Chapter 8:
Woodlands

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Key Findings*

Since 1945, the area of woodland has doubled to cover 12% of the UK, but still remains well below the EU average of 37%1. Much of this increase is due to afforestation for timber production, leading to the dominance of coniferous species. These comprise 81% of current woodland in Scotland, 55% in Wales and 35% in England. Recently, more broadleaved species have been planted, resulting in an increase of 6.9% in the area of broadleaved mixed and yew woodland in the UK between 1998 and 2007.

[1] *well established*

There is no primary woodland in the UK; all remaining woodland has been influenced by human activities[1]. Nevertheless, the woodland that remains contains significant biodiversity: a quarter of all UK Biodiversity Action Plan priority species are associated with trees and woods. Trends in the condition of habitats and species vary, but that of woodland SSSIs and seven priority native woodland habitats is improving. Short-term trends can be misleading, however[2]. Recent plantations gain native species, albeit with different assemblages from those of semi-natural woods[c].

[1] *well established*
[2] *established but incomplete evidence*
[c] *likely*

Many factors, at all scales, effect change in woodland ecosystems. Key drivers include climate change, pollution, government policy on land use, society, global trade and domestic markets, and the endogenous dynamics of ageing woodland. Although recent climate change has had little effect on woodland structure and composition, mobile species, such as insects and birds, have shown range changes, and increasing temperatures have led to faster tree growth and altered phenology in some areas[2]. Despite recent reductions in emissions, nitrogen deposition and ozone levels are still above 'critical loads' for habitats such as UK Atlantic Oakwoods. Wild herbivores, particularly deer, have increased in number over the past 30 years to the detriment of woodland habitats[1].

[1] *well established*
[2] *established but incomplete evidence*

Timber production is an important provisioning service from woodlands. Domestic production has increased from an estimated 4% in the 1940s to 20% of UK consumption of timber, pulp and panel products today[1]. In 2009, 8.5 million green tonnes of softwood was produced in the UK—approximately 60% of annual growth increment—and production is predicted to rise to 11–12 million tonnes in the 2020s. A total of 0.4 million tonnes of hardwood were produced from broadleaves—about 20% of annual growth increment. Non-timber products from woodlands can also be important; for example, game shooting is estimated to contribute £640 million per annum to the UK economy[2].

[1] *well established*
[2] *established but incomplete evidence*

Woodlands are highly valued by people for social and cultural services*; there are approximately 250–300 million day visits to woodlands per year. Woodland includes nearly 5,000 Scheduled Ancient Monuments, plus many areas managed for geological study. The social and environmental benefits of woodlands in Great Britain (GB) were valued in 2002 at more than £1.2 billion per annum (at 2010 prices), with the landscape value of woodland estimated at £185 million (2010), and recreational visits valued at £484 million (2010). However, only 55% of the population has access to woods larger than 20 ha within 4 km of their home.

[2] *established but incomplete evidence*

* Each Key Finding has been assigned a level of scientific certainty, based on a 4-box model and complemented, where possible, with a likelihood scale. Superscript numbers and letters indicate the uncertainty term assigned to each finding. Full details of each term and how they were assigned are presented in Appendix 8.1.

Carbon sequestration is one of the most important regulating services provided by woodlands*. The total carbon (C) stock in UK forests (including soils) is around 800 megatonnes (Mt) of carbon (2,900 Mt of carbon dioxide (CO_2) equivalent), and is estimated to be a further 80 Mt C in timber and wood products. The strength of the UK forest carbon sink increased from 1990 to 2004, but may now start to decline due to a fall off in planting rates in the last 20 years and harvesting of mature trees. At peak growth, coniferous forest can sequester around 24 tonnes of CO_2 per hectare per year, with a net long-term average of around 14 t CO_2/ha/yr. Rates of around 15 t CO_2/ha/yr have been measured in oak forest at peak growth, with a net long-term average likely to be around 7 t CO_2/ha/yr.

[2] *established but incomplete evidence*

The social value of net carbon sequestration by UK woodlands is currently at least double the market value of wood production per hectare; and the total value of net carbon sequestration by UK woodlands planted after 1921 increased more than six-fold over the period between 1945 and 2004, falling thereafter[2]. Carbon sequestration currently has the highest annual social value of the woodland ecosystem services considered; however, as it remains largely a non-market value, there is little incentive for landowners to increase its provision or to maintain existing carbon storage at present.

[2] *established but incomplete evidence*

Forest policy and woodland management have changed over time as different goods and services have been required[1]. There are both trade-offs and synergies between goods and services produced by woodlands. The diversification of forest structure for biodiversity benefits may improve cultural services (including aesthetics), while increases in forest cover may benefit carbon regulation and flood regulation. However, maximising provisioning services through the use of highly productive species and intensive site treatments may have negative effects upon the value of woodland for biodiversity and for cultural services.

