have reported that salt water inundation from sea-level rise may inhibit trees' abilities to recover from other episodic disturbances related to climate change, such as storms and drought (Williams et al. 2003). Salinization of coastal soils effectively extends the forest impacts of sea-level rise further inland.



Figure 9. Forest killed by saltwater intrusion, Seward, Alaska. In this case, the saltwater incursion occurred as a result of an earthquake.

While inland systems that become inundated will increase in salinity, oceans themselves will experience reductions in salinity from the diluting effects of fresh glacial melt water. These chemical/density changes will alter ocean circulation and hence heat delivery and weather patterns inland; and will contribute further to sea-level fluctuation (Meehl et al. 2007; UNFCCC 2007). Sea-level rise also leads to coastal erosion (Hinzman et al. 2005), and Asia is particularly at risk. In addition to the complete loss of soil, forest habitat and land (Adger et al. 2001), this erosion increases carbon fluxes into the ocean itself (Hinzman et al. 2005), which has implications for the carbon cycle and atmospheric CO₂ enrichment.



Figure 10. Mangrove forests, such as this one near Cairns, Queensland (Australia) may be vulnerable to rising sea-levels. The very fine species zonation within such forests suggests that each species has precise environmental requirements.

2.3 CO₂ enrichment and sequestration

Increased atmospheric CO₂ concentrations have been found to cause changes in vegetation communities. The most obvious effect is a potential increase in forest growth and biomass accumulation. Despite CO₂ being a principle component in photosynthesis and therefore plant growth, other factors of climate change influence forest response to increased atmospheric CO₂. For instance photosynthesis also requires sunlight and water, and as we have seen from the previous discussion, climate change can both increase cloud cover and decrease moisture availability, thereby limiting photosynthesis. Therefore whether or not increases in atmospheric CO₂ will act to stimulate forest growth depends on a number of variables; and CO₂ enrichment may have a myriad of additional effects on forests.

Over the past two decades there have been a number of FACE (Free-Air Carbon Dioxide Enrichment) experiments examining the effects of elevated CO₂ concentrations on plants in their natural environments. In general, these experiments have been restricted to crop and grassland species, with the exception of loblolly pine (*Pinus taeda*) in North Carolina, trembling aspen (*Populus tremuloides*) in Wisconsin, poplar (*Populus* spp.) in Italy (McLeod and Long 1999; Ainsworth and Long 2005), Sweetgum (*Liquidambar styraciflua*) in Tennessee, and five

temperate forests species in Switzerland (Asshoff et al. 2006). However, no FACE experiments have been conducted on forest species native to the AP. Despite this, results from FACE experiments provide insight into the nature of plant response to elevated CO₂. For instance CO₂ enrichment without the alteration of other factors – such as moisture or cloud cover – has been found to increase forest tree productivity by 23% (median) (Norby et al. 2005).

Most enrichment studies have focused on younger trees, and Asshoff et al. (2006) found that as trees mature they tend to respond less to CO₂ enrichment. While studies show that mature trees have much lower rates of CO₂ uptake and growth under conditions of CO₂ enrichment (Körner et al. 2005; Asshoff et al. 2006), it is old primary forests that hold the largest carbon pools (Thompson et al. 2009). The increase in forest productivity brought about by CO₂ enrichment varies greatly between species (Karnosky et al. 2005; Körner et al. 2005; Asshoff et al. 2006) and climatic variables. Some studies have found no overall stimulation in growth or biomass storage after years of CO₂ enrichment (e.g. Körner et al. 2005). Other studies have found that growth response is limited by nutrient (Oren et al. 2001) and moisture availability (McCarthy et al. 2010), and yet others that nutrient-use efficiency is increased under elevated CO₂ – meaning that growth can be enhanced without a concomitant increase in nitrogen uptake (Norby et al. 1986). Whether or not CO₂ enrichment will increase tree growth and biomass production appears to depend upon not only the amount of CO₂, but also species, stand age, climate, soil fertility, moisture availability and a multitude of other factors.

Although an increase in photosynthesis from increased light absorption promoted by CO₂ is the main cause of increased productivity, CO₂ enrichment may also increase leaf size and therefore the photosynthetic surface area in some species (Karnosky et al. 2005). Elevated atmospheric CO₂ can also enhance productivity by increasing root growth and providing a larger surface area for the absorption of soil water and nutrients (Norby et al. 2005). Whether or not CO₂ enrichment enhances the growth of AP's forests may be an important factor influencing their future. Overall, trees appear to be the most photosynthetically responsive of all plant types to elevated CO₂ (Ainsworth and Long 2004).

Increases in atmospheric CO₂ can also have non-growth related influences on forest of the AP. For instance isotopic carbon analyses in southern India's tropical montane forest found that increasing moisture and CO₂ levels favour C3 grassland plants over C4 evergreen forest types (Sukumar et al. 1995). Many studies have shown that elevated CO₂ can also reduce stomatal conductance, rubisco and N-content, while increasing starches and sugars (Ainsworth and Long 2004). Elevated CO₂ can also decrease carboxylation rates and the maximum rates of electron transport (Medlyn et al. 1999), and can influence leaf phenology such as leaf duration or the timing of bud break (Asshoff et al. 2006).

Because plants convert atmospheric CO₂ into carbohydrate that is stored until either decomposition or combustion occur, forest trees and forest products such as timber, represent an effective mechanism of atmospheric carbon storage. There are a number of political and economic initiatives by which forests and their appropriate management can gain “carbon credits”. These credits can be traded, sold, or used to offset anthropogenic GHG emissions. While a description of these programmes throughout the AP is beyond the scope of this review, it is important to understand the role that forests of the AP may be able to play in these market

mechanisms. Körner (2004) and Asshoff et al. (2006) warn that the increased growth of trees does not necessarily mean increased carbon sequestration or an enhanced carbon pool, because such growth may simply hasten a tree's lifecycle, and forest turnover, thereby releasing stored carbon rather than providing a long-term solution to carbon sequestration. While this is true in a natural forest setting, managed forests may benefit from accelerated lifecycles that allow for shorter rotation times. In addition, wood and other forest products continue to store carbon as long as their carbohydrates remain intact (i.e. until they become decomposed or combusted).

Stored carbon pools are comparatively large in some parts of the AP, particularly in the below-ground biomass of the region's northern boreal forests (Banfield et al. 2002; Stolbovi and Stocks 2002) that occur in Russia, China, Canada and the United States (Alaska); and in the above-ground biomass of the region's tropical forests (Lal 2005). According to Keith et al. (2009), the world's largest total biomass carbon stores, up to $2844 \text{ tC}\cdot\text{ha}^{-1}$, reside in Australia's temperate moist *Eucalyptus regnans* forests. Additionally, the temperate forests of the United States' Pacific Northwest are said to contain a relatively high biomass carbon density with an upper limit of $568\text{--}794 \text{ tC}\cdot\text{ha}^{-1}$, and the *Agathis australis* forests of New Zealand are said to store up to $400\text{--}982 \text{ tC}\cdot\text{ha}^{-1}$ (Keith et al. 2009). Furthermore, the forests of Siberia have been reported to account for 20% of the global net primary production of all forests (Valentini et al. 2000), and the forests of China are reported to have become a carbon sink in the 1980s after being a net carbon source earlier in the century (Fang et al. 2001). Although mature systems such as these take up less carbon and produce less new biomass than their immature counter-parts, in terms of limiting the emissions that drive anthropogenic climate change, it is imperative that mature forest ecosystems be protected and maintained. Furthermore, the peatlands that lie beneath western Canada's boreal forest, actively store $19.4 \text{ g}\cdot\text{m}^{-2}\cdot\text{y}^{-1}$ ($194 \text{ tC}\cdot\text{ha}^{-1}$) of new atmospheric carbon each and every year (Vitt et al. 2000). In general, cool and moist forests hold the largest carbon stores (Keith et al. 2008), while belowground carbon stores decrease from the poles to the tropics (Lal 2005), and aboveground biomass increases towards the equator (Keith et al. 2008).



Figure 11. Mountain ash (*Eucalyptus regnans*) forest at Toorong Forest, Victoria, Australia. Forests such as this have very large amounts of carbon stored above-ground.

Forests' carbon storage, carbon uptake, productivity, and responses to CO₂ enrichment are complex and interdependent. Many other elements of climate change affect how forests take up or release atmospheric carbon. For instance, boreal zones that have recently been damaged by wildfires can temporarily become sources of CO₂ due to the switch from primarily autotrophic to heterotrophic respiration, and a lack of primary productivity, as found in Northeast China (Wang et al. 2001). Reversals from forest carbon sequestration to forest carbon emissions have also been estimated in Western Canada following climate change induced forest mortality from a mountain pine beetle (*Dendroctonus ponderosae*) outbreak that turned a small carbon sink, into a significant carbon source (Kurz et al. 2008). Any cause of large-scale forest mortality is expected to cause the same reversal, leading to a positive climate change feedback when that mortality is climate-induced. Climate-induced disease outbreaks, drought, flooding and wind damage can all cause similar sink-to-source transitions in forests that may persist for many years. Fires themselves release the carbon stored in combusted biomass as atmospheric CO₂, and Rapalee et al. (1998) reported that fire scars may be a net carbon source for up to 30 years following a burn.



Figure 12. Damage to lodgepole pine (*Pinus contorta*) forests caused by the mountain pine beetle (*Dendroctonus ponderosae*), Barkerville, British Columbia, Canada. Over 18 million ha of forest in British Columbia have been affected by the outbreak, which has now spread to the neighbouring province of Alberta.

2.4 Air pollution interactions

Air pollution and climate change are mutually relevant, especially in regards to the forests of the AP. Not only do the effects of both climate change and air pollution interact to have a combined impact on forests (Dale et al. 2001; Bytnerowicz et al. 2007; Karnosky et al. 2007), but the emission sources of atmospheric greenhouse gases thought to be responsible for climate change (such as CO₂, CH₄, N₂O, etc.), are also usually sources of other harmful air pollutants. Even the cleanest of fuels (e.g., natural gas or CH₄) burn in a nitrogen-dominated atmosphere such as that of the Earth to form oxides of nitrogen (NO_x). NO_x are responsible for the formation of tropospheric ozone (O₃ – the main component of photochemical smog), and contribute to both eutrophication and acid rain (Bytnerowicz 2007). NO_x, which includes NO (nitrogen oxide), NO₂ (nitrogen dioxide) and NO₃⁻ (nitrogen trioxide or nitrate), may also cause direct phytotoxic injury to forest plants (Bytnerowicz et al. 2007; Bytnerowicz et al. 1999).

There are reactions between greenhouse gases (GHG) and air pollutants in both the atmosphere and in the forest system. For instance O₃, itself a potent GHG, is able to indirectly influence the atmospheric lifetimes of other GHG such as methane (CH₄). CH₄, a GHG approximately 20

times more potent than CO₂ (Brakkee et al. 2008), reacts on a transboundary and global scale with NO_x to produce O₃ (Fiore et al. 2002; Bytnerowicz et al. 2007). O₃ is also directly toxic to many forest plants (Fowler et al. 1999) as are other pollutants such as SO₂ (Legge et al. 1998) and NO_x (Bytnerowicz et al. 1999). However unlike O₃ and other GHG, SO₂ and NO_x have negative, rather than positive, global warming potentials due to their role in the formation of atmospheric aerosols that reflect incoming solar radiation back to space (Brakkee et al. 2008).

Tropospheric O₃ at levels capable of causing direct plant injury have been measured throughout the AP: in British Columbia, Canada (Krzyzanowski et al. 2006); Washington (Brace and Peterson 1999) and California, USA (Miller and McBride 1999); China (Wang et al. 2007); India (Roy et al. 2009); Russia (Elansky 2009); and Australia (Hart et al. 2006). Ozone at these levels can affect forest health by increasing tree mortality (Karnosky et al. 2007), slowing growth (Wang et al. 1986; Karnosky 1996), reducing insect and disease resistance (Manning and Tiedemann 1995; Percy et al. 2002; Jones et al. 2004) and overall, making forests more vulnerable to climate change (Bytnerowicz et al. 2007). Fowler et al. (1999) predicted that by the year 2100 at least half of the world's forests will be exposed to O₃ at phytotoxic levels.

