

SECTION 2

IMPACTS

Chapter 4 - Observed impacts of climate change on UK forests to date

Chapter 5 - An assessment of likely future impacts of climate change on UK forests



OBSERVED IMPACTS OF CLIMATE CHANGE ON UK FORESTS TO DATE

Chapter

4

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Key Findings

There is clear evidence that climate change is having an impact on some aspects of the composition and function of woodland. Leafing has advanced by 2–3 weeks since the 1950s in response to increased temperatures and there is some evidence that this is having a negative impact on woodland flora, particularly vernal species.

Evidence for increases in tree growth rates and forest productivity resulting from lengthening growing seasons, rising atmospheric CO₂ concentrations and climatic warming is limited for the UK. This is consistent with recent studies which attribute the increases reported across much of continental Europe to changes in forest management and nitrogen availability.

In most cases it is not possible to separate the impacts of climate change on forest ecosystems from the effects of forest maturation over the past century, changes in land cover and forest management and the effects of nitrogen deposition.

The likely effects of extreme drought years such as 1976 and 1989/90 provide an insight into some of the well-documented impacts of climate. The effects of drought are not restricted to a single year. Subsequent decline and, ultimately, mortality can occur over a period of more than 10 years.

Current monitoring networks have not adequately addressed climate change. Extensive monitoring networks such as Countryside Survey and the National Forest Inventory may not be suitable for monitoring the impacts of climate change or the effectiveness of adaptation measures. An annual monitoring platform is required to monitor the impacts of climate change, particularly extreme climatic events as they occur, and to provide further evidence for modelling future impacts and developing appropriate adaptation strategies.

Evidence is now compelling that the climate has changed over past decades in the UK, with nine of the ten warmest years on record having occurred since 1990 and a shift in the seasonality of rainfall now emerging.

Although it is difficult to prove that these changes are the direct result of anthropogenic greenhouse gas (GHG) emissions, modelling experiments provide the evidence: only when GHG forcing is included in those experiments can global climate models replicate the climate of the past century adequately (IPCC, 2007). We can therefore link climate-driven observed changes in woodland form and function to the direct effects of anthropogenic climate change; more importantly, we can use these observations, coupled to the fundamental knowledge of tree physiology

outlined in Chapter 3, to project the future impacts of climate change (see Chapter 5).

It is critical that observed impacts of climate change are well documented to enable conclusions on future impacts to be drawn. However, a note of caution should be sounded because the level of climate change witnessed to date (e.g. 0.8°C to 1°C change in mean UK temperatures; see Chapter 2) is small relative to the change that is likely over the course of the 21st century; we therefore

can neither predict how woodlands will be impacted nor design or implement adaptation responses to limit the future impacts on the basis of recent responses alone. An equally important point is that trends in response variables may not yet have become apparent as a result of either the limited timecourse of monitoring data or a functional threshold not having been reached; the lack of an observed trend in a response variable does not therefore provide proof that climate change is not having an impact or will not in the shorter or longer term. The corollary of this is that where impacts of climate change are evident, these can provide a powerful indicator of advancing climate change.

This chapter collates evidence of impacts of climate change on woodland ecosystems in the UK to date. It also assesses the impacts of extreme climatic events – whether a result of climate change or not – that are likely to become more common as climate change progresses. Such information represents a powerful resource for modelling studies, to provide an analysis of the likely future impacts of climate change.

4.1 Evidence of climate change impacts to date

A clear temperature trend has only emerged since around 1980 and the magnitude of change observed to date is only 0.8–1.0°C (Chapter 2 and Jenkins *et al.*, 2007). Clear trends in rainfall patterns and other climatic variables are more difficult to detect. These difficulties are compounded by changes in land management (Carey *et al.*, 2008), pollutant deposition (NEG-TAP, 2001) and atmospheric composition (IPCC, 2007) that have occurred over the same time period. Furthermore, trees are typically more resilient and slower to respond to environmental pressures than more dynamic elements of ecosystems such as ground flora, fungi and fauna. It therefore follows that unquestionable evidence of impacts of climate change on trees and forests are likely to be limited. However, the impact of recent extreme climatic events has been well documented and when considered with projections for the future and the quantification of uncertainty available through the UK Climate Projections (Chapter 2 and Murphy *et al.*, 2009) they provide an insight into the likely future effects of a changing climate.

A number of formal networks and reporting protocols that are relevant to forestry and woodland have been established to identify impacts of climate change. These

are reviewed here, together with a summary of the information they have provided.

4.1.1 Climate change indicators for the UK

A series of indicators covering aspects of the climate, economy and natural environment were published in 1999 (Cannell *et al.*, 1999) and reviewed in 2003 (Cannell *et al.*, 2003). Two ‘impacts indicators’ are directly relevant to woodlands (health of beech trees and leafing date of oak), with a further four providing information on the wider natural environment (date of insect appearance/activity, insect abundance, arrival date of the swallow and small bird population size).

4.1.2 Environmental Change Network

The UK Environmental Change Network (ECN) is an ecosystem monitoring and research programme with a range of contrasting sites across the UK established by the UK Government in 1992 and funded by a range of stakeholders that operate the individual sites. Seven of these have forested areas (Alice Holt, Wytham, North Wyke, Hillsborough, Rothamsted, Porton, Cairngorm). The programme has been operating since 1992 and Morecroft *et al.* (in press) have recently completed a review of trends in the data. The climate of the sites has been monitored and has shown a significant warming trend of 0.9°C over 15 years. There has also been a significant rise in precipitation during this period. There was, however, no consistent evidence of change in species composition of vegetation across sites. Local changes in the composition of functional types of plants within communities were found, but attribution to climate change was generally not possible. Morecroft *et al.* (2008) studied tree growth at Wytham Woods ECN site and found evidence of low growth rates of sycamore, *Acer pseudoplatanus*, during dry periods and this was associated with reduced photosynthetic rates. This contrasted with ash, *Fraxinus excelsior*, which is often identified as sycamore’s main native competitor. Morecroft and colleagues concluded that an increase in frequency of summer droughts might reduce the competitiveness of sycamore in future.

4.1.3 UK Phenology Network

The UK Phenology Network (UKPN) is a joint initiative between the Centre for Ecology and Hydrology and the Woodland Trust, using volunteer observers (~20 000) to report on the timing of a range of development processes (phenological indicators) in both spring and autumn

(*Nature's Calendar Survey*). The Network produces a powerful dataset for identifying the progressive impacts of climate change on the natural environment. However, its application to impacts and adaptation studies is currently limited because of its relatively short duration. Other activities of the Network, particularly the collation of historical datasets, address this need.

4.1.4 Countryside Survey

The Countryside Survey is the largest survey of soils and vegetation in the UK, to date, and provides evidence about the state of the UK's countryside today and details of land use and land use change. The most recent survey took place in 2007 and the findings can be compared with those of previous Countryside Surveys from 1998, 1990, 1984 and 1978. There are two parts to the Survey: the Field Survey and the Land Cover Map. The Land Cover Map uses data from satellites to form a digital map of the different types of land and vegetation across the UK. The Field Survey is a very detailed study of a sample of 600 1 km squares, located across England, Scotland, Wales and Northern Ireland. The individual squares are chosen so that they represent all major habitat types in the UK. The field survey includes soil and vegetation assessments.

4.1.5 Forest Health Survey

The Forest Health Survey (later renamed the Forest Condition survey) was established in 1984 (see Innes and Boswell, 1987) to assess, primarily, the crown condition of single species stands of forest trees across Great Britain. Five species (oak, beech, Scots pine, Sitka spruce and Norway spruce) were assessed on an annual basis across up to 350 sites. In addition to crown condition, a range of other metrics were recorded, including stem diameter (increment), level of fruiting (masting), biotic and abiotic damage. In 1993, the network of plots were incorporated within the EC/ICP-Forests Level I survey, providing the opportunity for comparison with wider changes in forest condition across Europe. The initial focus was on the impacts of air pollution, although latterly, climate change became of increasing importance. The survey was suspended in 2006.

4.1.6 National Forest Inventory

Extensive woodland surveys have been carried out across Great Britain in an *ad hoc* way over the past 80 years, with Woodland Censuses in 1924, 1947, 1965 and 1980

and the National Inventory of Woodland and Trees (NIWT) in 2003. These surveys collected data on the extent and nature of the woodland resource and, particularly in the case of NIWT, on its condition. The National Forest Inventory (NFI) is a rolling five-year programme with field assessments covering a minimum of 0.5% of the woodland resource. The field survey began in summer 2009 and the first cycle will report in 2015, with interim reports also to be made available.

4.2 Impacts on the timing of natural events

The study of the timing of natural events (phenology) has a long history, stretching back to 1736 through the 'Marsham Records' from Norfolk that spanned a period of more than 200 years to 1947 (Sparks and Carey, 1995). Because of the longevity of the record (and the availability of the Central England Temperature index), clear relationships can be demonstrated between spring/winter temperature and leafing date, for example. Further analysis of the Marsham record has enabled a temperature sensitivity of leaf flushing in different tree species to be calculated that will be critical for interpreting the likely impacts of climate change on the complex interactions that determine the composition of woodland ecosystems (Table 4.1) (Sparks and Gill, 2002).

Leafing date provides, for the forestry/woodland sector, the most straightforward indicator of the impact of climate change because there is a well-known relationship between the date of occurrence of phenological phases and temperature. There has been an advance in leafing date of oak by about 3 weeks since the 1950s, with leafing now consistently earlier than prior to 1990 (Figure 4.1). In theory, earlier leafing will increase net primary productivity through extending the growing season. Extending the growing season in spring will have a greater effect than in autumn because of the higher incoming solar energy in April (budburst) as compared with October (autumn senescence). However, earlier leafing may have negative impacts on the wider woodland ecosystem through affecting the synchrony between different trophic levels (for example, oak, winter moth caterpillar and blue tit: Buse and Good, 1996) or through reducing the amount of light available for characteristic woodland ground flora specialists such as bluebell, wood anemone and sanicle. This may have contributed to the decline in woodland specialists reported by Kirby *et al.* (2005), see 4.4 below.

Table 4.1

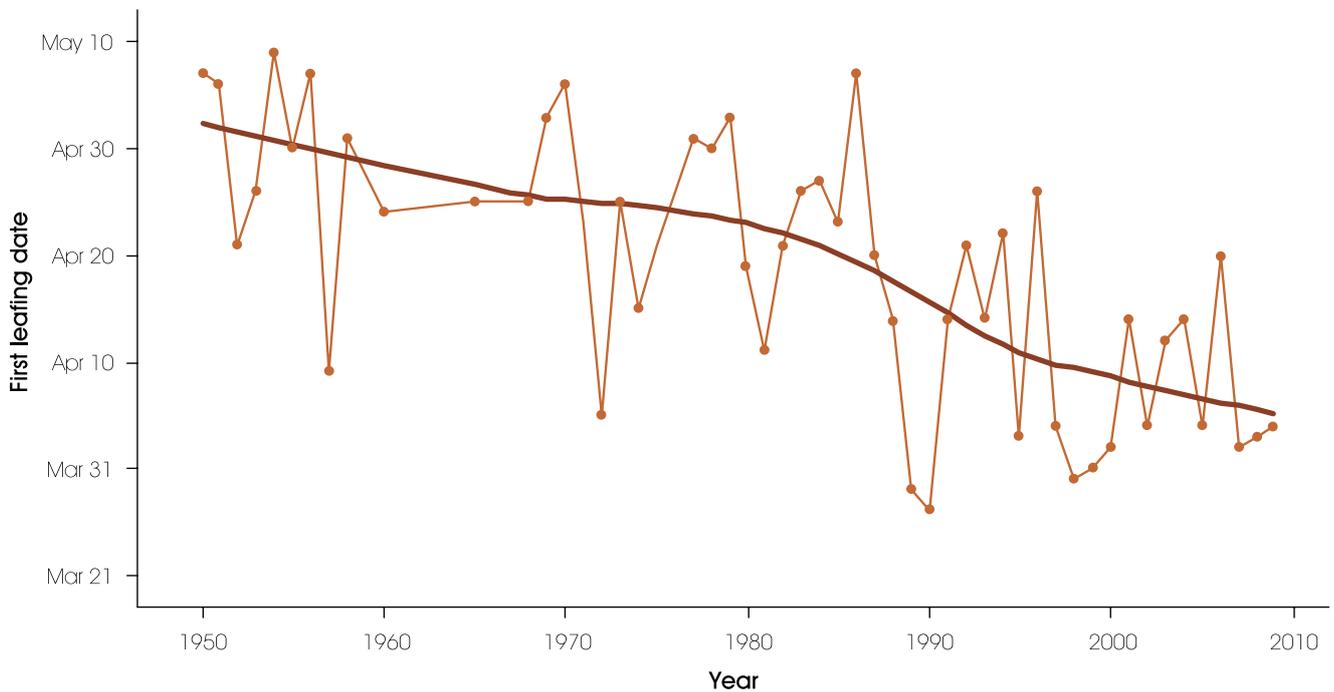
Estimated response of flushing to temperature in the first quarter of the year for 13 woody species. Data taken from the Marsham record (Sparks and Gill, 2002).

Common name	Scientific name	Temperature response (days °C ⁻¹)*
Hawthorn	<i>Crataegus monogyna</i>	9.9
Sycamore	<i>Acer pseudoplatanus</i>	6.7
Birch	<i>Betula pendula</i>	5.2
Elm	<i>Ulmus procera</i>	5.7
Mountain ash	<i>Sorbus aucuparia</i>	5.6
Oak	<i>Quercus robur</i>	5.6
Beech	<i>Fagus sylvatica</i>	3.0
Horse chestnut	<i>Aesculus hippocastanum</i>	4.8
Sweet chestnut	<i>Castanea sativa</i>	5.5
Hornbeam	<i>Carpinus betulus</i>	6.1
Ash	<i>Fraxinus excelsior</i>	3.5
Lime	<i>Tilia</i> spp.	5.2
Field maple	<i>Acer campestre</i>	4.4

*Advance in first leafing date in days per degree increase in January–March temperatures.

Figure 4.1

Leafing date of oak in Ashstead, Surrey between 1950 and 2009.



Data courtesy of Jean Coombs and Tim Sparks. See also Sparks and Gill, 2002.

Earlier leafing dates also enhance the risk of frost damage to trees because cold night-time temperatures are stochastic weather events. For example, although the modelled date of leafing advanced significantly between 1950 and 2005 in southern England, there has been no

corresponding trend in the date of the last spring frost (less than -2°C, below which tissue damage is likely). There has therefore been an enhanced risk of frost damage over this period, although mean daily temperatures have risen by nearly 1°C. There has been no evidence of increased

levels of spring frost damage to date, but the implications of changing vulnerability to frost should be an important consideration in provenance and species choice as part of developing adaptation strategies (see 5.1.1, Chapter 5). For some species, experimental studies have revealed the converse with a lack of winter chilling, leading to delayed budburst (Murray *et al.*, 1989).

A series of other phenological records are available that are relevant to the forestry sector, including appearance dates of woodland butterflies and birds and emergence or flowering dates of woodland flora. As in the case of leafing date of oak, there is clear evidence of enhanced precocity, confirming that climate change is already impacting on woodland ecosystems (Sparks and Gill, 2002).

4.3 Impacts of climate change on tree growth and forest productivity

The climate of the UK has changed considerably over the last century, as outlined in Chapter 2. Mean annual temperature has increased with high temporal consistency in the rate of change between regions of the UK (Jenkins *et al.*, 2007). Although the total annual rainfall has tended to increase in many parts of the UK, summer rainfall has shown a slight decrease, particularly in eastern regions and in southern and central England. These changes have resulted in an increasing frequency of drier and warmer summers. If soil moisture reserves are not depleted during the growing season, such changes would be expected to have increased plant growth, due to higher rates of photosynthetic CO₂ uptake and cellulose accumulation.

An index of seasonal warmth can be calculated from the sum of daily temperatures above a base of 5°C (accumulated temperature). This index has been calculated from monthly temperature records over the past 50 years (Figure 4.2). A sample of nine meteorological stations, three each in England, Scotland and Wales, show that the growing season has become warmer, based on normalised data, over the past 50 years in all regions. There is a direct relationship between accumulated temperature and yield and, if soil moisture deficit is not limiting, trees will grow faster in a warmer climate. Indeed, a number of studies have reported an increased yield from forests in Britain and Europe, which has been attributed to the warming climate, increased nitrogen availability largely because of pollutant deposition (see 3.3.4, Chapter 3) rising CO₂

levels (270 to 390 parts per million; see 5.1.1, Chapter 5) and improved silviculture (Worrell and Malcolm, 1990; Cannell *et al.*, 1998; Cannell, 2002; Magnani *et al.*, 2007). If summers have also been drier, this implies reduced cloud cover, and thus higher solar radiation, which could have caused increased photosynthesis and growth. However, photosynthesis in forest stands can be higher when light is more evenly distributed as under light overcast conditions than in cloudless conditions, so the net consequences of changes in cloud cover are uncertain.

Increased growth in warmer growing seasons is dependent on sufficient moisture. In dry summers, stem increment of beech has been shown to be reduced, particularly on surface water gley soils as the seasonally fluctuating water table tends to restrict rooting depth and thus moisture availability (see Figure 4.3; note dips of radial increment in 1976, 1984, 1989/90 and 1995/6).

Very dry summers have caused serious damage to tree stands, particularly in species that are not well suited to site conditions. In 1975 and 1976 two consecutive dry summers caused more serious damage and die back to beech than to oak (Mountford and Peterken, 2003), in the Denny Inclosure of the New Forest. Very dry summer periods have also caused abiotic damage to Sitka spruce on shallow freely-draining soils in eastern Scotland. Drought causes the xylem to collapse resulting in stem lesions and cracks appearing through the cambium (Green and Ray, 2009). Affected trees have been shown to exhibit shake (stem cracking), rendering the timber poor quality and of no structural use. Examples of this type of damage occurred in spruce forests in eastern Scotland in 2003.

Dendrochronology studies have demonstrated increasing growth rates across much of Europe over the past century (Briffa, 1991; Becker *et al.*, 1994; Spiecker *et al.*, 1996; Kahle *et al.*, 2008). The evidence is compelling, particularly for natural stands, which have shown an increased growth rate in central Europe. However, these studies have focused on conifer species and there has been little examination of broadleaved tree species. Briffa (1991) found no increase in growth (as indicated by diameter increment) for four Scots pine stands in Scotland and one in southern England, contrasting with a significant enhancement across much of central and southern Europe between 1860 and 1975. Kahle *et al.* (2008) reported a similar lack of response in Great Britain (three sites in Scotland, one in the English Pennines) and northwest Europe. For this reason, the increase in height and diameter increment that was observed across much

Figure 4.2

Accumulated temperature (above a base of 5°C) trend over 50 years for nine meteorological recording stations, three each in England, Scotland and Wales. The trend is shown as the yearly anomaly, i.e. the difference each year from the overall 50-year mean for each station, in order to facilitate comparison between stations.

