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ORIGINAL PAPER

Plantation forests, climate change and biodiversity

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Abstract Nearly 4 % of the world's forests are plantations, established to provide a variety of ecosystem services, principally timber and other wood products. In addition to such services, plantation forests provide direct and indirect benefits to biodiversity via the provision of forest habitat for a wide range of species, and by reducing negative impacts on natural forests by offsetting the need to extract resources. There is compelling evidence that climate change is directly affecting biodiversity in forests throughout the world. These impacts occur as a result of changes in temperature, rainfall, storm frequency and magnitude, fire frequency, and the frequency and magnitude of pest and disease outbreaks. However, in plantation forests it is not only the direct effects of climate change that will impact on biodiversity. Climate change will have strong indirect effects on biodiversity in plantation forests via changes in forest management actions that have been proposed to mitigate the effects of climate change on the productive capacity of plantations. These include changes in species selection (including use of species mixtures), rotation length, thinning, pruning, extraction of bioenergy feedstocks, and large scale climate change

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driven afforestation, reforestation, and, potentially deforestation. By bringing together the potential direct and indirect impacts of climate change we conclude that in the short to medium term changes in plantation management designed to mitigate or adapt to climate change could have a significantly greater impact on biodiversity in such plantation forests than the direct effects of climate change. Although this hypothesis remains to be formally tested, forest managers worldwide are already considering new approaches to plantation forestry in an effort to create forests that are more resilient to the effects of changing climatic conditions. Such change presents significant risks to existing biodiversity values in plantation forests, however it also provides new opportunities to improve biodiversity values within existing and new plantation forests. We conclude by suggesting future options, such as functional zoning and species mixtures applied at either the stand level or as fine-scale mosaics of single-species stands as options to improve biodiversity whilst increasing resilience to climate change.

Keywords Conservation · Forestry · Landscape ecology · Indirect effects · Climate/global change

Introduction

Climate change is expected to be one of the main future causes of biodiversity loss worldwide (Sala et al. 2000; Millennium Ecosystem Assessment 2005), and there is compelling evidence that climate change will result in the extinction of species from many taxa (Thomas et al. 2004). Trees (and therefore forest ecosystems) are particularly sensitive to climate change as they are relatively long-lived compared to other organisms and have limited adaptive capacity to respond to rapid environmental change (Lindner et al. 2010). Furthermore, their longevity may paradoxically prevent managers and society from detecting changes before important changes have already occurred. The potential impacts of climate change on natural forests and their capacity to provide numerous ecosystem services has been the subject of intensive research. There is evidence that links biodiversity conservation to the improved provision of ecosystem services across a range of ecosystem types (Balvanera et al. 2006), however the lack of data from forest systems has hindered the adoption of these concepts by forest policy makers (Thompson et al. 2011).

Climate change is predicted to alter: the relative abundance of tree species within forests (Condit et al. 1996); tree phenology (seasonality of flowering, bud burst and fruiting) (Schwartz et al. 2006; Beaubien and Hamann 2011) that could disrupt producer-herbivore dynamics (Visser and Holleman 2001)); frequency and intensity of key forest disturbance mechanisms, including wind and fire events; and the population dynamics of forest pests and pathogens (Logan et al. 2003). While many of these mechanisms have been studied in natural and semi-natural forests, the broad concepts are also relevant to plantation forests. However, some mechanisms will differ significantly in the effects they induce in plantations compared to natural forests. For example, plantation forests are normally dominated by only one or few tree species with limited genetic diversity, which may render them more susceptible to the consequences of climate change, such as changing pest dynamics. As afforestation, reforestation and forest restoration are key strategies to mitigate climate change, it is important to anticipate how climate change may affect new plantations and their ability to provide habitat for biodiversity.

While the establishment of plantation forests that replace natural vegetation typically causes biodiversity losses locally, plantations established on former agricultural or otherwise degraded land may provide significant opportunities for biodiversity conservation (Parrotta et al. 1997; Carnus et al. 2006; Loyn et al. 2007; Brockerhoff et al. 2008; Paquette et al. 2009; Pawson et al. 2010a) and deliver important ecosystem services (Winjum and Schroeder 1997; Bauhus and Schmerbeck 2010), even in the most intensively managed monocultures (Updegraff et al. 2004). In particular plantation forests are most important in highly fragmented landscapes where they may represent a large proportion of remaining forest habitat (Brockerhoff et al. 2005) and can provide corridors between habitats.

Despite their relatively small extent at the global scale, plantation forests are the focus of much debate regarding forest sustainability and biodiversity conservation. Plantation forests have been called "biological deserts" (Stephens and Wagner 2007) and some even argue that "plantations are not forests" (Carrere 2004). The poor opinion of plantation forests is sometimes deserved but just as often it is simply misinformed (Paquette and Messier 2010), and the impact of plantation forests on biodiversity will depend on what land use they replace. In a world where there are large areas of degraded (formerly forested) land suitable for reforestation, those plantations that replace natural forests rightly deserve criticism. However, those established on formerly forested, now anthropogenic grasslands, are more likely to confer net environmental benefits. These plantation forests, while predominantly managed for economic timber-related products, can provide important habitat for biodiversity, particularly in fragmented landscapes and where trees are grown on long rotations. An emerging area of research, and policy development, relates to the economic valuation of biodiversity within plantation forests. For example, trading systems, e.g., subsidies and incentives, for biodiversity have been proposed by The Economics of Ecosystems and Biodiversity study (TEEB 2010).

The potential of unmanaged natural forests to adapt to climate change impacts is somewhat limited, particularly when the dispersal of their component species (to more compatible latitudes, altitudes or aspects) is severely restricted by biological constraints and/or anthropogenic modification of landscapes relative to the potentially rapid changes in climate parameters (Travis 2003). In contrast, the adaptive potential of plantation forests is far greater as forest managers can alter silvicultural regimes and tree species composition to maintain the productive and thus economic capacity of these forests to adapt to, or mitigate, the effects of climate change. As a result, the impact of climate change on biodiversity in plantation forests will be the product of an interaction between the direct impacts of climate change and the indirect effects of new management approaches taken to address these predicted impacts. This warrants an in-depth analysis of climate change impacts with a focus on plantation forests.

Although, the outcome of proposed adaptation strategies (Johnston et al. 2006; Howden et al. 2007; Watt et al. 2008; Lindner et al. 2010; FAO 2012) is uncertain, alternative management actions have the potential to improve biodiversity in existing plantation forests. For example, afforestation to mitigate climate change could employ multi-purpose plantation designs (potentially incorporating new species) that confer greater resistance of these plantation forests to stressors, and require less management intervention (Paquette and Messier 2010).