Air quality issues persist and are even worsening in much of the AP. In particular, developing nations in Asia are experiencing increases in air pollution emissions (Ohara et al. 2007). Increases in emissions, however, are not restricted to the developing world; parts of western Canada are also experiencing increases in the emission of compounds such as SO₂ and NO_x (Environment Canada 2010), released by a booming upstream oil and gas sector. Furthermore, despite recent and future reductions in O₃ precursor emissions in the USA, increased temperatures and changes in transboundary atmospheric transport associated with climate change, may actually increase the length and intensity of the (summer) O₃ season (Fiore et al. 2002). A reduction in O₃ surface removal provided by reduced forest canopy conductance in a warmer climate will further assist in increasing future tropospheric O₃ concentrations (Fowler et al. 1999).

The permanence and likelihood of both climate change and air pollution, and the interaction between these stressors – particularly in terms of O₃, makes it imperative that the two be approached in concert. For example, air pollutants such as tropospheric ozone may further increase the tree mortality associated with climate change (Karnosky et al. 2007; Allen et al. 2010), and air pollutants increase the vulnerability of forests to climate change impacts such as insects, disease, frost and drought (UNECE 2005; Bytnerowicz et al. 2007).

Particulate matter and aerosols have strong impacts on climate. For instance sulphate aerosols, formed in the atmosphere from SO₂ emissions, are very effective at reflecting incoming solar radiation and thus have a cooling effect on climate (Zhao et al. 2006; Brakkee et al. 2008). This cooling is so effective that some have even suggested (e.g., Wigley et al. 2006; Rasch et al. 2008) increasing atmospheric SO₂ emissions as a 'geoengineering' technique to help combat climate change. Atmospheric aerosols and particles also act as condensation nuclei to promote cloud formation (Rasch et al. 2008; Fowler et al. 2009), which again has been touted as a geoengineering strategy for climate change mitigation throughout the Pacific, due to the cooling effect of cloud cover (Jones et al. 2009). Due to the toxic nature of SO₂ emissions on both plants (Legge et al. 1998; Bytnerowicz et al. 2007) and animals, and the lung damaging capability of

fine particles such as dust and soot (Kampa and Castanas 2008; Yang and Omaye 2009), these modes of mitigation are not recommended. However in many parts of the AP, SO₂ and other emissions are increasing – particularly in Asia (Ohara et al. 2007; Fowler et al. 2009) and western Canada (Environment Canada 2010). These and additional aerosol forming emissions such as NO_x and NH₄ may not only help to counteract surface heating, but increase cloud cover. This increased cloud cover is particularly important in tropical Asia, which faces a loss of montane cloud forests as a result of global climate change (Foster 2001).



Figure 13. Upland evergreen rainforest, Java, Indonesia. Forests such as these are very dependent on cloud moisture; the incidence of cloud cover is changing in many parts of the tropics.

Strong winds, as associated with a changing climate, increase the transport distances of pollutants and increase dispersion and mixing to reduce concentrations at the surface. However short bursts of high winds can carry high concentrations of pollutants to more remote northern regions (Derome et al. 2006). Altered wind patterns change the fate of pollutants. For instance the particulate matter (PM) originating from the deserts of Asia (“Asian dust”) that is currently transported across the Pacific to western North America, is controlled by wind speed and precipitation. Rainfall controls the amount of dust emitted from dry regions in Asia, while wind determines where it may end up. Although transpacific dust transport has been decreasing in recent decades (Zhang et al. 2003), it is not known how climate change will ultimately affect this phenomenon.

In addition to being controlled by climatic variables, atmospheric dust is also, like the sulphate aerosol, a climate forcer that can alter the radiative properties and moisture characteristics of the atmosphere (Zhao et al. 2006). Atmospheric particles are so efficient at the scattering of sunlight that areas with polluted skies may receive 15–20% less solar radiation than their ‘clean’ counterparts (Pierrard 1969). A reduction in solar radiation would reduce rates of photosynthesis and growth. Mineral dust and other more inert atmospheric particles such as soot, can also clog plant stomata, interfere with pollen germination and increase pathogen susceptibility. They can also increase the likelihood of insect pests by eliminating or damaging the health of their predators. Depending on the chemical make-up of the dust or particles, further damage can occur. For instance very alkaline dusts can cause alkaline damage to leaves or needles, and soluble particles can be absorbed into leaf tissue where they may have toxic effects. Acidic particles can cause necrotic stippling (US Department of Health, Education and Welfare 1969) a symptom also seen with high ozone exposure (Karnosky et al. 2007). Both acidic and alkaline particles can contribute to the alteration of soil pH in areas where they deposit.

The atmospheric deposition of acidic particles, aerosols or gases (also known as “acid rain” but which can also occur as dry deposition) can lead to soil acidification which has purportedly led to the death of forests in some parts of the world – particularly in Europe and North America. If the deposition of acidic ions, such as SO_4^{2-} or NO_3^- formed via the atmospheric oxidation of SO_2 and NO_x , respectively, exceeds the base cations supplied by chemical weathering of parent material, then the soil begins to acidify. Alkaline soils are the most resistant. Some of the most susceptible soils occur in parts of Russia, Alaska and northern Canada (Derome et al. 2006). When carbonate (HCO_3^-) based alkaline rocks are weathered to neutralise acidity, CO_2 is emitted to the atmosphere. Conversely when silicate rocks are weathered in soils sensitive to acidity, CO_2 is taken up from the atmosphere and precipitated as carbonate (HCO_3^-) (Batjes and Bridges 1992). Carbonate rocks are estimated to store over 1000 times more CO_2 than is contained within the Earth’s atmosphere, oceans and biota combined (Walker 1994). Acidic deposition of sulphate and nitrate (wet or dry) also affects natural GHG emissions (CH_4 and N_2O) from forest soils (Brink et al. 2001). The higher temperatures, altered precipitation patterns and changes in primary productivity associated with climate change are generally expected to lower acidification risk, at least in Europe. However, in mountainous and arid areas, acidification risk is expected to increase with climate change (Posch 2000). It is not known how climate change may affect the acidification risk for forests of the AP, although enhanced forest growth will increase the uptake of both nitrogen and base cations, an acidifier and neutraliser, respectively. In addition, managed forests can provide long-term sinks for excess N.

N-enrichment by atmospheric deposition has long been known to result in changes in plant community composition (Berendse et al. 2001; Bobbink et al. 2002). However, changes in climate can also influence the release of nitrogen bound to the organic soil fraction during times of elevated atmospheric deposition (Belyazid et al. 2011). N mineralization (the conversion of organic N into plant usable forms) is largely controlled by soil temperature and moisture availability (Crohn 2004; Rixen et al. 2008). Excess soil nitrate becomes denitrified and as a result, N_2O (a GHG 310 times more potent than CO_2) emissions from soil increase (Berendse et al. 2001). Wet or dry acidic deposition can affect forest plants in a number of ways. For instance in addition to soil acidification and associated base cation depletion, the wet deposition of acidic ions inhibits pollen germination and pollen tube growth – particularly in broad-leaved forest tree

species (Cox 1983), which may further the impacts of climate change on seedling establishment and forest regeneration.

The timing of corresponding weather events is also important in terms of pollutant exposure. For instance snowfall in combination with arctic haze will increase the deposition of pollutants (Derome et al. 2006) to the forest canopy and soil surface. Additionally rain and snowfall events effectively ‘scrub’ pollutants from the atmosphere and deposit them on soil and leaf surfaces. Snowmelt can provide a soil influx of acidifying sulphate (SO_4^-) anions deposited to snowpack through the atmospheric deposition of pollutants (Derome et al. 2006). A change in the timing or magnitude of snow (or glacial) melt, as associated with climate change, will alter the receipt of acidifying sulphate by the soil. This occurs in the spring and could impact seed germination or seedling survival since young plants, particularly seedlings, are particularly sensitive to acidification. For instance, the germination and establishment of conifer seedlings is known to be pH dependent (Braekke 1976). The increased temperatures associated with climate change also initiate the opening of plant stomata, which increases the uptake of pollutants by forest plants and therefore increases forest sensitivity to pollutants (Bytnerowicz et al. 2007).

Trees, understory vegetation and forest soils are a sink for pollutants in addition to being a sink for CO_2 . The uptake of pollutants into leaf or needle tissue depends on a number of internal and external factors. For instance the presence of liquid water on a leaf (or soil) surface increases pollutant absorption, but both the chemistry of the leaf (or soil) surface water and the chemical composition of the atmosphere (including CO_2) strongly control canopy (or surface) resistance to pollutant deposition (Fowler et al. 2009). In addition, because ozone and other air pollutants inhibit photosynthesis in adequately exposed leaves, air pollution reduces a forest’s ability to store carbon (Felzer et al. 2005).

3. Forest Management

It is not sufficient to understand how ecosystems such as forests will change and adapt to a changing climate. Rather we must also understand how humans will cope and adapt to the resulting forest changes. The specific climate change adaptation mechanisms employed by humans will be influenced by the ecological, economic and social values that a particular forest ecosystem holds. However, due to the vast number of purposes that forests hold for people in the Asia Pacific – from religious and spiritual, to commercial timber and non-timber forest products – it is unrealistic to explore all potential adaptation mechanisms. Instead we focus on the commercial value of these forests, and in particular the main commercial species within each area of the AP.

3.1 Examples of important economic species

Chinese fir (*Cunninghamia lanceolata*) is native to China and is the country’s most important commercial species. Although this species has been cultivated in China for over 2000 years (Yeh et al. 1994), populations of *C. lanceolata* have declined over the past few hundred years due to over-harvesting and land-use changes such as the conversion of forest to farmland. Despite the

popularity of this species in managed forest rotations, Zhang et al. (2004) found that soil organic carbon and stand biomass decreased by 10% and 24% from first to second rotation, and by 15% and 40% from second to third rotations, on a per hectare basis, respectively. This loss of carbon storage has drastic implications if Chinese fir plantations are to be used as part of climate change mitigation strategies. Reductions in biomass as high as 68% have been observed in a 12-year old third rotation (Ma et al. 2002), and 72% in a 6-year-old second rotation of Chinese fir (Zhang et al. 2001). Reductions in soil nitrogen and increases in soil bulk density accompany these carbon and biomass reductions. Soil bulk density increases, attributed to erosion, and nutrient losses following traditional slash-and-burn site preparation, are thought to be the cause of these growth restrictions (Zhang et al. 2004). Therefore, forest management and site preparation practices have strong implications for carbon storage and climate-change mitigation in China's managed *C. lanceolata* forests. Improper management practices constitute human-induced forest degradation, which can lead to carbon imbalances and increased CO₂ emissions. From the viewpoint of the United Nations Framework Convention on Climate Change (UNFCCC 2012), these imbalances must be accounted for in climate change mitigation strategies, along with emissions arising from other forest management or land conversion activities (e.g., from natural to planted forests).



Figure 14. Chinese fir (*Cunninghamia lanceolata*) plantation in Fujian, China. Second and third rotations of Chinese fir often show productivity declines, a factor that may be related to the allelopathic effects of the litter.