[1] *well established*

A spectrum of techniques within a framework of sustainable forest management can deliver different goods and services[2]. Certification schemes encourage appropriate action. Around half of the UK's woodlands are certified under independent sustainability assessment schemes. There are multiple spatial (and temporal) scales at which choices can be made, limited evidence for some of the consequences, and a variety of planning frameworks to assist with choices. Achieving coordinated action across multiple ownerships at broad scales is challenging, but has become the target of recent forest policy and research; coordination across land uses to secure integrated landscapes now needs to be pursued.

[2] *established but incomplete evidence*

8.1 Introduction

"[Trees] …Nothing can compete with these larger-than-life organisms for signalling the changes in the natural world." (Roger Deakin 2007)

8.1.1 Woodlands of the UK

The forests and woodlands of the UK provide an important range of ecosystem services and associated goods and benefits, such as timber, soil protection, amenity and biodiversity (Sections 8.2 & 8.3).

The climate of the UK has a strong maritime influence that, over time, has led to the development of a number of distinctive cool temperate and boreal native forest types, which are a subset of those found in continental Europe (Barbatti *et al.* 2007) and observable despite the substantial loss of natural woodland cover. There is considerable variation in composition in response to climatic gradients, lithology and soil type. Distinctive 'Atlantic' woodland, dominated by oak (*Quercus petraea*) and birch (*Betula* species), occurs in wetter and cooler north and west areas, with Scottish native Scots pine (*Pinus sylvestris*) woodland on nutrient-poor acid soils. In the south and east of the UK, the dominant native woodland habitat is mixed lowland broadleaved woodland consisting of oak (*Q. petraea* and *Q. robur*) and ash (*Fraxinus excelsior*), and with localised areas of beech (*Fagus sylvatica*) and hornbeam (*Carpinus betulus*). Wet woodlands of alder (*Alnus glutinosa*), willows (*Salix* species) and birch (*B. pubescens*) occur in sites with regularly wet soils (Rodwell 1991; Malcolm *et al.* 2005; Barbatti *et al.* 2007). The same climatic constraints have influenced the development of woodland management, for example, by enabling a wide range of potential temperate species to be considered, but also by presenting some particular issues, such as wind, rather than fire, being a dominant abiotic disturbance (Quine & Gardiner 2006).

The post-glacial history of native woodland in the UK is largely one of loss, degradation and fragmentation (Rackham 1986). Tree species, such as oak, recolonised from refugia in southern Europe (e.g. Iberia) (Petit *et al.* 2002) and possibly from the west, but others like Norway spruce (*Picea abies*) failed to establish, despite being present in previous interglacial periods (Rackham 2003). From a post-glacial high of perhaps 70–80%, there was a progressive decline in woodland cover, partly due to climate and partly human-driven; by medieval times there was little extant native woodland, especially in Scotland and Northern Ireland (Section 8.2). A number of notable woodland species, such as bear, wolf, wild ox, beaver, lynx and capercaillie, became extinct (Corbett & Yalden 2001; Smout 2002), although some have since been reintroduced. Soil changes towards increased podzolisation also occurred in many upland areas after the loss of woodland cover (Dimbleby 1962; Chapter 5). By the beginning of the 20th Century, woodland comprised less than 5% of the country (Rackham 1986; Peterken 1996). Major forest types associated with tree-lines and floodplains were largely lost, as was the lime-dominated forest over much of southern Britain. Nevertheless, many other species contributing to our woodland biodiversity were conserved through the retention of ancient broadleaved woodland, some of which was managed on a coppicing system (Rackham 1986; Peterken 1991), the preservation of larger tracts of wood pasture and parkland with ancient trees (Rackham 2003) and the survival of areas of old-growth native pinewood in Scotland (Mason *et al.* 2004).

Concern over the further loss and degradation of ancient and native woodland in the decades following the Second World War (WWII) led to the development of policies, firstly, for the protection of key sites (e.g. National Nature Reserves and Sites of Special Scientific Interest), and later, for the protection, management and expansion of priority woodland habitats (Kirby 2003; Latham *et al.* 2005; UK BAP 2006; Section 8.2).

Notable woodland planting by private estate owners began in the late 17th and 18th Centuries, but substantial re-afforestation efforts began in the 20th Century (Linnard 2000; Smout 2002). Successive governments attempted to address the shortage of timber by encouraging the creation of large plantations of non-native conifer species, but this effort was compounded by wartime fellings. There was considerable criticism of such conifer-planting on open moors and bogs (Avery 1989) and on existing ancient woodland sites (NCC 1984; Humphrey & Nixon 1999) due to the loss of valued habitats and the rapid pace of change in upland areas; afforestation, together with development and agriculture, contributed to major reductions in the extent and fragmentation of lowland heath (Webb 1986). Opposition to 'commercial forestry' from the conservation sector and other changes, such as those relating to government taxation policy, have led to a dramatic decrease in new planting of conifers over the last 20 years (Section 8.2), although they continue to be used extensively in the restocking of existing forests. In contrast, over this period, there has been an increase in the area of native woodland and the use of broadleaved tree species for planting or natural regeneration (Section 8.2).