In this paper we structure our appraisal of climate change impacts on biodiversity in plantation forests into two broad themes. Firstly we summarise the potential abiotic and biotic impacts of climate change on biodiversity in existing plantation forests that could occur irrespective of forest management practices. Secondly we consider two broad groups of potential forest management responses to climate change and how they may affect opportunities for biodiversity conservation in plantation forests: (A) changes in the management of existing plantations to protect them from the effects of climate change, and (B) how climate-mitigation driven changes in plantation extent (i.e., afforestation/reforestation or deforestation programmes) may affect biodiversity in plantation forests. For the most part we focus on biodiversity impacts at the species level, or at the habitat scale when referring to landscape level changes in biodiversity. In part, this restriction reflects the available literature as most studies in plantation forests examine species level responses to forest management or climate change.

Scope of literature considered

When gathering reference material to support the concepts expressed in this paper we combined formal searches, using the ISI Web of Knowledge database and the Google Scholar search engine, with the experience of individual co-authors to compile relevant and pertinent literature. When searching databases the following terms were used singularly or in combination: Climate change and related terms (fire, rainfall, temperature, mitigation), biodiversity and related terms (riparian, aquatic, bird, mammal, plant, understorey), plantations and related terms (plantation forests, thinning, stocking, pruning, planted), pests, pathogens, diseases, bioenergy, residue removal. Where possible, references were restricted to those relating specifically to plantation forests. However many potential impacts of climate change on biodiversity have not been studied in plantation forests, but have been researched within the broader concept of a planted forest (see Box 1 for concepts and definitions). In these circumstances we have presented evidence from planted forests to provide an indication of what the impact may be in plantation forests. However, we recognise that plantation forests may not respond in the same manner as planted forests given the differences in management practices. These knowledge gaps are identified where relevant and provide an indication of future research requirements.

Potential impacts of climate change on biodiversity in current plantations

Abiotic disturbance

Climatic

In addition to effects of rising temperatures on forest biodiversity (Table 1), current climate change scenarios predict significant changes to regional rainfall and storm patterns (IPCC 2007; Seppälä et al. 2009). Future temperature-driven changes in biodiversity values within plantation forests are likely to be the result of an interaction between temperature and other limiting climatic factors, such as rainfall. For example, the diversity of understorey plant species has been shown to be highly dependent on available rainfall in New Zealand plantations (Brockerhoff et al. 2003), thus the effects of any future rise in temperatures is likely to be dependent on changes in rainfall patterns. Additional secondary effects are also predicted, such as altered wildfire severity and frequency (Table 1). It remains difficult to accurately predict the effects that such climatic-induced changes will have on biodiversity in plantation forests. Outcomes are likely to be context dependent due to such variables as tree species (and genotypes) and their silvicultural management, e.g., differences in pruning, thinning, harvesting, and pre-planting site preparation activities, such as herbicides for weed control and harvest slash removal.

Box 1 A matter of definition planted forest versus plantation forest

The Food and Agriculture Organisation (FAO) recognises that plantation forests and planted semi-natural forests are not materially different from each other, and unite these two forest categories under a single concept of planted forests (FAO 2006). Planted forests are broadly defined by the FAO, as forests established by planting, seeding or coppicing (FAO 2006). In this review we focus on the plantation forest sub-set of planted forests, which is defined by the FAO as "A forest of introduced species and in some cases native species, established through planting or seeding *mainly for production of wood or non-wood forest products*" (FAO 2006). In the media, "plantation" is a term frequently used without differentiating between particular types of plantations. For example, the Indonesian oil-palm industry is routinely compared with other types of "man-made forests" (e.g. reforestation, mixed-species plantations, enrichment planting) without clearly defining the former as agricultural plantations and the latter as plantation forests. The key criteria of the FAO plantation forest sub-category is the objective of the forest and the intensity of management, i.e., those managed primarily for timber and other wood products (i.e., pulp fibre, bioenergy), plus those that in the future may be established/managed primarily for carbon sequestration. Such plantations form a continuum from short-rotation (i.e., 7–8 years) to older (i.e. 50+ years) aged stands, and include even-aged stands of exotic or native species that are intensely managed, to planted 'semi-natural' stands of mixed native species (Brockerhoff et al. 2008). While many plantation forests are established by government agencies and timber companies, extensive plantings of small holdings (<5 ha) now exceed the areas of plantation forest established by governments and corporations, particularly in tropical areas (FAO 2006). Although the global forest area is declining (~13 million ha per annum), the area of planted forests has been increasing annually by an average of 5 million ha between 2000 and 2010 and now represents ~7 % (264 million ha) of the global forest area (FAO 2010). Similarly the extent of the plantation forest subset has been increasing in area by ~2 % annually between 1990 and 2005 to an estimated 140.8 million ha, or ~4 % of the global forest area (FAO 2006). Future afforestation/reforestation rates may increase the proportion of plantation forests due to recent economic incentives to sequester carbon, i.e., "carbon forestry".

A key characteristic of climate change is the projected increase in climatic variability and the greater risk of extreme weather events in some regions (Table 1). Extreme rainfall events and associated flooding are predicted to increase under future climate projections (IPCC 2007), and although this has not been studied in detail in plantation forests it is known that these floods significantly damage riparian forest (Kramer et al. 2008). Riparian areas provide important (frequently native/natural) habitat for terrestrial biodiversity within plantation forests (Langer et al. 2008; Nasi et al. 2008; van Halder et al. 2008; Baker et al. 2009; Hsu et al. 2010). Given that riparian strips can represent a significant proportion of the available natural forest habitat in plantations of exotic tree species (Nasi et al. 2008; van Halder et al. 2008), greater flood disturbance would have a disproportionately large effect on biodiversity in plantation forest dominated landscapes compared to natural forests. However, we acknowledge that floods are important natural processes and more frequent extreme rainfall may in fact restore riparian processes in landscapes where the high water demand of some plantation forest species has lowered river flows. However, this concept has not been tested yet, and it is likely to be difficult to assess at this time.