Some of the losses in biomass, soil nitrogen and soil carbon associated with pure stands of *C. lanceolata* can be alleviated by planting another (broadleaved) species. For instance Wang et al. (2009), found that carbon storage was significantly increased when *C. lanceolata* was planted with either *Kalopanax septemlobus* (Thunb.) Koidz or *Alnus cremastogyne* Burk. Compared with pure stands. Wang (2005) found that in addition to significantly increasing the height, diameter, soil organic and nutrient content, the planting of the broad-leaved tree *Camptotheca acuminata* alongside Chinese fir also increased the stand's resistance to fire through both physical and chemical means, and increased overall forest moisture content (Wang 2005). Being able to naturally suppress fire and hold greater soil and canopy moisture are two beneficial traits for managing forests in a climate with an increased incidence of drought and fire. Therefore, the planting of mixed stands maybe considered a beneficial adaptation mechanism. Furthermore Yang et al. (2010) found that the enhanced carbon storage and cycling offered by enhanced litterfall in stands of *C. lanceolata* mixed with *Schima superba*, offers further benefits over pure Chinese fir stands in light of a changing climate and carbon economy. Despite its inferiority to mixed forests in terms of biomass accumulation and carbon storage, pure stands of Chinese fir are superior to single-species stands of some other species (specifically *Pinus elliottii*, and *Camellia oleifera*) in terms of carbon sequestration and soil nutrient availability (Zheng et al. 2008). However, there are other factors that affect the ability of Chinese fir to store carbon. For instance enhanced atmospheric nitrogen deposition to Chinese fir forests increases carbon storage. However at higher levels of N-deposition ($>20\text{--}30\text{ kg ha}^{-1}$) carbon (and N) uptake stabilises and N-saturation and leaching (Wei et al. 2012) may promote the eutrophication of both soil and freshwater systems. Therefore, although some excess N can be beneficial to *C. lanceolata* growth and carbon sequestration, it is imperative that climate change mitigation strategies also account for, and manage, associated atmospheric emissions.

Chinese populations of *C. lanceolata* have undergone extensive selection and sub-division throughout their 2000-year history of cultivation. The resulting gametic disequilibria among such populations mean that the selective breeding of particular individuals may lead to genetic disadvantage. Therefore repeated cycles of within and between population selection are suggested to maintain genetically diverse populations (Yeh et al. 2004). Genetic diversity is a major factor explaining forest resilience and can endow characteristics of resistance or adaptivity in the face of climate change and its associated outcomes.



Figure 15. Experimental plantation of Chinese fir (*Cunninghamia lanceolata*), Fujian, China. Experimental plantations such as this provide important information on growth and yield trends in Chinese fir.

C. lanceolata may have a variety of responses to a changing climate and related stressors. Despite the wealth of research on this species' carbon storage capabilities, there is little information on how Chinese fir may respond to other factors such as increased temperature, drought, or niche-shifts associated with climate change. One study found the genetic diversity within Chinese fir populations to be linearly related to both temperature and altitude (Yeh et al. 1994), implying that genetic diversity within populations of this species may increase with climate change as temperatures increase and forests migrate to higher elevations.

In Australia, and other parts of the South Pacific, eucalypts – species from the genus *Eucalyptus* – are commercially the most important. *Eucalyptus regnans* is the world's largest angiosperm (Wood et al. 2010) and dominates some of the cool temperate forests of Australia's southeast. According to Keith et al. (2009) these *E. regnans* dominated forests store more carbon in aboveground biomass than any other forest type. This means that appropriately managed eucalypt forests provide substantial atmospheric carbon mitigation potential. *E. regnans* can also live for over 500 years, providing long-term carbon storage potential (Wood et al. 2010). However, this currently cool moist forest may become warmer and drier putting these Australian forests at risk of reduced biomass production and subsequent decline (Booth 2012). *E. regnans* dominated systems are also marked with infrequent yet catastrophic fire events that may lead to

succession from even-aged eucalypt stands to rainforest (Wood et al. 2010). This is due to the duality of *E. regnans* being physiologically sensitive to fires, which threatens stands that have not yet reached reproductive maturity (Ashton 1976); and being an obligate seeder, that cannot germinate without fire (Wood et al, 2010). Summer precipitation and measures of the El Niño – Southern Oscillation are factors influencing the area of *E. regnans* forests burnt each year in Tasmania (Nicholls and Lucas, 2007), and variation in the frequency and intensity of both rainfall and El Niño events are some of the expected outcomes of climate change. Furthermore, climate change may lead to increased fire frequency in the region (Wood et al. 2010), which may promote stand-replacement and cause further decline of these forests, depending on the timing of these fire events.

Of the ca. 800 species of eucalypt that grow in Australia, 41% have a narrow temperature range of only about 2°C; however the few commercially important species have wider ranges (Hughes et al. 1996) making them less susceptible to changes in temperature. Furthermore, eucalypt forests may or may not experience growth enhancements from increases in atmospheric CO₂. However, research has shown that CO₂ enrichment increases water use efficiency in eucalypts (Booth 2012), which will prove useful if moisture limitation results from climate change within the current eucalypt range or niche. In addition, eucalypts are fast growing and therefore good at storing atmospheric carbon as biomass. Modelling studies have estimated the annual carbon storage potential of *Eucalyptus tereticornis* Sm. at 6 MgC·ha⁻¹·yr⁻¹ (Kaul et al. 2010).



Figure 16. Forest red gum (*Eucalyptus tereticornis*), the subject of modelling studies of annual carbon storage potential.

In addition to numerous pine species, Douglas-fir (*Pseudotsuga menziesii*) is a commercially important species in western North America, particularly in British Columbia, Canada, and Washington and Oregon in the USA. At least two subspecies are known to exist; *P. menziesii* var. *menziesii* in coastal regions, and *P. menziesii* var. *glauca* in the continent's western interior region. *P. menziesii* may suffer from specific climate change-induced impacts. For instance, Apple et al. (2000) found (in a laboratory setting) that elevated temperature slightly, but significantly, increased stomatal conductance and transpiration in Douglas-fir, whereas elevated CO₂ did not significantly influence any aspect of needle morphology. Guak et al. (1998) found that elevated CO₂ and temperatures can advance the timing of bud burst, slow and reduce the growth of buds, and reduce the overall cold hardiness of Douglas-fir. The absence of cold winters can also drastically limit seedling establishment in this species (Shafer et al. 2001). However, Douglas-fir may also be capable of adapting to a modified climate via genotypic

selection, which has been observed in this species in response to additive climatic and other selective pressures (Eckert et al. 2009).

The amount of growing season precipitation has been found to restrict the growth of Douglas-fir, however at the limits of its range, temperature was found to be a more important growth-limiting factor (Zhang and Hebda 2004). Olszyk et al. (1998; 2003) also found that warming influenced growth factors such as seedling height; however biomass allocation and stem diameter were unaffected by either warming or enhanced atmospheric CO₂. Species such as *P. menziesii*, that inhabit bands of elevation in mountainous regions, are expected to lose habitat faster than new habitat is gained. Therefore, trees such as *P. menziesii* are expected to decline with the elevational habitat and niche shifts that are expected as an outcome of climate change (Hamann and Wang 2006). Various climate scenarios and modelled biophysical climate-response surfaces have predicted declines in Douglas fir within their current ranges, and also predict large spatial gaps between their current distribution and predicted future distribution (Shafer et al. 2001). These climatic responses could mean drastic consequences for Douglas-fir forests. However, others have predicted increases in Douglas fir ranges, particularly the northward expansion into areas currently dominated by white spruce (*Picea glauca*) or lodgepole pine (*Pinus contorta*) (Bunnell and Kramsater 2012).

Although *P. menziesii* is known to be a relatively drought-tolerant species (Franklin and Dyrness, 1988), giving it a potential competitive advantage over other species in North America's Pacific Northwest, Ferrell and Woodard (1966) reported significant genetic variation and differences in drought tolerance between coastal and mountain populations. More specifically, interior mountain populations displayed increased drought tolerance over coastal populations. Conditions of drought also increase fire frequency. Although Douglas-fir trees have developed physiological mechanisms to survive low- to medium-intensity ground fires (namely thick bark), their forests may become encroached by Ponderosa pine (*Pinus ponderosa*) and Western larch (*Larix occidentalis*) following severe fire disturbance (Habeck and Mutch 1973). In addition Douglas-fir seeds are sensitive to heat, and fires of any intensity can lead to reductions in the viable seed bank of these forests (Clark and Wilson 1994), limiting natural regeneration following catastrophic fire or other disturbance.

Other species in the AP region are also of economic or ecological importance and may be particularly sensitive to a change in climate or a shift in climatic envelope of niche. For instance teak (*Tectona grandis*), which is native to south and southeast Asia, but is now grown throughout the tropics for its valuable wood, is especially sensitive to changes in temperature and moisture. This sensitivity is reflected in the development of wood and tree rings, and has led to teak's widespread use in dendrochronology for the reconstruction of past climate, particularly rainfall, throughout south Asia (Jacoby and D'Arrigo 1990; Sinha 2012). Changes in rainfall and moisture availability may therefore affect the development of cambium and the quality or grain of teak wood. Teak is also grows faster than other hardwoods and has been estimated to store 2 MgC·ha⁻¹·yr⁻¹ of atmospheric carbon (Kaul et al. 2010).



Figure 17. Teak (*Tectona grandis*) forest at Ramnagar in Uttarakhand, India. Teak may be an important way to sequester carbon in forest products for relatively long periods.

Bamboo is a diverse group of fast-growing grasses in the family Gramineae. Nearly 1500 species of bamboo have been identified worldwide, about 40% of which are found in China (Ohrnberger, 1999). Bamboos are also increasingly important plants to the global economy. Bamboo is now used extensively for building products, textiles and other ‘sustainable’ products due to its relatively short rotation period and strength of fibre, and for this reason bamboo plantations are now being established outside their natural range. However, with a warming global climate the natural limit of bamboo forests is expected to move northward (Rui 2002), which may open up new areas for the cultivation of these unique grasses. Although bamboo can form dense tall stands throughout much of the AP, bamboo forests do not store carbon effectively. Rather mature stands are in equilibrium between carbon taken up for growth, and carbon released from stem death and decomposition (Düking et al. 2011).

A number of pine (*Pinus*) species are grown commercially throughout the AP. Lodgepole pine (*Pinus contorta* var. *latifolia*), which grows throughout much of North America’s Pacific Northwest, is resistant to both shade and fire (Clements 1910), however it is also the species that has been most devastated by mountain pine beetle outbreak in British Columbia, Canada (Smith et al. 2012). Radiata pine (*Pinus radiata*) is native to the south-western United States, but is now cultivated around the world over a much larger area than its native habitat ever covered (Lavery and Mead 1998) – including in Australia and New Zealand where this species has replaced

native forest stands (Turner and Lambert 2000). Radiata pine is usually grown in large monocultures, and is considered by some to be an invasive species due to changes (such as soil chemistry) it may cause in neighbouring native eucalypt systems (Baker and Murray 2012). Most pines are fast growing and therefore have substantial carbon storage potential under appropriate management regimes. Radiata pine is thought to have a competitive advantage in this regard, and due to its ability to adapt beyond its environmental niche, holds much promise for plantations in light of climate change. However, as Lavery and Mead (1998) point out, the limitations of this species are in the use of its products, and in its ‘biologically safe growth environment’ – i.e. so as not to threaten other species.



Figure 18. Radiata pine (*Pinus radiata*) plantation in the Wombat State Forest, Victoria, Australia. This species plays an important role in many temperate parts of the Asia-Pacific region, especially in Australia and New Zealand.

3.2 Management specific issues and strategies

‘Ecological resilience’ is defined by Gunderson (2000) as ‘the ability of a system to absorb impacts before a threshold is reached’. Resilience is similar to resisting change, but is applied across a larger scale and is aimed at surviving climate-induced disturbances (Dale et al. 2001). The concept of ‘resilience’ is of great importance when considering the ability of AP’s forests to flourish (or sometimes survive) in times of climate change. In essence, forest managers and

policy makers need to ensure that forest resilience is achieved or maintained. The complex relationships between individual forest species (or sometimes individual trees) and climate change, can make forest planning and management very complicated in light of a changing climate. However, there are some management issues that need to be approached throughout the AP region that are neither species nor population specific, but may have specific influences on, or repercussions for, the population or species at hand. These issues must be managed appropriately in order to ensure that forests of the AP are resilient to climate change.