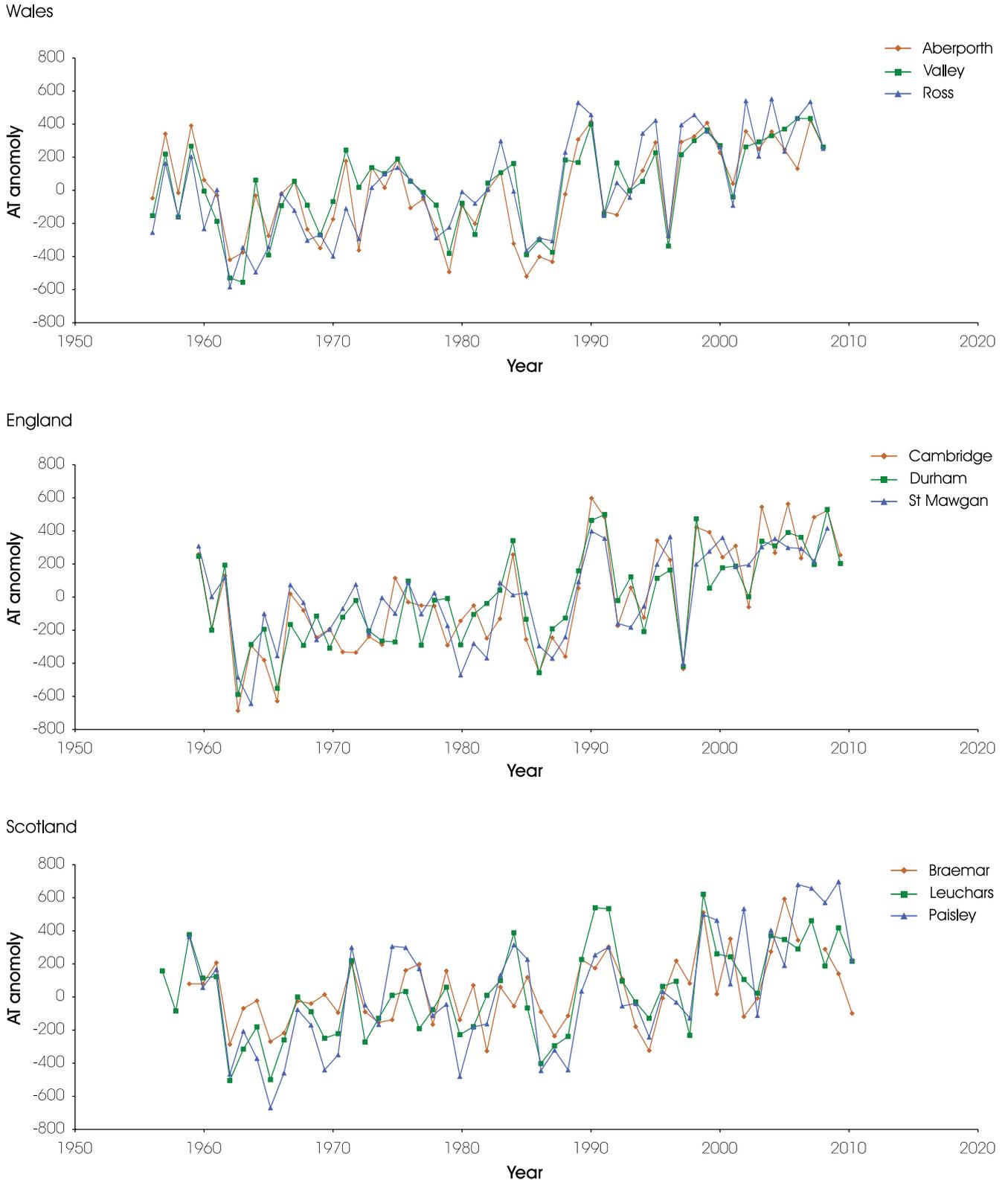
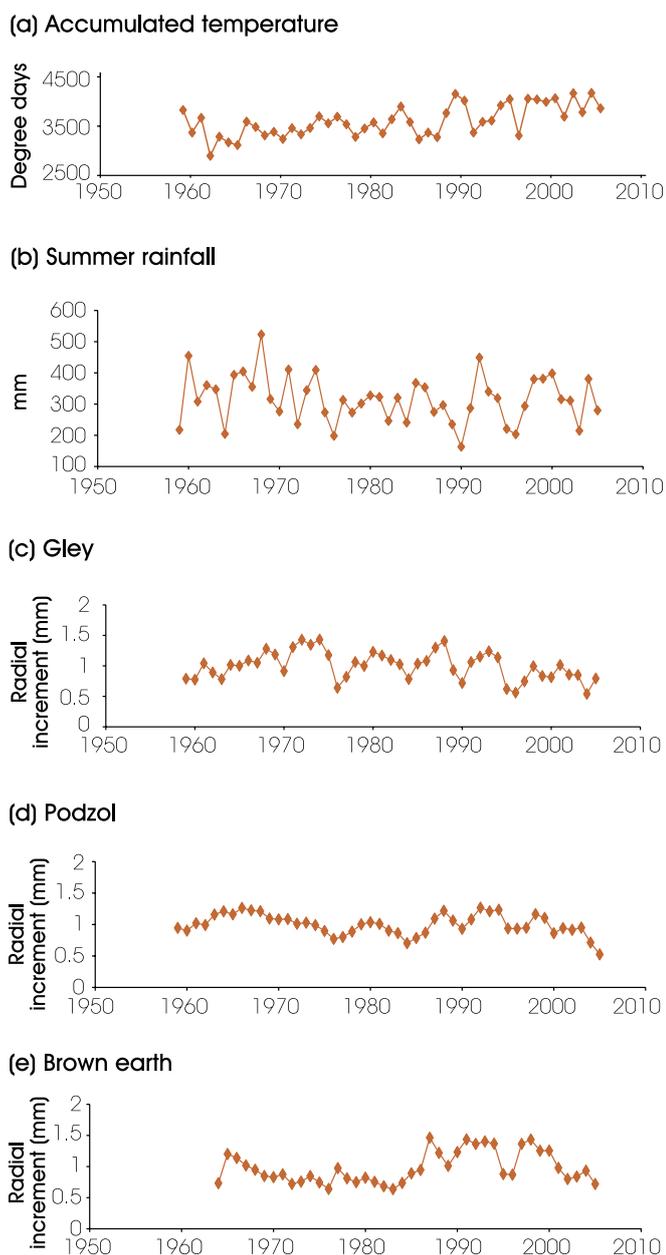


Figure 4.3
Relationship between warm and dry summers and the normalised stem diameter increment of beech (*Fagus sylvatica*) growing on three soil types (c) surface-water gley, (d) podzol and (e) brown earth in southern England. Accumulated temperatures (above a base of 5°C) (a) and summer rainfall totals (b) are shown for a site in Oxford (Wilson *et al.*, 2008, Sanders *et al.*, in press).



of the remainder of Europe was ascribed to changes in nitrogen availability resulting from changes in atmospheric nitrogen deposition and forest management. No effect of rising carbon dioxide concentration, temperature or rainfall patterns could be identified. Matthews *et al.* (1996) analysed mensuration sample plot collected over the

period 1920–85, reporting an increase in growth rate over this period. However, survey design and planting year had over-riding impacts on the analysis, with observed effects ascribed to improvements in planting practices, site type selection and approaches to management.

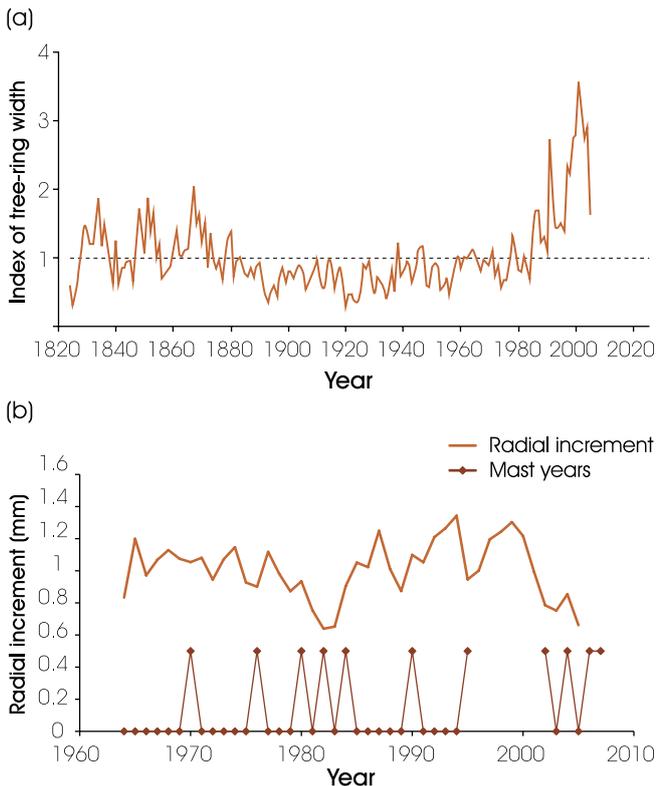
More recent analyses of broadleaf growth in southern England have revealed trends and climatic impacts that were not apparent in earlier studies. Sanders *et al.* (2007) reported an increase in tree ring width of old growth (planted 1820) woodland since the mid-1970s (Figure 4.4a). They also report a positive impact of mild spring temperature and above average early summer rainfall and a negative impact of lower than average July–August rainfall. Wilson *et al.* (2008) assessed the response of tree-ring width of beech stands in southern England to summer soil moisture deficit. The strength of the response was dependent on soil type. Growth on the gley soil, in particular, showed a large negative response to the summer droughts of 1976, 1989/90 and 1995 that was maintained for a number of years in each case. On a number of the sites included in the analysis, a decline in growth has been evident since 2000. Sanders *et al.* (in press) report an interaction with frequent masting that has been evident in recent years and earlier periods of reduced growth apparent in the chronology (Figure 4.4b).

4.4 Impacts on woodland flora

A 2001 re-survey of woodland plots first surveyed in 1971 (Kirby *et al.*, 2005) showed some potential evidence of impacts of climate change, as the frequency of 47 out of 332 plant species showed a significant positive relationship with early season temperature change over the period, whereas four species showed negative correlations. Those species which increased in frequency or abundance tended to be associated with locations with higher precipitation and lower July maximum temperatures, but were otherwise hard to categorise. There was also evidence of contrasting relationships between percentage cover and temperature in different species.

The most recent Countryside Survey, carried out in 2007 (Carey *et al.*, 2008), showed that there had been a change in the character of broadleaved woodlands, with tree and shrub species tending to increase in frequency, while ground flora, particularly grasses, tended to decrease (Carey *et al.*, 2008). There were also increases in competitive species and a decline in ruderal (weedy) species. This is indicative of the maturing of woodlands,

Figure 4.4
(a) Annual growth ring increment for oak trees at Alice Holt forest between 1820 (planting) and 2005. $n = 11$; data have been de-trended for age effects with standard dendroclimatological approaches (Sanders *et al.*, 2007). **(b)** Comparison of annual stem increment of five dominant, 55-year-old beech trees at Covert Wood in Kent, assessed through tree-ring analysis, and the occurrence of repeated heavy 'masting' as determined from various sources (Sanders *et al.*, in press).



confirming the findings of Kirby *et al.* (2005) and the previous Countryside Survey (Haines-Young *et al.*, 2000). No effects of climate change were evident in CS2007, although the report notes that 'factors such as land-use surrounding the sampled woods, climate change or pollution (sulphur and nitrogen deposition) may also be important. More detailed analysis is required to determine and distinguish the possible roles of the drivers of the changes reported here for the woodlands of Great Britain'.

Where changes in ground flora are apparent, it is important to note that other drivers may be responsible, as outlined by Carey *et al.* (2008) and Hopkins and Kirby (2007). Foremost are changes in levels of management in semi-natural woodland leading to a change in structure. In an analysis of successive woodland surveys, Hopkins and Kirby (2007) report that in 1947, only 51% of the broadleaved resource was classed as high forest,

compared with 97% in 2002. Coppice and 'scrub' made up 21% and 28%, respectively, of the remainder in 1947. The decline in the level of management that this implies was confirmed by Kirby *et al.* (2005) who reported that 68 of the 103 woods visited in 2001 showed no evidence of recent management activity. They also reported both an increase in stem size and basal area of the overstorey tree species. The observation that the 10 sites that were in the track of the 1987 storm that affected southern England tended to show an increase in plant species richness as compared to sites outside the storm track, emphasises the importance of natural disturbance to species composition (see 5.1.5, Chapter 5). The impacts of atmospheric nitrogen deposition are well documented (NEG-TAP, 2001) with fast-growing plant species expected to be favoured at the expense of slower growing woodland species (Grime *et al.*, 1988). This trend is evident in the work of Haines-Young *et al.* (2000), Kirby *et al.* (2005) and Carey *et al.* (2008), and is not unsurprising given that 95% of broadleaved woodland in the UK is reported to receive excess nitrogen deposition (Hall *et al.*, 2004). If the impacts of climate change are to be detected at an early stage, those effects will need to be distinguished from changes in flora resulting from the drivers outlined above. This will require that a monitoring system designed for that purpose, such as the Environmental Change Network (see 4.1 above), remains in place to provide a continuous record measured using methods that remain common over time.

4.5 Impacts on woodland fauna

There is clear evidence of changes to emergence and first flight times for a range of butterflies and moths (Burton and Sparks, 2002) and hoverfly species (Morris, 2000). There have also been changes in the arrival times of migrant bird species (Sparks and Gill, 2002). The abundance of moth species has also shown a significant decline over the period 1966–2001 with an average loss of about two species per year (Benham, 2008). Although land use, land management and other factors are implicated, relationships with summer temperature and winter precipitation are evident and help to explain the 60% decline in macrolepidoptera numbers since 1930 (Woiwod, 2003). The BTO Bird Atlas provides an unparalleled resource for the interpretation of changes in bird distributions (www.bto.org/birdatlas and Eaton *et al.*, 2009). For example, there is clear evidence of expanding ranges of some heathland species such as the Dartford warbler (*Sylvia undata*), while there has been a contraction

in the range of some upland species such as the twite (*Carduelis flavirostris*) and snow bunting (*Plectrophenax nivalis*). However, changes in the distribution of many species are difficult to interpret because of the complexity of the underlying drivers.

Populations of deer have a significant impact on establishment and regeneration of both plantation forests and semi-natural woodlands. Sparks and Gill (2002) evaluated the relationship between climate and deer survival/recruitment, concluding that populations would be expected to increase in response to climate change. Although data on deer numbers are limited and do not allow a thorough evaluation of trends, deer-cull data for Scotland confirm the more anecdotal evidence that deer populations are increasing, with the cull rising from 35 000 in 1987/88 to 63 000 in 2004/05 (R. Gill and J. Irvine, pers. comm.). Although this trend is consistent with current understanding of the likely effects of climate change, a causal relationship cannot be demonstrated.

4.6 Interactions between climate change and tree health

There is little evidence in the UK that the prevalence and severity of outbreaks of existing forest pests and pathogens have been directly impacted by climate change, to date. There is, however, evidence from elsewhere in the world of changes to the range of destructive outbreaks of forest insect pests. A good example is the southern pine beetle in the southeast United States, for which the effective range has spread north and westwards as minimum winter temperatures have increased (Evans *et al.*, 2002).

Over the past decade, several new pests and diseases have been found in Great Britain, and some have established with potentially serious economic consequences. New pathways for pests to travel and establish are being provided by the ever-growing global trade. Of particular importance is the increasingly rapid trade in live plants and plant products, as well as the timber trade and goods of all kinds shipped in wood packaging material. These pathways provide increasing opportunities for pests to transfer from their native habitats and successfully establish in new areas.

During 2006, several pests made national headlines. On horse chestnut, a significant ornamental tree species, both bleeding canker (caused by the recently discovered bacterium *Pseudomonas syringae* pv *aesculi*) and the

horse chestnut leaf miner (*Cameraria ohridella*) were widely reported. First described in Macedonia in 1985, leaf miner was detected in Great Britain in 2002 in Wimbledon and has spread rapidly over the past seven years. Oak processionary moth (*Thaumetopoea processionea*), native to southern Europe made headlines when it was found in London due to its irritating hairs and adverse reactions it can provoke when in contact with people, as well as its potential to severely damage oaks. Introduced species of the fungal pathogens *Phytophthora* that include *P. ramorum*, the causal agent of the disease known as 'sudden oak death' in California, are a continuing cause for concern because of their potential to adversely affect trees in urban and rural environments. None of these significant pests are native to Britain and have been imported from different parts of the world, often with warmer climates than Britain.

In addition to dealing with the above, the citrus longhorn (*Anoplophora chinensis*) and Asian longhorn (*A. glabripennis*) beetles continue to be intercepted; both of which have the potential to be very damaging. Southern European cicadas now occur in Kent, and the plane lace-bug (*Corythuca ciliata*) was recently found by Defra plant health and seeds inspectors to be breeding and causing damage to plane trees in Bedfordshire. The lists of intercepted and, more significantly, established pests and pathogens grow each year and these changes are accelerating, as global trade expands.

4.6.1 Imported plant material and horticulture

The global trade in live plant material is on an upward trend. The UK is importing more and more plant material from around the world, much of it for use in urban planting. In 2005, tree species imported into the UK had a total value of £65 million, with a further £69 million spent on importing a large variety of outdoor plants. Interest in gardens and gardening is linked to activity in the housing market and this has helped fuel a booming garden-centre trade over the last decade. Of particular concern is the expanding trade in 'instant trees', with giant root balls. Often these are 'salvaged' rather than produced under controlled production conditions. Some of the genera involved are not restricted by quarantine controls and are therefore not subject to any form of inspection, either on arrival or when planted.

The discovery of *Phytophthora ramorum* and *P. kernoviae* in both forest trees and a wide range of ornamental trees and shrubs links the world of gardens and horticulture

to the woodland environment. Gardens are a very highly valued aspect of national life and the discovery of *P. kernoviae*, which is not native to Britain, in a national Magnolia collection in Cornwall has highlighted the potential dangers of imported pathogens. An increasing proportion of imported plants are grown in the Far East and then imported into the EU. Once the plants have entered the EU, the scope for health checks and inspections is severely restricted.

4.6.2 Plant health and import controls

The sea represents a barrier to natural colonisation for most pests, and Britain is fortunate in not having many of the major pests prevalent on the European mainland. Under the EU Protected Zone provisions, the UK is able to maintain controls against named pests and pathogens present elsewhere in the EU.

Three levels of risk are identified by import controls: prohibited (plants or plant materials cannot be imported due to a high risk of pest introduction); controlled (materials are inspected before they are permitted into circulation) and unrestricted (material is not checked). The controlled list relates to material that could host pests indigenous to mainland Europe, or elsewhere in the world where climatic conditions are similar and the same hosts are grown. These factors could readily result in pest establishment in this country. The restricted list is very limited and there is a wide range of genera which, because there is no recorded information about pest risk associated with them, are not subject to any regulation. Therefore, although these controls undoubtedly reduce the risk of the harmful introduction of pests, the risk is by no means eliminated.

4.7 Impacts on woodland soils

Forest management and species choice is, in many cases, dependent on soil form and function. Principal concerns over the effects of climate change relate to nutrient sustainability and enrichment (eutrophication), recovery from past acidification, changes in soil carbon levels and the ability to retain moisture. In particular, soil organic carbon content is a key area of concern because of its role in water retention, in binding soil particles together limiting erosion and ability to retain nutrients. Soil carbon stocks also represent a large potential source of GHG emissions to the atmosphere in that soils in Great Britain store more than 10 billion tonnes of carbon – equivalent to 70 times the total GHG emissions from the UK (Milne and Brown, 1997).

Bradley *et al.* (2005) reviewed the likely impacts of climate change on soil form and function, identifying the following effects that are likely to be observed as climate change progresses:

- In isolation, rising temperatures would be expected to increase soil respiration rates and lead to a decline in soil carbon content. Reduced summer rainfall coupled with higher rates of evapotranspiration may lead to drying of moist organic soils for a greater proportion of the year leading to significant soil carbon loss.
- Increased litter inputs as a result of increased tree productivity in response to rising CO₂ levels and climatic warmth may counter the effects of rising temperatures on soil respiration and enhance soil carbon levels.
- Increased tree growth might contribute to nutrient depletion, possible acidification and reduced productivity in second/third rotation crops.
- As a result of changes to soil carbon levels coupled with changes in nitrogen deposition and management practice, the carbon to nitrogen ratio may change with consequent impacts on woodland ground vegetation composition.
- Increased soil disturbance as a result of more frequent/severe windstorms and consequent windthrow.
- Increased soil erosion on vulnerable sites as a result of heavier rainfall events.
- Dissolved organic carbon losses from forest soils may increase as a result of higher temperatures and heavier winter rainfall leading to impacts on water quality and treatment costs.
- More frequent and widespread forest fires resulting in soil carbon loss with consequent impacts on water-holding capacity and nutrient retention.
- The relationship between mycorrhiza, ground flora and host trees is affected by atmospheric CO₂, levels pollutant deposition and soil moisture content. Given the importance of such relationships in nutrient cycling and conferring resistance against pathogenic soil fungi, this is potentially an area of concern, although poorly documented to date.

The majority of the likely impacts of climate change outlined above result from interactions between vegetation, soils and climate. Detecting a climate change signal for any of these potential impacts in isolation is therefore challenging, particularly given the limited change in climate to date. However, the two aspects of soil form and function outlined below provide some indication of the extent of climate change impacts on the soil environment to date.