Changes to abiotic factors in soils and their impacts on soil fauna

The complex network of abiotic and biotic interactions occurring in soils is integral to forest health and provides habitat for many endogenous species. Future climate scenarios predict changes to abiotic factors (increasing soil temperatures and regional changes to soil moisture, including the frequency of wetting and drying cycles) that will impact on the biodiversity of soil organisms (Table 1). Increasing atmospheric CO_2 concentration is predicted to "fertilise" forests and thus increase productivity. However, this fertilisation effect is likely to be limited by available nitrogen (Hyvonen et al. 2007) and water. Initially

Table 1 Potential direct effects of climate change on biodiversity in forest ecosystems that are likely to occur in plantation forests

Factor	Direct effects	Particular examples
Rising temperature	Adverse effects on heat-sensitive species (Welbergen et al. 2008)	Increasing abundance of thermophilic species (Reid 2006). Changes in species richness patterns (Roder et al. 2010)
Fire frequency	Changing fire frequency (Overpeck et al. 1990; Pitman et al. 2007)	Increase in disturbance adapted species (Aubin et al. 2008)
Extreme storm events	Greater frequency and magnitude (Leckebusch et al. 2006; IPCC 2007) and Shifting storm tracks (Bengtsson et al. 2006)	Large storms in Europe, USA, and Australia have devastated plantations, see Box 1, (Oswalt et al. 2008; Lindroth et al. 2009; Blennow et al. 2010; Kanowski et al. 2008).
Soil abiotic factors	Increased soil temperature and altered frequency of wetting and drying cycles	Changes in the abundance and community composition of soil fauna (Lindberg et al. 2002; Allison and Treseder 2008; Briones et al. 2004; Taylor et al. 2004; David and Handa 2010)
	Changing quantity and quality of key soil carbon inputs, such as leaf litter (Beedlow et al. 2004; Couteaux and Bolger 2000)	
Pests and diseases	Shifts in pest range (Battisti et al. 2005; Netherer and Schopf 2010)	Latitudinal and altitudinal spread of the pine processionary moth beyond its previous range in Europe (Battisti et al. 2005; Netherer and Schopf 2010)
	Reduced winter mortality and increased number of pest generations per year leading to larger and more persistent pest outbreaks (Bale et al. 2002; Carroll et al. 2004)	Outbreak of the mountain pine beetle in pine forests in British Columbia and other parts of western North America (Bale et al. 2002; Carroll et al. 2004)
	Increasing rainfall may lead to increased disease severity (Woods 2011)	Dothistroma needle blight in pine plantations in Canada is correlated with increasing amounts of rainfall (which may be caused by climate change in some regions) (Woods 2011)
Biocontrol	Potential for reduced effectiveness of current biocontrol agents due to phenological or other changes under climate change	

there was concern that nitrogen limitation would affect soil macroinvertebrates and litter decomposition rates due to a reduction in nitrogen concentration and subsequent litter quality (Couteaux and Bolger 2000). However, recent research suggests that invertebrates can adapt to changes in resource quality and that changes in abiotic factors will have a greater impact on biodiversity (David and Gillon 2009).

Biotic impacts

Pest and disease organisms are influenced by climate change (Table 1), and this is likely to affect biodiversity in a number of ways, both directly and indirectly. Range shifts have already been documented for the pine processionary moth (*Thaumetopoea pityocampa*) in Europe, and outbreaks are likely to affect new regions and larger areas than they have in the past (Battisti et al. 2005; Netherer and Schopf 2010). Indeed, outbreaks of the pine processionary moth in the Landes plantation forest in 2009 and 2010 have caused extensive defoliation that is further exacerbating the post-storm recovery of Europe's largest pine plantation. The unprecedented severity and size of the outbreak of the mountain pine beetle (*Dendroctonus ponderosae*) in pine forests in British Columbia is thought to be caused, in part, by warmer winter temperatures and increased annual heat sums (Carroll et al. 2004). The resulting large-scale tree mortality is likely to affect insectivorous bird diversity as spikes and troughs of food and nest-site availability occur following the beetle outbreak (Martin et al. 2006). Although this outbreak is clearly changing the structure and species composition of the forest (Vyse et al. 2009), the effects on plant diversity in the regenerating forests have not yet been established. However, it is clear that the widespread and considerable increase in early successional habitat and the loss of mature forests will affect many species that occur in British Columbia's pine forests. Although the observations on the pine processionary moth and the mountain pine beetle are not specific to plantation forests, their impacts will apply to natural forests, planted forests, and plantation forests.

Range shifts and changes in severity induced by climate change also occur for a number of diseases. For example, outbreaks of Dothistroma needle blight (*Dothistroma pini*) in pine plantations in Canada are correlated with increasing amounts of rainfall, a possible outcome of climate change in some regions (Woods 2011). Changes in *Mycosphaerella* leaf disease severity may affect rotation lengths in *Eucalyptus globulus* plantations (Pinkard et al. 2010), and changing (shortened) rotation lengths are expected to impact understorey forest biodiversity (Brockerhoff et al. 2008). Similarly, the severity of Swiss needle cast disease in Douglas fir plantation forests in New Zealand is expected to increase with warmer winter temperatures (Watt et al. 2010b). Indirect effects of Swiss needle cast on biodiversity are probable as a reduction in needle density will affect understory light levels and therefore, understory plant communities. Such impacts of tree diseases could have negative or positive effects on plantation forest biodiversity. Similarly, in some circumstances insect outbreaks may be temporarily beneficial for biodiversity. For example, outbreaks of defoliating insects can provide an abundant food source for bird species that feed on them (Pimentel and Nilsson 2009; Barbaro and Battisti 2010) and, potentially insect predators.

Climate change could affect pest impacts by influencing relationships between pests and their natural enemies. Biological control has been used widely in plantation forests, with many cases of successful control. For example, biocontrol agents have been introduced for vegetation management (Scott and Evans 2002), particularly to reduce vegetation competition from invasive weeds during plantation tree seedling establishment (Watson et al. 2011). Similarly, parasitoids have been introduced to control a number of invasive insect pests such as the woodwasp *Sirex noctilio* (Hurley et al. 2007). Controlling invasive plants and insects can create significant economic (Jarvis et al. 2006) as well as biodiversity benefits (Hanula and Horn 2011). However, there is concern that the performance of existing biological control agents could be compromised in the future, due to climatic mismatch (Barbaro and Battisti 2010), yet there is optimism that biocontrol agents will respond in synchrony with target pest species to changes in climate (Watt et al. 2010a). It is difficult to predict the impact of climate change on the population dynamics of both biocontrol agents and target pest species. However, it is possible that some pest and weed species that are currently under biocontrol will again be problematic as these delicate systems are increasingly influenced by climate change.

Finally, the threat from new pest species is likely to increase as climate change may enable some species to persist in regions where current climatic conditions prevent their establishment. For instance, New Zealand's current temperate climate precludes the establishment of numerous pests occurring in warmer regions, but under future climates some of these pests will probably encounter climatically suitable conditions (Kriticos 2012). As alien invasive species are a significant global threat to biodiversity (Sala et al. 2000), these changing pest threats will present additional challenges for biodiversity.