For instance changes in temperature and moisture have been shown to influence the relative abundance of age classes within a species by reducing seedling recruitment (Lloret et al. 2009). If climatic changes also result in increased adult tree mortality, either through shifts in niche, or via disease, fire, flooding, or other climate-induced disturbance, seedling establishment may not be able to compensate for losses, and shifts in forest community composition may occur. As mentioned above, stand or stem age also influences responses to heat, shade, wind, fire, and drought or moisture. Stand age is also important in terms of air pollution injury (Krupa and Legge 1999; Karnosky et al. 2007) and carbon sequestration potential (Asshoff et al. 2006) with younger trees and seedlings being particularly susceptible to both. However, Karnosky et al. (2007) point out that the majority of air pollution research has been conducted on young trees and seedlings, and their resultant injury may differ significantly from that of mature trees and stands. Because different ages of trees respond to climate and associated stressors in distinct ways, a best practice would be to ensure that stands consist of multiple age-classes so as to ensure forest resilience in light of unknown climate outcomes, ensuring that a common stressor does not equally threaten all individuals.

There is also evidence that mixed-species forests of the AP are in general more resilient than single-species stands (Singh 2008; Thompson et al. 2009). Mixed-species and mixed-age planting can reduce the scale of devastation resulting from most climate-induced stressors and disturbances such as disease or insect outbreaks, and damage from air pollution or windstorms. Species, age and height classes are important determinants in forest resilience. This can become a hurdle for forest managers accustomed to the often mono-culture plantations of the AP. Either the resilience of monocultures needs to be artificially increased, or the practices of sustainable forest management, such as planting and maintaining biologically diverse multi-aged stands, need to be integrated as part of climate change adaptation strategies. In terms of increasing resilience, genetic association mapping holds promise for gene conservation and the development or propagation of more adaptive tree phenotypes (Aitken et al. 2008). Additionally, assisted migration can be used to maximise traits such as drought hardiness and increase the resistance offered by population genotypes (Chmura et al. 2011).

Climate change adaptation includes carbon management as a part of the forest management paradigm. Kaul et al. (2010) found that in the teak and sal (*Shorea robusta* Gaertn. f.) forests of India, the length of rotation and the thinning regime utilised by forest managers can influence the carbon stored by forests and the value of timber harvested. For instance maximum primary productivity was seen at 60-year rotations, but declined as the rotation length was extended. However average carbon stock increased by approximately 12% when rotation length was increased from 120 years to 150 years. Seely et al. (2002) found that in western Canada's boreal forests, longer rotations also resulted in increased carbon storage, regardless of the species being

considered. In Australia, Turner and Lambert (2000) found that soil organic carbon stores were much lower in plantations than in the natural forests, and that surface soil carbon stores depleted rapidly within the first 12 years following plantation establishment. These lost soil carbon stores were eventually offset by vegetative carbon accumulation, however the rate at which carbon loss was offset differed substantially between stands. Canary et al. (2000) found that the fertilisation of Douglas-fir stands with urea increased total carbon storage by an average of $34.7 \text{ Mg}\cdot\text{C}\cdot\text{ha}^{-1}$. However the risks associated with forest fertilisation, especially when combined with atmospheric organic nitrogen deposition, may outweigh the benefits of carbon sequestration and associated credits, in parts of the AP. Excess nitrogen leaches from soil systems into ground and surface water and can lead to soil and freshwater eutrophication or other undesirable environmental and human health impacts.



Figure 19. Sal (*Shorea robusta*) forest in the Himalayan foothills of northern India. This is an important timber species, and the rotation length used in forests has important implications for the amount of carbon that they store.

Certain management techniques may be used to maximize the benefits, or minimise the impacts, of climate change. For instance, density management can be used to aid fire prevention, and to reduce the threats posed by wind or storms (Chmura et al. 2011). Although Kaul et al. (2010) found that stand thinning did not aid in carbon sequestration, it was deemed necessary as a forest sector source of fuel and for disease, pest or fire control. Thinning and density management can also optimise canopy openings, promote natural regeneration and growth factors, and be used to

maintain multiple age classes and species richness.

Extreme weather events, such as enhanced storms and heavy rainfall can increase runoff and erosion in forested basins – particularly in areas of complex terrain, or where recent logging has left soils exposed. Erosion and runoff can lead to flooding or mass movements that may not only affect forests, but also affect forest practices by limiting or removing access. Similarly increased fire frequency not only reduces the standing stock of affected forests, but can damage infrastructure and also limit forest access. In northern regions, where permafrost is relied upon to provide winter access in otherwise saturated and soft ground, the melting of winter permafrost, or a change in the timing of freeze, can also limit access. Certain strategies such as fire breaks, bridges or slope stabilization may be required by forest managers in order to limit the impacts of these sorts of outcomes. Planting or maintaining trees and other forest vegetation can minimise the risk of landslides (Dale et al. 2001; Santoso et al. 2008) that are predicted to occur more frequently due to increased rainfall amounts and intensity. In addition the maintenance of (salt tolerant) vegetation on coastlines can help to prevent coastal erosion and associated land or habitat loss.

Tools such as process-based models can also be used to aid in effectively predicting and managing climate change impacts on the forests of the AP. The diversity and complexity of such models make their review beyond the scope of this paper. However, there are other simple techniques by which forest managers may track and prepare for climatic change in forest stands. For example indicators such as seasonal changes in plant phenology can be used to track and assess climate change impacts in forests of the AP (Kozlov and Berlina 2002; Beaubien and Hall-Beyer 2003; Chen et al. 2005; Gallagher et al. 2009). Visual indicators can also be used to identify symptoms of air pollution (Krupa nad Legge 1999; Bytnerowicz et al. 2007) and drought injury. These sorts of indicators, and many of those associated with sustainable forest management (e.g. Kotwal et al. 2008), can help to identify and track climate change and its associated forest impacts.

Forests are not always managed by skilled professionals; rather communities throughout the AP region manage and utilise forests for timber and non-timber forest product. Traditional knowledge, particularly in regards to past forest history, can be an important factor in understanding forests under a changing climate, particularly in parts of the AP lacking historical forest monitoring and data. For example in Nepal traditional knowledge had been used to address many climate change related forest issues including carbon storage, landslides and water availability (Rijal 2008). Other non-timber forest uses, such as tourism or the collection of non-timber forest products, can help in maintaining forest biodiversity and other features that maintain forest resilience. These forest uses may be seen as beneficial in light of climate uncertainty and changes in forest composition or health. However, not all human responses to climate change may be positive, especially at a local level. Some ecosystems may experience increased pressure as a result of altered human behaviour. For instance increased human pressure is predicted to lead to declines in the boreal forests of China (Preston et al. 2006). Similarly some tropical areas of the region may experience dieback and increases in wildfires (Preston et al. 2006) from changes in land-use.



Figure 20. Boreal forest near Yichun, Heilongjiang Province, China. Such forests are under considerable anthropogenic pressure, increasing their vulnerability to climate change.

4.0 Conclusion

As a result of the potential impacts of climate change discussed above, some forests or forest communities may disappear entirely, while others may experience drastic reductions in size. For instance, China's boreal forest is predicted to lose 70% of its area (Preston et al. 2006) and tropical montane cloud forests may completely vanish (Foster 2001).

Due to the diversity of the AP's forests and other ecosystems, there are numerous ecotones or boundaries between forests and other systems across the landscape. Because of the drastic impacts climate change is expected to have and is having in these community margins, there are likely to be major impacts of climate change on the forests in the region. However, not all of these changes will be negative for forests. Forest migration to increased altitudes and latitudes means that total forest area in the AP may actually increase, which is beneficial from a forestry perspective. An increase in forest area is also a helpful mechanism in reducing GHG-caused climate change through carbon storage mechanisms. However if the existing forests are lost to other non-forest ecosystems, total forest cover may decline because there will be less viable area to inhabit with increasing altitude on the mountain (Foster 2001) and towards the poles.

Large-scale and severe forest disturbances caused by fire, flood or windstorms may trigger seedling establishment and regeneration (Dale et al. 2002) and may promote succession or the natural adaptation of forests to a changing climate by forcing complete renewal and regrowth (Foster 2001), creating more resilient and adapted forests.

In general, any change in climatic variables (moisture, temperature, wind) will alter ecosystems or forests. The magnitude of the climatic change and its variability will largely dictate the changes in forests or ecosystems. However, whether the change is considered positive or negative, is subjective and dependent on one's perspective and goals. For instance, from the perspective of forest management, anything that increases trees, tree habitat, or forest cover may be viewed as a positive outcome.

Although global climate model outputs are often referred to as “predictions”, each output really represents one possible outcome, or ‘scenario’ of the future. One of the most difficult things to predict is human behaviour, and therefore most climate change modelling exercises include different runs based on various emission scenarios. Whether global GHG levels, represented as ‘carbon dioxide equivalents’, increase or decrease over the next century, and the magnitude of that increase or decrease, depends on a multitude of political, social, and economic factors that are not easily predicted. In essence it depends on human behaviour and both political and personal will to change. The magnitude, extent and nature of atmospheric emissions, agriculture, energy, transport, human population growth and land-use or land-cover change will all affect the impact climate change has on forests of the AP, as much as forest resilience does.

Although climate change in the AP presents many challenges for forest managers and users alike, it also presents opportunity. Forests store carbon efficiently and effectively, while providing additional goods and services – from recreation, to timber and non-timber forest products. While there are numerous potential impacts that climate change may have on forests of the AP, the fate of the region's forests depends on our ability to reduce GHG emissions, mitigate anthropogenic climate forcing, and adapt our forest management practices. In general, the research reviewed has shown that the treatment and design of managed forests to be more like natural forests, increases both carbon storage and resilience, and can provide forests of the AP the qualities they need to survive and flourish in an uncertain and changing climate.

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6. References

- Adger, W.N., Adhikary, S., Ando, M., Anokhin, Y., Cruz, R.V., Ilyas, M., Kopaliani, Z., Lansigan, F., Li, C., Patwardhan, A., Safriel, U., Suharyono, H., Zhang, X., 2001. Chapter 11: Asia. In: McCarthy, J.J., Canziani, O.F., Leary, N.A., Dokken, D.J., White, K.S. (Eds.), *Climate Change 2001: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge, pp. 535–590.
- Agrios, N.G., *Plant pathology*. 4th edition. – Academic Press San Diego. 635 pp.
- Ainsworth, E.A., and Long, S.P., 2005. What have we learned from 15 years of free-air CO₂ enrichment (FACE)? A meta-analytic review of the responses of photosynthesis, canopy properties and plant production to rising CO₂. *New Phytologist* 165: 351-372.
- Aitken, S. N., S. Yeaman, J. A. Holliday, T. Wang and S. Curtis-McLane, 2008 Adaptation, migration or extirpation: climate change outcomes for tree populations. *Evolutionary Applications* 1: 95–111.
- Allen, C.D., and Breshears, D.D., 1998. Drought-induced shift of a forest-woodland ecotone: Rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences of the United States of America* 95: 14839-42.
- Allen, C.D., Macalady, A.K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D.D., Hogg, E.H., Gonzalez, P., Fensham, R., Zhang, Z., Castro, J., Demidoca, N., Lim, J.-H., Allard, G., Running, S.W., Semerci, A., and Cobb, N., 2009. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*. 259(4): 660-684.
- Apple, M. E., Olszyk, D. M., Ormrod, D. P., Lewis, J., Southworth, D., and Tingey, D.T., 2000. Morphology and Stomatal Function of Douglas Fir Needles Exposed to Climate Change: Elevated CO₂ and Temperature. *International Journal of Plant Sciences* 161(1): 127-132.
- Ashton, D.H., 1976. The development of even aged stands of *Eucalyptus regnans* F. Muell. in central Victoria. *Australian Journal of Botany* 24, 397–414.
- Asia Development Bank, 2005. *Climate Proofing: A Risk-Based Approach to Adaptation*. Asian Development Bank, Manila, Philippines. 191pp.
- Asshoff, R., Zotz, G. & Körner, C. 2006 Growth and phenology of mature temperate forest trees in elevated CO₂. *Global Change Biology* 12(5): 848–861.
- Australian Bureau of Meteorology 2006. *The South Pacific Sea Level & Climate Monitoring Project - Monthly Data Report No. 132*. June 2006. National Tidal Centre, Bureau of