4.7.1 Soil carbon

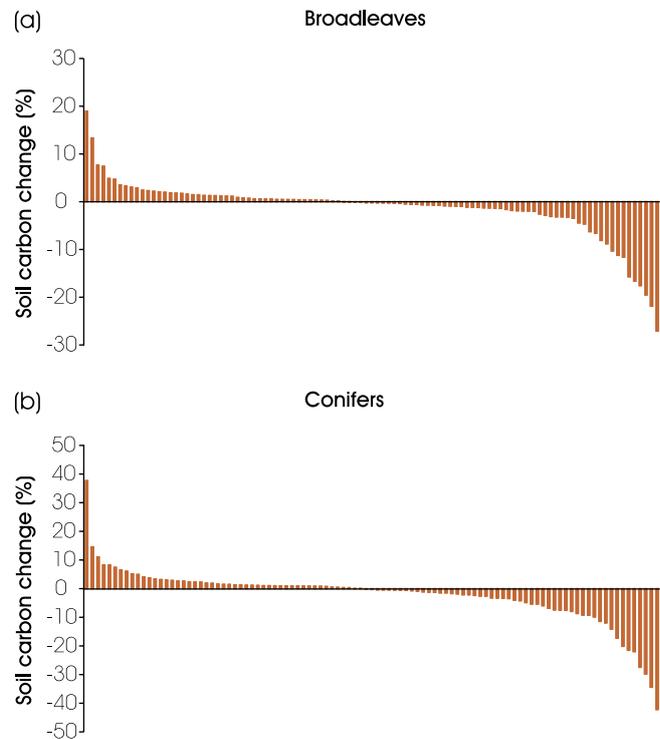
Forest soils typically accumulate carbon during stand development due to woody and other inputs (Poulton, 1996). However, an analysis of soil carbon change from more than 2000 sites across England and Wales between 1978 and 2003 reported significant loss of soil organic carbon from all land covers, including woodland (Bellamy *et al.*, 2005). Mean carbon content of the upper 15 cm declined from 6.8% and 11.0% to 5.46% and 8.74% for broadleaf and conifer woodland soils, respectively. The losses reported for individual land covers were scaled up to a UK level, suggesting that soil carbon losses could amount to up to 13 MtC, or 8%, of total fossil fuel emissions. The authors suggested that climate change might, in part, explain the results. However, Schulze and Freibauer (2005) concluded that rising temperatures can only have had a limited impact. Although the mean carbon content showed a clear decline over the 25-year period, there was close to a 50:50 split between those soils that showed an increase in soil carbon content and those that showed a decline (Figure 4.5). A limitation of the study was that impacts of changes in land management (including harvesting, restocking or changes in woodland type) were not assessed. These changes could explain some of the unexpectedly large losses in soil organic carbon of up to 90%. Powelson (2006) commented that the reported rate of carbon loss was extraordinarily rapid for temperate forest soils in the absence of major management or land use change. The National Soil Research Institute (NSRI) soil survey has been re-evaluated for woodland soils in Wales and revealed no change in soil carbon content between the two sampling points of those soils (Alton *et al.*, 2007).

The Countryside Survey provides an alternative source of information on changes in soil carbon content (Carey *et al.*, 2008), based on a similar methodology sampling to 15 cm at three time-points (1978, 1998 and 2007). Between 1978 and 2007, the carbon content of broadleaf woodland soils showed a significant increase, with no change reported for coniferous woodland soils. A combined analysis also revealed a significant increase. However, between 1998 and 2007 both broadleaf and conifer woodland soils showed a small but non-significant decline. The authors could not distinguish a climate change signal from the effects of changes in land management or atmospheric pollutant deposition.

The ability to interpret both the NSRI and Countryside Survey datasets (sampled to 15 cm) for woodland soils is limited because many woodland soils have deeper active

Figure 4.5

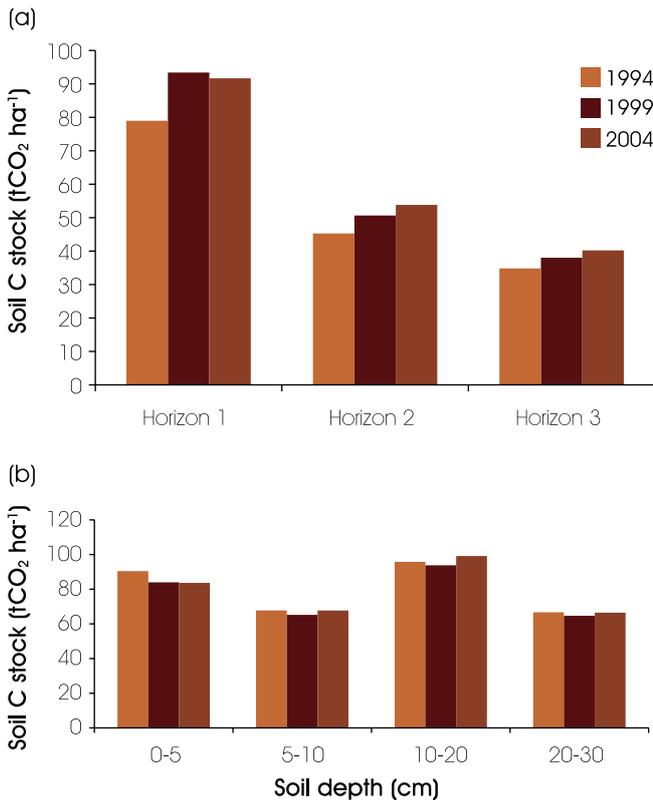
Distribution of values of percentage soil carbon changes under woodland (a) broadleaves and (b) conifers from the NSRI soil re-survey data sets (1978–2003; Morison *et al.*, 2009). Each bar represents an individual site. For further details and description of methodology, see Bellamy *et al.*, (2005).



rooting profiles than comparable agricultural soils. The Environmental Change Network provides an alternative source of data, for which the soils have been extensively assessed and characterised to 80 cm depth on a five-yearly basis (1994, 1999 and 2004). Soil organic carbon stocks in the top two (A and B) horizons at the Alice Holt woodland site increased over the three sampling periods at a rate of 1.1 tCO₂ ha⁻¹ year⁻¹ (Morison *et al.*, 2009; Figure 4.6). A separate study across a chronosequence of oak stands at Alice Holt reported a similar rate of carbon accumulation in woodland soils (0.7–1.1 tCO₂ ha⁻¹ year⁻¹ (Pitman *et al.*, 2009). The similarity in the reported rates suggests that woodland succession rather than climate change is the primary driver of this increase in soil carbon. An important observation is that when the data were reported by depth rather than by horizon, the increase in soil carbon stocks was not apparent (Figure 4.6). This highlights the inability of sampling the top 15 cm of woodland soils to adequately reflect changes in soil carbon stocks; any changes in soil carbon stocks resulting from the development of the soil profile will not be reported. In

part, this may explain some of the conflicting evaluations in soil carbon exchange that have emerged over recent years (see 3.5.1, Chapter 3).

Figure 4.6
Soil carbon stocks measured in (a) soil horizons and (b) soil depths in the ECN oak site at Alice Holt in 1994, 1999 and 2004.



4.7.2 Dissolved organic carbon

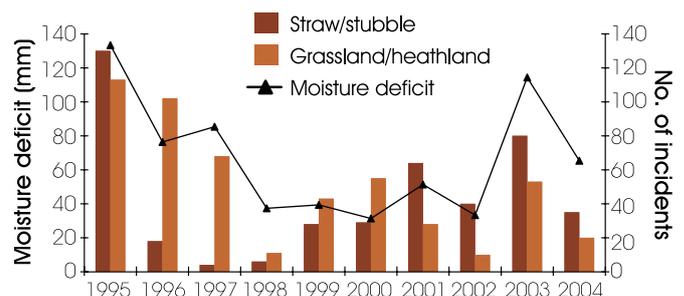
Some organic carbon can become dissolved in the soil water (DOC). There is clear evidence of an increase in levels of DOC over the past 20 years (Monteith and Evans, 2000), although the cause of this is unclear. It has been suggested that higher soluble carbohydrate levels resulting from rising carbon dioxide levels in the atmosphere may be responsible. There is also some evidence that rising temperatures leading to increased mineralisation may be implicated, while more frequent and intense wetting/drying cycles have also been shown to result in increased DOC losses (Hentschel *et al.*, 2007).

4.8 Damage to woodland

4.8.1 Forest fires

An increasing trend for outdoor ‘grassland’ fires has been observed over recent decades (CLG, 2008). This contrasts with the number of forest fires (on Forestry Commission land) which have declined over the period for which data are available (1975–2004: Forestry Commission, 2004). The decline is likely to result primarily from the reduction in the area of thicket stage conifer woodland which is at greatest risk of fire. Cannell and McNally (1997) also noted that climate has generally been the catalyst for forest fires but rarely the actual cause, as most fires are human-induced whether intentional or not. The area of woodland burned increased in the well-documented ‘drought’ years such as 1976, 1989–90 and 1995, although no increase was evident in 2005. The very large area burned in 1976 was, in part, a result of the ground flora drying out in late summer, leading to a second fire season additional to the normal late-spring fire season where the previous year’s litter provides the fuel source. Gazzard (2006) clearly demonstrated the link between soil moisture deficit and number of outdoor fires reported in southern England (Figure 4.7). In most cases, forest fires in the UK are low temperature burns (Mitchell *et al.*, 2007) and do not significantly damage the wider woodland ecosystem, although economic damage to forestry crops is clearly serious. In contrast to the UK, forest fires in more southerly regions of Europe represent a much greater threat with, for example, an area in excess of 800 000 ha burned in 2003. Although this indicates that forest fires are likely to become a greater risk with drier weather, it is unlikely that such large-scale forest fires would result in the UK because of the highly fragmented nature of woodland cover.

Figure 4.7
Incidence of straw/stubble (brown) and grassland/heathland (light brown) fires in southeast England compared with calculated soil moisture deficit for the Alice Holt climatological station. (After Gazzard, 2006.)



4.8.2 Wind

Wind damage to UK forests has been reviewed by Quine and Gardiner (2002), who concluded that no trend in wind damage has been discernible over the course of the 20th century. However, Lindroth *et al.* (2009) reported a significant increasing trend of forest damage caused by wind storms in continental Europe over the past 60 years. The most significant single storm event in terms of damage to growing stock was the 1987 storm in southern England, during which 3.9 million m³ was lost – a volume equivalent to a little less than half of annual timber production for the UK – thus highlighting the seriousness of the risk.

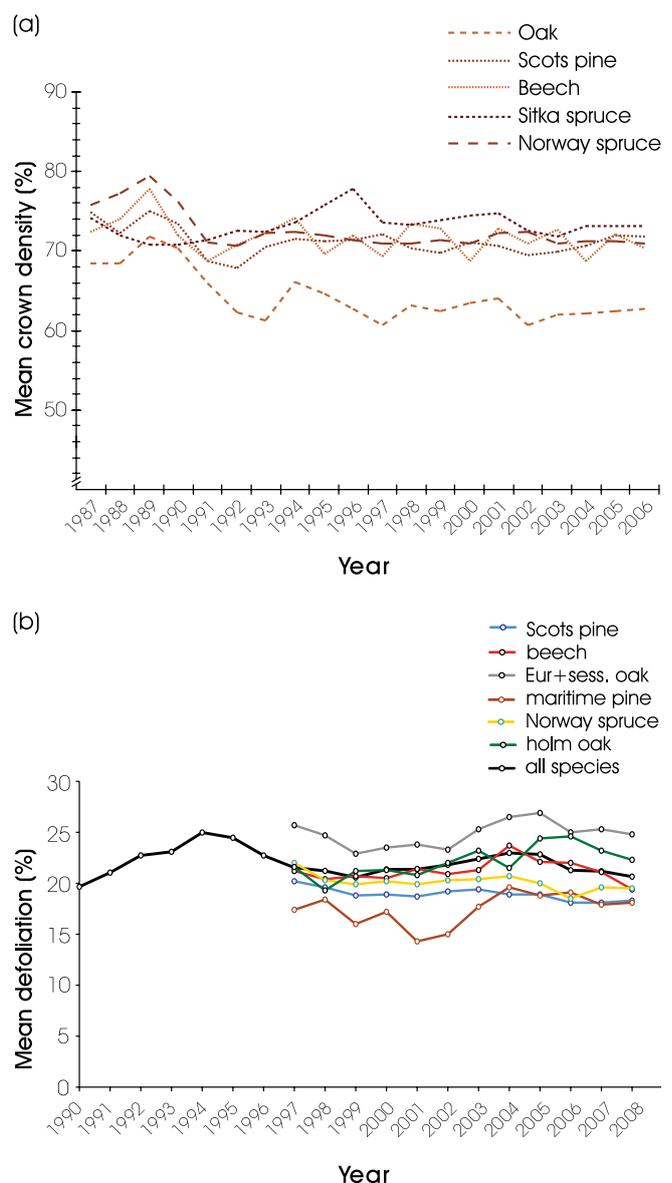
4.8.3 General condition

Although the Forest Condition Survey was suspended in 2006, it provided a time series reporting mean crown density over nearly 20 years across ~350 plots (Figure 4.8a; Hendry, 2005). Importantly, the methodology was common across the EC ICP-Forests Level I Network of which the Forest Condition Survey was a component. Therefore, a picture of how forest condition in the UK relates to trends across Europe can be drawn (Figure 4.8b; Lorenz *et al.*, 2008).

Across Britain, crown condition of oak showed the greatest decline between 1987 and 2004, in line with observations across Europe that deciduous oak species show the greatest level of ‘defoliation’ of the six species groups reported. Oak dieback (Gibbs, 1999) was evident over a number of years, while more severe, acute, dieback has been reported more recently on a number of plots. However, abiotic influences (wind and hail damage), together with defoliating insect damage were generally the reason of poor crown condition. Norway spruce also showed a statistically significant decline in crown density over the 20-year period that the survey operated, although there has been no further decline since 1991. Beech has shown large inter-annual fluctuation in crown condition, generally related to the well-documented response of heavy fruiting to weather conditions (Piovesan and Adams, 2001) that has been implicated in the decline of growth in the species (see 4.3 above and Figure 4.4b). Across Europe, beech also showed a marked decline in crown density in 2004, following the severe summer drought conditions experienced over much of Europe in 2003. There has been limited variation in the condition of Scots pine, with no trends evident. Similarly, there has been no trend evident in the condition of Sitka spruce, although the inter-annual variability is large, in part related to green spruce aphid-driven defoliation episodes.

Figure 4.8

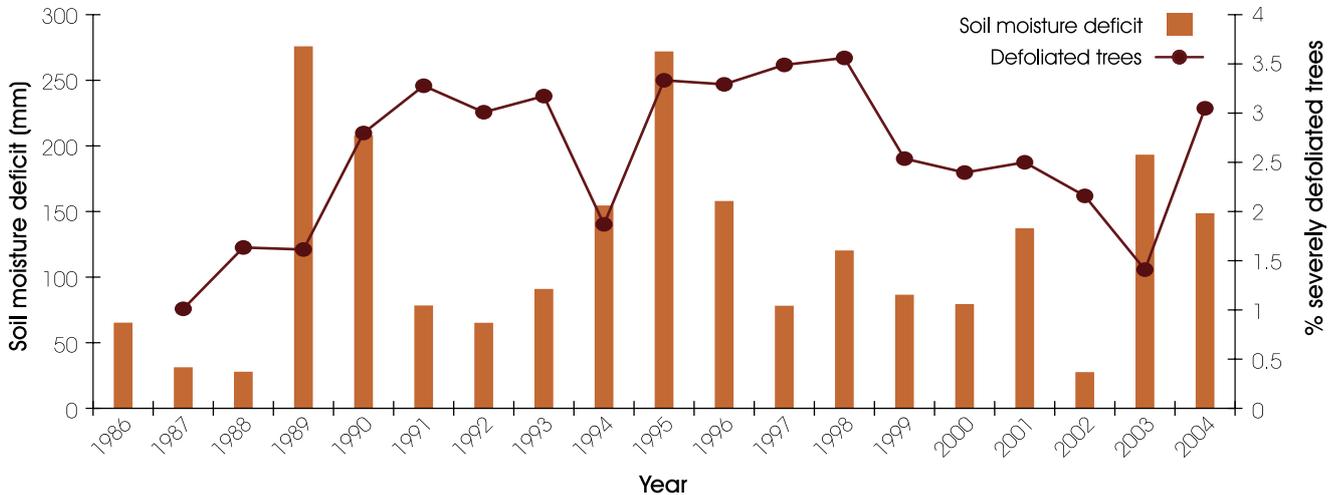
(a) Changes in crown density since 1987 for five tree species surveyed annually. The crown density compared with that of an ‘ideal’ tree with a completely opaque crown is shown for each species. (b) Mean defoliation index in forests in Europe for commonly reported tree species and for the total of all tree species. Samples only include countries with continuous data submission. Sample size for the selected main tree species varies between 1950 and 26 788 trees per species and year. The time series, starting in 1990, is available for a smaller number of countries and is based on between 38 026 and 45 204 trees depending on the year.



There is therefore no evidence from the Forest Condition Survey of direct impacts of climate change, although an indirect climate change signal could be inferred from the decline in the condition of oak. In reality, the time series of the Forest Condition Survey is too short and the range

Figure 4.9

Comparison between the proportion of severely defoliated trees and soil moisture deficit between 1986 and 2004. Proportion of defoliated trees representative of all five species (oak, beech, Scots pine, Sitka spruce, Norway spruce) in south and east England from the Forest Condition Survey (Hendry, 2005) and Soil moisture deficit calculated for the climatological station at Alice Holt, Surrey.



of interacting factors too wide to deduce a clear climate change signal at this stage. However, the impact on crown condition of the well-documented summer droughts of 1989/90, 1995 and 2003 is clear (Figure 4.9), providing insight into the likely impacts on woodland condition as climate change progresses. It is notable that the increase in crown transparency induced by drought the previous year is maintained for 3–4 years after the year in which the drought conditions prevailed.

4.9 Research priorities

- **Monitoring the changes.** There is no established framework for collecting or evaluating information on forest growth and productivity which has contributed to the lack of clear evidence of growth trends of British forests. If an appropriate framework is not put in place, it will therefore be difficult to identify when climate change starts to impact – either negatively or positively – on forest growth. This, in turn will delay the implementation of more extreme adaptation measures and changes in species choice or management systems, should that prove necessary. The development of a framework for monitoring and evaluating changes in increment on an annual basis is therefore an urgent requirement. We suggest that this is developed as a UK climate change indicator for the forestry sector and reported on annually.
- **Understanding the effect of rising temperatures.** There is clear evidence that rising temperatures are having an impact on a range of physiological and ecological functions. In particular, the timing of natural events

shows a strong temperature response. Further research is required to evaluate how the phenological changes observed to date have interacted and impacted on the wider woodland ecosystem.

- **Understanding the impacts of extreme events.** The impacts of extreme events on forest ecosystems needs to be better understood, particularly the effects of prolonged drought on tree physiology and mortality. Responsive monitoring programmes should be planned to ensure that the impacts of extreme climatic events are captured to provide an insight to the likely effects of future climate change. In particular, the early detection and anticipation of forest dieback will be essential to allow early adaptation to limit the impacts of climate change.

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AN ASSESSMENT OF LIKELY FUTURE IMPACTS OF CLIMATE CHANGE ON UK FORESTS

Chapter

5

M. S. J. Broadmeadow, J. F. Webber, D. Ray and P. M. Berry

Key Findings

An increased frequency and severity of summer drought is likely to represent the greatest threat to woodlands from climate change. There is a very high likelihood that there will be serious impacts on drought-sensitive tree species on shallow freely-draining soils, particularly in southern and eastern Britain. These impacts will be widespread in plantations already established. They will necessitate a reassessment of the suitability of species for use in commercial forestry in all regions. The species currently available for use, assuming that appropriate provenances are selected, will remain suitable across much of the UK. However, by the end of the century, impacts in the south and east of England may be sufficiently severe to necessitate the introduction of new species. Planning of which species and species mixtures to plant where, will be the challenge for forest managers.

Models suggest that widespread impacts of climate change on the suitability of most major species currently planted in the UK only become apparent by the middle of the century under the UKCIP02 'High emissions scenario' and towards the end of the century under the UKCIP02 'Low emissions scenario'. Typical conifer rotations currently in the ground are therefore likely to reach maturity before serious impacts are apparent. However, this means that appropriate modification of species choice must be undertaken from now on when restocking and creating new woodlands.

Pests and diseases of forest trees, both those that are currently present in the UK and those that may be introduced, themselves represent a major threat to woodlands. These threats may be increased by interactions with the direct effects of climate change on tree function.

By the end of the century, some native tree species are likely to lose 'climate space', particularly in southern England. The southern limit of the range of many species that retain climate space will have moved close to the UK suggesting that they will struggle on many sites, their regeneration and successful establishment will decline and they could be out-competed by introduced species. The distribution of tree species will inevitably change in response to climate warming. However, if the trees are to stay within appropriate climatic envelopes there will be a requirement for species migration rates to be more than 10 times faster than those achieved in reaching present distributions after the last ice age.

Increased winter waterlogging may render increased areas of upland plantation forest liable to possibly catastrophic wind disturbance, whether or not climate change leads to a change in the wind climate of the UK.

The impacts of climate change are likely to be first seen on establishment, but widespread mortality is initially, unlikely. However, as climate change progresses, the levels of mortality of mature trees will increase as a result of direct and indirect impacts; street, hedgerow and free-standing/isolated trees will show higher levels of mortality because of their higher water demand.

There are likely to be significant changes to the composition, structure and character of woodland ground flora; current species descriptions of native woodland communities are unlikely to remain valid. The range, and ability of the majority of priority woodland species (both flora and fauna) to persist will change as a result of climate change; some will decline further, others will benefit. The current species-based approach to nature conservation will be difficult to maintain in the long term because of the likely degree of change.

Predicting the future can only be an uncertain science, particularly given the interactions between the components – both fauna and flora – of woodland ecosystems. However, the link between climate and tree growth has been well documented, from as far back as 1662 (Evelyn, 1729). The knowledge gained, mostly over the course of the 20th century, on the specific requirements of individual tree species and woodland ecosystems has enabled a range of models of tree growth and suitability to be developed.