Management of existing plantations to adapt to climate change and their potential impacts on biodiversity

A critical feature of plantation forests as opposed to semi-natural or unmanaged natural forests is the degree and intensity of management activities. To prepare for potential climate change a variety of mitigation and adaptation strategies have been suggested (Johnston et al. 2006; Howden et al. 2007; Watt et al. 2008; Lindner et al. 2010; FAO 2012). The extent to which these mitigation strategies are implemented will vary between countries due to differences in financial and technical resources (Lindner et al. 2010). However, we suggest that in the short-to-medium-term, changes to forest management in plantation forests that are implemented as part of mitigation and adaptation strategies could have a greater impact on biodiversity than the direct effects of climate change on such plantation forests. This hypothesis is based on the contrast between the largely slow, cumulative, direct effects of climate change discussed above compared to the immediate changes that can occur when plantation forest owners alter their management of large areas (discussed below). Unlike other productive land uses, e.g., agriculture, such change will be longer-lasting given the rotation length of plantation forests.

Risk-spreading approaches to tree species selection for climate change mitigation

Climate change is predicted to shift the bioclimatic envelope of many important tree species currently used in plantations (Watt et al. 2008; Lindner et al. 2010). For example, a decline in the productivity of key European plantation forest species, such as *Picea abies* or *Pinus pinaster*, is predicted due to a combination of rising temperatures and hydric stress at low latitudes and altitudes (Lexer et al. 2002; Mora et al. 2012). To maintain timber production in the face of climatic uncertainty, the adoption of a risk-spreading approach (analogous to the 'portfolio effect' (Tilman et al. 1998) or insurance hypothesis (Yachi and Loreau 1999)) that relies on a diversification of tree species is frequently proposed (Bauhus and Schmerbeck 2010; Milad et al. 2012). Diversification is important at both inter- and intraspecific genetic scales with new species or genotypes planted as monocultures and the expansion of mixed species stands (Erskine et al. 2006; Kelty 2006; Knoke et al. 2008; Schueler et al. this issue).

At the stand level, the capacity to adapt while maintaining both wood production and biodiversity could benefit from increasing crop species diversity. The current resistance of many foresters to expanding mixed species plantings is the perception that it reduces yield (Knoke et al. 2008) and complicates forest management operations. However, there is increasing evidence from both natural forests (Paquette and Messier 2011) and plantations (Erskine et al. 2006; Vila et al. 2007; Plath et al. 2011) that tree diversity can either improve, or has no detrimental effect on, stand productivity. Mixed plantings have the additional benefit that they are likely to be more resilient to future societal (cultural, economical) and environmental (stability facing global change, biodiversity conservation) challenges, including climate change (c.f. Thompson et al. 2009). For example, mixed species stands have been found to be more resistant to various forms of damage, and to be more diverse in their fauna and flora than pure, single-species stands (Spiecker 2003). Less diversified systems, such as monoculture plantations, may also be less resistant and resilient to natural disturbance (Drever et al. 2006) or pests (Jactel and Brockerhoff 2007), something that global change may exacerbate (Woods et al. 2005). Mixed species can also have some financial advantages that make them more attractive, especially to smallholders. For example, crops can be grown in the understory or some fast-growing species can be harvested earlier than slower-growing species to generate an early return (Paquette et al. 2008; Rivest 2009). Greater diversity of forest products provides economic certainty for local communities, which promotes land use stability that has subsequent beneficial effects for biodiversity conservation (Paquette et al. 2009; Lamb 2011).

Trials are also underway at large, operational scales to demonstrate the feasibility of increasing diversity locally using multi-species plantation forests, even of the intensive, fast-growing type where spacing within and between rows are designed a priori to optimize the rotation length of all components and allow tending and harvest operations with present equipment (Paquette and Messier 2013). In other cases diversity is achieved within the landscape using mosaics of patches or corridors of forest remnants or plantations of different species, or even simply by varying stand age and rotation length (Lamb 1998; Carnus et al. 2006; Paquette and Messier 2010).

The biodiversity conservation outcomes from changing canopy composition from a monoculture of one species to another will be context-dependent. While some tree species provide habitat for particular species (of birds, mammals, insects and understory plants), a change in planted tree species may be detrimental to these species but potentially beneficial for a range of other species.

Animal diversity is frequently related to plant diversity, thus the presence of a more diverse canopy (or understory) is likely to result in greater biodiversity opportunities within plantations (Recher et al. 1987; Lindenmayer and Hobbs 2004). Empirically species mixtures have been shown to increase resilience to changing conditions (Yachi and Loreau 1999), and in some instances they are known to improve productivity in both planted (Forrester et al. 2006; Potvin and Gotelli 2008; Kanowski and Catterall 2010) and natural forests (Paquette and Messier 2011). In addition to productivity and stabilising influences, mixed species plantation forests can enhance biodiversity opportunities by diversifying habitat within stands (Hartley 2002; Lindenmayer and Hobbs 2004; Erskine et al. 2006; Brockerhoff et al. 2008; Wardell-Johnson et al. 2008; Felton et al. 2010).

Changes to silvicultural management

Plantation managers are re-examining basic silvicultural strategies, such as pre-planting site preparation, stocking rates, weed control, thinning, and pruning to ensure forest resilience to climate change (Puettmann et al. 2009; Puettmann 2011; Messier et al. 2013), and to take advantage of market instruments created for climate change mitigation (Adams and Turner 2012). Predicting the biodiversity impacts of such silvicultural changes is difficult. Apart from harvesting, thinning is probably the most studied silvicultural practice and in most cases neutral or positive effects on biodiversity have been observed (Ohsawa 2004; Maleque et al. 2010; Verschuyl et al. 2011). Reduced thinning (e.g., stands planted specifically for carbon sequestration purposes (Adams and Turner 2012)) can decrease understorey plant diversity (Brockerhoff et al. 2003, however see Taki et al. 2010) and community composition (Taki et al. 2010) in plantations. Flow-on effects on other groups, such as invertebrates and birds (Taki et al. 2010; Verschuyl et al. 2011) have been observed; thus the full impact of reduced thinning on biodiversity is not yet known.

In certain circumstances, changes to pruning regimes may be detrimental to individual species. For example, in Chilean pine plantations the Andean tapaculo (*Scytalopus magellanicus fuscus*) and the ochre-flanked tapaculo (*Eugralla paradoxa*) are both positively associated with the presence of dead branches that are less abundant in pruned stands (Vergara and Simonetti 2006). However, in general changes to pruning regimes are likely to have a smaller impact on biodiversity than altering stocking or thinning rates.