- Meteorology, Kent Town, Australia. 32pp.
<http://www.bom.gov.au/ntc/IDO60101/IDO60101.200606.pdf> [Accessed 22 May 2012].
- Baker, A.C., and Murray, B., 2012. Seasonal intrusion of litterfall from non-native pine plantations into surrounding native woodland: Implications for management of an invasive plantation species. *Forest Ecology and Management* 277: 25-37.
- Banfield, G.E., Bhatti, J.S., Jiang, H., Apps, M.J., Karjalainen, T., 2002. Variability in regional scale estimates of carbon stocks in boreal forest ecosystems: results from west-central Alberta. *Forest Ecology and Management* 169: 15–27.
- Batjes, N.H., and Bridges, E.M., 1992. A review of soil factor and processes that control fluxes of heat, moisture and greenhouse gases. Technical Paper 23, International Soil Reference and Information Centre, Wageningen, The Netherlands. 197pp.
- Battaglia, L. L., S. A. Fore, and R. R. Sharitz. 2000. Seedling emergence, survival and size in relation to light and water availability in two bottomland hardwood species. *Journal of Ecology* 88: 1041–1050.
- Beaubien, E.G., and Freeland, H.J., 2000. Spring phenology trends in Alberta, Canada: links to ocean temperature. *International Journal of Biometeorology* 44(2): 53-59.
- Beaubien, E.G., and Hall-Beyer, M., 2003. Plant phenology in western Canada: trends and links to the view from space. *Environmental Monitoring and Assessment* 88: 419-429.
- Belyazid, S., Kurz, D., Braun, S., Sverdrup, H., Rihm, B., Hettlingh, J.-P., 2011. A dynamic modelling approach for estimating critical loads of nitrogen based on plant community changes under a changing climate. *Environmental Pollution* 159: 789-801.
- Benjamin, L.P., Ramasamy, S., Ian, M., Janice, B., 2006. Climate change in the Asia/Pacific region. Climate change impacts and risk. .CSIRO Marine and Atmospheric Research, Australia. 93pp.
- Berg, E.E., Henry, J.D., Fastie, C.L., De Volder, A.D., Matsuoka, S.M., 2006. Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: relationship to summer temperatures and regional differences in disturbance regimes. *Forest Ecology and Management* 227, 219– 232.
- Boerner, R.E.J., 1982. Fire and Nutrient Cycling in Temperate Ecosystems. *BioScience* 32(3): 187-192.
- Brace, S., and Peterson, D.L., 1999. Spatial patterns of tropospheric ozone in the mount rainier region of the cascade mountains, USA. *Atmospheric Environment* 32(21): 3629-3637.
- Braekke, F. H. 1976. Impact of acid precipitation on forest and freshwater ecosystems in Norway. Fagrapport FR6/76. SNSF Project 1432, Ås-NLH, Norway, 111 pp.

- Brakkee, K.W., Huijbregts, M.A.J., Eickhout, B., Hendriks, A.J., Van de Meent, D., 2008. Characterization factors for greenhouse gases at a midpoint level including indirect effects based on calculations with the IMAGE model International Journal of Life Cycle Assessment. 13(3): 191-201.
- Bunnell, F., and Kremsater, L.L, 2012. Migrating like a herd of cats – climate change and emerging forests in British Columbia. Journal of Ecosystems and Management 13(2): 1-24.
- Bytnerowicz, A., Padgett, P., Percy, K., Krywult, M., Riechers, G., Hom, J., 1999. Direct effects of nitric acid on forest vegetation. In: Miller, P.R., McBride, J. (Eds.), Oxidant Air Pollution Impacts in the Montane Forests of Southern California: The San Bernardino Mountains Case Study. Springer Ecological Series 134. Springer, New York, pp. 270-287.
- Bytnerowicz, A., Omasa, K., Paoletti, E., 2007. Integrated effects of air pollution and climate change on forests: a northern hemisphere perspective. Environmental Pollution 147: 438–445.
- Canary, J.D., Harrison, R.B. Compton, J.E. Chappell, H.N., 2000. Additional carbon sequestration following repeated urea fertilization of second-growth Douglas-fir stands in western Washington. Forest Ecology and Management 138: 225-232.
- Cayan DR, Kammerdiener SA, Dettinger MD, Caprio JM, Peterson DH. 2001. Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society 82:399–415
- Chapin FS III, Lovecraft A L, Zavaleta E S, Nelson J, Robards M D, Kofinas G P, Trainor S F, Peterson G D, Huntington H P and Naylor R L 2006 Policy strategies to address sustainability of Alaskan boreal forests in response to a directionally changing climate. Proceedings of the National Academy of Sciences 103: 16637–16643.
- Chen, I.-C., Hill, J. K., Ohlemüller, R., Roy, D. B., Thomas, C. D., 2011. Rapid range shifts of species associated with high levels of climate warming. Science 333: 1024 .
- Chen, X., Hu, B., Yu, R., 2005. Spatial and temporal variation of phenological growing season and climate change impacts in temperate eastern China. Global Change Biology 11(7): 1118-1130.
- Chen, X., and Xu, L., 2011. Phenological responses of *Ulmus pumilla* (Siberian Elm) to climate change in the temperate zone of China. International Journal of Biometeorology. DOI 10.1007/s00484-011-0471-0
- Chmura, D.J., Anderson, P.D., Howe, G.T., Harrington, C.A., Halofsky, J.E., Peterson, D.L., Shaw, D.C., and St. Clair, J.B., 2001. Forest responses to climate change in the

- northwestern United States: Ecophysiological foundations for adaptive management. *Forest Ecology and Management* 261: 1121-1142.
- Clark, D.L., and Wilson, M.V. 1994. Heat-treatment effects on seed bank species of an old-growth Douglas-fir forest. *Northwest Science* 68(1): 1-5.
- Clements, F.E., 1910. *The Life History of Lodgepole Burn Forests*. USDA Forest Service Bulletin 79. USDA Forest Service, Washington, DC.
- Cochrane, M.A., 2003. Fire Science for Rainforests. *Nature* 421: 913-919.
- Cohen, S.J., 2011. Chapter 29 Overview: Climate Change Adaptation in Rural and Resource-Dependent Communities. In: J.D. Ford and L. Berrang-Ford (eds.), *Climate Change Adaptation in Developed Nations: From Theory to Practice*, Advances in Global Change Research Vol 42. Pp. 401-412.
- Cox, R.M., 1983. Sensitivity of forest plant reproduction to long range transported air pollutants: in vitro sensitivity of pollen to simulated acid rain. *New Phytologist* 95(2): 269-276.
- Cruz, R.V., Harasawa, H., Lal, M. & Wu, S., 2007. Chapter 10 Asia. In: *Climate Change 2007: Impacts, Adaptation and Vulnerability IPCC Fourth Assessment Report*. Working Group II Report Cambridge, UK, Cambridge University Press.
- Dale, V.H., Joyce, L.A., McNulty, S., Neilson, R.P. Ayres, M.P. Flannigan, M.D. Hanson, P.J., Irland, I.C., Lugo, A.E., Peterson, C.J., Simberloff, D., Swanson, F.J., Stocks, B.J., and Wotton, M.B., 2001. Climate change and forest disturbances. *BioScience* 51: 723-34.
- Day, M.D., Broughton S, Hannan-Jones MA 2003. Current distribution and status of *Lantana camara* and its biological control agents in Australia, with recommendations for further biocontrol introductions into other countries. *BioControl* 24: 63N-76N.
- Desprez-Loustau, M., Marcais, B., Nageleisen, L., Piou, D., Vannini, A., 2006. Interactive effects of drought and pathogens in forest trees *Annals of Forest Science* 63: 597-612.
- Devi N, Hagedorn F, Moiseev P, Bugmann H, Shiyatov S, Mazepa V and Rigling A 2008 Expanding forests and changing growth forms of Siberian larch at the Polar Urals treeline during 20th century *Glob. Change Biol.* **14**: 1581-1591.
- Doi, H., 2007. Winter flowering phenology of Japanese apricot *Prunus mume* reflects climate change across Japan. *Climate Research* 34: 99-104.
- Düking, R., Gielis, J., and Liese, W., 2011. Carbon Flux and Carbon Stock in a Bamboo Stand and their Relevance for Mitigating Climate Change. *The Journal of the American Bamboo Society* 24(1): 1-7.
- Eckert AJ, Bower AD, Wegrzyn JL, Pande B, Jermstad KD, Krutovsky KV, Clair JBS, Neale

- DB (2009a) Association genetics of coastal Douglas fir (*Pseudotsuga menziesii* var. *menziesii*, Pinaceae). I. Cold-hardiness related traits. *Genetics* 182:1289–1302.
- Elansky, N.F., 2009. Russian Studies of the Atmospheric Ozone in 2003-2006. *Izvestiya, Atmospheric and Oceanic Physics* 45(2): 207-220.
- Environment Canada, 2010. Acid Rain FAQ. URL: <http://www.ec.gc.ca/air/default.asp?lang=En&n=7E5E9F00-1> [Accessed 26 April 2012].
- Esper J, Schweingruber FH (2004) Large-scale treeline changes recorded in Siberia. *Geophys Res Lett* 3: L06202.
- Etkin DA. 1995. Beyond the year 2000, more tornadoes in western Canada? Implications from the historical record. *Natural Hazards* 12: 19–27.
- Fang, J., A. Chen, C. Peng, S. Zhao and L. Ci, 2001. Changes in forest biomass carbon storage in China between 1949 and 1998. *Science*, 292:2320–2322.
- Felzer, B., Reilly, J., Melillo, J., Kicklighter, D., Sarofim, M., Wang, C., Prinn, R., and Zhuang, Q., 2005. Future effects of ozone on carbon sequestration and climate change policy using a global biogeochemical model. *Climate Change* 73: 345-373.
- Fensham, R.J., Fairfax, R.J., Butler, D.W., Bowman, D.M.J.S., 2003. Effects of fire and drought in a tropical eucalypt savanna colonized by rain forest. *Journal of Biogeography* 30, 1405–1414.
- Fensham, R. J., and Fairfax, R. J., 2007. Drought-related tree death of savanna eucalypts: Species susceptibility, soil conditions and root architecture full access. *Journal of vegetation science*, 18 1: 71-80.
- Ferrell, W.K., Woodard, E.S., 1966. Effect of seed origin on drought resistance of Douglas-fir (*Pseudotsuga menziesii*) (Mirb.) Franco. *Ecology* 47, 499–503.
- Fiore, A.M., Jacob, D.J., Field, B.D., Streets, D.G., Fernandes, S.D., Jung, C., 2002. Linking ozone pollution and climate change: The case for controlling methane. *Geophysical Research Letters* 29 (19), 1919.
- Fowler, D., Cape, J.N., Coyle, M., Flechard, C., Kuylenstierna, J., Hicks, K., Derwent, D., Johnson, C., Stevenson, D., 1999. The global exposure of forest ecosystems to air pollutants. *Water Air and Soil Pollution* 116, 5-32.
- Fowler, D., Pilegaard, K., Sutton, M. A., Ambus, P., Raivonen, M., Duyzer, J., Simpson, D., Fagerli, H., Fuzzi, S., Schjoerring, J. K., Granier, C., Nefel, A., Isaksen, I. S. A., Laj, P., Maione, M., Monks, P. S., Burkhardt, J., Daemmgen, U., Neirynek, J., Personne, E., Wichink-Kruit, R., Butterbach-Bahl, K., Flechard, C., Tuovinen, J. P., Coyle, M., Gerosa, G., Loubet, B., Altimir, N., Gruenhage, L., Ammann, C., Cieslik, S., Paoletti, E.,