Methods of woodland management have evolved, reflecting woodland type, markets and labour availability and affordability. Economic production of a narrow range of timber and wood products has led to the neglect of multiple products and services in favour of the simplification of practices. In the latter part of the 20th Century, there was an almost complete cessation of traditional coppice management systems in native woodlands (Buckley 1992). Commercial plantations were, and in many cases still are, managed on an even-aged basis, with large-scale felling of stands at economic maturity to maximise timber production (Section 8.5).

In recent decades, there has been a shift in forestry policy and practice with the adoption of the principle of Sustainable Forest Management (SFM) for multiple benefits (Mason 2007) (Section 8.5). Woodlands are managed as a resource for people, providing timber, wood products, recreation, amenities and well-being, as well as being managed for the benefit of local wildlife.

8.1.2 What is Woodland?

In this chapter, we use the term 'woodland' interchangeably with 'forest'. **Table 8.1** summarises the definitions used in the various major survey and reporting schemes covering woodland in the UK. The National Inventory of Woodlands and Trees (NIWT) provides the most comprehensive information on woodland in GB (Forestry Commission 2003a), but only recently have NIWT data been combined with inventory data from Northern Ireland to give an annual UK picture (Forestry Commission 2009a). NIWT also provides the British information for the Food and Agriculture Organization's (FAO) Global Forest Resource Assessments (FAO 2005).

The UK NEA broad habitat 'woodlands' is based on the broad habitat definitions in the UK Biodiversity Action Plan (BAP) (**Box 8.1**). These definitions are also used by the UK-wide Countryside Survey (Carey *et al.* 2008) with slight variation. Two woodland habitats are recognised in the UK BAP: coniferous woodland and broadleaved mixed and yew woodland. Within these two categories, further priority

Table 8.1 Definitions of woodland used in recent surveys.

UK BAP Broad Habitats*	Countryside Survey 2007	National Inventory of Woodlands and Trees†	Food and Agriculture Organization 2005	Native Woodland Survey of Scotland	Ancient Woodland Inventories (AWI)
Definition of woodland					
Vegetation dominated by trees >5 m in height when mature; >20% canopy cover.	Trees and shrubs >1 m in height (from vegetation key 2007) with >25% canopy cover; felled or recently planted woodland not included. Minimum area of woodland 400 m², minimum width 5 m.	A minimum area of 0.5 ha; and a minimum width of 20 m; tree crown cover ≥20% or the potential to achieve it; a minimum height of 2 m, or the potential to achieve it.	Trees >5 m in height in areas >0.5 ha; canopy cover >10%; minimum width 20 m or able to make these thresholds *in situ*. Does not include agro-forestry or parks and gardens.	Wooded polygons larger than 0.5 ha with a canopy cover of ≥20% of which ≥40% is native species.	Areas ≥2 ha marked as woodland on 1920s base maps and supporting woodland since at least 1600 in England and Wales, 1750 in Scotland, 1830 in Northern Ireland¶ (date under review in Wales). Woods less than 2 ha were considered in the Northern Ireland inventory and in more recent revisions to the inventory in south-east England and Wales.
Definition of woodland types					
Coniferous woodland >80% of canopy comprising conifer species; includes areas temporarily cleared of woodland. **Broadleaved, mixed and yew woodland** >20% canopy to be dominated by broadleaved species or yew; woody scrub <5 m tall included in some circumstances.	Divided by UK BAP woodland broad habitats; coniferous woodland and broadleaved woodland.	Indicative forest types interpreted from aerial photographs: broadleaved, conifer, mixed conifer, mixed broadleaved, young trees, scrub, felled.	**Other wooded land** Land not classified as forest, spanning >0.5 ha; with trees >5 m tall and a canopy cover of 5–10%, or trees able to reach these thresholds *in situ*; or with a combined cover of shrubs, bushes and trees above 10%. It does not include land that is predominantly under agricultural or urban land use. **Other land with tree cover** Agricultural land, meadows and pastures, built-up areas, barren land, with groups of trees >0.5 ha; canopy cover >10% of trees capable of >5 m height at maturity.	Polygons are ascribed to HAP and NVC‡ types.	Ancient semi-natural woodland—no recent evidence of planting. Ancient replanted woodland Long-established woodland category in Scotland and Northern Ireland used for sites wooded since the middle of the 19th Century.

* From Jackson (2000).
† Based on Patenaude *et al.* (2005).
‡ HAP = Habitat Action Plan; NVC = National Vegetation Classification (Rodwell 1991).
¶ The Northern Ireland AWI was undertaken separately to the GB AWI and included areas 0.5 ha and more; where the origin of woodland in Scotland and Northern Ireland is unknown, but presence can be verified in 1750 or 1830 respectively the woodland is termed "long-established".