Rotation length

The rotation length is usually determined by economic circumstances such as whether the timber is destined for pulpwood or biomass energy (in which case a short rotation <10 years may be chosen) or whether timber will be used for saw logs (in which case rotations may span several decades). Different species are used in each case. Fast-growing species are usually grown on short rotations while slower growing (but often more valuable) species are grown on longer rotations. Although they are planted as single species stands, many monocultural plantations are colonized by other plant species and wildlife as the stands age (Hartley 2002; Lindenmayer and Hobbs 2004; Carnus et al. 2006; Aubin et al. 2008; Brockerhoff et al. 2008). Young stands are often more suitable for open-habitat or ruderal species (Eycott et al. 2006; Pawson et al. 2009; Archaux et al. 2010), whereas older stands are often colonised by a diverse range of understorey species and support forest adapted wildlife species, particularly when moisture is not limiting (Keenan et al. 1997; Brockerhoff et al. 2003; Archaux et al. 2010; Pawson et al. 2011). In some circumstances 'carbon farming' could result in benefits for biodiversity, such as increased understorey plant diversity if their management involves increases in rotation length. However, this will occur only if areas of natural forest are nearby to provide propagules, or if partial cuts retain mature plantation stands as source populations. Nevertheless, rotation lengths are unlikely to extend to the point that old-growth stand conditions are created, which are a key constraint for extending biodiversity gains (Aubin et al. 2008). Furthermore, benefits from greater rotation lengths in plantation forests designed for carbon farming may be offset by the implementation of increased stocking rates and reduced thinning to maximise carbon (see above).

Conversely, strategies to adapt plantation forest management to climate change, or to mitigate climate change directly, could result in dramatic reductions to the rotation length of existing plantation areas. For example, shorter rotations may be adopted to avoid abiotic risks, such as increased future storm frequency (see Box 2, and Kanowski et al. (2008)); or traditional long-rotation timber plantations may be converted to produce short-rotation biomass tree crops to substitute fossil fuels and reduce greenhouse gas emissions. Such reductions in the rotation length of existing plantation forests will decrease opportunities for biodiversity (Lindenmayer and Hobbs 2004; Carnus et al. 2006; Brockerhoff et al. 2008). However, the impact of large scale plantings of new short-rotation woody biomass crops on biodiversity is more difficult to predict (Webster et al. 2012). Benefits may accrue if tree crops are planted on degraded agricultural land, but biodiversity opportunities will be context-dependent, e.g., displacing food production could promote further clearance of natural forests, and limited by the intensity of the silvicultural management applied in such systems.

Box 2 Extreme storms reshape the Landes Forest

The Landes Forest, Europe's largest plantation forest (located in the south-west of France) was devastated by cyclone Martin in 1999 (uprooting or breaking 23 million m^3 of standing wood that was salvage harvested where possible) and cyclone Klaus in 2009 (37 million m^3 of wood requiring salvage) (Fig. 1). Within 10 years, the Landes region had lost almost 45 % of its standing timber volume. Furthermore, subsequent bark beetle outbreaks facilitated by the increased volume of dead wood are compounding the initial damages by attacking wind stressed trees. As of 2012, ca. 4 million m^3 of standing wood is infested, primarily by *Ips sexdentatus* and *Orthotomicus erosus*. Such severe storms have significant short term impacts on biodiversity in plantation forests by altering the availability of resources, e.g. dead wood, and stand structure (Peterson 2000; Bouget and Duelli 2004; Bouget 2005). However, the potential increase in extreme storm frequency due to climate change in the Landes may result in substantial long-lasting impacts on biodiversity at a regional scale. For instance, the two recent storms have led to a government-established expert panel to evaluate the future of the Landes Forest. Maritime pine (*Pinus pinaster*) is likely to remain the predominant tree species, although diversification is suggested at the landscape scale, with increased planting of loblolly pine (*Pinus taeda*, L.), *Eucalyptus* spp. and black locust (*Robinia pseudoacacia*, L.), all exotic species (Mora et al. 2012). Diversification may provide new biodiversity opportunities; however it could also facilitate invasion of additional exotic species into remaining native habitat, e.g., black locust (Katona et al. this issue). The conservation and expansion of areas of native deciduous tree species that are not used for timber production is also recommended to increase the resistance of pine stands to pests and diseases (Koricheva et al. 2006; Jactel and Brockerhoff 2007), and will have substantial benefits for biodiversity (van Halder et al. 2008). In addition to diversification, new management schemes that shorten rotations (to reduce risk of storm damage or to produce biomass) could be considered (Mora et al. 2012). It is well known that the value of plantations for biodiversity increases with stand age (Brockerhoff et al. 2008), thus the proposed amendments would have negative effects on biodiversity in the Landes.

Removal of residual biomass for bioenergy

The extraction of wood residues from traditional logging operations in plantation forests is a comparatively simple cost-effective method to obtain bioenergy feedstocks, as it utilises a 'waste' product that is available immediately. However, extraction of harvest residues reduces the quantity and temporal availability of large diameter coarse woody debris in forests (Verkerk et al. 2011), which is an important habitat for saproxylic (deadwood dependent) invertebrates. In Sweden, Dahlberg et al. (2011) estimated that the extraction of logging residues for bioenergy will reduce above-ground fine woody debris by as much as 35–45 %, which represents a 20 % decline in the available substrate for 50 % of saproxylic species in Norway spruce forests. Despite these dramatic reductions in habitat, Dahlberg et al. (2011) conclude that residue utilisation at current rates will have a minimal impact on regional extinction risk because log residues in Norway spruce plantations are not the primary habitat for any red listed taxa. However, in North America, Riffell et al. (2011) conducted a broad meta-analysis of 745 biodiversity data sets from 26 studies involving manipulation of coarse woody debris (i.e. removal or addition). They found consistent reductions in invertebrate biomass and the diversity and abundance of bird species as a result of harvest residue removal. Rifell et al. (2011) and Bouget et al. (2012) both highlight significant knowledge uncertainty surrounding the biodiversity impacts of removing coarse woody debris from managed forests. They stress the need for long term research trials and an understanding of the representativeness of small scale experimental trials as compared to current and future commercial biomass harvesting operations. Given the lack of current knowledge and the potential for long-term cumulative impacts, such research gaps are of great concern given that residue extraction operations can be quickly expanded. For example, in 2001 only 13 % of harvested areas in Sweden were subjected to residue harvesting after tree removal, but by 2009 this had risen to 53 % (Dahlberg et al. 2011).