A number of such models (see Ray *et al.*, 2002) have been used to identify how forest productivity and ecosystem function may be influenced by projected climate change. Although it is impossible to make such predictions with certainty (and their output should be treated with caution – see Walmsley *et al.*, 2007), these models do represent a useful tool for developing adaptation strategies.

An alternative approach is to interpret the documented impacts of extreme climatic events or past climatic trends in terms of future climate projections. Again, these provide a useful insight into the future, but such analyses should be treated with some caution because the combination of conditions in extreme climate events of the past is unlikely to be replicated in the future.

A third option for informing us of the likely impacts of climate change is to investigate the composition and performance of forest plantations and woodland ecosystems growing in climates more representative of that of the future for a given region (see Broadmeadow *et al.*, 2005). Although ‘climate analogues’ for the range of climate variables that influence tree growth and ecosystem function rarely exist, the approach has merit in providing information at a broad level on change impacts that might be expected and changes in management practice that would be appropriate.

This chapter therefore considers information on the future impacts of climate change on trees and woodlands from a range of approaches. Evidence from experimental impact studies, spatial analyses linked to climatic variation, and model predictions is evaluated. This then allows a broad assessment of the likely impacts of climate change on UK forestry and woodland ecosystems to be made, and enables a commentary on the robustness of the predictions.

The likely changes in climate for the UK have been outlined

in Chapter 2, which takes into account key results from the recent release of the UK Climate Projections (UKCP09: Murphy *et al.*, 2009). However, much of the existing analysis of future climate impacts has used the UKCIP02 projections (published in 2002), and impact studies using UKCP09 are not yet available. Where appropriate, the High and Low scenarios for these earlier projections can give an indication of the uncertainty associated with such impact studies.

5.1 Direct impacts of climate change on forest productivity

5.1.1 Direct impacts of rising carbon dioxide levels

Pre-industrial concentrations of carbon dioxide are sub-optimal for tree photosynthesis (see 3.2.2, Chapter 3). Rising concentrations will therefore have a ‘growth-stimulating’ effect and there is a large body of supporting experimental evidence (see Broadmeadow and Randle, 2002). Most experiments have been carried out on young trees under controlled environment conditions (Curtis and Wang, 1998). An average growth enhancement for above ground biomass of 51% across 21 studies for a doubling of CO₂ from 350 to 700 parts per million volume (ppmv), corresponds with a 51% increase in maximum photosynthetic rates (ECOCRAFT, 1999: see Broadmeadow and Randle, 2002). Other aspects of tree physiology were also affected, including a small reduction in respiration rates; a larger reduction in stomatal conductance and corresponding water use; reduced leaf nitrogen (and other nutrients) content; larger leaf area index (leaf area per unit ground area); and a tendency to greater below-ground biomass allocation. In particular, the effects of CO₂ concentration on function have implications for tree responses to the changing climate. The reduction in stomatal conductance for some species will enhance

water use efficiency and also reduce the damaging effect of ozone by reducing uptake (see below); the larger leaf area may increase risk of wind damage and also reduce light transmission to the forest floor affecting ground flora community structure. There may also be interactions with insect pests as a result of higher soluble carbohydrate levels in the leaf and phloem sap (Broadmeadow and Jackson, 2000; Evans *et al.*, 2002), but also other effects through higher concentrations of plant secondary metabolites (Penuelas *et al.*, 1997), some of which have effects on insect herbivores.

A recent meta-analysis of experimental data concluded that an average 23% increase in net primary productivity would be expected for a doubling of CO₂ from the pre-industrial concentration to 550 ppmv (Norby *et al.*, 2005). However, for practical reasons the majority of experiments have been carried out on young trees. Four free air carbon enrichment (FACE) experiments in mature woodlands of sweetgum (*Liquidambar styraciflua*), loblolly pine (*Pinus taeda*) and aspen (*Populus tremuloides*) show that there is significant photosynthetic downregulation (Schafer *et al.*, 2003; Norby and Luo, 2004; Körner *et al.*, 2005), so that growth rate stimulation is not as large as with young trees. These FACE studies have also demonstrated increased carbon storage in the soil, although it is uncertain what the overall balance will be between CO₂-driven increases and temperature-driven decreases in soil carbon. One FACE system has been operating in the UK, investigating competition between native trees and impacts of elevated CO₂ on soil processes, but this study was also with young trees (see: www.senr.bangor.ac.uk/research/themes/ess/climate_change.php). Karnosky *et al.* (2007) reviewed the published results from all FACE experiments identifying an overall increase of forest productivity but with considerable variability determined by genotype, tree age, climate, air pollution and nutrient availability. The authors also identified a number of research gaps.

Modelling systems and operational decision support systems that are used to predict the impacts of climate change on forest growth, tree species distribution or woodland community composition (e.g. ESC: see Ray *et al.*, 2002; SPECIES: see Pearson *et al.*, 2002) are based on relationships between past/current climate and plant performance. While these models can accommodate climate change scenarios with relative ease, they do not incorporate the direct and indirect effects of rising carbon dioxide levels outlined above. It is therefore likely that they underestimate the positive effects of future climate and

atmospheric composition change and may provide an unduly negative picture of the impacts. The limitations of the modelling approaches are further explored in 5.2 and 5.3 below.

5.1.2 Impacts of ground-level ozone pollution

Models of climate change and atmospheric chemistry suggest that levels of ozone pollution are likely to increase significantly during this century (NEG-TAP, 2001; Royal Society, 2008). Unlike most other pollutants, ozone affects foliage directly, and its harmful effects are primarily caused by ozone molecules entering the leaf through the stomata (see Chapter 3). There is some evidence that ozone can degrade leaf surface waxes increasing water loss. However, ozone principally affects the internal components of leaves through an acceleration of cell senescence and local cell necrosis in the photosynthetically active tissues. The physical disruption of the photosynthetic apparatus often results in reduced levels of chlorophyll, a lower photosynthetic capacity and advanced leaf senescence. The degradation of chlorophyll can be seen as generalised, diffuse 'chlorosis', i.e. yellowing of the foliage. The lower photosynthetic capacity and the continuing costs (in energy terms) of repairing cellular damage often results in reduced growth, even at current ambient exposure levels in the UK. In conifer species, advanced leaf senescence appears as reduced needle retention, with fewer age-classes of needles retained in areas experiencing ozone pollution (NEG-TAP, 2001).

Changes in carbon allocation resulting from diversion to ozone damage repair also leads to reduced root biomass with the result that ozone exposure can increase the vulnerability to drought. Furthermore, ozone impairs the functioning of the stomata with the result that stomatal closure in response to drought is compromised, thus compounding any effects of water shortage. Several statistical analyses (outside the UK) have shown links between ozone exposure and forest productivity and condition supporting the findings of research in controlled environment facilities. Dose-response relationships have recently been developed for a limited number of tree species. Further research may enable a quantitative assessment of the cost of ozone pollution to be made.

Emissions control policies for volatile organic compounds (VOCs) and nitrogen oxides, the main precursors to the formation of ozone (see Chapter 3), have led to a recent reduction in peak ozone concentrations. However, against

this fall in peak concentrations has been a steady rise in background concentrations. Model predictions are for this trend to continue, with background concentrations rising to approximately double the current value by the end of this century. This increase should be viewed in the context of the impacts on tree health and productivity that current ozone concentrations already have, which may be reducing the productivity of sensitive tree species in some parts of the UK, and in some years, by up to 10%. Ozone pollution may also make trees more susceptible to biotic and abiotic damage, thus compounding the direct effects on growth.

5.1.3 Direct impacts of rising temperature

Most aspects of the direct impacts of climate change (wind, rainfall patterns and rising CO₂ levels) as well as indirect impacts (through changing pest/pathogen risk, for example) are dealt with either in specific sections or as part of the overall impacts on growth and distribution assessed in 5.2.1 and 5.3 below. However, two effects of rising temperature are worthy of further consideration here.

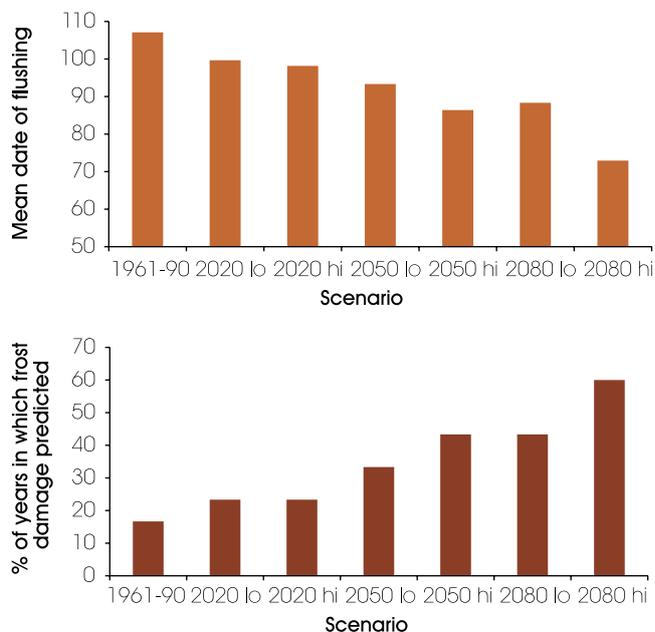
Leafing date and frost risk

Rising temperatures would be expected to lead to a decline in spring frost risk. However, late spring frosts are unpredictable and there is little evidence that the date of the last spring frost has changed in recent decades. In contrast, the advance in leafing is well documented (see 4.2, Chapter 4). If the trend in leafing date was to continue, as seems likely on the basis of models of budburst available in the literature (Sparks and Gill, 2002), but with no or little corresponding change in the date of the last frost, the risk of late spring frost could increase (Figure 5.1). Spring frost damage is an area of concern because of its effect on form and resulting timber quality. Furthermore, repeated frost damage can reduce productivity significantly and, in extreme circumstances, result in mortality, particularly of young trees (Redfern and Hendry, 2002).

Chilling requirement

Some growth stages of some tree species have an obligate chilling requirement. These processes include bud set and the breaking of seed dormancy. The majority of tree seeds in the UK exhibit one of two sorts of dormancy. A few (alder, birch and Scots pine) have seeds that exhibit 'shallow dormancy'. In this type of dormancy a varying proportion of seeds germinate at different temperatures and all seeds respond to a relatively short pre-chill which stimulates faster germination at all temperatures and improves the maximum germination percentage at most

Figure 5.1
Impact of projected climate change (UKCIP02 low and high emissions scenarios) on (a) leafing date of oak (expressed as day of year from 1 January) and (b) proportion of years in which leafing will occur before the date of the last -2°C spring frost, assuming no change in the date of the last frost. See Sparks and Gill (2002) for details of the leafing model.



temperatures (Gosling and Broadmeadow, 2006). If climate change brings warmer autumn temperatures, there is a risk that the seeds of these shallow-dormancy species may germinate too soon in autumn and be vulnerable to frost that winter. However, the native tree species that are most likely to be affected by climate change are those with 'deeply dormant' seeds, including juniper, yew, and nearly all broadleaves (e.g. ash, beech, cherry). Freshly shed seeds of these species have a complete metabolic block to germination at any temperature and there is an absolute requirement for a relatively lengthy and unbroken period of cold moist conditions to bring about any germination at all. If climate change brings about winters that are warmer or shorter or both, and these are succeeded by faster rising spring and summer temperatures, then many of these species may not be as well suited to natural regeneration in the projected climate of the future. Variability exists within populations and natural selection will favour those individuals that obtain sufficient chilling. However, variation between populations is likely to mean that more southerly provenances that require less winter chilling may be better adapted to the climate of the future than local/native populations. This potentially has very serious implications for native woodlands and requires further investigation

through controlled experimentation and field-based 'reciprocal transplant' trials.

5.2 Assessing future impacts of climate change on tree species suitability and forest productivity using Ecological Site Classification

5.2.1 Modelling approach

The Forestry Commission has developed a knowledge-based model, the Ecological Site Classification (ESC; Pyatt *et al.*, 2001), to map the suitability of tree species to a site. Suitability in this context is defined in terms of growth relative to maximum growth rates achieved in the UK. ESC also has the capability to model the suitability of native woodland ecosystems on the basis of climatic and edaphic conditions. It is important to note in this and subsequent sections the differences between suitability for commercial timber production, the ability to persist in the medium to long term and other ecological functions.

The ESC approach uses six biophysical factors to describe tree species suitability and yield potential for application at individual site level. Two are soil factors – fertility and moisture availability; the remaining four are climatic factors including accumulated temperature (warmth index), moisture deficit (droughtiness index), wind exposure, and continentality. The ESC system has also been developed as a spatial tool for selecting tree species in design plans at the operational scale and for assessing regional suitability of species for different forest types (Ray and Broome, 2003). The regional scale spatial tool has also provided a framework to evaluate the likely impacts of climate change on tree species suitability.

When interpreting the output of the Ecological Site Classification model, particularly in the context of climate change, a number of important caveats (most of which equally apply to the SPECIES model: see 5.3 below) should be considered:

- ESC evaluations are based on mean climate, and extreme climatic events such as windstorms and unseasonal frost (see 4.2, Chapter 4) are not considered;
- Climatic and, in some cases, edaphic factors are represented at coarse resolution (5 km grid), which does

not fully represent the variability in site and micro-climate that can be exploited by trees;

- The impacts of pests and diseases are not considered;
- The beneficial effects of rising atmospheric carbon dioxide levels are not considered;
- Competition between tree species is not considered;
- ESC is an empirical model based on tree performance under British conditions. Climate change projections, particularly the more extreme scenarios, are beyond the knowledge-base of the model and extrapolation has therefore been necessary.

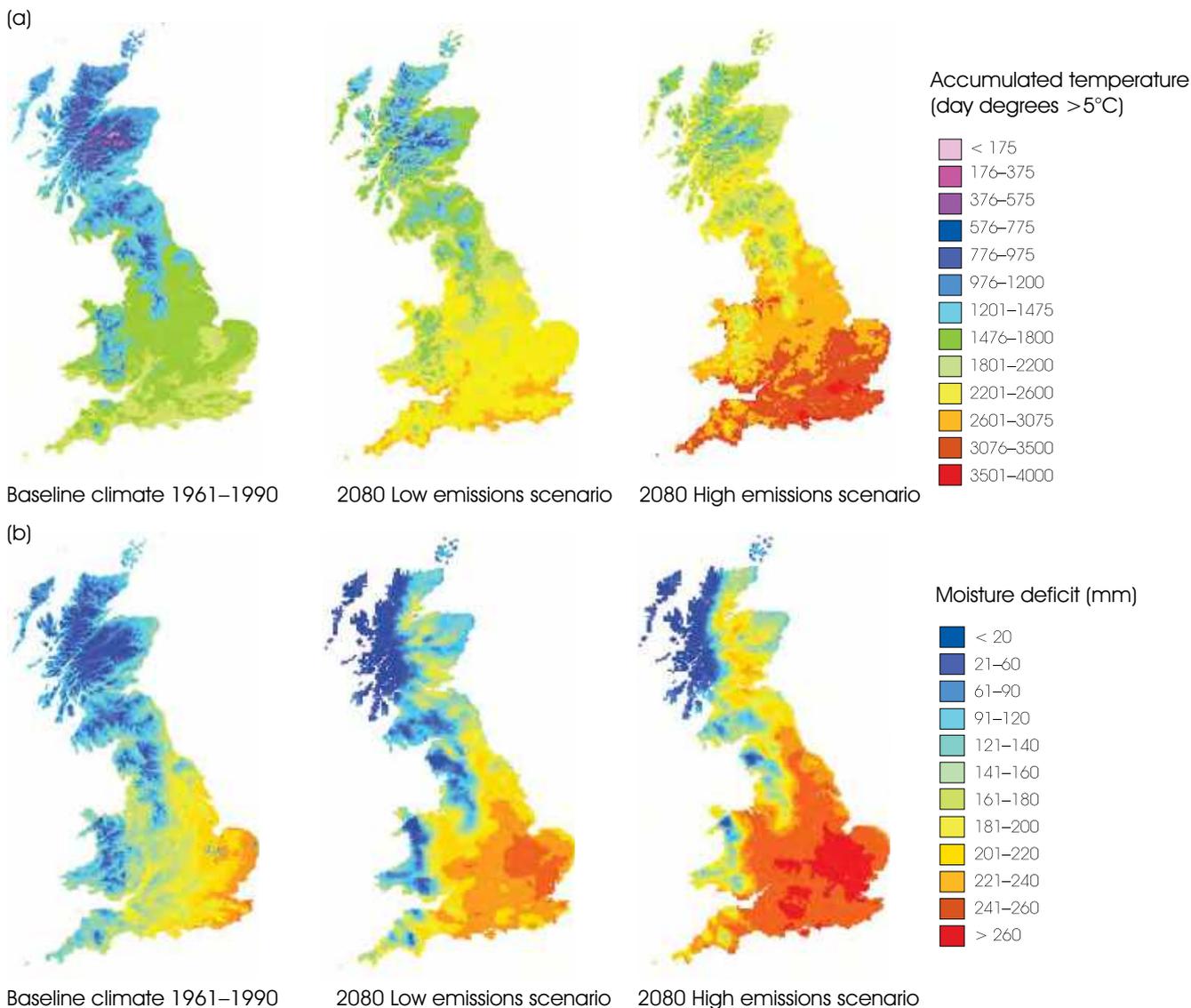
5.2.2 Model results

Climate projections for the UK (UKCIP02: Hulme *et al.*, 2002) have been used to adjust accumulated temperature and moisture deficit, two of the climate factors in the ESC model (Ray *et al.*, 2002; Broadmeadow *et al.*, 2005) with dynamic linking between climatic moisture deficit and summer soil moisture regime. The data and ESC model have provided a low resolution, strategic planning tool to indicate the kinds of changes that may occur in the climatic factors used to specify tree species suitability and site yield potential.

By the 2080s, climate change is projected to have a profound effect on accumulated temperature and moisture deficit in Britain (Figure 5.2). UKCIP02 high emissions projections (consistent with the IPCC A1FI greenhouse gas (GHG) emissions scenario; Nakicenovic and Swart, 2000) show accumulated temperature could increase by as much as 40–50% in south Wales and central Scotland, and by as much as 60% in south east England. For a relatively cool climate, such as the baseline climate of Britain, large increases in accumulated temperature occur due to small increases in the mean daily temperature above 5°C. There are two reasons for this, warmer seasonal temperatures and a longer growing season. Moisture deficits are also likely to increase by up to 40% in eastern and southern Wales and central Scotland, and increases of up to 50% are likely in southeast England. This would cause forest soils in affected areas of Britain to become depleted of moisture for tree growth and, on freely draining and shallow soils, would be serious for species that are sensitive to drought conditions in excess of 180–200 mm (e.g. Sitka spruce, beech, ash). Indeed, on such sensitive sites, the High emissions scenario projections of frequent and extreme moisture deficit would seriously reduce the growth and suitability for most tree species currently grown in Britain. Work by Herbst *et al.* (2007) showed that evapotranspiration at forest edges is

Figure 5.2

Projected changes in (a) accumulated temperature above a base of 5°C and (b) calculated moisture deficit for future climate projections simulating the UKCIP02 2080s low (IPCC B1) and high (IPCC A1FI) emissions scenarios.



much higher than from the middle of the forest. Where the impacts of climate change are primarily related to water availability, it would be expected that small fragmented woodlands would be more susceptible than large, continuous forest areas.

Outputs from the ESC spatial model show changes in the suitability of different tree species by region. Figure 5.3 shows Britain divided into eight regions. For the area within a region described as woodland (Forestry Commission, 2003), the suitability of five major forestry tree species has been calculated for the baseline climate, and the projected climates of 2050 and 2080 simulated for both Low (IPCC B1) and High (IPCC A1FI) emissions scenarios. It is

important to note that these projections do not differentiate between where a particular species would or would not be planted; they are projected changes in average suitability for an individual species across the current total planted area, much of which in the uplands is marginal land. As a consequence, they do not predict performance of these key species on less marginal land.

Several points emerge from this analysis of predicted changes to suitability relative to the baseline climate scenario. In the case of beech under all but the 2080s High emissions scenario, the area of woodland falling into the 'very suitable' category in east and west Wales, northern England, and eastern and western Scotland is predicted

to rise. In contrast, the areas designated 'suitable' for this species decline in eastern England to the extent that, under the 2080s High emissions scenario, there is virtually no area deemed 'suitable' for beech in the region. The model thus predicts the likelihood of a significant regional shift in the occurrence of beech as a productive forest species. In the case of the major commercial timber species, Sitka spruce, there are projected increases in the very suitable category in west Wales, west Scotland and northern England. A decline in suitability of this species, relative to the baseline climate scenario is seen in east Wales and western England. Under most scenarios, with the exception of the 2080s High emissions scenario, Scots pine maintains its suitability in all regions, although it is noteworthy that under this extreme scenario it is predicted to be completely unsuitable for use in western and eastern England. The pronounced decreases in the areas of existing woodland designated as suitable or very suitable for most of the five species in east and west England is attributable to the higher soil moisture deficits projected for the growing seasons in these areas under all future climate scenarios. It should be emphasised here that although the suitability of many of our tree species is likely to decline under these scenarios, (particularly on freely draining shallow soils) the expression of these effects is likely to be seen in the form of reduced timber productivity rather than complete loss of the species from our woodlands. Further, under the projected conditions of climatic stress it is likely that there will be associated biotic damage.