Box 8.1 Woodland habitats and species within the UK Biodiversity Action Plan (UK BAP).

The UK BAP recognises two **broad** woodland habitats and eight *priority* habitat types. In addition, about a quarter of the UK BAP priority species are associated with woodland or tree habitats to varying degrees. For example, in England 256 species are associated with tree and woodland habitats (Webb *et al.* 2009).

Bluebells under mixed broadleaved woodland. *Photo courtesy of FC Picture Library / George Gate.*

Conservation area. Non-native tree species have been removed to allow the Caledonian pines and sensitive ground vegetation to expand. *Photo courtesy of FC Picture Library / Isobel Cameron.*

Broadleaved, mixed and yew woodland	Coniferous woodland
Lowland mixed deciduous woodland	*Native pine woods* (Scotland only)
Lowland beech and yew woodland	
Wet woodland	
Wood-pasture and parkland *	
Upland mixed ash woodland	
Upland oak woodland	
Upland birch woodland (Scotland only)	

* Note that wood-pasture and parkland is considered in Chapter 6.

habitats are recognised (Section 8.2). Habitat Action Plans (HAPs), which suggest measures for the conservation and restoration of priority habitats, cover the range of native woodland types in the UK; these may either be planted, semi-natural, ancient or recent stands.

There is not a clear division between provision of goods and services by priority habitats and those from other types of woodland. As a generalisation, however, priority habitats contribute more to biodiversity and some cultural services, and less to the provision of wood fibre and carbon sequestration. Despite this, there are many overlaps between them, so we have not separated out the contribution that specific priority habitats make to ecosystem service provision, but have tried to distinguish the difference between the two broad habitat types where appropriate.

8.1.3 Interactions with Other UK NEA Broad Habitats

Woodlands and, perhaps most notably, native woodlands share many of the species and vegetation assemblages of non-wooded broad habitats; for example, native pinewood vegetation is very similar to heathland vegetation (Rodwell 1991). Riparian zones form a key ecotone between woodland and the aquatic environment, so recently developed guidelines recommend the amount and type of woodland that should be grown alongside watercourses in order to ensure the continuation of this habitat (Forestry Commission 2003b). There is a dynamic interplay between woodlands and other habitats, with successional pathways operating at multiple temporal and spatial scales, and directions (Hester *et al.* 1991a, 1991b, 1991c). In addition, organisms may move between different habitats to secure their full range of food and shelter requirements.

During the 20th Century, there was considerable afforestation of upland heath and bog habitats (Thompson *et al.* 1995), and in the lowlands, many heathland ecosystems were converted to conifer plantations (Mason 2007). Restoration has recently been undertaken to repair some of the damage incurred by these changes, including extensive removal of woodlands in the Flow Country and other peatlands (Patterson & Anderson 2000; Lindsay 2010). In England, a new framework policy has been developed to guide woodland clearance to meet UK BAP targets for priority open habitats (Forestry Commission 2010a). However, there remains pressure upon woodlands from other uses such as urban development and energy development (e.g. windfarms). The Countryside Survey (Carey *et al.* 2008) provides some data on stocks and flows (which are summarised below), while net gains and losses of woodland can be derived from successive national forest inventories.

More recently, there has been a focus on landscape-scale planning as a framework for deciding on the balance and configuration of woodland and other habitats over large areas (Humphrey *et al.* 2009). Tools such as BEETLE (Watts *et al.* 2010) and the Native Woodland Model (Towers *et al.* 2002), have been developed to provide an ecological basis for planning frameworks within the context of regulations such as the EU Water Framework Directive (WFD) and Strategic Environmental Impact Assessments.

Table 8.2 Extent of ancient and semi-natural woodland in the UK ('000 ha). Source: Forestry Commission (2009a) based on data from Pryor and Peterken (2001) with Northern Ireland data from Woodland Trust (2007).

Woodland type	England	Scotland	Wales	Northern Ireland	UK
ASNW *	206	89	31	0	326
PAWS *	135	59	30	1	225
OSNW *	210	44	52	15	320
Total ancient †	**341**	**148**	**61**	**1**	**551**
Total semi-natural †	**416**	**133**	**82**	**15**	**646**

* ASNW (Ancient Semi-Natural Woodland) is both ancient and semi-natural; PAWS (Plantation on an Ancient Woodland Site) is ancient but not semi-natural; OSNW (Other Semi-Natural Woodland) is semi-natural but not ancient.
† Ancient woodland is woodland that has been in continuous existence since 1600 (1750 in Scotland, 1830 in Northern Ireland); semi-natural woodland is woodland with natural characteristics (predominantly native species of trees, ground plants and animals).