- Mikkelsen, T. N., Ro-Poulsen, H., Cellier, P., Cape, J. N., Horváth, L., Loreto, F., Niinemets, U., Palmer, P. I., Rinne, J., Misztal, P., Eitzinger, E., Nilsson, D., Pryor, S., Gallagher, M. W., Vesala, T., Skiba, U., Brüggemann, N., Zechmeister-Boltenstern, S., Williams, J., O'Dowd, C., Facchini, M. C., de Leeuw, G., Flossman, A., Chaumerliac, N., and Erisman, J. W.: Atmospheric composition change: ecosystems – atmosphere interactions, *Atmospheric Environment* 43: 5193–5267.
- Foster, P., 2001. The potential negative impacts of global climate change on tropical montane forests. *Earth-Science Reviews* 55: 73-106.
- Franklin, J.F., Dyrness, C.T., 1988. *Natural Vegetation of Oregon and Washington*. Oregon State University Press, Corvallis, OR, USA.
- Gallagher, R.V., Hughes, L., and Leishman, M.R., 2009. Phenological trends among Australian alpine species: using herbarium records to identify climate-change indicators. *Australian Journal of Botany* 57(1): 1–9.
- Glenz, C., Schlaepfer, R., Iorgulescu, I., Kienast, F., 2006. Flooding tolerance of Central European tree and shrub species. *Forest Ecology and Management* 235: 1–13.
- Goldammer, J.G., and Price, C., 1998. Potential impacts of climatic change on fire regimes in the tropics based on MAGICC and a GISS GCM-derived lightning model. *Climatic Change*, 39: 273–296.
- Goto, Y., Yoshitake, T., Okano, M., and Shimada, K., 1996. Seedling regeneration and vegetative resprouting after fires in *Pinus densiflora* forests. *Vegetatio* 122:157-165.
- Guak, S., Olszyk, D.M., Fuchigami, L.H., and Tingey, D.T., 1998. Effect of elevated CO₂ and temperature on cold hardiness and spring bud burst and growth in Douglas-fir (*Pseudotsuga menziesii*). *Tree Physiology* 18(10): 671-679.
- Guarín, A., and Taylor, A.H., 2005. Drought triggered tree mortality in mixed conifer forests in Yosemite National Park, California, USA. *Forest Ecology and Management* 218 (1-3): 229-244.
- Habeck, J.R., and Mutch, R.W., 1973. Fire-dependent forest in the Northern Rocky Mountains. *Quaternary Research* 3(3): 408-424.
- Hamann, A., and Wang, T., 2006. Potential effects of climate change on ecosystems and tree species distribution in British Columbia. *Ecology* 87: 2773-2786.
- Hart, M., de Dear, R., Hyde, R., 2006. A synoptic climatology of tropospheric ozone episodes in Sydney, Australia. *International Journal of Climatology* 26: 1635–1649.
- Harvell, C.D., Mitchell, C.E., Ward, J.R., Altizer, S., Dobson, A.P., Ostfeld, R.S., and Samuel, M.D., 2002. Climate warming and disease risks for terrestrial and marine biota. *Science*,

296, 2158–2162.

- Hinzman L.D., Bettez, N.D., Bolton W.R., Chapin, F.S., and Dyurgerov, M.B. 2005 Evidence and implications of recent climate change in northern Alaska and other Arctic regions *Climate Change* 72: 251-298.
- Hogg, E.H., Brandt, J.P., Michaellian, M., 2008. Impacts of a regional drought on the productivity, dieback, and biomass of western Canadian aspen forests. *Canadian Journal of Forest Research* 38: 1373–1384.
- Hubbard, N.N, 1995. In search of regional paleoclimates: Australia, 18,000 yr BP. *Paleogeography, Paleoclimatology, Paleoecology* 116: 167-188.
- Hughes, L., 2003. Climate change and Australia: Trends, projections and impacts. *Austral Ecology* 28(4): 423-443.
- IPCC, 2001a. *Climate Change 2001: The Scientific Basis*. Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., van der Linden, P.J., Dai, X., Maskell, K., and Johnson, C.A. (eds.). . Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change, World Meteorological Organisation and United Nations Environment Programme. Cambridge University Press. 94 pp.
- IPCC, 2001b. *Climate Change 2001: Impacts, Adaptation and Vulnerability*. McCarthy, J., Canziani, O., Leary, N., Dokken, D and White, K. (eds.). Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change, World Meteorological Organisation and United Nations Environment Programme. Cambridge University Press. 1032 pp.
- IPCC, 2007a. *Climate Change 2007: Synthesis Report*. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Core Writing Team, Pachauri, R.K and Reisinger, A. (eds.). IPCC, Geneva, Switzerland, 104 pp.
- IPCC, 2007b. *Climate Change 2007: The Physical Science Basis*. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 996 pp.
- Ishizuka, M., Toyooka, H., Osawa, A., Kushima, H., Kanazawa, Y., and Sato, A., 1997. Secondary Succession Following Catastrophic Windthrow in a Boreal Forest in Hokkaido, Japan. *Journal of Sustainable Forestry* 6(3-4): 367-388.
- Jacoby, G.C., and D'Arrigo, R.D., 1990. Teak (*Tectona grandis* L.F.), a tropical species of large-scale dendroclimatic potential. *Dendrochronologia*. 8: 83-98.

- Jakob, M., and Lambert, S., 2009. Climate change effects on landslides along the southwest coast of British Columbia. *Geomorphology* 107: 275-284.
- Jones, M.E., Paine, T.D., Fenn, M.E., Poth, M.A., 2004. Influence of ozone and nitrogen deposition on bark beetle activity under drought conditions. *Forest Ecology and Management* 200: 67-76.
- Jones, A., Hatwood, J., Boucher, O., 2009. Climate impacts of geoengineering marine stratocumulus clouds. *Journal of Geophysical Research* 114: D10106, doi:10.1029/2008JD011450
- Juday, G.P., Barber, V., Duffy, P., Linderholm, H., Rupp, S., Sparrow, S., Vaganov, E., and Yarie, J., 2005. Forests, Land Management, and Agriculture. *In: Arctic Climate Impact Assessment, Scientific Report*. Cambridge University Press. Pp. 781-862.
- Karlsson, J.M., Bring, A., Peterson, G.D., Gordon, L.J., and Destouni, G., 2011. Opportunities and limitations to detect climate-related regime shifts in inland Arctic ecosystems through eco-hydrological monitoring. *Environmental Research Letters* 6: 1-9.
- Kampa, M., and Castanas, E., 2008. Human health effects of air pollution. *Environmental Pollution* 151: 262-367.
- Karnosky, D.F., Gagnon, Z.E., Dickson, R.E., Coleman, M.D., Lee, E.H., Isebrands, J.G., 1996. Changes in growth, leaf abscission, and biomass associated with seasonal tropospheric ozone exposures of *Populus tremuloides* clones and seedlings. *Canadian Journal of Forest Research* 26: 23-37.
- Karnosky, D.F., Pregitzer, K.S., Zak, D.R., Kubiske, M.E., Hendrey, G.R., Weinstein, D., Nosal, M. & Percy, K.E. 2005. Scaling ozone responses of forest trees to the ecosystem level in a changing climate. *Plant, Cell and Environment* 28(8): 965–981.
- Karnosky D., Skelly J, Percy K, Chappelka A, 2007. Perspectives regarding 50 years of research on effects of tropospheric ozone air pollution on US forests. *Environmental Pollution* 147: 489–506.
- Kauhanen, H., Wallenius, T., Kuuluvainen, T., Aakala, T., and Mikkola, K., 2008. Extensive mortality of spruce forests in Arkhangelsk region: satellite image analysis. In: *International Conference on Adaptation of Forests and Forest Management to Changing Climate with Emphasis on Forest Health: A Review of Science, Policies, and Practices, Book of Abstracts and Preliminary Programme*. Umeå, Sweden, FAO/IUFRO, 25–28 August 2008.
- Kaul, M., Mohren, G.M.J., and Dadhwal, V.K., 2010. Carbon storage and sequestration potential of selected tree species in India. *Mitigation and Adaptation Strategies for Global Change* 15(5): 489-510.

- Kawamura, R., Suppiah, R., Collier, M.A., and Gordon, H.B., 2004. Lagged relationships between ENSO and the Asian Summer Monsoon in the CSIRO coupled model. *Geophysical Research Letters* 31, doi:10.1029/2004GL021411.
- Kelly, A.E., and Goulden, M.L., 2008. Rapid shifts in plant distribution with recent climate change. *Proceedings of the National Academy of Sciences of the United States of America* 105 (33): 11823-11826.
- Khurana, E., and Singh, J.S., 2001. Ecology of tree seed and seedling: Implication for tropical forest conservation and restoration. *Current Science* 80(6): 748-757.
- Kieth, H., Mackey, B.G., and Lindenmayer, D.B., 2009 Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proceedings of the National Academy of Sciences* www.pnas.org cgi doi: 10.1073/pnas.0901970106. 6 pp.
- Kleinbauer, I., Dullinger, S., Peterseil, J., and Essl, F., 2010. Climate change might drive the invasive tree *Robinia pseudacacia* into nature reserves and endangered habitats. *Biological Conservation* 143(2): 382-390.
- Körner, C., 2004. Through enhances tree dynamics carbon dioxide enrichment may cause tropical forests to lose carbon. *Philosophical Transactions of the Royal Society, London B* 359: 493-498.
- Körner C, Asshoff R, Bignucolo O Hättenschwiler, S., Keel, S. G., Pelaez-Riedl, S., Pepin, S., Siegwolf, R. T. W. & Zotz, G., 2005. Carbon flux and growth in mature deciduous forest trees exposed to elevated CO₂. *Science* 309: 1360–1362.
- Kotwal, P.C., Omprakash, M.D., Gairola, S., and Dugaya, D., 2008. Ecological indicators: Imperative to sustainable forest management. *Ecological Indicators* 8(1): 104-107.
- Kovats S, Ebi KL, Menne B, Campbell-Lendrum D, Canziani OF, Githeko A, 2003. Methods of assessing human health vulnerability and public health adaptation to climate change. *Health and Global Environmental Change. Series no.1. Copenhagen, Denmark: WHO/EURO.*
- Kozłowski, T. T. 1997. Responses of woody plants to flooding and salinity. *Tree physiology monograph*, No. 1. Heron Publishing, Victoria, British Columbia, Canada.
- Kozlov, M. V. and Berlina, N. G. 2002, Decline in Length of the Summer Season on the Kola Peninsula, Russia, *Climate Change* 54: 387-398.
- Krawchuk, M.A., Cumming S.G., and Flannigan, M., 2009. Predicted changes in fire weather suggest increases in lightning fire initiation and future area burned in the mixedwood boreal forest. *Climatic Change* 92: 83-98.
- Kreuzwieser, J., Papadopoulou, E., and Rennenberg, H., 2004. Interaction of Flooding with

Carbon Metabolism of Forest Trees. *Plant Biology* 6: 299-306.