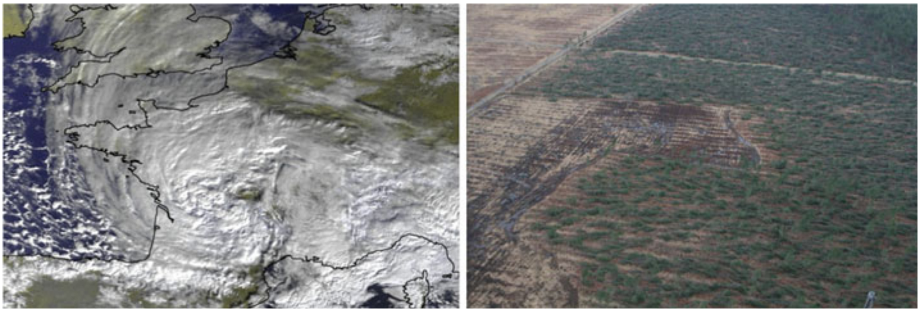

Fig. 1 Impacts of cyclone Klaus on the Forêt des Landes in France. In the last 10 years two devastating cyclones have destroyed almost 45 % of the standing timber volume in this 1 million ha plantation of *Pinus pinaster*. *Photo credits* Satellite image courtesy of Météo-France/Centre de Météorologie Spatiale (Lannion) and forest stand image courtesy of DRAAF Aquitaine, France

Establishing new plantations to enhance forest carbon stocks, and their effects on biodiversity

Afforestation/reforestation: reducing native deforestation and degradation

The provision of timber and other wood based products from plantation forests provide a significant indirect benefit to biodiversity by reducing the need to extract resources from natural forests (Brockerhoff et al. 2008). The world's most comprehensive mechanism for mitigating climate change, the Kyoto Protocol, will influence this relationship through the range of market instruments developed in response to a price being placed on carbon. Afforestation or reforestation for 'carbon farming' and the production of wood-based biofuel crops is likely to accelerate the expansion of highly productive plantation forests (see the REDD/REDD + example below). Current projections of forest plantation area and global round wood supply from plantation forests are forecasted to increase by 32 and 53 percent respectively by 2030 over 2005 levels (Carle and Holmgren 2008). Combining this with projections of wood demand over the same period (Turner et al. 2006) it is possible that round wood supply from plantation forests could increase from 66 % of global demand to 80 % thus significantly reducing logging pressure on natural forests and hence potential adverse biodiversity impact. Although the impact of climate change mitigation strategies on these projections has not yet been quantified, it has been estimated that more than 2 billion ha of degraded or deforested agricultural land exists that could be restored to forest cover (Laestadius et al. 2012). Some of this area is suitable for plantation forests, which may provide opportunities to enhance biodiversity in degraded landscapes (Loyn et al. 2007). A comparison of regional afforestation/reforestation potential by Carle and Holmgren (2008) with the global biodiversity hotspots analysis (Myers et al. 2000) indicates a reasonable alignment between areas suitable for plantation forest expansion and hotspots, apart from in Africa where plantation forest area is not expected to increase. Given that all hotspots have lost more than two-thirds of their original vegetation cover (Brooks et al. 2002) new plantation forests (on degraded agricultural areas that are no longer suitable for agriculture in these regions) are one strategy that could reduce the economic pressure on remaining habitat and significantly increase total forest cover and improve forest connectivity by creating low-contrast matrix habitat between remaining native forest remnants (Tomasevic and Estades 2008; Prevedello and Vieira 2010). Unplanted areas, such as firebreaks, riparian strips, and road buffers can also be used to diversify habitat opportunities for biodiversity at the landscape scale (Barbaro et al. 2007; van Halder et al. 2008). Because unplanted areas are often long narrow corridors adjacent to riparian areas or roads, they can improve the resilience of biodiversity to climate change by increasing connectivity throughout the landscape.

The expansion of highly productive plantations that relieve logging pressure on natural forest systems may qualify as a 'carbon credit' as part of the REDD/REDD + (Reducing Emissions from Deforestation and Forest Degradation in Developing Countries) schemes due to avoided deforestation or degradation that would have otherwise occurred (Paquette et al. 2009). Such market mechanisms have been discussed in detail at a global level; however it remains uncertain as to how these could be implemented in forested landscape. The idea is also known in the agriculture world as "land-sparing", where increased yields on less area are promoted as having less environmental impacts than lower-yield agriculture requiring larger areas (at the expense of natural systems such as forests) for the production of the same units or food (Balmford et al. 2005; Hodgson et al. 2010).

Functional zoning

To protect biodiversity within the landscape it is generally accepted that new plantation forests should not be established via the conversion of native forest, particularly in fragmented landscapes where native forest remnants are critical for maintaining biodiversity (Hodgson et al. 2011). However, the establishment of new plantation stands within the same landscape as natural forests can improve the conservation of biodiversity via functional forest zoning at the landscape scale (Seymour and Hunter 1992; Messier et al. 2009; Ranius and Roberge 2011). One example of this is the TRIAD (forest functional zoning) project in Quebec that uses plantations to improve biodiversity conservation by avoiding forest degradation, whilst maintaining harvest levels and employment at the landscape or management unit level. A current trial of the TRIAD approach implemented on 0.86 million ha aims to increase conservation areas in the landscape from nothing to a minimum of 11 % (Côté et al. 2010). The TRIAD approach will achieve this by increasing the intensity of forest management in ~20 % of the landscape and establish new high-yield plantations on <2 % of land. The remaining forest (~70 %) will be managed on the basis of sustainable forest management plans (i.e. ecosystem-based management). However, from a landscape perspective the possible loss of diversity at the stand level in the intensively managed and high-yield plantation areas is offset by the creation of large dedicated conservation forests. The resulting zoned forest landscape is predicted to be more resilient to climate change impacts (Nitschke and Innes 2008). Indeed simulations clearly demonstrate a dramatic increase in the proportion of older forests in the long term following the adoption of a functional zoning approach due to the increase in areas set aside for conservation (Côté et al. 2010). Medium term increases in attributes of older forests managed extensively (ecosystem-based management—in the bulk of the managed areas) are also expected (Côté et al. 2010), with noticeable benefits for biodiversity, namely for dead-wood dependant species (Ranius and Roberge 2011). New plantation forest stands created by functional zoning also offer an opportunity to introduce new phenotypes adapted to future expected climates, as well as the re-establishment of declining species.

Retaining native habitat

New large scale afforestation/reforestation programmes should maximise the retention of existing native forest remnants as they are critical for conserving biodiversity

(Lindenmayer et al. 2002; Loyn et al. 2007; Prevedello and Vieira 2010; Hodgson et al. 2011). The concept of conserving a significant proportion of native habitat within a landscape has been incorporated into some large tropical plantations. For example, one large concession area in Sabah, Malaysia, covers 288,000 ha, a monoculture of *Acacia mangium* occupies 38 % of this with the remainder (hills and riverine areas) being natural forest regrowth that is recovering from historical badly managed logging. In the absence of the timber plantation the whole area would probably have been cleared and used for oil palm. In this case the conversion of some land to a plantation monoculture has enabled the retention of a large areas of secondary forest and the resulting landscape mosaic is probably now better able to adapt to climate change (Lamb 2010).