8.2 Trends and Changes in Woodlands

"The great accomplishments of foresters in deforested Great Britain are admirable... The first step was to establish the material base of the forest, i.e. to create biomass. It is possible to progress only after such a base has been created...." (Prof. D. Mlinsek 1979)

8.2.1 Current Extent, Location and Composition

Woodland area in the UK currently amounts to 2.84 million hectares, representing 12% of the total land area. Approximately 9% of England is wooded, 17% of Scotland, 14% of Wales and 6% of Northern Ireland according to the Forestry Commission (2009a); the Countryside Survey (Carey *et al.* 2008) suggests that 9% of England, 15% of Scotland, 13% of Wales and 10% of Northern Ireland are wooded. Despite the small discrepancies between estimates, it is widely accepted that there is substantially less woodland cover in the UK in comparison to the global average of 30% and the EU average of 37% (FAO 2005).

There are major concentrations of planted coniferous woodlands in Wales, south and west Scotland, Northumberland, and in Thetford Forest, Norfolk. The Scottish Highlands have significant cover of native woodland of various HAP types, as does the New Forest, Forest of Dean/Wye Valley and the south-east of England; elsewhere native woodland is largely fragmented and dispersed (**Figure 8.1a,b**).

All UK woodland has been modified by management to some extent. There are no areas of primary woodland left in the UK (FAO 2005): the majority of woodland area (66.8%) is classed as Productive Plantation, with Modified Natural and Semi-natural representing 32.3% of woodland area, 0.7% is classed as Protective Plantation. Each of these categories delivers a different set of ecosystem services (Section 8.3).

The Modified Natural category (22.7% of woodland area) can be sub-divided further into Ancient Semi-natural Woodland (ASNW), Plantations (of mostly non-native tree species) on Ancient Woodland Sites (PAWS) and Other Semi-natural Woodland (OSNW), i.e. non-ancient semi-natural woodland (FAO 2005) (**Table 8.2**). Within the UK, ancient semi-natural woodland has long been recognised as being of the highest value for nature conservation and biodiversity (Peterken 1977), but more recent semi-natural woodland and plantations can also be of value in delivering other services, such as providing recreational and educational opportunities.

8.2.2 Stand Age and Structural Stages

Any one point in a forest or woodland might naturally go through a series of structural stages over time (**Figure 8.2a,b**). The extent and distribution of the different stages across the landscape is then largely determined by natural disturbance regimes, or their emulation and replacement by patterns of woodland management (Hopkins & Kirby 2007; Mason 2007), in particular, the patterns of felling. In general, management truncates the age-class distribution. Conifers in commercial stands are felled at economic maturity (e.g. for Sitka spruce 40–50 years old) and only 1% of commercial forests are managed as natural reserves to develop old-growth features (Forestry Commission 2004; UKWAS 2008). Such management decisions result in different structures (Quine *et al.* 2007) and different combinations of goods and services (Sections 8.3, 8.4 & 8.5).

The most recent data for forest structure and age suggest that 7% of the current forest area is less than 15 years old; 31% 15–50 years; 43% 51–100 years and 20% over 100 years (Mason 2007). These age classes equate broadly with the ecological definitions of stand stages developed in North America (Oliver & Larson 1996; Frelich 2002). Stands less than 15 years old are typically in the 'stand initiation' phase, where the canopy is not yet closed and understorey vegetation has not been shaded out. These stands can have considerable value for biodiversity (Warren & Key 1991). Stands of 15–50 years represent the 'stem exclusion' stage, where there is a closed canopy and little understorey. Stands of 51–100 years old characterise the 'understorey reinitiation' or 'demographic transition' phase, where shrubs and ground vegetation recolonise. Stands older than 100 years are entering the last phase, termed 'old-growth' or 'multi-aged', which is characterised by a high frequency of large-diameter trees, a mix of tree ages and significant accumulations of fallen and standing deadwood of importance for biodiversity (Warren & Key 1991).

The precise age demarcation of the stand stages, especially with respect to old-growth, varies between tree species, soil types and localities (Mason *et al.* 2004; Humphrey 2005). Stands older than 100 years are generally dominated by broadleaves; although native Scots pine woodland contains current stands of 250 years or more (Humphrey *et al.* 2003).