- Krzyzanowski, J., I.G. McKendry, and J.L. Innes, 2006. Evidence of elevated ozone concentrations in the Lower Fraser Valley, British Columbia. *Water, Air and Soil Pollution* 73 (1-4): 273-287.
- Kumar, K.K., Rajagopalan, B., Hoerling, M., Bates, G., and Cane, M., 2006. Unraveling the mystery of Indian monsoon failure during El Niño. *Science* 314(5796): 115-119.
- Kurz, W.A., Dymond, C.C., Stinson, G., Rampley, G.J., Neilson, E.T., Carroll, A.L., Ebata, T., and Safranyik, L., 2008. Mountain pine beetle and forest carbon feedback to climate change. *Nature* 452: 987 -990.
- Lavery and Mead, 1998. *Pinus radiata*: a narrow endemic from North America takes on the world. In: *Ecology and Biogeography of Pinus*. D.M. Richardson (ed.), University of Cambridge Press, Cambridge UK. Pp: 432-449.
- Legge, A.H., Jager, H.-J., Krupa, S.V., 1998. Sulfur dioxide. In: Flagler, R.(Ed.), *Recognition of Air Pollution Injury to Vegetation: A Pictorial Atlas*. Air and Waste Management Association, Pittsburgh, PA, pp. 3/1e3/42.
- Lim, J.H., Chun, J.H., Woo, S.Y., Kim, Y.K., 2008. Increased declines of Korean fir forest caused by climate change in Mountain Halla, Korea. In: *International Conference on Adaptation of Forests and Forest Management to Changing Climate with Emphasis on Forest Health: A Review of Science, Policies, and Practices, Book of Abstracts and Preliminary Programme*. Umeå, Sweden, FAO/IUFRO, 25–28 August 2008.
- Lloret, F., Peñuelas, J., Prieto, P., Llorens, L., and Estiarte, M., 2009. Plant community changes induced by experimental climate change: Seedling and adult species composition. *Perspectives in Plant Ecology, Evolution and Systematics* 11: 53–63.
- Lusk, C. H., 2004. Leaf area and growth of juvenile temperate evergreens in low light: species of contrasting shade tolerance change rank during ontogeny. *Functional Ecology* 18:820–828.
- Mack MC, D'Antonio CM., 1998. Impacts of biological invasions on disturbance regimes. *Tree* 13: 195–198.
- Manning, W.J., Tiedemann, A.V., 1995. Climate change: potential effects of increased atmospheric carbon dioxide (CO₂), ozone (O₃) and ultraviolet-B (UV-B) radiation on plant diseases. *Environmental Pollution* 88: 219-245.
- Matsumoto K, Ohta T, Nakamura T, 2003. Climate change and extension of the Ginkgo bioloba L. growing season in Japan. *Global Change Biology* 9:1634–1642.
- McCarthy H.R., Oren R., Johnsen K.H., Gallet-Budynek A., Pritchard S.G., Cook C.W., LaDeau

- S.L., Jackson R.B., Finzi A.C. Reassessment of plant carbon dynamics at the Duke free air CO₂ enrichment site: interactions of atmospheric [CO₂] with nitrogen and water availability over stand development. *New Phytologist* 185:514-528.
- McLeod, A.R., and Long, S.P., 1999. Free-air Carbon Dioxide Enrichment (FACE) in Global Change Research: A Review. *Advances in Ecological Research* 28:1-56.
- Meehl, G.A., T.F. Stocker, W.D. Collins, P. Friedlingstein, A.T. Gaye, J.M. Gregory, A. Kitoh, R. Knutti, J.M. Murphy, A. Noda, S.C.B. Raper, I.G. Watterson, A.J. Weaver and Z.-C. Zhao, 2007. Global Climate Projections. In: *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- Michaelian, M., Hogg, E.H., Hall, R.J., and Arsenault, E., 2011. Massive mortality of aspen following severe drought along the southern edge of the Canadian boreal forest. *Global Change Biology* 17: 2084-2094.
- Miller, P.R., McBride, J.R., 1999. Oxidant Air Pollution Impacts in the Montane Forests of Southern California: A Case Study of the San Bernardino Mountains. In: *Ecological Studies*, Vol. 134. Springer, New York, 441 pp.
- Moore, R.D., Fleming, S.W., Menounos, B., Wheate, R., Fountain, A., Stahl, K., Holm, K., and Jakob, M., 2009. Glacier change in western North America: influences on hydrology, geomorphic hazards, and water quality. *Hydrological Processes* 23: 42-61.
- Mpelasoka, F., Hennessy, K., Jones, R., and Bates, B., 2008. Comparison of suitable drought indices for climate change impacts assessment over Australia towards resource management. *International Journal of Climatology* 28: 1283-1292.
- NOAA, 2012. National Oceanic and Atmospheric Association. Tides & Current. Regional Mean Sea Level Trends. URL: <http://tidesandcurrents.noaa.gov/sltrends/slrmap.html> [Accessed 29 May 2012].
- National Assessment Synthesis Team, 2000. Change Impacts on the United States. The Potential Consequences of Climate Variability and Change. US Global Change Research Program, Washington, DC, 522 pp.
- Nitschke, C.R. and Innes, J.L., 2008. Climatic change and fire potential in South-Central British Columbia, Canada. *Global Change Biology* 14: 841-855.
- Niinemets, U., Valladares, F., 2006. Tolerance to shade, drought, and waterlogging of temperate Northern Hemisphere trees and shrubs. *Ecological Monographs* 76: 521-547.
- Norby, R.J., Pastor, J., and Melillo, J., 1986. Carbon-nitrogen interactions in CO₂-enriched white

- oak: physiological and long-term perspectives. *Tree Physiology* 2 (1-2-3): 233-241.
- Norby, R.J., DeLucia, E.H., Gielen, B., Calfapietra, C., Giardina, C.P., King, J.S., Ledford, J., McCarthy, H.R., Moore, D.J.P., Ceulemans, R., De Angelis, P., Finzi, A.C., Karnosky, D.F., Kubiske, M.E., Lukac, M., Pregitzer, K.S., Scarascia-Mugnozza, G.E., Schlesinger, W.H. & Oren, R. 2005. Forest response to elevated CO₂ is conserved across a broad range of productivity. *Proceedings National Academy of Sciences* 102(50): 18052–18056.
- Ohara, T., Akimoto, H., Kurokawa, J., Horii, N., Yamaji, K., Yan, X., Hayasaka, T., 2007. An Asian emission inventory of anthropogenic emission sources for the period 1980-2020. *Atmospheric Chemistry and Physics Discussions* 7: 4419-4444.
- Ohsawa, M., 1995. Latitudinal comparison of altitudinal changes in forest structure, leaf-type and species richness in humid monsoon Asia. *Vegetation* 123, 3 – 10.
- Olszyk, D., Wise, C., VanEss, E., Apple, M., Tingey, D., 1998. Phenology and growth of shoots, needles, and buds of Douglas-fir seedlings with elevated CO₂ and (or) temperature. *Canadian Journal of Botany* 76: 1991–2001.
- Olszyk, D.M., Johnson, M.G., Tingey, D.T., Rygiewicz, P.T., Wise, C., VanEss, E., Benson, A., Storm, M.J., King, R., 2003. Whole-seedling biomass allocation, leaf area, and tissue chemistry for Douglas-fir exposed to elevated CO₂ and temperature for 4 years. *Canadian Journal of Forest Resources* 33: 269–278.
- Oren, R., Ellsworth, D. S., Johnsen, K. H., Phillips, N., Ewers, B. E., Maier, C., Schafer, K. V. R., McCarthy, H., Hendrey, G., McNulty, S. G., and Katul, G.G. 2001. Soil fertility limits carbon sequestration by forest ecosystems in a CO₂-enriched atmosphere. *Nature* 411, 469–472.
- Ohrnberger, O. 1999. *The bamboos of the world - Annotated Nomenclature and Literature of the Species and the Higher and Lower Taxa*. Amsterdam, the Netherlands, Elsevier. 585pp.
- Osawa, A. 1992. Development of mixed-conifer forest in Hokkaido, northern Japan, following a catastrophic windstorm: A “parallel” model of plant succession. In: M.J. Kelty (ed.) *The Ecology and Silviculture of Mixed-Species Forests*, Kluwer Academic Publishers, Netherlands, pp. 29-52.
- Osterkamp, T. E., Viereck, L., Jorgenson, M.T., Racine, C., Doyle, A. and Boone, R.D., 2000. Observations of thermokarst and its impact on boreal forests in Alaska, USA Arctic, Antarctic and Alpine Research 32: 303–15.
- Park, P.S., Lee, K.H., Jung, M.H., Shin, H., Jang, W., Bae, K., and Lee, J., 2008. The effect of environmental changes on forest distribution in South Korea. In: *International Conference on Adaptation of Forests and Forest Management to Changing Climate with Emphasis on*

Forest Health: A Review of Science, Policies, and Practices, Book of Abstracts and Preliminary Programme. Umeå, Sweden, FAO/IUFRO, 25–28 August 2008.

- Pierrard, J.M., 1969. Environmental Appraisal-Particulate Matter, Oxides of Sulfur and Sulfuric Acid. *Journal of the Air Pollution Control Association* 19(9): 632-636.
- Percy, K.E., Awmack, C.S., Lindroth, R.L., Kubiske, M.E., Kopper, B.J., Isebrands, J.G., Pregitzer, K.S., Hendrey, G.R., Dickson, R.E., Zak, D.R., Oksanen, E., Sober, J., Harrington, R., Karnosky, D.F., 2002. Altered performance of forest pests under CO₂- and O₃-enriched atmospheres. *Nature* 420: 403-407.
- Peterson, D.W., and Peterson, D.L., 2001. Mountain hemlock growth responds to climatic variability at annual and decadal time scales. *Ecology* 82(12): 3330-3345.
- Posch, M., 2002. Impacts of climate change on critical loads and their exceedances in Europe. *Environmental Science and Policy* 5(4): 307–317.
- Preston, B.L., Suppiah, R., Macadam, I. and Bathols, J., (2006) “Climate Change in the Asia/Pacific Region: A Consultancy Report Prepared for the Climate Change and Development Roundtable“, Melbourne, Climate Change Impacts and Risk, CSIRO Marine and Atmospheric Research, October, 89 pp.
- Price, C., Rind, D., 1994. The impact of a 2 x CO₂ climate on lightning-caused fires. *Journal of Climate* 7: 1484–1494.
- Quaas, J., Ming, Y., Menon, S., Takemura, T., Wang, M., Penner, J. E., Gettelman, A., Lohmann, U., Bellouin, N., Boucher, O., Sayer, A. M., Thomas, G. E., McComiskey, A., Feingold, G., Hoose, C., Kristjánsson, J. E., Liu, X., Balkanski, Y., Donner, L.J., Ginoux, P. A., Stier, P., Grandey, B., Feichter, J., Sednev, I., Bauer, S. E., Koch, D., Grainger, R. G., Kirkevåg, A., Iversen, T., Seland, Ø., Easter, R., Ghan, S. J., Rasch, P. J., Morrison, H., Lamarque, J.-F., Iacono, M. J., Kinne, S., and Schulz, M., 2009. Aerosol indirect effects general circulation model intercomparison and evaluation with satellite data, *Atmospheric Chemistry and Physics* 9: 8697–8717.
- Ranson, J., Kharuk, V., Ostorbin, P., Im, S., 2011. Drought-Caused Forest Decline In The Trans-Baikal Lake Area. American Geophysical Union, Fall Meeting 2011, Abstract #GC31A-1012.
- Rapalee, G., S.E.Trumbore, E.A. Davidson, J.W. Harden and H. Veldhuis, 1998. Soil carbon stocks and their rates of accumulation and loss in a boreal forest landscape. *Global Biogeochemical Cycles*, 12(4):687–702.
- Rasch, P. J., Tilmes, S., Turco, R. P., Robock, A., Oman, L., Chen, C.-C. J., Stenchikov, G. L. and Garcia, R. R., 2008. An overview of geoengineering of climate using stratospheric sulphate aerosols. *Phil. Trans. R. Soc. A* 366: 4007–4037. (doi:10.1098/rsta.2008.0131)