The Forest Stewardship Council (FSC) certification scheme is an important global initiative that requires the retention of natural forest reserves in forest systems, including plantation forests. As a global forest certification body FSC now certifies the management of ~57 million ha of plantation forest or semi-natural and mixed plantation forests (FSC 2012b). According to FSC principles and criteria, set-aside areas (conservation zones and protected areas actively managed for conservation) are mandatory to protect biodiversity as part of the sustainable management of plantation forests (FSC 2012a). Given the increasing requirement for FSC certification as a pre-requisite to market access for plantation grown wood products, large-scale afforestation efforts to mitigate climate change will need to incorporate significant conservation areas in new plantation forests.

Impacts of afforestation/reforestation on aquatic ecosystems

Plantations can improve water quality on land that was historically forested (van Dijk and Keenan 2007). Improvements are achieved by a variety of mechanisms operating on different spatial and temporal scales, including (1) stream bank consolidation to reduce sediment inputs, (2) reduction of large-scale erosion/landslides during extreme rainfall events (Marden and Rowan 1993), (3) reducing water temperature by providing stream channel shade. These factors can result in an in-stream community composition that is more similar to unmanaged natural forest systems than streams in agroecosystems (Rowe et al. 1999). As such, potentially large-scale changes in land use that may occur to mitigate climate change (i.e., carbon sequestration) could have significant impacts on aquatic ecosystems. Expanding plantation forests on previously forested marginal agricultural land is likely to restore aquatic communities to their pre-deforestation state. The benefits of plantations for in-stream biodiversity can be compromised by periodic harvesting, however, when designed correctly, riparian 'buffers' can significantly reduce such impacts (Quinn et al. 2004). In addition to changes in water quality within plantations, large-scale afforestation/reforestation can reduce downstream water availability (Farley et al. 2005). However, the negative effects of reduced water yield on biodiversity are context dependent and can vary significantly between different plantation species (Andréassian 2004; Jackson et al. 2005; van Dijk and Keenan 2007). For example, if the landscape was originally forested, new plantations are likely to mimic past flow regimes.

Facilitating the spread of invasive species

Climate change is predicted to alter the geographic range of many tree species (Iverson and Prasad 1998). Some planted tree species have already spread extensively creating highly invasive populations (Richardson and Rejmanek 2004). The impact of such invasive trees on biodiversity is variable, but can affect both abundance and community composition

(Samways et al. 1996; Ledgard and Paul 2008; Pawson et al. 2010b; Dickie et al. 2011). Climate change may exacerbate the impact of invasive tree species on biodiversity in a number of ways. Firstly, it may facilitate the expansion of existing infestations beyond their current limits (Kleinbauer et al. 2010). Secondly, altered climatic conditions may facilitate the spread from existing plantations of tree species that were not previously considered to be invasive. Finally, significant new afforestation for timber, carbon sequestration, or biofuel feedstocks could dramatically increase 'propagule pressure' of potentially invasive tree species into new regions, thus increasing the probability of invasion success (Rouget and Richardson 2003; Richardson and Blanchard 2011).

Unintended consequences of climate change mitigation strategies

By establishing carbon as a commodity, it immediately becomes subject to broad scale market forces, which tend to focus on 'value to shareholders' and can result in unforeseen (or "perverse") outcomes. In the wider context of managed forests it is known that optimising carbon sequestration does not necessarily optimise biodiversity values (Hatanaka et al. 2011). Indeed, well-intentioned climate change mitigation policies that were intended to promote carbon sequestration, e.g., The New Zealand Emissions Trading Scheme (NZ-ETS), may have significant unintended consequences for biodiversity in plantations due to the actions of forest owners. Adopted in 2008 the NZ-ETS catalysed a period of accelerated deforestation where at least 50,000 ha of plantation was converted to dairy pasture between 2004 and 2008 to avoid the potential future liability of deforestation penalties post 2008 (summarised by Cox and Peskett 2010). This unintended deforestation represents a significant regional loss of biodiversity, as the plantation forest stands provided important alternative habitat for native forest species (Brockerhoff et al. 2003; Pawson et al. 2008, 2009, 2010a) in regions that had lost a significant proportion of their native forest cover due to historical land clearance (Ewers et al. 2006). Although we are not likely to experience such Kyoto Protocol-inspired conversion of plantations to pasture in New Zealand in the future due to the enforcement of deforestation penalties, extensive deforestation still occurs elsewhere in both tropical and temperate regions (Wilson et al. 2005; Bradshaw et al. 2009). In some cases natural forest conversion is driven by plantations established to produce wood products (Echeverria et al. 2006; Cyranoski 2007). However, this is prohibited by key forest management certification bodies, e.g., FSC (2012a). Currently we lack a comprehensive global assessment of the relative importance of plantation forestry as a driver of native forest conversion and to what extent this has affected biodiversity. Climate change mitigation strategies may accelerate the conversion of natural forest to high-yield plantation forests to provide cellulosic feedstocks for future bio-based fuels. Biofuels are increasingly promoted by various national bioenergy policies, although significant controversy still surrounds the greenhouse gas emission profiles of biofuels, and the environmental consequences of producing feedstocks (Groom et al. 2008; Koh and Ghazoul 2008). Yet, biofuels are increasingly promoted by various national bioenergy polices. A wide range of feedstocks have been proposed, and some, e.g., oil palm, have already been planted on a large scale, with significant impacts on biodiversity (Koh et al. 2011). However, the technology to produce second generation lignocellulosic biofuels is still developing and extensive planting for biofuel tree plantations has not yet occurred. It is likely that such biofuel plantations will be short-rotation species, thus the potential biodiversity benefits may be more limited than other types of plantation forests. However, careful planning of biofuel tree plantations on degraded agricultural land, as opposed to the conversion of native forests, could benefit biodiversity at a regional scale by providing additional habitat and increasing connectivity between native forest patches (Webster et al. 2012). By comparison, the alternative scenario of further conversion of natural forests to create plantations would have a significant negative impact on biodiversity.

Future importance of plantations: how to maintain biodiversity despite climatic uncertainty?