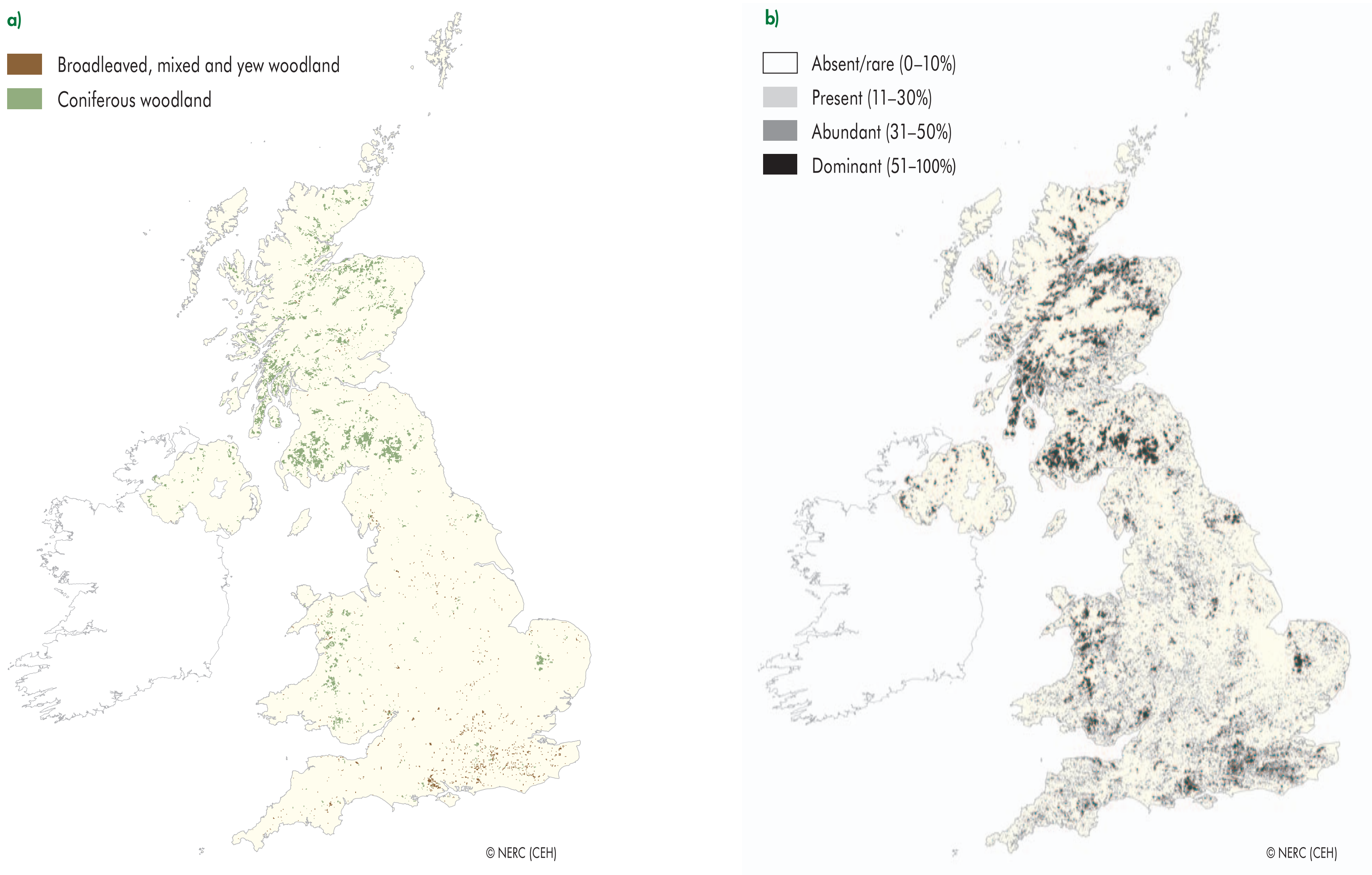

Figure 8.1. Distribution of UK NEA Woodland habitat in the UK by a) dominate (>51% area per 1 km cell) woodland type and b) percent cover per 1 km cell.
Source: Land Cover Map (2000).

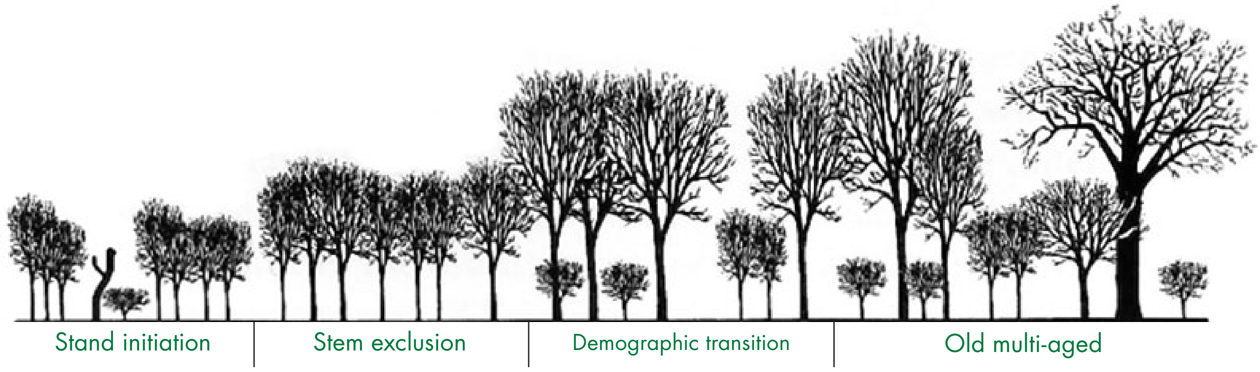

Figure 8.2a Four stand stages. Source: reproduced from Frelich (2002).