- Rigal, A., 2008. Exploring forest management solution in traditional knowledge system to address climate change issues. In: International Conference on Adaptation of Forests and Forest Management to Changing Climate with Emphasis on Forest Health: A Review of Science, Policies, and Practices, Book of Abstracts and Preliminary Programme. Umeå, Sweden, FAO/IUFRO, 25–28 August 2008.
- Roy, S.D., Beig, G., and Ghude, S.D., 2009. Exposure-plant response of ambient ozone over the tropical Indian region. *Atmospheric Chemistry and Physics* 9: 5253-5260.
- Rui, I., 2002. response to the Distribution of Bamboo to Climate Change in China During Historic Periods. *Journal of Bamboo Research* 2002-02. [In Chinese with English Abstract].
- Sánchez-Gómez , D., M. A. Zavala, and F. Valladares. 2006. Survival responses to irradiance are differentially influenced by drought in seedlings of forest tree species of the temperate-Mediterranean transition zone. *Acta Oecologica* 30(3): 322-332.
- Santoso, H., Hairiah, K., Tohari, A., Soedradjat, G.M., and Herawati, H., 2008. Impacts of climate change on landslides and using vegetation as an adaptation option to reduce the landslide risk. In: International Conference on Adaptation of Forests and Forest Management to Changing Climate with Emphasis on Forest Health: A Review of Science, Policies, and Practices, Book of Abstracts and Preliminary Programme. Umeå, Sweden, FAO/IUFRO, 25–28 August 2008.
- Seely, B., Welham, C., Kimmins. H., 2002. Carbon sequestration in a boreal forest ecosystem: results from the ecosystem simulation model, FORECAST. *Forest Ecology and Management* 169(1-2): 123-135.
- Seppälä, R., Buck, A., and Katila, P., (eds.) 2009. Executive Summary. In: Adaptation of Forests and People to Climate Change. A Global Assessment Report. IUFRO World Series Volume 22. Helsinki, Finland 224 pp.
- Shafer, S.L., Bartlein, P.J., and Thompson, R.S., 2001. Potential Changes in the Distributions of Wester North America Tree and Shrub Taxa under Future Climate Scenarios. *Ecosystems* 4:200-215.
- Singh, P.P., 2008. Exploring biodiversity and climate change benefits of community-based forest management. *Global Environmental Change* 18(3): 468-478.
- Sinha, S.K., 2012. Tree ring width of Teak (*Tectona grandis* L.f.) and its relationship with rainfall and temperature. *Indian Journal of Ecology* 39(39): 1-5.
- Smith, A.L., Hewitt, N., Klenk, N., Bazely, D.R., Yan, N., Wood, S., Henriques, I., MacLellan, J.I., and Lipsig-Mummé, C., 2012. Effects of climate change on the distribution of invasive alien species in Canada: a knowledge synthesis of range change projections in a

- warming world. *Environmental Reviews*: 20: 1-16.
- Stern, N., 2007. *The Economics of Climate Change*. Cambridge University Press, Cambridge UK, 712pp.
- Still, C., Foster, P.N., Schneider, S.H., 1999. Simulating the effects of climate change on tropical montane cloud forests. *Nature* 398, 608 – 610.
- Stocks, B. J., Fosberg, M. A., Lynham, T. J., Mearns, L., Wotton, B. M., Yang, Q., Jin, J.-Z., Lawrence, K., Hartley, G. R., Mason, J. A., and McKenney, D. W., 1998. Climate change and forest fire potential in Russian and Canadian boreal forests. *Climatic Change* 38: 1–13.
- Stocks, B.J., M.A. Fosberg, M.B. Wotton, T.J. Lynham and K.C. Ryan, 2000. Climate change and forest fire activity in North American boreal forests. In: E.S. Kasichke and B.J. Stocks (eds.). *Fire, Climate Change, and Carbon Cycling in the Boreal Forest*, Springer-Verlag. pp. 368–376.
- Stolbovoi, V. and Stocks, B.J., 2002. Carbon in Russian soils. *Climatic Change*, 55: 131–156.
- Sukhinin, A. I., Soja, A. J., McRae, D. J., Cahoon, D. R., Stocks, B. J., Dubrovskaya, O. A., Ji-Zhong, J., Flannigan, M., DeGroot, B., Westberg, D., Stackhouse, P. W., Conard, S. G., Hao, W. M., 2011. Catastrophic Fires in the Russian Federation. American Geophysical Union, Fall Meeting 2011, Abstract #GC41C-0833.
- Sukumar, R., Suresh, H. S., and Ramesh, R., 1995. Climate Change and Its Impact on Tropical Montane Ecosystems in Southern India. *Journal of Biogeography* 22 (2/3): 533-536.
- Taylor S, Kumar L, Reid N, Kriticos DJ, 2012. Climate Change and the Potential Distribution of an Invasive Shrub, *Lantana camara* L. *PLoS ONE* 7(4): e35565. doi:10.1371/journal.pone.0035565
- Thompson, I., Mackey, B., McNulty, S., Mosseler, A., 2009. *Forest Resilience, Biodiversity, and Climate Change. A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems*. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series no. 43, 67 pages.
- Turner, J., and Lambert, M., 2000. Change in organic carbon in forest plantation soils in eastern Australia. *Forest Ecology and Management* 133(3): 231-247.
- UNECE, (United Nations Economic Commission) 2005. *Europe's Forests in a Changing Environment*. Federal Research Centre for Forestry and Forest Products, Geneva, Switzerland. 60 pp.
- UNFCCC (United Nations Framework Convention on Climate Change), 2007. *Climate change impacts, vulnerabilities, and adaptation in developing countries*. Climate Change

- Secretariat. Bonn, Germany. 64pp.
- USAID, 2010. United States Agency for International Development. Asia-Pacific Regional Climate Change Adaptation Assessment. Final Report: Findings and Recommendations. International Resources Group, Connecticut, NW. 145pp.
- US Department of Health, Education, and Welfare, 1969. Air Quality Criteria for Particulate Matter. Washington, DC, 211pp.
- van Mantgem, P.J., Stephenson, N.L., Byrne, J.C., Daniels, L.D., Franklin, J.F., Fule, P.Z., Harmon, M.E., Larson, A.J., Smith, J.M., Taylor, A.H., Veblen, T.T., 2009. Widespread increase of tree mortality rates in the western United States. *Science* 323: 521–524.
- Valentini, R., S. Dore, G. Marchi, D. Mollicone, M. Panfyorov, C. Rebmann, O. Kolle and E.-D. Schulze, 2000. Carbon and water exchanges of two contrasting central Siberia landscape types: regenerating forest and bog. *Functional Ecology* 14(1): 87–96.
- Veblen, T.T., Kulakowski, D., Eisenhart, K.S., and Baker, W.L., 2001. Subalpine forest damage from a severe windstorm in northern Colorado. *Canadian Journal of Forest Research* 31(12): 2089-2097.
- Vitt, D.H., Halsey, L.A., Bauer, I.E., and Campbell, C., 2000. Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. *Canadian Journal of Earth Science* 37: 683-693.
- Walker, J.C.G., 1994. Global Geochemical Cycles of Carbon. In: Tolbert, N.E., and Preiss, J. (eds.) *Regulation of Atmospheric CO₂ and O₂ by Photosynthetic Carbon Metabolism*. Pp: 75-89.
- Walther, G.-R., Post, E., Menzei, A., Parmesan, C., Beebee, T.J.C., Fromentin, J.-M., Hoegh-Guldberg, O., and Bairlein, F., 2002. Ecological responses to recent climate change. *Nature* 416: 389-395.
- Wang, C., Gower, S.T., Wang, Y., Zhao, H., Yan, P., and Bond-Lamberty, B.P., 2001. The influence of fire on carbon distribution and net primary production of boreal *Larix gmelinii* forests in north-eastern China. *Global Change Biology*, 7(6):719–730.
- Wang, D., Bormann, F.H., Karnosky, D.F., 1986b. Regional tree growth reductions due to ambient ozone: evidence from field experiments. *Environmental Science & Technology* 20: 1122-1125.
- Wang, G., Innes, J.L., Yusheng, Y., Shanmu, C., Krzyzanowski, J., Jingsheng, X., and Wenlian, L., 2011. Extent of soil erosion and surface runoff associated with large-scale infrastructure development in Fujian province, China. *Catena* 89(1): 22-30.
- Wang, H.B., Zhang, Z., Kong, X.B., Lui, S.C., Shen, Z.R., 2007a. Preliminary deduction of

- potential distribution and alternative hosts of invasive pest, *Dendroctonus valens* (Coleoptera: Scolytidae). *Scientia Silvae Sinicae* 143(10): 71–76.
- Wang, X., Manning, W., Feng, Z., and Zhu, Y., 2007. Ground-level ozone in China: Distribution and effects on crop yields. *Environmental Pollution* 147(2): 394-400.
- Wang, Y., Flannigan, M., and Anderson, K., 2010. Correlations between forest fires in British Columbia, Canada, and sea surface temperature of the Pacific Ocean. *Ecological Modelling* 221(1): 122-129.
- Wardle, P., and Coleman, M. C., 1992. Evidence for rising upper limits of four native New Zealand forest trees. *New Zealand Journal of Botany* 30, 303-314.
- Wearne L. J. & Morgan J. W. (2001) Recent forest encroachment into subalpine grasslands near Mount Hotham, Victoria, Australia. *Arctic Antarctic Alpine Res.* 33: 369–77.
- West, J.J., Fiore, A.M., Naik, V., Horowitz, L.W., Schwarzkopf, M.D., Mauzerall, D.L., 2007. Ozone air quality and radiative forcing consequences of changes in ozone precursor emissions. *Geophysical Research Letters* 34, L06806.
- Wigley, T.M.L., 2006. A Combined Mitigation/Geoengineering Approach to Climate Stabilization. *Science* 314(5798): 452-454.
- Williams, K., M. MacDonald, and L. D. L. Sternberg. 2003. Interactions of storm, drought, and sea-level rise on coastal forest: A case study. *Journal of Coastal Research* 19: 1116–1121.
- Wood, S.W., Hua, Q., Allen, K.J., and Bowman, D.M.J.S., 2010. Age and growth of a fire prone Tasmanian temperate old-growth forest stand dominated by *Eucalyptus regnans*, that world's tallest angiosperm. *Forest Ecology and Management* 260(4): 438-447.
- Woods, A., Coates, K.D., and Hamann, A., 2005. Is an unprecedented dothistroma needle blight epidemic related to climate change? *Bioscience* 55, 761–769.
- Wrona, F.J., Prowse, T.D., and Reist, J.D., 2005. Chapter 8. Freshwater ecosystems and fisheries. *In: Arctic Climate Impact Assessment, Scientific Report*. Cambridge University Press. pp. 353-452.
- Wu, J., Krutovskii, K., and Strauss, S.H., 1999. Nuclear DNA diversity population differentiation and phylogenetic relationships in the California close-cone pines based on RAPD and allozyme markers. *Genome* 42:893–908.
- Yang, W. and Omaye, S.T., 2009. Air pollutants, oxidative stress and human health. *Mutation Research* 674: 45-54.

- Zhang, Q.-B. and Hebda, R.J., 2004 Variaiton in radial growth patterns of *Pseudotsuga menziesii* on the central coast of British Columbia, Canada. *Canadian Journal of Forest Research* 34(9): 1946-1954.
- Zhang, X.Y., Gong, S.L., Zhao, T.L., Arimoto, R., Wang, Y.Q., Zhou, Z.J., 2003. Sources of Asian dust and role of climate change versus desertification in Asian dust emission *Geophysical Research Letters* 30(24): 2272 (ASC 8-1 – ASC 8-4).
- Zhao, T.L., Gong, S.L., Zhang, X.Y., Blanchet, J.-P., McKendry, I.G., and Zhou, Z.J., 2006. A simulated climatology of Asian dust aerosol and its trans-Pacific transport 1. Mean climate and validation. *Journal of Climate* 19: 88–103.