Climate change will have direct effects on future biodiversity in plantation forests due to changes in regional temperature and moisture balances. However, in the short-to-medium term, forest management actions that are likely to be implemented to mitigate climate change could have a greater impact on biodiversity than the direct effects of climatic change in plantation forests. Greater understanding of such forest management impacts on biodiversity is required, particularly the potential for synergistic interactions between management activities and climate change (Brook et al. 2008). Furthermore, any change to forest management that is implemented to mitigate predicted climate change will continue to affect biodiversity for several decades where plantations are to be managed on long rotations for timber or carbon.

To maximise future biodiversity opportunities (and protect existing biodiversity values) in plantations, forest managers must carefully consider the potential impacts of new strategies that have been proposed to increase the resilience and adaptive capacity of plantations to changing climatic conditions. In some circumstances this may require new types of multi-species plantation stands or fine-scale mosaics of single-species stands within the landscape. To succeed forestry must move from the current stand-focussed management approach and consider plantations as part of the wider landscape context (Brockerhoff et al. 2008; Puettmann et al. 2009; Paquette and Messier 2010; Paquette and Messier 2013), particularly their roles in landscape scale processes, e.g., erosion control and connectivity of natural forest remnants.

Conclusion

Climate change will have direct impacts on biodiversity in both natural and plantation forests. However, the indirect effects on biodiversity, associated with climate change mitigation strategies in the world's existing plantation forests [140.8 million ha (FAO 2006)] could exceed the direct short-to-medium term impacts of climate change on biodiversity in plantation forests. Although this hypothesis has not yet been formally tested, forest managers worldwide are considering new approaches to plantation forestry in an effort to create forests resilient to the effects of changing climatic conditions (FAO 2012). These mitigation strategies can be grouped into those that affect existing plantations, and those that influence future patterns of afforestation and deforestation. Managers of plantation forests should carefully consider the long-term consequences of strategies to adapt to, or mitigate, the effects of climate change. Such caution is warranted as any effects on biodiversity (both positive and negative) could endure for decades depending on the rotation length. For a visual summary of key points see Fig. 2.

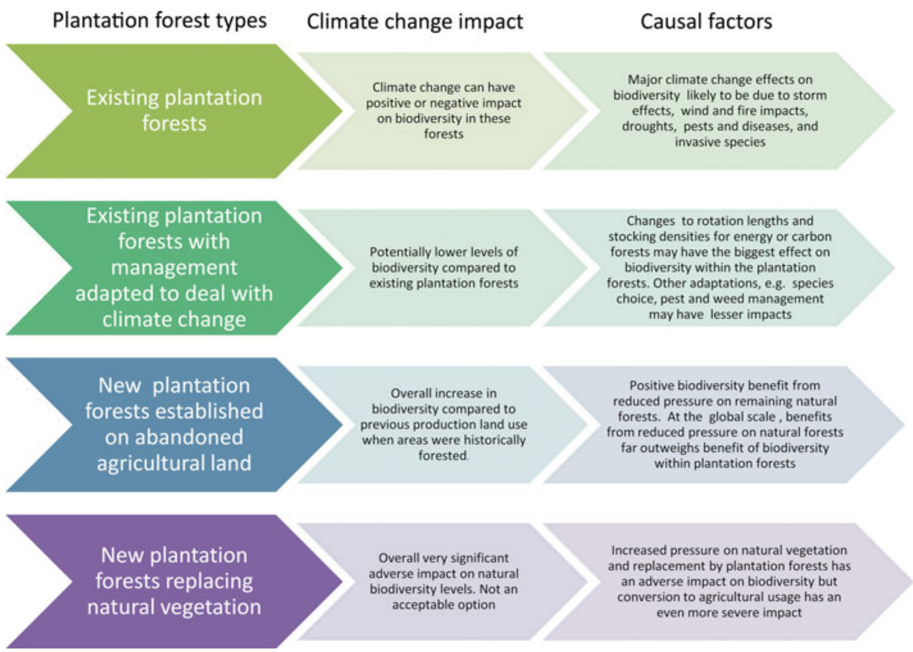

Fig. 2 Biodiversity outcomes in existing plantation forests, plantations subjected to climate change-inspired alterations of forest management, and from future afforestation/reforestation initiatives

Effects on existing plantations

Given that future plantation management practices for climate change mitigation and adaptation, e.g., shorter rotations, different species, changes in weed control, and other silvicultural operations, are likely to have significant, long-lasting, impacts on biodiversity, it is imperative that we consider the potential effects of new silvicultural practices on biodiversity. The most severe impacts on biodiversity will stem from changed silvicultural regimes, such as the expansion of short rotation energy forests or carbon forests with longer rotations and higher stocking rates. However, in many countries little is known about the inherent (and relative) biodiversity values of different forest stand types, let alone the effect of changing current silvicultural regimes. In some locations new tree crop species will be better suited to future climatic conditions, although their effect on biodiversity within plantations will be context dependent. Multi-species plantations are an alternative to hedge against unknown future climates. Applied as either mixed-species stands and/or fine-scale mosaics of a variety of single-species stands within plantation landscapes they are likely to be more resilient to unexpected climate change effects.

Effects from afforestation/reforestation/deforestation

In contrast to the ongoing decline in the extent of natural forests, the global area of plantation forests is increasing. Climate change mitigation strategies are likely to accelerate reforestation and afforestation rates due to financial incentives for reducing greenhouse gas emissions by increasing forest carbon stocks (Canadell and Raupach 2008). The overall effect of these new plantation forests on biodiversity within the landscape will depend on the prior land use history, e.g., replacement of existing native forest versus reforestation of degraded agricultural land. Although afforestation will compete with the demand for agricultural land in many parts of the world as the human population grows, changes in the location of agricultural areas caused by climate change may lead to enhanced opportunities for reforestation of abandoned agricultural lands. If designed appropriately, afforestation or reforestation of such areas to create new plantation forests are likely to provide greater opportunities for biodiversity than the degraded agricultural lands they replace.

In the future, greater thought must be given to the ways in which new plantation forests fit within the landscape in order to maximise opportunities for biodiversity conservation. However obvious this may seem, it demands a profound change in the way forestry has been conducted for centuries, moving from an exclusively stand-focussed science and practice, to one that considers the landscape as its management unit. Well-situated new forests could have significant positive effects on biodiversity. However the economic potential of such forests must be better understood and defined to facilitate their future establishment. This may require new ways of thinking about forestry, particularly the shift to a landscape focus that incorporates non-timber values. Such landscape level plantation designs intended to improve resilience in the face of climate change impacts are unlikely to generate ecologically optimal outcomes if such decisions are left solely to individual landowners or to 'the market'. Smart policy frameworks and landscape-level planning processes that incorporate the opportunity cost of reforestation to landholders need to be developed that will promote economically viable plantations that also optimise biodiversity at landscape, national and regional scales.

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