8.2.3 Composition and Management Types

In the past, the natural composition of UK woodland would have been predominantly broadleaved, with local concentrations of yew and juniper, and more significant areas of pine in the highlands of Scotland and possibly as part of tree-line woods on wooded bogs and acid sands further south (Peterken 1996). According to the FAO (2005), the UK has 66 native tree and shrub species. Among these are a number of whitebeams which are unique to the UK and Ireland and are considered by the International Union for Conservation of Nature to be threatened: three are classified as Critically Endangered (*Sorbus leptophylla, S. leyana, S. wilmottiana*), one is Endangered (*S. bristoliensis*), and six are Vulnerable (*S. anglica, S. arranensis, S. eminens, S. pseudifennica, S. subcuneata, S. vexans*) (see www.iucnredlist.org for definitions of the categories).

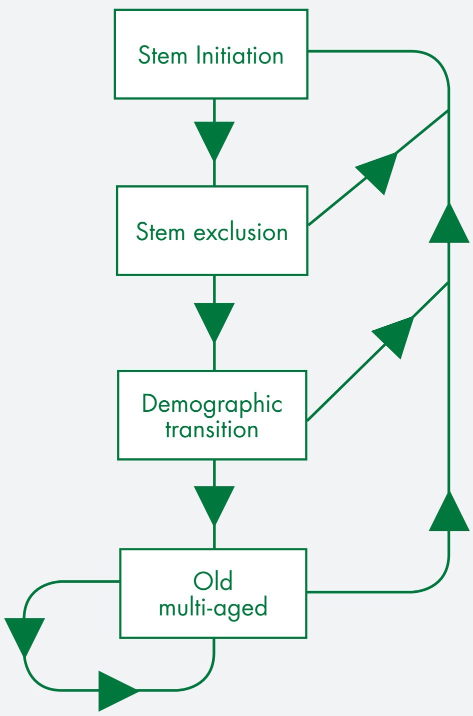

Figure 8.2b Stand stages—multiple pathways between stages that may be brought about by natural processes or management interventions.

Climate, soils, land availability and productive potential have all influenced the location of plantations and the selection of species within them. In the 20th Century, Sitka spruce (*Picea sitchensis*) was the dominant choice of commercial species in northern and western areas, with extensive forests planted on upland heaths and grasslands. Scots pine (now only native in Scotland) and Corsican pine (*Pinus nigra* subspecies *Laricio*) were more popular in the south and east, but Douglas fir (*Pseudotsuga menziesii*) and various species of larch and other conifers were also planted extensively across the UK. Sitka spruce is the commonest tree species in Britain (29% by area) followed by Scots pine (9.5%) and oak (9.4%) (**Table 8.3**); in Northern Ireland, Sitka spruce accounts for the largest volume of growing stock (6.5 million m^3 in 2010), followed by Scots pine (0.7 million m^3), Norway spruce (0.5 million m^3) and oak (0.4 million m^3) (FAO 2010). The vast majority of woodland in the UK is managed as high forest, with clear-felling and restocking on a 40- to 50-year rotation for conifers. Thinning of woodland has been limited by threat of windthrow and by lack of markets for small dimension produce. The small amount of coppicing (0.9% by area) that takes place in the UK is undertaken almost exclusively in England.

Veteran trees (Read 2000), often managed in the past by pollarding, tend to be commoner in non-woodland situations or in open parks and wood-pastures, but may still be found in forested landscapes; for example, in Savernake, Wiltshire, over 5,000 veteran trees have been mapped (P. Crow pers comm.). Such trees, often several hundred years old, are not only living evidence of past land management, but provide habitats for rare and specialist organisms. The UK is widely believed to have a higher density of veteran trees than most other northern European countries and, hence, has a particular responsibility for protecting them for their high conservation value.

All trees in the UK form mycorrhizal associations; these are mutualistic associations between the trees and some soil fungi, where the relationship is based upon transfer of soil-derived nutrients from the fungus to the host in exchange for photosynthate. These relationships between trees and mycrorrhizal fungi are essential to the functioning of the whole system (Smith & Read 2007; Chapter 13), and yet, are rarely used in measures of woodland biodiversity status or condition.

Table 8.3 Area of woodland in GB by main tree species ('000 ha). Source: Forestry Commission (2003a).