Wetter winters throughout Britain, and particularly in the uplands on imperfectly- and poorly-draining soil types, will cause increased areas, and longer periods, of anaerobiosis. This is very likely to reduce the rooting depth for many shallow-rooting tree species, as well as tree species that are intolerant of a fluctuating water table. The impact of a reduced rooting depth is a reduction in tree stability in forest stands. Whether or not the winter wind climate becomes increasingly stormy, with higher magnitude and/or more frequent damaging events, the reduction in stability through increased water-logging will lead to greater areas of endemic and occasionally catastrophic wind disturbance. Forest management systems must be adapted to reduce the potential for increased damage through using self thinning mixtures, short rotations and perhaps a reversion to native woodland or open habitat on the most affected forest sites.

A summary of the results for an extended range of forestry species is presented in Table 5.1, but on the basis of all

land within a region, rather than being restricted to existing woodland areas as is the case in the analysis presented in Figure 5.3. Because all land within a region is considered in the analysis, the indicative suitabilities given in Table 5.1 are low, reflecting the inherent unsuitability of much of the land for most tree species in some regions. The important point is to consider the change relative to the baseline scenario. Suitability is presented for three contrasting regions in Great Britain (southeast England, north Wales and Perth and Argyll) to indicate the direction of change of suitability of individual species. Further details are available at www.forestresearch.gov.uk/climatechange.

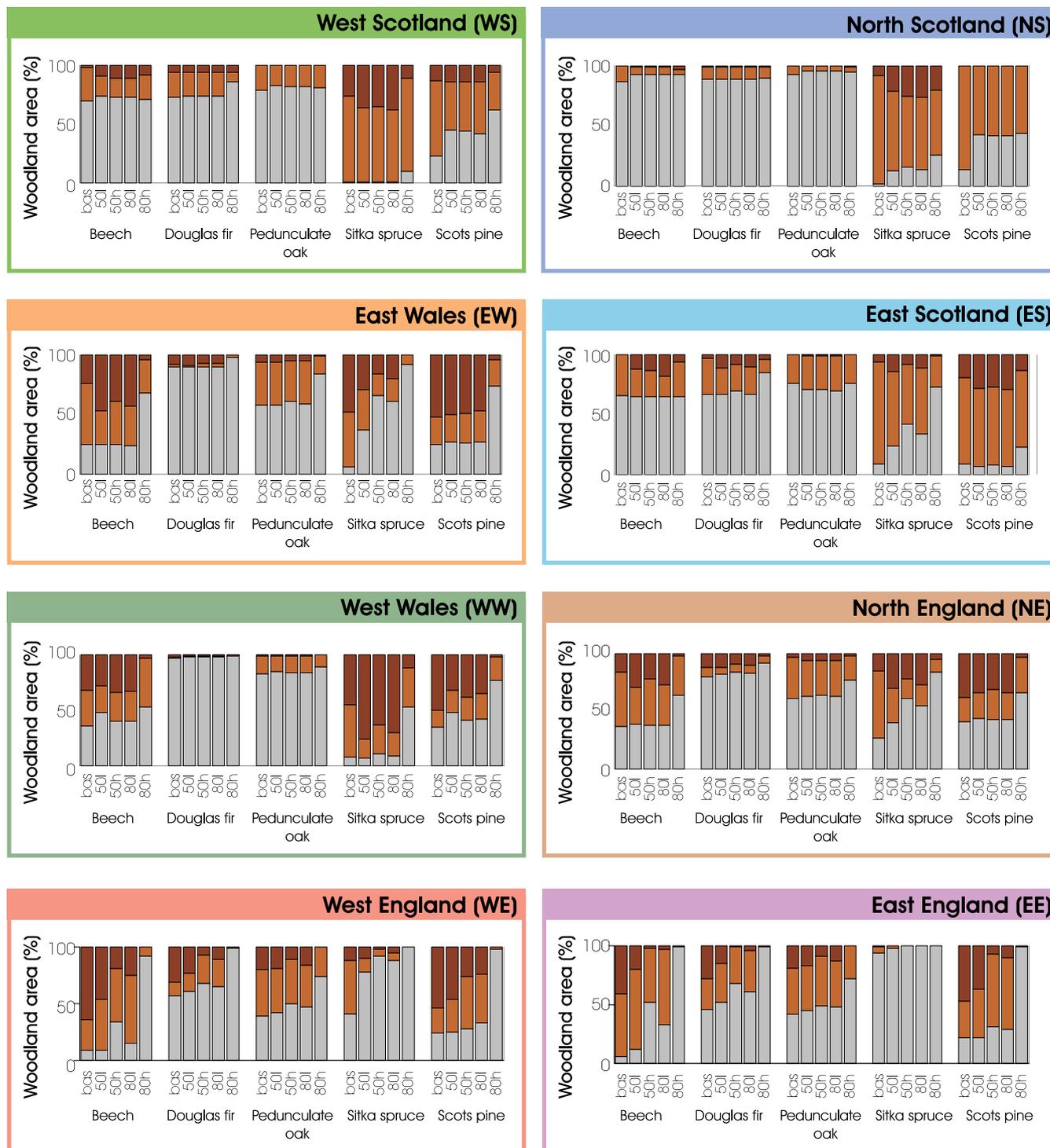
The analysis presented in Figure 5.3 suggests that in most regions all five of the tree species considered – whether grown for commercial timber production or as productive components of semi-natural woodland – will be challenged by climate change. However, the broader picture presented in Table 5.1 suggests that for many species in some regions, particularly 'minor' or less commonly planted species, the consequences of climate change may be less extreme. Furthermore, for both North Wales and Perth and Argyll, the analysis suggests that an increase in productivity for many species – both conifers and broadleaves – is likely as a result of climate change. Indeed, in Perth and Argyll, the suitability of 20 out of the 28 species assessed is predicted to increase under the 2080s High emissions scenario relative to the baseline. In contrast, the implications for southern England are of real concern and suggest that a different approach to silviculture will be required, including the use of alternative species. Here, the suitability of all but two species (Norway maple and sweet chestnut) is predicted to decline under the 2080s High emissions scenario and of the conifer species, Corsican pine is the only one predicted not to be unsuitable under this more extreme scenario.

Although there will clearly be challenges for British forestry, there will also be real opportunities for the forestry sector to exploit:

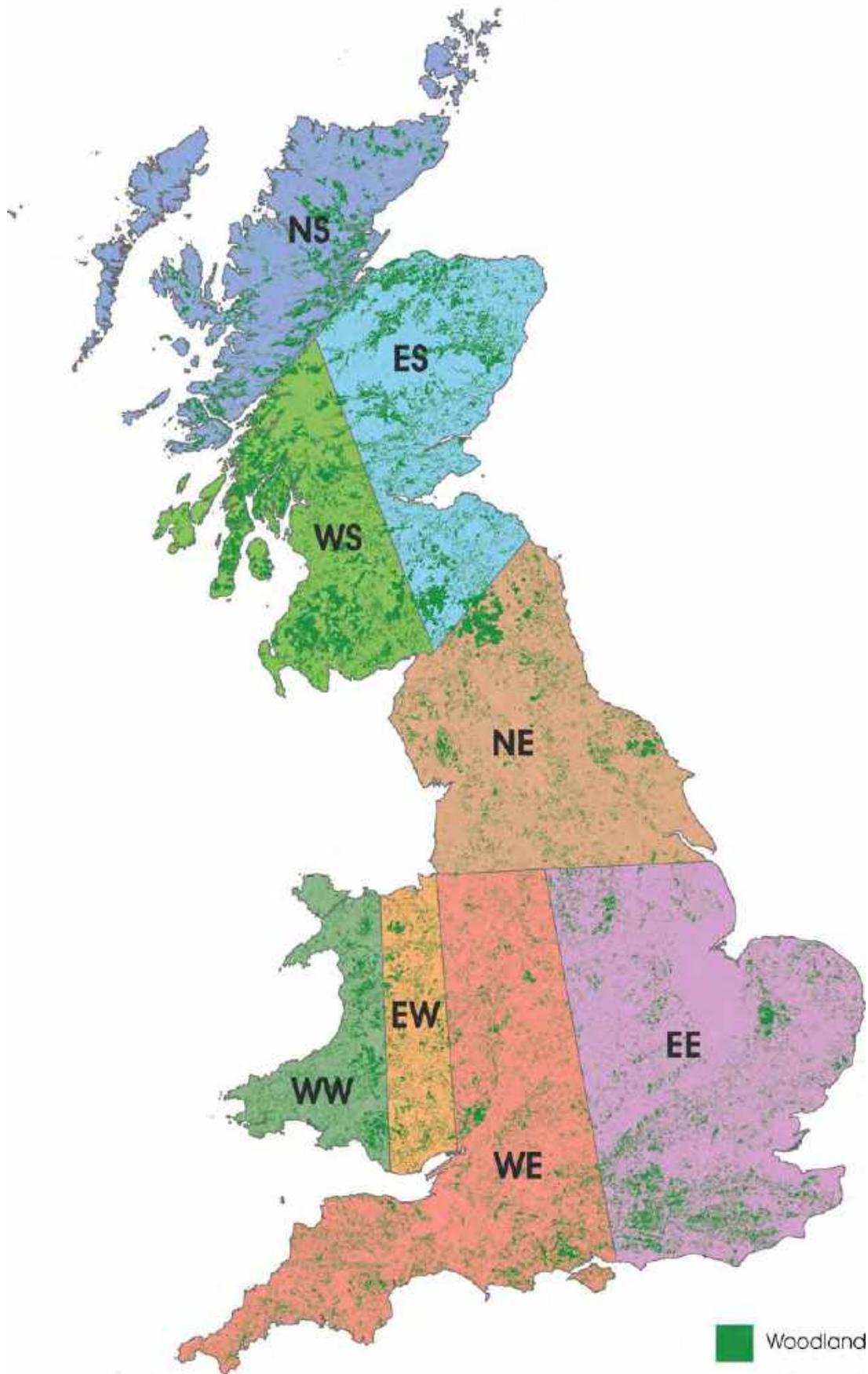
- increases in productivity for some existing commercial species in the north and west;
- changes in the range of existing commercial forestry species;
- maintenance of productivity of major commercial species conferring competitive advantage in comparison with many other areas of Europe;
- opportunities to plant new species and provenances.

Figure 5.3

Suitability of five tree species, described as the proportion of the current woodland area that is unsuitable, suitable or very suitable for that species in eight regions of Britain (see map on page 75), and simulated for the baseline climate, and the future climate scenarios 2050 Low (50 l), 2050 High (50 h), 2080 Low (80 l) and 2080 High (80 h) scenarios of UKCIP02 (Hulme *et al.*, 2002).



Very suitable Suitable Unsuitable



Section 2: Impacts

Table 5.1

Suitability as predicted by Ecological Site Classification (ESC) for a range of tree species in southeast England, north Wales and Perth and Argyll under four climate change scenarios. See text for guidance on interpretation of results. Values show productivity (YC) relative to the maximum currently achievable in the UK and is the average for all land within the region.

Key: ■ Very suitable (>70%) ■ marginal (40–50%) ■ unsuitable (less than 30%) ■ Suitable (50–70%) ■ poor (30–40%)

	Suitability											
	Southeast England				North Wales				Perth and Argyll			
	Baseline	2050 Low	2050 High	2080 High	Baseline	2050 Low	2050 High	2080 High	Baseline	2050 Low	2050 High	2080 High
Broadleaves												
Alder (<i>Alnus glutinosa</i>)	0.56	0.62	0.28	0.25	0.44	0.51	0.3	0.24	0.31	0.32	0.23	0.25
Ash (<i>Fraxinus excelsior</i>)	0.61	0.64	0.52	0.38	0.4	0.44	0.41	0.46	0.23	0.23	0.22	0.32
Aspen (<i>Populus tremula</i>)	0.65	0.69	0.45	0.38	0.55	0.61	0.48	0.45	0.36	0.38	0.33	0.42
Beech (<i>Fagus sylvatica</i>)	0.67	0.64	0.46	0.15	0.54	0.57	0.59	0.53	0.32	0.34	0.38	0.51
Downy birch (<i>Betula pubescens</i>)	0.39	0.38	0.16	0	0.55	0.6	0.34	0.24	0.46	0.46	0.28	0.3
Norway maple (<i>Acer platanoides</i>)	0.64	0.71	0.63	0.66	0.43	0.46	0.45	0.53	0.27	0.27	0.28	0.39
Pedunculate oak (<i>Quercus robur</i>)	0.63	0.68	0.56	0.37	0.36	0.42	0.41	0.41	0.21	0.23	0.23	0.33
Poplar (<i>Populus</i> spp.)	0.66	0.63	0.45	0.02	0.39	0.46	0.41	0.28	0.18	0.24	0.24	0.32
Rauli (<i>Nothofagus procera</i>)	0.38	0.46	0.23	0	0.25	0.34	0.29	0.28	0.09	0.11	0.11	0.17
Roblé beech (<i>Nothofagus obliqua</i>)	0.6	0.66	0.35	0.21	0.35	0.41	0.3	0.25	0.17	0.2	0.16	0.23
Silver birch (<i>Betula pendula</i>)	0.56	0.63	0.46	0.41	0.44	0.49	0.42	0.46	0.3	0.3	0.27	0.39
Sessile oak (<i>Quercus petraea</i>)	0.57	0.59	0.44	0.27	0.45	0.5	0.44	0.41	0.24	0.26	0.27	0.38
Sweet chestnut (<i>Castanea sativa</i>)	0.48	0.61	0.57	0.67	0.22	0.31	0.32	0.45	0.09	0.13	0.15	0.26
Sycamore (<i>Acer pseudoplatanus</i>)	0.65	0.53	0.35	0.16	0.52	0.54	0.46	0.4	0.31	0.33	0.3	0.38
Wild cherry (<i>Prunus avium</i>)	0.75	0.72	0.63	0.25	0.47	0.49	0.48	0.45	0.28	0.28	0.3	0.4
Wych elm (<i>Ulmus glabra</i>)	0.61	0.44	0.21	0.00	0.51	0.50	0.43	0.23	0.28	0.29	0.28	0.30
Conifers												
Corsican pine (<i>Pinus nigra</i>)	0.73	0.78	0.9	0.6	0.48	0.47	0.6	0.75	0.24	0.28	0.36	0.57
Douglas fir (<i>Pseudotsuga menziesii</i>)	0.67	0.65	0.56	0.28	0.48	0.51	0.56	0.5	0.22	0.22	0.34	0.44
European larch (<i>Larix decidua</i>)	0.54	0.52	0.27	0	0.49	0.48	0.42	0.35	0.28	0.26	0.25	0.34
Grand fir (<i>Abies grandis</i>)	0.30	0.24	0.01	0.00	0.36	0.41	0.29	0.18	0.21	0.22	0.16	0.19
Japanese larch (<i>Larix kaempferi</i>)	0.24	0.08	0.08	0	0.59	0.57	0.57	0.2	0.42	0.4	0.4	0.29
Lodgepole pine (<i>Pinus contorta</i>)	0.69	0.62	0.55	0.07	0.73	0.67	0.68	0.57	0.56	0.54	0.54	0.67
Noble fir (<i>Abies procera</i>)	0.59	0.53	0.25	0.05	0.64	0.65	0.37	0.14	0.39	0.40	0.29	0.28
Norway spruce (<i>Picea abies</i>)	0.59	0.52	0.26	0	0.63	0.62	0.54	0.39	0.4	0.4	0.37	0.45
Western red cedar (<i>Thuja plicata</i>)	0.41	0.29	0.10	0.00	0.55	0.54	0.42	0.22	0.39	0.39	0.34	0.32
Scots pine (<i>Pinus sylvestris</i>)	0.65	0.57	0.49	0.03	0.62	0.57	0.6	0.47	0.43	0.41	0.45	0.58
Sitka spruce (<i>Picea sitchensis</i>)	0.36	0.32	0.14	0	0.57	0.59	0.48	0.34	0.46	0.46	0.39	0.43
Western hemlock (<i>Tsuga heterophylla</i>)	0.51	0.41	0.32	0.01	0.54	0.55	0.18	0.29	0.39	0.4	0.38	0.38

5.3 Projected changes in tree species distribution

5.3.1 Modelling approach

The availability of species distribution maps and climate and soil data across Europe has allowed the development of models of potential species distribution (modelled on suitable 'climate space' – the geographical area that a given species could occupy on the basis of its climatic requirements alone) that can be used to examine the effects of projected climate change. Foremost among these is the SPECIES model (SPatial Estimator of the Climate Impacts on the Envelope of Species; Pearson *et al.*, 2002). Soil water availability, growing degree-day and temperature indices are used to define current climate space for individual species, with statistical comparison with distribution data enabling model performance to be assessed. Figure 5.4 presents the outputs from this model using climate projections from the Hadley Centre HADCM3 climate model assuming the IPCC SRES A2 GHG emissions profile (equivalent to the Medium-High emissions scenario of UKCIP02). Further details are available at: www.branchproject.org.uk and in Berry *et al.* (2007a). Although the maps presented in Figure 5.4 depict changes in climate space at a 10' resolution, this level of detail is considered by many as inappropriate for interpretation of necessary responses to meet conservation or forestry objectives because of uncertainties in climate and biological responses, as outlined in Walmsley *et al.* (2007). A number of important caveats should also be considered in interpreting the results. The key caveat is that such maps represent projections of where future climate space may be located for each species. They do not attempt to simulate the future distribution of species in response to climate change, and they do not take account of a species' capacity to disperse, or the presence of suitable habitat. Other caveats are identified in Walmsley *et al.* (2007). There are some additional points that also apply for the maps' interpretation for forestry:

- The maps assume that the genotypes (i.e. provenances) of individual species currently present in the UK are as well adapted to the climate of the future as genotypes from more southerly regions.
- Current species distributions are determined by reproductive capacity and/or dispersal. This may underestimate the climatic and/or geographic range over which they can successfully be grown. Beech is a good example, with its commercially viable range extending

well beyond its 'ecological distribution' of southern England well into Scotland.

- The current (and therefore projected future) distribution may be the result of competition. Forest management, for example single species stands, may enable a species to endure beyond its ecological climate envelope.

5.3.2 Model results

Putting the caveats outlined above aside, projected changes in species envelopes provide a powerful picture of how species' suitability might change. These should not be used as the sole basis on which to base forest plans and the fine detail of the maps should not be explored. However, as presented in Figure 5.4, they provide a broad assessment of those species that should continue to be well matched to the UK's climate towards the end of the century, those that are likely to be towards their range limits and those that are likely to struggle as a result of climate change.

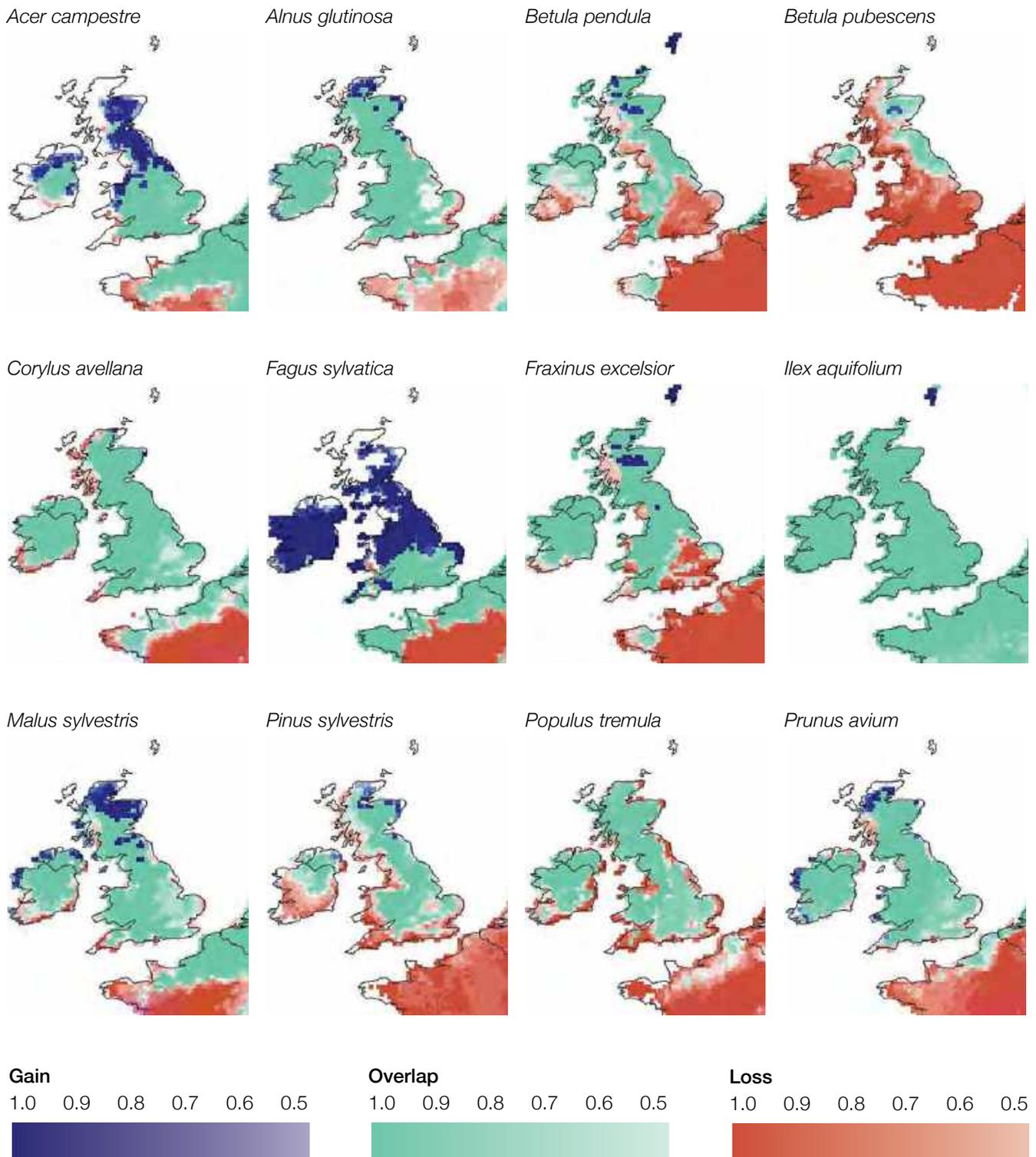
The results presented in Figure 5.4 provide a 'snapshot' of possible future patterns of species distribution at a single timepoint (the 2080s) based on a single global climate model (GCM: HADCM3) and GHG emissions scenario (IPCC SRES A2). However, as demonstrated in Figure 5.5 for *Quercus robur*, one of the species examined in Figure 5.4, a range of different outcomes can be obtained for different timepoints (the 2020s, 2050s and 2080s), utilising the output of different GCMs (PCA and HADCM3) and assuming different GHG emissions scenarios (IPCC SRES A2 and B1). The significant losses in climate space for the species in the 2080s predicted from the HADCM3 GCM output and IPCC SRES A2 GHG emissions scenario (see Figure 5.4) are not apparent if the output from the PCA GCM is used (assuming the IPCC SRES A2 emissions scenario). The predicted loss of climate space is also much reduced when the IPCC SRES B1 GHG emissions scenario is assumed for the HADCM3 GCM. The implications that arise from uncertainty in GHG emissions scenarios and Global Climate Model output must be recognised and considered alongside the uncertainties inherent in the ecological models.

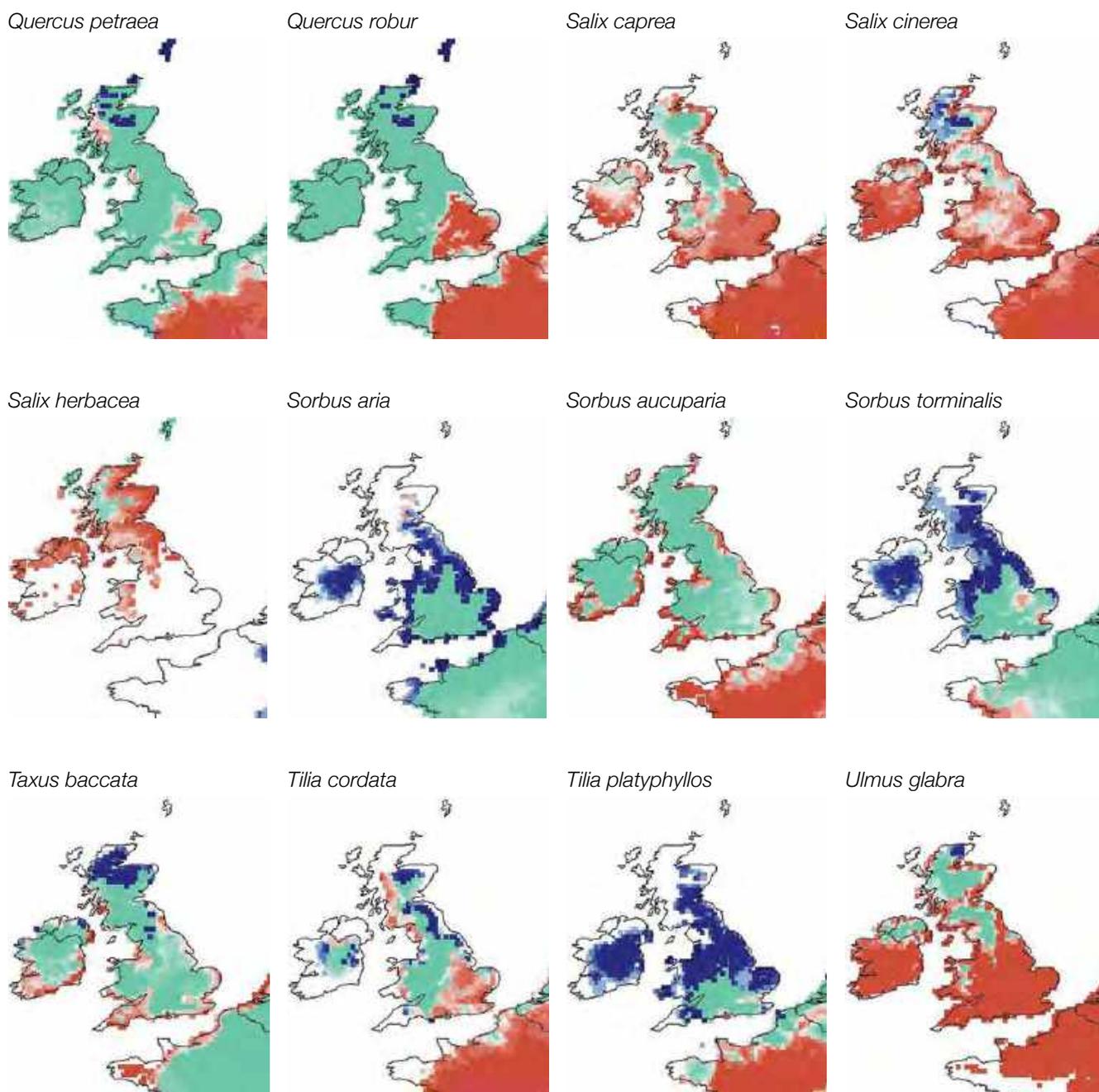
Of the main species grown for timber production, beech shows the greatest benefit from climate change, with its distribution extending in the north, and with minimal contraction of its range in southern England. However, this outlook should be tempered by the well-documented impact of drought on the species (Peterken

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Figure 5.4

Projected changes in climate space for 24 native tree species according to the SPECIES model (Harrison *et al.*, 2001) for the 2080s. Projections assume the IPCC SRES A2 scenario (equivalent to UKCIP02 Medium-High scenario) and are based upon the output from the HADCM3 global climate model. Red depicts loss of climate space, blue gain of climate space and green, the maintenance of climate space. See Berry *et al.* (2007a) for further details.

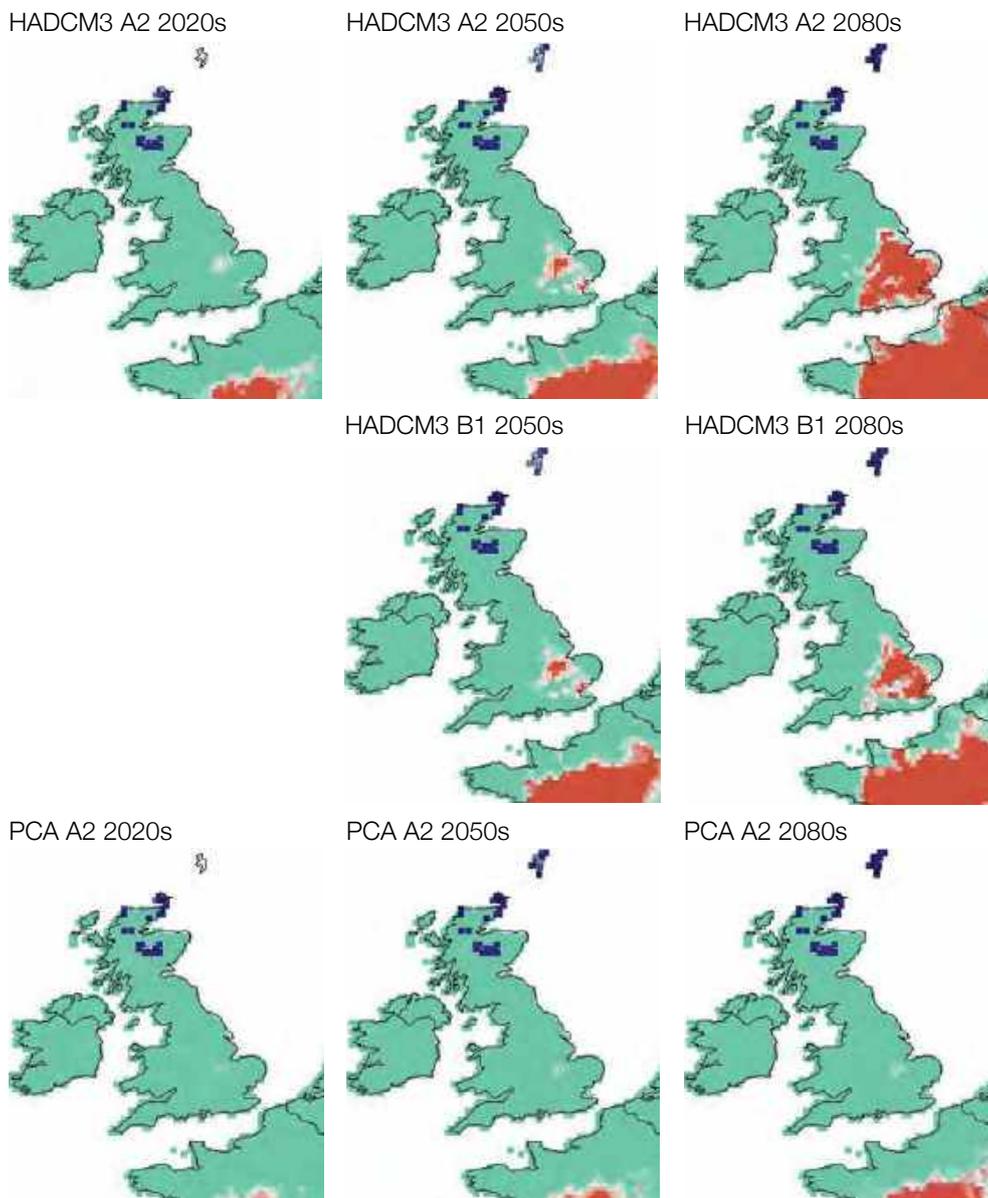




and Mountford, 1996) that has been shown to be linked to soil type (Wilson *et al.*, 2008). It is therefore likely that as a commercially viable species, beech will be limited to good soils in southern and eastern England, but with real promise further north. This northward expansion of its range may have implications for nature conservation policy and management and further consideration of its management outside its 'historical native range' is required, as outlined by Kirby (2009). This future distribution also highlights the point that, in contrast with many sensationalist popular articles, beech is highly

unlikely to disappear from the landscape of southern England. The two native species of oak both show a contraction in their range in south and east England, pedunculate oak (*Quercus robur*) more so than sessile oak (*Q. petraea*). It is noteworthy that the relative changes in *range* of the two species predicted by SPECIES differs from the relative changes in *suitability* projected using ESC, based on their current performance (Table 5.1). The climatic range of ash (*Fraxinus excelsior*) is also projected to move outside parts of southern England, presumably because of limited water availability. However this could

Figure 5.5
Comparison of projected changes of climate space for *Quercus robur* for different timeframes, global climate models (HADCM3 and PCA) and GHG emissions scenarios (IPCC SRES B1 and A2). Key as for Figure 5.4.



be complicated by the European distribution of ash being limited by competition with other species (principally *F. angustifolia*), rather than climate; silver birch (*Betula pendula*) shows a similar but greater decline in range, which is not surprising given its well documented sensitivity to moisture deficit (Peterken and Mountford, 1996). An important point to note is that most of the 'minor' native species of British woodland maintain climate space in these model projections, with the exception of relict boreal species such as dwarf willow (*Salix herbacea*), downy birch (*Betula pubescens*) and wych or Scots elm (*Ulmus glabra*). Apart from beech, the other potential timber species to gain climate space is European lime (*Tilia platyphyllos*),

in contrast with small-leaved lime (*Tilia cordata*) for which climate space contracts in southern England.

An additional consideration for nature conservation policy and the development of adaptation strategies is whether species will be able to disperse and move through the landscape as their climate space moves – assuming that suitable habitat is available. For tree species, this is highly unlikely as the typical change in climate space depicted in Figure 5.4 will require that species move at a rate of 2–5 km per annum, in line with the 'anecdotal rule of thumb' of a one degree temperature difference being equivalent to a 100 km shift in latitude. This rate of

migration is well in excess of the 500–1000 m per year that tree species moved after the last ice age to reach present distributions (Huntley and Birks, 1983) a rate that is now considered an over-estimate (Birks and Willis, 2008). For this reason, Aitken *et al.* (2008) suggest that some tree species, particularly those with fragmented ranges, small populations, low fecundity or suffering decline, should be candidates for facilitated migration.

The overall conclusion from this modelling work is that the majority of native species maintain climate space with the possible exception of in coastal areas where moisture deficits are exacerbated by high wind speeds. The HADCM3 A2 scenario includes relatively severe summer moisture deficits and the projections based on this model are for a number of native species to lose climate space. Climate models with less extreme water deficits than those assumed in the HADCM3 A2 scenario, but which are consistent with UKCP09, indicate that the majority of native species will retain existing climate space in the UK, although these analyses suggest that the edge of the climate envelope for most of those species will move to northern France (data not shown, see: www.branchproject.org.uk). However, most native species in southern England will be at the edge of their climate space by the end of the century and will therefore be challenged. They will not disappear from the landscape, even in southern England where the impacts of climate change are likely to be most severe. The model outputs also confirm the risk to timber production outlined in the ESC analysis.

5.4 Impacts of climate change on woodland habitats and priority species identified in the UK Biodiversity Action Plan

The UK Biodiversity Action Plan (UKBAP) was published in 1994 (Department of the Environment, 1994) as a response to the United Nations Convention on Biological Diversity (United Nations, 1992). It identifies and prioritises actions to protect threatened, and nationally important species and habitats. Of the 400 species listed in the original Plan, around 130 were associated with woodlands, often as their main habitat. The list has now been extended to cover 1150 species and 65 habitats. Of the 65 priority habitats which have UK-wide plans, eight are for native woodlands.

Mitchell *et al.* (2007) assessed the risk of future climate

change to woodland habitats in England, concluding that summer drought rather than temperature represented the greatest risk to broadleaved mixed and yew woodland, particularly in southeast England. However, change in species composition rather than loss of woodland would be the main impact. Rising sea level was unlikely to represent a major threat, with less than 1% of woodland and only 154 ha of ancient and semi-natural woodland falling within the Environment Agency's tidal flood risk map. An earlier evaluation by Hossell *et al.* (2000) rated the vulnerability of this broad habitat as low–medium, although Mitchell *et al.* note that revised climate change scenarios (UKCIP02: Hulme *et al.*, 2002) published in 2002 probably mean that Hossell *et al.* underestimated the risk. However, the most recent climate projections (UKCP09: Murphy *et al.*, 2009) suggest a less extreme reduction in summer rainfall in southeast England, so the earlier evaluation of Hossell *et al.* may be appropriate. This changing risk assessment over time highlights the difficulty in making firm predictions on the future of woodland and, particularly, in developing appropriate adaptation strategies. For the coniferous woodland habitat, Mitchell *et al.* (2007) concluded that although the suitability of different species for different sites would be affected by climate change, the persistence of the habitat would be dependent on management decisions rather than climatic conditions. The potential impacts of climate change during this century on priority woodland habitats in England, Scotland and Wales are summarised below. Further details are provided by Mitchell *et al.* (2007) and Ray (2008a,b).

5.4.1 Lowland mixed deciduous woodland

This priority habitat covers a broad range of species and woodland types on heavier soils. As identified above, the majority of native tree species will persist, although there will clearly be differential effects of climate change resulting in changing competitive advantage and a shift in species composition. In particular, sycamore and beech may tend to increase at the expense of oak and ash (Ray, 2008b). Bramble, nettle and other species comprising 'rank' vegetation may become more dominant on the heavier, rich soils that are characteristic of the habitat, at the expense of forbs and grasses. Although it is impossible to be precise over its future, lowland mixed deciduous woodland is likely to continue as a functional system across the UK, even if the tree species and ground flora communities within them may change. The greatest threat to the habitat, in common with others, probably comes from the potential impacts of pest and disease outbreaks, particularly where species diversity is limited.

5.4.2 Lowland beech and yew woodland

The impacts of drought on beech are well documented (e.g. Peterken and Mountford, 1996) and a number of studies (e.g. Harrison *et al.*, 2001; Berry *et al.*, 2002a b; Broadmeadow and Ray, 2005; Berry and Paterson, 2009) have suggested that the species will struggle for climate space, or as a commercially viable species. However, the natural distribution of the species coupled to more recent modelling results suggest that the species will persist across southern England. Soil type will be a key determinant of which sites it continues to thrive on (Wilson *et al.*, 2008), as will other site conditions including aspect and topography. Roberts and Rosier (2006) highlight the ability of beech to access water at greater depth on some chalk soils as a result of the nature of the chalk matrix. Stribley and Ashmore (2002) also note the interaction between ozone levels and climate on the condition of beech. Harrison *et al.* (2001) and more recent work (Berry *et al.*, 2007b) indicates that yew is unlikely to lose climate space. Although the individual species are likely to persist as climate change progresses, beech is likely to be challenged on some sites, particularly those with free-draining thin soils. This is likely to allow ash and, in time, oak to colonise and the nature of the habitat is likely to change, and will probably also extend beyond its native range (Kirby, 2009). There is already an increased acceptance of the species beyond its native range (Wesche, 2003; Kirby, 2009), although continued expansion has the potential to impact negatively on other priority habitats, particularly upland oakwoods, if it is not appropriately managed on those sites.

5.4.3 Lowland wood pasture and parkland

The persistence of the habitat will be determined by past and future management decisions, including species choice, and this is reflected in Hossell *et al.*'s assessment of vulnerability as 'low', as compared to medium for other priority habitats in England. Broadmeadow (2000) also noted that the impacts of climate change are likely to become evident first on young trees and isolated trees because of their higher water requirement.

5.4.4 Upland mixed ashwoods

There is little evidence to suggest that upland ashwoods will be negatively impacted by climate change with the main species (ash, hazel, oak, birch) highly likely to persist in the wetter north and west where the habitat occurs. However, Mitchell *et al.* (2007) note that species

composition may change, suggesting that the northward movement of small-leaved lime may be a possibility. Ray (2008a) highlights the particular importance of the habitat in Wales (25% of the semi-natural woodland area) and concludes that the habitat, in both Wales and Scotland, may show slower changes in species composition than other habitats because of the ability of ash to regenerate in dense shade.

5.4.5 Upland oakwood

The distribution of tree species in upland oakwoods is unlikely to be affected by projected climate change this century. However, the habitat, which is restricted to high rainfall areas, is highly valued for its fern, bryophyte and lichen flora which are sensitive to changes in rainfall and humidity. ESC NVC analysis suggests that W11 would replace W10 as the most suitable of the oak dominated woodland types in the less wet upland oakwood range (Broadmeadow and Ray, 2005). Of particular note is the very restricted range of W11 under the 2050s high emissions scenario, particularly in Wales, confirming concerns over the possible impact on the community composition of upland oakwoods. Ray (2008a) also notes the possibility that more frequent disturbance events coupled to the changing climate may allow other species such as birch, hazel and rowan to colonise Atlantic oakwoods. Of particular concern in this context is the potential for beech to colonise upland oakwoods with the deeper shade cast by the species impacting negatively on the ground and epiphytic flora.

5.4.6 Wet woodland

Mitchell *et al.* (2007) suggest that the persistence of wet woodland will depend on local factors rather than regional climate change. Ray (2008a,b) highlighted the role that wet and riparian woodland might play in flood risk management and the maintenance of freshwater temperature to protect fisheries. The extent of these communities may therefore be increased as a management response to climate change. In southern England, the increasing frequency and severity of summer drought represents a threat to these communities, but such threats may be countered by increased winter rainfall maintaining ground water supplies through the early part of the summer. If individual sites do become drier, ash may colonise at the expense of alder (*Alnus glutinosa*), which is currently the dominant species of many sites. Ray (2008a,b) suggests that water supply is unlikely to limit the habitat in Scotland and Wales, and that wet woodland might expand into upland mires and flushes

as a result of changes in climate and land management. The greater fluctuations in water levels that would be expected to result from climate change could enhance the risk of significant dieback of alder through *Phytophthora* infection (Lonsdale and Gibbs, 2002).

5.4.7 Native pinewoods

The persistence of the principal tree species, Scots pine, is unlikely to be affected by climate change. However, the composition of the ground flora is likely to be affected with plant communities associated with the drier sub-communities of the west and central Highlands favoured (Ray, 2008a). There may also be colonisation by other tree and shrub species (e.g. oak, birch, rowan) and the appearance of other flora not generally associated with native pinewoods. Disturbance through fire may represent an enhanced risk, particularly in woodlands used extensively for recreation. Where grazing levels permit, there may be a colonisation of scrub (juniper and montane willows), and ultimately pine, above the current tree-line. This is consistent with the average rise of 80 m (maximum 200 m) observed in the tree-line across sites in Sweden over the past century (Kullman and Oberg, 2009) and more widely (Grace *et al.*, 2002).

5.4.8 Upland birchwoods

The persistence of birch in upland birchwood habitats is unlikely to be challenged by climate change as currently projected. However, Ray (2008a) noted the recent planting and limited tree species diversity that might render these woodlands vulnerable to pest/disease impacts or the effects of extreme climatic events. On drier sites, silver birch may become more competitive at the expense of downy birch.

5.4.9 Priority species

Of the 1150 priority species identified in the UK Biodiversity Plan, 243 have been classified as woodland species, although not confined solely to woodlands. The impacts of projected climate change (using the HADCM3 model and IPCC A2 GHG emissions scenario) on the distribution of a number of woodland priority species (for England) have been modelled using the SPECIES model (see 5.3 above). The same caveats outlined in 5.2.1 apply to the changes in distribution given in Table 5.2, as well as the fact that many of these species had poor European distribution data for model training and thus the projections are subject to a high level of uncertainty.

From the limited range of priority species modelled, higher plants and lichens show a large loss of climate space in contrast to most invertebrate species which show a large expansion in climate range across England, Scotland and Wales. The terrestrial vertebrates show a range of responses, varying from the current very limited distribution and simulated climate space projected to expand to cover the majority of Great Britain in the case of the barbastelle bat, to an almost complete loss of climate space for the black grouse. Very few of the species show a small change to their climate space (and modelled potential distribution) as a result of climate change.

If climate change progresses as projections suggest and if these changes affect priority species distributions as models of climate space indicate, nature conservation policy will need to be revisited as a matter of urgency. Many of the species given as examples in Table 5.2 will no longer be at the edge of their climatic range and pressures on existing populations would be expected to decline. In contrast, other priority species (e.g. the lichen *Biatordium monasteriense*) will lose all 'climate space' as a result of climate change and efforts to maintain the species on all but a handful of sites are likely to fail. The fact that climate space (and inferred distribution) in the UK is likely to be unaffected for very few species and that the scale of these impacts will vary with time, suggests that a habitat-focussed approach to nature conservation will better address climate change than a species-based approach.

5.5 Impact of changing frequency of wind storms

There is high uncertainty in the wind speed element of future climate change scenarios, with different global climate models showing disparity in both the magnitude and direction of change (see Chapter 2). The methodology behind the recently published UKCP09 projections does not allow projections of changes in wind speed (Murphy *et al.*, 2009). However, the impact on windthrow risk of the changes in mean climate as depicted in the UKCIP02 climate change scenarios (Hulme *et al.*, 2002) have been assessed (Ray, pers. comm.). The projected changes in winter wind speed (up to ~10% increase) resulted in minimal change to the DAMS (Detailed Aspect Method of Scoring) wind hazard classification system developed by the Forestry Commission for UK conditions. However, Quine and Gardiner (2002) outline the importance of the distribution of wind speeds, concluding that a small change in mean wind speed could result in a very large increase

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Table 5.2

Projected change in climate space for English woodland priority species as modelled by the SPECIES model (Berry *et al.*, 2007a) using the HADCM3 climate model and the IPCC SRES A2 GHG emissions scenario.

Species	Common name	Projected change in distribution		
		England	Scotland	Wales
Vascular plants				
<i>Arabis glabra</i>	Tower mustard	--	(L) ---	(A) +
Lichens, fungi and bryophytes				
<i>Schismatomma graphidioides</i>	Lichen	--	+	+
<i>Biatoridium monasteriense</i>	Lichen	---	---	---
<i>Catapyrenium psoromoides</i>	Tree Catapyrenium (lichen)	---		
Terrestrial invertebrates				
<i>Andrena ferox</i>	Mining bee	-	-	
<i>Lucanus cervus</i>	Stag beetle	+	+++	+++
<i>Limoniscus violaceus</i>	Violet click beetle	+	++	+++
<i>Argynnis adippe</i>	High brown fritillary (butterfly)	++	(A) +++	(A) +++
<i>Boloria euphrosyne</i>	Pearl-bordered fritillary (butterfly)	-	--	
<i>Erynnis tages</i>	Dingy skipper (butterfly)	+	+	++
<i>Leptidea sinapis</i>	Wood white (butterfly)	+++	+++	+++
<i>Melitaea athalia</i>	Heath fritillary (butterfly)	+++		+
Terrestrial vertebrates				
<i>Triturus cristatus</i>	Great crested newt (amphibian)	---	--	-
<i>Caprimulgus europaeus</i>	Nightjar (bird)	(L) +++	(L) +	(L) ++
<i>Emberiza schoeniclus</i>	Reed bunting (bird)	---		--
<i>Lullula arborea</i>	Wood lark (bird)	+++	+	+++
<i>Muscicapa striata</i>	Spotted flycatcher (bird)	-		--
<i>Passer montanus</i>	Tree sparrow (bird)	+	-	+
<i>Pyrrhula pyrrhula</i>	Bullfinch (bird)	---		-
<i>Streptopelia turtur</i>	Turtle dove (bird)	++	+	++
<i>Tetrao tetrix</i>	Black grouse (bird)	---	---	---
<i>Turdus philomelos</i>	Song thrush (bird)	---		-
<i>Barbastella barbastellus</i>	Barbastelle bat (mammal)	++	(A) ++	(A) +++
<i>Lepus europaeus</i>	Brown hare (mammal)	--	+--	
<i>Muscardinus avellanarius</i>	Dormouse (mammal)	--+		-
<i>Myotis bechsteinii</i>	Bechstein's bat (mammal)	-++	+	+
<i>Rhinolophus hipposideros</i>	Lesser horseshoe bat (mammal)	+++	+++	++
<i>Sciurus vulgaris</i>	Red squirrel (mammal)	--		+

Key: (A) currently absent; (L) currently very localised distribution; --/-- --- contraction in climate space; +/+ +++ expansion in climate space.

in wind risk if the frequency or speed of extreme gusts increases. Changes in wind direction that were not included in the UKCIP02 scenarios could also result in changed wind risk; particularly if the storm track across the UK became more southerly, as suggested by Hulme *et al.* (2002). Improved data availability including global climate model (GCM) pressure field outputs may allow the risk associated with a change in wind direction to be further evaluated.

Climate change may also lead to an indirect effect on wind risk, even if the wind climate itself remains unchanged. The projected increase in winter rainfall is likely to result in more frequent and long-lasting periods of water-logging, which may reduce tree stability. The increase in leaf area that has been reported in many experiments indicate that increased carbon dioxide levels could also enhance the risk of wind damage, as would an extension to the duration of the leafed period for deciduous trees (Broadmeadow and Ray, 2005). Finally, changes to root:shoot allocation, structural properties of wood or tree form/taper could also affect wind risk. It could therefore be argued that amending the ForestGales decision support system to reflect the potential indirect effects of climate change is more of a priority than further evaluation of highly uncertain changes in wind speed. However, if the output from different GCMs converge to give greater confidence in wind speed and storm track projections, a re-evaluation of the impacts of changes in windstorms would be a high priority, given the potentially catastrophic impacts exemplified by recent major storms in the UK and Europe. When new information on a wind speed projections becomes available, it will be important to evaluate vulnerability of woodlands and risks to infrastructure from falling trees. In both cases, contingency planning should be examined.

5.6 Impacts of pests and diseases under future climate change scenarios

Predicting changes to the impact of specific insect pests and tree diseases on woodlands is difficult because of the fine balance between pest/pathogen, host tree and natural enemies. However, it is possible to make two generalisations – stressed trees are more susceptible to insect pests and diseases, and the majority of insect pests that currently affect UK forestry are likely to benefit from climate change as a result of increased activity and reduced winter mortality (Straw, 1995). The impact of facultative pathogens may worsen, while some insect pests that are

present at low levels, or currently not considered important, may become more prevalent. Examples of the latter include defoliating moths and bark beetles. In addition, the ‘effective’ range of existing pests or pathogens may change, including a northwards expansion of those with a southern distribution and the likely appearance of some from continental Europe. Firm predictions cannot be made, although expert judgement of forest pathologists and entomologists allows some assessment to be made of changes in the prevalence of certain diseases and insect pests, based on their current distribution and associated climatic conditions, known biology and epidemiology. Considerable caution should be exercised in extrapolating this analysis to a future climate. For some pests and diseases, likely trends cannot be predicted even on the basis of expert judgement; in this category, and of particular concern, is *Phytophthora ramorum*, the agent responsible for sudden oak death (see 4.1.6, Chapter 4). The higher level of uncertainty associated with the biology of fungi compared to insect pests is reflected in the less specific predictions of future trends in the incidence of fungal diseases and disorders.

5.6.1 Forest insect pests

Climate change will influence the distribution, abundance and performance of forest insect pests, with their impact dependent, in part, on their feeding habits, life cycle characteristics and the relationship between individual climatic variables and population dynamics (Masters *et al.*, 1998). As a general guide, rising temperature will have its greatest impact on the development rate of insect populations, leading to faster insect development, a larger number of generations (voltinism) in a year, as well as range extensions. Insect groups most likely to be affected are multi-voltine aphids, semi-voltine bark beetles, sawflies, weevils and wood-boring lepidoptera. Apart from being linked to drought stress, thereby making trees more vulnerable to the effects of insect pest outbreaks, changes in precipitation patterns are likely to be associated with changes in resin flow of conifers and the nutritional quality of foliage (Wainhouse, 2005). Insect groups most likely to be affected include bark beetles, aphids and longhorn and buprestid beetles. More general forest damage resulting from windstorms or forest fires is also likely to promote outbreaks of bark and ambrosia beetles (Långström, 1984; Fernandez, 2006). A more detailed risk assessment of possible interactions between climate change and insect pest outbreaks is given in Table 5.3 (after Wainhouse, 2008), with a commentary on individual insect groups given below, based on the review of Wainhouse (2008).

Table 5.3

Risk of increased damage by UK forest insect pests as a result of projected climate change. Risk is rated as low, moderate, high or very high, based on an assessment of life history characteristics, population dynamic, historical patterns of damage and likely changes in planting. Where species have a wide host range, risk is assessed for the main host only.

Insect pest	Host	Risk	Main risk factors	Geographical area
Bark beetles, weevils and related species				
Spruce bark beetle (<i>Dendroctonus micans</i>)	Spruce	High	Range extension, reduced generation time, drought stress of host trees	Northern Britain
	Pine	Low		
Pine weevil beetle (<i>Hylobius abietis</i>)	Spruce	High	Reduced generation time, forest management	Large forest areas managed by clearfell and re-planting throughout the UK
	Pine	High		
Pine shoot beetle (<i>Tomicus piniperda</i>)	Pine	High	Windblow, sister broods, heat stress through drought or defoliation	Southern Britain
Larch bark beetle (<i>Ips cembrae</i>)	Larch	Moderate	Windblow, sister broods, heat stress, range extension, defoliation	Northwest Scotland, northern England
Oak pinhole borer (<i>Platypus cylindrus</i>)	Oak	High	Link to oak decline, reduced generation time, increased importance of oak	Southern Britain
Oak buprestid (<i>Agrylus pannonicus</i>)	Oak	High	Link to oak decline, reduced generation time, increased importance of oak	Southern Britain
Aphids and scale insects				
Green spruce aphid (<i>Elatobium abietinum</i>)	Spruce	Very high	Reduced generation time, increased winter survival, drought stress of host trees	Most spruce growing areas
Beech scale (<i>Cryptococcus fagisuga</i>)	Beech	Low	Increase in young plantations	
Defoliators				
Pine looper moth (<i>Bupalus piniaria</i>)	Pine	Moderate	Low rainfall sites	Most pine growing areas
Pine beauty moth (<i>Panolis flammea</i>)	Pine	Low	Lodgepole pine	Northern Britain
Winter moth (<i>Operophtera brumata</i>)	Spruce	Moderate	Phenological synchrony on oak. Host range extension	
	Oak	Moderate		
Oak processionary moth (<i>Thaumetopoea processionea</i>)	Oak	High	Range extension	Southern Britain
Gypsy moth (<i>Lymantria dispar</i>)	Oak	Low	Range extension	Southern Britain
Larch budmoth (<i>Zeiraphera diniana</i>)	Spruce	Low	Phenological synchrony with host tree	
	Pine	Low		
	Larch	Low		
Lesser pine sawfly (<i>Neodiprion sertifer</i>)	Pine	Low	Possible increased risk on dry nutrient-poor sites	
European spruce sawfly (<i>Gilpinia hercyniae</i>)	Spruce	Moderate	Reduced generation time, range extension	
Web-spinning larch sawfly (<i>Cephalcia lariciphila</i>)	Larch	Low	Range extension, overlap with <i>Ips cembrae</i>	Northwest Scotland, northern England

Aphids, scale insects and adelgids

Insects in these groups can reduce growth and lead to cosmetic damage to high value trees through gall production (Bevan, 1987). They typically have a close and sometimes highly specialised relationship with the host tree which can, in turn, significantly influence the timing of the life cycle and population dynamics.

Many are relatively small, sedentary, insects that are often exposed on the surface of the plant. As a result, they are vulnerable to the effects of heavy rainfall, unseasonal cold temperatures and to natural predators. Projected climate change would therefore be expected to have both direct and indirect impacts on populations (Evans *et al.*, 2002; Straw, 1995), resulting in a general increase in the damage caused by aphids and related insect pests. More specifically:

- Higher temperatures will increase the reproductive rate and those species that have multiple generations or can remain active throughout the winter will benefit the most;
- Drought stress of host trees through changes in rainfall patterns and increases in evapotranspiration will favour many species of insect.

Bark beetles, weevils and related species

Bark beetles are among the most important forest insect pests because they attack mature trees and can introduce harmful pathogens (e.g. Redfern *et al.*, 1987; Gibbs and Inman, 1991). The abundance of species and their population dynamics is primarily determined by environmental influences on the availability of suitable breeding substrate. As a consequence, climate change would generally be expected to lead to an increase in the level of damage caused by bark beetles and related insects through:

- more widespread and frequent forest fires;
- increased summer drought stress leading to greater tree mortality and a larger proportion of stressed or compromised living trees that would be vulnerable to attack;
- changes to the wind climate, although projections are highly uncertain; if there is more frequent and widespread windblow this will increase the prevalence of bark beetles.

Defoliators

Defoliators are a highly diverse group of insects for which the population dynamics and abundance are driven by complex interactions between climate, site factors, host tree suitability and predators. Most have a single generation a year, but sawflies have the potential for two or more a year (Knerer, 1993). The most damaging defoliators have 'eruptive' population dynamics. Broadleaved tree species are generally affected early in the growing season and are able to re-flush. Significant damage and reductions in productivity generally only occur as a result of repeated outbreaks in successive years (*Tortrix* and winter moth, *Operophtera brumata*; Gradwell, 1974). Conifers are particularly vulnerable to defoliators that can feed on more than one age-class of needle (Watt and Leather, 1988) and those that feed late in the growing season and affect flushing the following year. Again, as a general rule, projected climate change is likely to lead to an increase in the prevalence and severity of defoliation by insect pests through:

- an increase in the number of generations per year for sawflies and other multi-voltine insects;
- drought-stress increasing susceptibility to defoliating pests;
- range extensions of some species, particularly those with a southerly distribution.

Changes in management

Changes in forest management practice, in part as a direct response to the impacts of climate change – or to its mitigation – may affect the fine balance between host trees and insect pests. Such changes include increases (or decreases) in rotation length, altered stocking densities and different age structures across large forests. Changes to stand structure through conversion to continuous cover systems of management may also have an impact. Finally, increased rates of woodland planting, larger areas of young woodlands and new species or provenances may also affect the frequency and severity of insect pest outbreaks.

The threat of insect pests that have not yet been introduced to the UK is poorly understood, but the risk is potentially very significant. Examples of devastating introductions in other parts of the world, for example the emerald ash borer in North America (see: www.emeraldashborer.info), highlight the potential impacts of such interactions. Climate change will make the climate

of the UK suitable for an increasing range of forest insect pests, and there is an urgent need to review the global literature on these potential threats to enable appropriate surveillance and contingency plans for forest management to be drawn up.

5.6.2 Tree pathogens

Although there has been some analysis of the impact of climate change on insect populations and the associated damage in forests, in general there has much less consideration of the effect on pathogens. Typically, the key factors in the development of plant disease epidemics are temperature and moisture, and it is well known that rainfall patterns affect the frequency and severity of certain diseases from year to year. Inevitably therefore, climate change will alter the activity of tree pathogens, both through direct and indirect effects. Changes in temperature, precipitation, soil moisture and relative humidity will all have a direct influence on the infection success of pathogens. Indirect impacts will also result as trees suffer episodes of climatic extremes which cause water stress or undermine resistance mechanisms, thus making them more susceptible to latent or opportunist pathogens. If there are more frequent storm events this will increase levels of physical injury on trees, and thereby create wounds that allow pathogen entry. A summary of the risks posed by major pathogens in the context of a changing climate is given in Table 5.4, together with an assessment of current and future risks associated with bacterial pathogens in Table 5.5.

Latent pathogens

As a particular group of organisms that include significant tree pathogens, latent pathogens or 'endophytes' have long been predicted to pose a heightened threat under conditions of climate change. These organisms are highly specialised, able to infect trees and remain asymptomatic for years until environmental factors – often drought stress – trigger the development of disease. Examples include *Biscogniauxia* species which cause damaging strip cankers on beech and oak; these are already observed to be more common on beech in areas of low rainfall and high temperatures (Hendry *et al.*, 1998). Sooty bark disease of sycamore (*Cryptostroma corticale*) is another temperature dependent latent pathogen which only becomes active after hot dry summers (Dickenson and Wheeler, 1981), particularly when the mean monthly temperature of more than one summer month equals or exceeds 23°C (Young, 1978). However, latent pathogens are not just limited to broadleaved trees. The disease

known as Diplodia blight (*Diplodia pinea*) is one of the most common and widely distributed pathogens of conifers worldwide (Burgess *et al.*, 2004). Although considered a southern fungus, its impact becomes visible in the north during drought periods as it is released from its quiescent stage in hosts under water stress (Stanosz *et al.*, 2001). Severe drought in 2003 encouraged *D. pinea* to become epidemic in central Europe, and reports of damage caused by this pathogen have become much more frequent in Britain over the past 10 years (Brown and MacAskill, 2005).

Foliar pathogens

The most immediate and visible changes in tree health in response to climate change could result from increased activity of foliar pathogens. In fungi, sporulation and infection are strongly linked to changes in temperature and precipitation (Peterson, 1967) and, by their nature, foliar pathogens are directly exposed to fluctuations in air temperatures and moisture. Consequently, these agents which often require free moisture for host infection, sporulation and spore dissemination are likely to become more damaging, particularly in western parts of the UK where increased spring rainfall is indicated in climate projections.

Typical examples expected to worsen on broadleaved tree species include *Marssonina* and *Melampsora* species, while some of the warm temperature *Melampsora* species which are currently damaging in central Europe are predicted to become problematic in southern Britain as they extend their range (Lonsdale and Gibbs, 2002). Warmer weather will probably also favour attacks of powdery mildew on foliage (e.g. oak mildew, *Erysiphe alphitoides*), making outbreaks more intense and longer lasting. Moreover, if heavy infestations occur over successive years the vigour and productivity of affected trees is likely to be reduced. Conifer needles and shoots are also susceptible to various foliar pathogens. The most striking example is *Dothistroma* needle blight, which has escalated in incidence markedly over the past decade in Britain (Brown and Webber, 2008) and one of the drivers of this increase is thought to be climate change (Woods *et al.*, 2005; Archibald and Brown, 2007). The shift towards an increased frequency of prolonged periods of rainfall in eastern England combined with temperatures greater than 18–20°C in spring and early summer since the mid- to late-1990s appears to have favoured the spread and intensification of *D. septosporum* and this could be replicated elsewhere.

Table 5.4
Analysis of risk posed by major pathogens present in Britain in relation to climate change (1–5 = low to high risk).

Pathogen	Disease symptoms	Affected genera/species	Likelihood of increased activity	Potential impact	Level of risk
Foliar pathogens					
<i>Marssonina</i> spp.	Leaf spots, shoot blights, branch and stem cankers on young trees	Poplar, birch and willow	Low–moderate	Moderate ³	2
<i>Erysiphe</i> spp.	Mildew causing leaf and shoot blight	Oak	Moderate–high	Moderate	3
<i>Venturia</i> spp.	Death of leaves and shoots	Poplar and willow	Low	Moderate	2
<i>Melampsora</i> spp.	Death of leaves, premature leaf fall	Poplar and willow	High	Moderate–high ⁴	4
<i>Dothistroma septosporum</i>	Needle death, premature defoliation and tree mortality	Pines, especially Corsican, lodgepole and now Scots pine	High	High	5
<i>Diplodea pinea</i>	Shoot blight, top dieback, and cankers on stem and branches	Pines, particularly black pine	Moderate–high ¹	Moderate	4
<i>Phytophthora</i> spp., e.g. <i>P. ramorum</i> , <i>P. kernoviae</i>	Leaf and shoot blights and stem cankers	Broadleaf species	Moderate ²	High	4
Root rots					
<i>Heterobasidion annosum</i>	Decay and mortality (particularly of young trees)	All conifers	Moderate–high	High	4
<i>Armillaria</i> spp.	Decay, tree decline and mortality	Wide range of conifers and broadleaves	Moderate–high	Moderate ⁵	4
<i>Collybia fusipes</i>	Decay, tree decline and mortality	Predominantly oak	Moderate	Low–moderate ⁶	3
<i>Phytophthora</i> spp., e.g. <i>P. cinnamomi</i> , <i>P. alni</i> , <i>P. cambivora</i>	Root death, bleeding canker and tree mortality	Wide range of broadleaves	High	High ⁷	5
Stem cankers					
Bacterial diseases	Bleeding stem canker, shoot tip die back, gummosis	Broadleaf species, e.g. alder, ash, cherry, horse chestnut and oak	High	Moderate–high	5
Stress related or latent pathogens					
<i>Biscogniauxia</i> spp.	Strip cankers and dieback	Beech and oak	Moderate–high	High ⁸	3
<i>Botryosphaeria stevensii</i>	Cankers and dieback	Ash and oak	Moderate	Low ¹	2
<i>Cryptostroma corticale</i>	Bark death and dieback	Sycamore	High	Low–moderate	2
<i>Nectria coccinea</i>	Bark death	Beech	Moderate	Moderate ⁹	3
<i>Phomopsis</i> spp.	Bark cracking and stem lesions	Spruce and larch	Moderate	Low	2
Other established diseases					
<i>Phacidium coniferarum</i>	Bark killing and cankers	Conifers	Low	Low	1
<i>Ophiostoma novo-ulmi</i>	Vascular wilt (Dutch elm disease)	Elm	Moderate	High	2
<i>Ophiostoma</i> and <i>Ceratocystis</i>	Bluestain of wood and bark death	Pines and other conifers	High	Moderate–high ¹⁰	4

¹ Impact could be especially high in nurseries, but also recognised as an endophyte which can be very damaging to stressed trees. ² Some of the aerial *Phytophthoras* infect best at moderate temperatures (18–22°C) with high humidity, but can persist over hot summers via resting spores. ³ Moderate but localised impact, dependent on species involved.

⁴ Likely to have a high impact on clonal polar/willow biomass crops but this will be localised. ⁵ Main impact may still be on ornamental rather than commercial plantation/woodland species, although increased incidence of *Armillaria* attack on commercial forestry species is being recorded in the Disease Diagnostic and Advisory Service database. ⁶ Low to moderate impact reflects the long time-scales before damage becomes apparent. ⁷ High impact anticipated because of root death interacting with drought. ⁸ Common species affected although the impact may be high but localised. ⁹ Losses likely to be increased by stems snapping in severe wind and rain storms. ¹⁰ High temperatures likely to favour not only the fungi but also the insect vectors.

Table 5.5

Analysis of the risk posed by bacterial diseases to British forests in the context of climate change.

Pathogen species	Host	Disease type	Known in UK	Risk of entry to Britain	Potential risk (losses) to British trees
<i>Brenneria nigrifluens</i> (syn. <i>Erwinia nigrifluens</i>)	<i>Juglans</i> spp. (walnut)	Shallow bark canker; affects stems and scaffold branches	Not in Britain but present in Europe	Moderate	Low
<i>Brenneria quercina</i> (syn. <i>Erwinia quercina</i>)	<i>Quercus</i> spp.	Drippy nut disease affecting acorns and twigs	Not in Britain or Europe	Moderate–high (dependent on host range)	Cross-pathogenicity is unknown but if cross-pathogenic then high risk
<i>Brenneria rubifaciens</i> (syn. <i>Erwinia rubifaciens</i>)	<i>Juglans regia</i>	Deep bark disease forming bark splits and bleeding lesions on stems	Not in Britain or Europe	Moderate	Low
<i>Erwinia amylovora</i>	Many <i>Rosaceae</i> including <i>Prunus</i> , <i>Crataegus</i> and <i>Sorbus</i> spp.	Fire blight, attacks blossoms, leaves, shoots and stems causing wilt, cankers and tree death in very susceptible species, e.g. <i>Sorbus</i>	Present and widespread in Britain and continental Europe	N/A	Disease incidence is expected to increase and losses would become moderate–high
<i>Erwinia salicis</i>	<i>Salix</i> spp.	Watermark disease. This disease is important on the cricket bat willow <i>Salix alba</i> var. <i>coerulea</i> but also affects other <i>Salix</i> spp.	Present and widespread in Britain and The Netherlands	N/A	Disease incidence is expected to increase but control measures should keep losses in check
<i>Pseudomonas avellanae</i>	<i>Corylus avellana</i>	Cankers and dieback		N/A	
<i>Pseudomonas savastanoi</i>	<i>Fraxinus excelsior</i>	Galls and cankers on stems	Present and widespread in Britain but at a low incidence	N/A	Although disease incidence may increase it is not expected to cause major losses
<i>Pseudomonas syringae</i> pv. <i>aesculi</i>	<i>Aesculus</i> spp.	Bleeding stem canker	Present and widespread in Britain and continental Europe, especially central Europe	N/A	Disease incidence is expected to increase and losses, which are already moderately high, will increase
<i>Pseudomonas syringae</i> (including a range of pathovars such as pv. <i>morsprunorum</i> and pv. <i>syringae</i>)	Broad host range including: <i>Acer</i> , <i>Alnus</i> , <i>Cornus</i> , <i>Fraxinus</i> , <i>Pinus</i> , <i>Populus</i> , <i>Prunus</i> , <i>Quercus</i> , <i>Salix</i> and <i>Tilia</i>	Bacterial blight (kills leaves and shoots), and bacterial canker (stems) and gummosis of fruit, twigs and stems	Present and widespread in Britain, especially damaging on wild cherry (<i>Prunus avium</i>)	N/A	Disease incidence is expected to increase but losses could be controlled with appropriate management practices
<i>Xanthomonas populi</i>	<i>Populus</i> spp.	Bacterial canker	Present in Britain and continental Europe	N/A	Disease incidence is expected to increase but losses could be controlled by planting resistant cultivars

Information sourced from Phillips and Burdekin, 1982; Sinclair and Lyon, 2005.

Root pathogens

In contrast to foliar pathogens, the interaction between root rot pathogens and climate change may be much more cryptic with trees on drought-affected sites liable to show an increased predisposition to infection by root attacking pathogens such as *Heterobasidion*, *Armillaria* and *Collybia*. The most important root rot pathogen of conifer forestry, *Heterobasidion annosum*, has become increasingly common and damaging on drier sites and the risk of infection is considered to be greatest on well drained mineral soils (Redfern *et al.*, 2001; Pratt, 2003). A changing climate is likely to favour it even more. Another root rotter, *Armillaria*, affects both conifer and broadleaved tree species and is ubiquitous throughout the British Isles. Although some *Armillaria* species are only weakly pathogenic, they are opportunists and attack and kill trees already weakened by other biotic agents or abiotic factors such as drought (Gregory and Redfern, 1998). Consequently their potential to cause damage is expected to increase as trees suffer more frequent episodes of elevated temperatures and drought stress (Desprez-Loustau *et al.*, 2006). Long term (chronic) declines of mature trees such as beech, oak and ash (oak decline is perhaps the best known) are often another visible sign of root rot pathogen activity aggravated by climatic extremes (Auclair *et al.*, 1992). As some *Armillaria* species as well as *H. annosum* have an optimum temperature for growth of around 25°C, increased ambient temperatures could well enhance the process of infection and spread. Moreover, the ability of these root rot fungi to persist on infected sites for decades and increase over successive rotations increases their potential for damage in the future, both through group killing in young restock sites as well as in older plantations and woodlands.

Phytophthoras

Climate models suggest that the impact of *Phytophthora* species is likely to be significantly enhanced under future climate scenarios. Many *Phytophthora* pathogens already have the capacity to be fast acting aggressive pathogens, often with a wide host range, and this behaviour makes them a formidable threat under conditions of climate change. *Phytophthoras* are a mainly introduced group of pathogens and around 10 species are now widespread in Britain with the potential to be highly damaging to tree species (Brasier, 1999; Jung *et al.*, 2009). Some attack aerial plant parts with activity favoured by mild moist springs, but the majority infect and kill the roots of susceptible tree species. The latter require moist soil

conditions, even periods of flooding, for infection and spread. The damage they cause tends to be most visible in the summer, especially if water availability is limited. A build up of soil-inhabiting *Phytophthoras* can result in the death of fine feeder roots, even root and stem girdling, so trees may die suddenly or show signs of marked decline. In addition, many species of *Phytophthora* can over-winter in soil providing winters are mild but also persist in dry soils for decades in the form of resistant spores, becoming active again under more favourable conditions

Bacterial pathogens

Until recently, the number of bacterial pathogens known to be damaging to woodland and forest trees in Britain was considered to be relatively modest and the most significant pathogens mainly affected fruit and ornamental trees. However, over the past 5–8 years, bacterial diseases on trees appear to be more common. For example, both horse chestnut and native oak species now suffer from bacteria-related disorders (Webber *et al.*, 2008); symptoms on both include stem bleeding, and mortality is by no means uncommon. The extent to which these disorders have been exacerbated by changing weather regimes is uncertain. However, if more extreme rainfall and wind storms occur, there may be increased opportunities for dispersal and infection (Boland *et al.*, 2004).

Implications for woodlands in the UK

For aggressive pathogens such as Dutch elm disease, climatic effects on tree physiology are likely to be small compared with the importance of the susceptibility of some tree genera and species which is under genetic control. For other pathogens, climate change is likely to play a major role in defining levels of damage. With foliar pathogens such as *Dothistroma septosporum* and some aerial *Phytophthoras*, the impact is likely to be direct and rapid, especially when environmental optima for sporulation and infection are reached. Moreover, once disease levels reach critical thresholds due to combinations of extreme events, they may not return to earlier levels when the weather returns to more normal conditions. This may be most critical for tree pathogens, particularly root rot and decay fungi which are usually long-lived, highly persistent once established on site, and operate over long time periods. These include pathogens such as *Heterobasidion*, *Armillaria* and *Collybia*, so the serious damage they can cause may take years to appear. In these instances the indirect influence of climate change

on tree physiology may be more crucial, although trying to predict how specific climate change scenarios are likely to influence tree resistance to pathogens is still largely a matter of speculation.

5.6.3 Deer

Climate change would be expected to favour the expansion of deer populations, as the carrying capacity of many habitats will increase due to longer growing seasons, rising carbon dioxide levels and increased warmth promoting plant productivity. Direct effects of climate change are also likely to increase recruitment and reduce winter mortality leading to further increase in deer populations. However, it is difficult to predict the effects on deer populations in any particular area of the UK. For example, as climate change progresses, southern England will experience conditions that may impact negatively on roe deer through reduced forage availability during the key reproductive period (Irvine *et al.*, 2007). This is supported by current roe deer populations in southern Europe being largely restricted to higher elevations. Furthermore, average body sizes are smaller than north European populations and juvenile growth is slower (Andersen and Linnell, 1998). There is also evidence (Raganella-Pelliccioni *et al.*, 2006) that juvenile survival is positively correlated with spring rainfall suggesting that forage availability is limiting population size. Broadmeadow (2004) suggested that, based on current European distribution, the effect of climate change on red and roe deer populations is uncertain. However, climate envelope modelling suggests that climate space for Roe deer may be lost in drier areas of east England by the 2050s, and across much of England by the 2080s under the UKCIP02 medium-high climate change scenarios. The population ecology of non-native species of deer (fallow, muntjac and sika) has been less well studied. However, in the UK they have been introduced at more northerly latitudes than their native ranges and are thus likely to benefit from climate change.

Where deer populations do increase, their impact on the establishment and growth of young trees and shrubs will increase. Deer browsing also alters the composition and structure of vegetation resulting in a more open understorey and increasing dominance of grasses at the expense of forbs (Kirby, 2001). Even if deer populations do not increase significantly, climate change may affect their habitat selection with, for example, the potential for deer to concentrate on areas where soil moisture is not limiting (Irvine *et al.*, 2007).

5.7 Impacts on timber quality

Potential impacts of climate change on timber quality can be separated into those that relate to the direct effects of climate change (temperature, carbon dioxide concentration, water availability, wind speed) and those that are the result of changes in growth rate. It should be noted that different species and provenances will be affected in different ways, particularly those impacts relating to water availability.

5.7.1 Direct effects of climate change on timber quality

Temperature

Milder winters could have a negative impact on tree growth form through late or incomplete winter hardening making trees more susceptible to frost damage. The same impact could result from early flushing, placing the frost tender new growth at risk of spring frost damage, as outlined in 5.1 above. Selecting more southerly, earlier flushing, provenances in anticipation of climate change could enhance this risk in the short to medium term and care should be taken in provenance selection for this reason. Earlier flushing may, however, have a positive impact on timber quality through countering the apparent relationship between late flushing and shake in oak (Savill and Mather, 1990). This would clearly be of economic benefit for producing high value logs over the longer term. A particular issue for conifer species is the tendency to produce lammas growth during mild, wet autumns, particularly if following summer drought. The shoots formed in this way are especially vulnerable to early autumn frost, which could lead to stem forking. Furthermore, the lammas growth sets a second whorl on the stem leading to increased knottiness in sawn timber.

Water availability

Apart from affecting timber quality through changes in growth rate, projected changes in water availability may lead to drought crack and ring shake in some species, as has been reported for Sitka spruce in eastern Scotland in recent years (Green and Ray, 2009). In contrast, Scots pine can produce poor form and lower timber quality as a result of forking and heavy branching on sites that are too wet. Since winter waterlogging is the principal problem, it is likely to become more widespread as a result of climate change.

Wind and exposure

The UK experiences a very windy climate that places limitations on forestry in some regions. Although projections of changes to the wind climate of the UK are far from certain (Chapter 2), potential impacts on timber quality are a concern alongside the more catastrophic effects of windthrow. Wind storms often result in leader loss which in turn leads to crooked stems as one of the side-branches takes over apical dominance from the lost leader. This poorer stem form in turn leads to a lower recovery of desirable straight 'green' logs used in construction grade timber production. Background wind loading can cause conifers to produce compression wood and broadleaved trees to produce tension wood. Both these types of wood lead to increased difficulties in processing and poorer performance in service. In particular compression wood can lead to increased distortion and failure under loading. Currently, compression wood is not a serious problem in British grown conifers but could be if windiness increases significantly.

5.7.2 Indirect, growth-rate mediated effects of climate change on timber quality

The most important climatic variable for plant growth is temperature, and when water and nutrients are not limiting, most species will produce increased growth (height or diameter or both) in warmer growing seasons. It has been suggested (Broadmeadow and Randle, 2002) that rising CO₂ levels and climate warming will result in an increase in productivity (defined here as Yield Class (YC): maximum mean annual wood volume increment over a rotation in cubic metres per hectare per year) of 1–2 units for most species (e.g. from YC 6 to YC 8 for oak; YC 14 to YC 16 for Sitka spruce). Trees growing faster will tend to have larger heights, diameters and inter-whorl distances. Changes in growth rate will affect wood properties and, in turn, timber performance. Properties affected include wood density, knot spacing and knot size. Typical values for Sitka spruce are given in Table 5.6 (Ray *et al.*, 2008).

Table 5.6
Comparison of Sitka spruce timber quality properties with yield class

Property	Unit	Yield class		
		14	16	18
Wood density	kg m ⁻³	425	422	420
Between whorl spacing	m	0.45	0.50	0.55
Knot size for largest whorl	cm	3.40	3.65	3.90

Although spruce and fir tend to have reduced wood density with increased growth rate, pine, larch and Douglas fir show little or no reduction in wood density with faster growth. Therefore, for these species, there is likely to be no change in the timber performance with increased growth rates resulting from climate change.

Hardwoods vary in response to increased rate of growth. The oaks, ash and elm are 'ring porous' producing harder and stronger timber when grown fast. In contrast, sycamore and birch are 'diffuse porous' and do not respond in this way. Chestnut and beech show an intermediate type of response.

Evidence on the direct effects of rising CO₂ levels on wood properties is conflicting, and it is difficult to distinguish between the direct effects of CO₂ and the indirect effects of CO₂ acting through enhanced growth. Evidence that CO₂ has little impact on wood anatomical properties is provided by Donaldson *et al.* (1987), Telewski *et al.* (1999) and Overdieck and Ziche (2005). However, this contrasts with the work of Conroy *et al.* (1990) and Ceulemans *et al.* (2002) who reported increases in tracheid wall thickness and tracheid width and resin canal density, respectively.

5.7.3 Interactions between timber quality and forest management

The negative impacts of climate change outlined above can, in part, be reduced by appropriate species or provenance and site selection. Equally, the opportunities presented by climate change can be taken through the same route of good silvicultural practice. If growth rates do increase as a result of climate change, this will clearly lead to increased sawlog production and higher value crops with more enduring uses. The future of commercial plantations will also be challenged in some regions – by either the direct effects of climate change or the impacts of pest and disease outbreaks. This may require the introduction of novel species for which there is a lack of processing and use information available in the UK. This should not be seen as a barrier for planting those species if there is clear evidence of the timber value of the species elsewhere in the world.

5.8 Preliminary commentary on the UK climate projections

In broad terms, the current UK climate projections (UKCP09) differ little in the magnitude of change from

the UKCIP02 Climate Change scenarios (Chapter 2). The publication of new projections does not therefore invalidate previous analyses such as those outlined in 5.2 and 5.3 above. The main differences are that the changes in precipitation (both summer and winter) are reduced in UKCP09 compared with UKCIP02, while the projected changes in summer and winter temperature are slightly larger. There is also a small shift in the location of the more extreme changes in temperature, with the area of greatest change extending from the southeast towards the southwest peninsula. The implications of climate change for forestry in the UK are therefore unlikely to change significantly. However, the way in which the climate projections are presented together with the availability of the weather generator will enable uncertainty to be quantified and a risk-based approach to be applied to adaptation strategies. Although such an approach will be far from straightforward to apply to complex systems such as forests that are affected by interactions between a range of climate variables, it will provide justification for any modification of current biodiversity or forestry policy.

5.9 Research priorities

- **Developing modelling capacity at the operational level.** There is a need to integrate modelling capabilities for UK conditions. A modelling system should be developed that combines the practical applicability of knowledge-based decision support systems with the more theoretical stand-level process-based models that can represent the effects of changing atmospheric composition on tree physiology and be extended outside the evidence-base of empirical models.
- **Identifying the possible effects of rising temperatures.** Rising temperatures may mean that the chilling requirement for successful germination of some native tree species fails to be met. There is a need to (1) identify whether this is the case and (2) identify whether there is a relationship between the chilling requirement of specific provenances and ambient temperature. The impact of rising temperatures on other processes (e.g. winter hardening and leafing date/frost risk) should also be further considered.
- **Learning from climate analogues.** ‘Climate matching’ analysis can identify broad regions that currently experience a climate similar to that projected for the UK in the future. This provides an opportunity to explore tangible impacts of likely climate change to accompany predictions based on model simulations. The approach should be adopted to explore likely changes in

woodland ecosystems, the suitability of tree species for commercial forestry, and to inform changes to forest management that might be required in response to the changing climate. It must, however, be understood that such an approach can only provide broad guidance as complete analogues of future conditions, particularly when the distribution of extremes is considered, do not exist.

- **Understanding risks to biosecurity.** Pest risk analysis is required for a range of insect pests that potentially represent a risk to British forests if they are introduced. The *ex-situ* temperature response of growth should also be determined for a range of tree pests and pathogens to provide the basis for epidemiological modelling of future outbreaks under a changing climate.

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