Species	England	Scotland	Wales	GB*
Conifers				
Scots pine	82	140	5	227
Corsican pine	41	2	3	47
Lodgepole pine	7	122	6	135
Sitka spruce	80	528	84	692
Norway spruce	32	35	11	79
European larch	14	9	1	23
Japanese/hybrid larch	33	56	22	111
Douglas fir	24	10	11	45
Other conifer	19	5	6	30
Mixed conifer	9	8	0	18
Total conifers	**340**	**916**	**149**	**1,406**
Broadleaves				
Oak	159	21	43	223
Beech	64	10	9	83
Sycamore	49	11	7	67
Ash	105	5	19	129
Birch	70	78	13	160
Poplar	11	0	1	12
Sweet chestnut	12	0	1	12
Elm	4	1	0	5
Other broadleaves	84	18	18	120
Mixed broadleaves	91	62	8	160
Total broadleaves	648	206	118	971
Total—all species	**988**	**1,123**	**266**	**2,377**
Felled	15	23	9	47
Coppice †	22	1	0	24
Open space ‡	72	134	11	217
Total woodland	**1,097**	**1,281**	**287**	**2,665**

* Note no equivalent data are available for Northern Ireland.
† Coppice includes coppice with standards.
‡ Areas of integral open space, each <1 ha.

8.2.4 Trends and Indicators of Change in Woodland

The following sections explore changes in the condition of woodland in the UK, and how such changes are currently assessed by a number of indicators.

The European Environment Agency (EEA) defines an environmental indicator as "a measure, generally quantitative, that can be used to illustrate and communicate complex phenomena simply, including trends and progress over time" (EEA 2005). The EEA distinguish indicators of driving forces, pressures, states, impacts and responses. In relation to forests and woodlands, indicators have been adopted which relate mostly to states and impacts. Conceptually, these indicators represent easily measured features, such as an organism, forest structure or productivity, which can be used as an index of attributes (e.g. diversity) that are too difficult or expensive to measure for other aspects of woodland ecosystem supply and services (Williams & Gaston 1998; EEA 2009). Following the Ministerial Conference on the Protection of Forests in Europe (MCPFE 1993), countries have developed indicator sets which assess progress in sustainable forestry and relate to ecosystem supply (biodiversity, condition and extent) and ecosystem services (supporting, regulating and cultural). The UK indicators, some of which form subsets of more general indicators of sustainability both at national and international scales, are incorporated into the UK Forestry Standard (Section 8.5).

We do refer to the condition and status of UK BAP habitats and of protected Sites/Areas of Special Scientific Interest (SSSIs/ASSIs), but place less emphasis on these data than in some other chapters because our scope is all forests, not just the semi-natural component.

8.2.4.1 Change in extent and connectivity of woodlands

Tree cover of one sort or another is considered to have dominated the landscape in the UK in the pre-Neolithic period, although there are disputes as to how much of this was closed high forest and how much was a more open wooded system (Vera 2000; Rackham 2003; Hodder *et al.* 2005, 2009). Throughout the Middle ages, forest cover declined, until it reached an all-time low of 4.7% around the beginning the 20th Century (**Table 8.4**). Since 1945, there has been a significant increase in forest cover through new planting and forest creation (**Table 8.5**). This has also led to changes in the distribution of forest cover across the UK and within regions. The largest increases have been of coniferous woodland in Wales and Scotland, although changes have been positive throughout the country (**Figure 8.3**).

Periodic forestry and land-cover surveys (Forestry Commission 2003a; Carey *et al.* 2008) provide regular information on gains and losses of trees and woodland.

Since the mid 1980s, the rate of increase has slowed (the increase from 2001 to 2006 being only 40% of that in the period 1971 to 1976). There has been a shift towards expansion of broadleaved/native woodland (Forestry Commission 2003a), rather than coniferous woodland, and usually in smaller blocks (**Table 8.5**). For example, the development of the 'National Forest' in the English Midlands has been largely through relatively small, scattered, new woods (Anon 2009). Countryside Survey data show that the area of broadleaved woodland increased by 6.9% (from 5.6–6.0% of land-cover) in the UK between 1998 and 2007 (**Table 8.6a**) and that there was no detectable change in the area of coniferous woodland in the UK, although it decreased by 7.2% in Scotland (from 12.9–11.9% of land-cover) between 1998 and 2007 (**Table 8.6b**). Reporting of the UK BAP priority woodland habitats also suggest that change is modest and largely positive in character, and showed that most were stable or increasing extent (**Table 8.7**).

There has been a revival of interest in afforestation as part of the future climate change adaptation/mitigation programmes, which may shift the balance back towards conifers and highly productive broadleaved species (Read *et al.* 2009). This renewed interest is consistent with the increased emphasis on conifer-planting in England over the last eight years (**Table 8.5**), but rates of planting are influenced by many factors.

Ancient and semi-natural/native woodland—a particular concern from a biodiversity perspective (Peterken 1977)—has declined due to losses to agriculture and, to a lesser extent, development. In addition, before 1985, large areas of ancient semi-natural woodland were converted to plantations predominantly of non-native species, generally conifers (Spencer & Kirby 1992; Roberts *et al.* 1992). Since then, policy changes have reduced the rates of clearance and encouraged restoration of replanted stands to native species (Forestry Commission 1985; Defra & Forestry Commission 2005; Goldberg *et al.* 2007). Individual sites do still come under threat from development (Woodland Trust 2010) and from insidious loss through overgrazing, especially in the Scottish Highlands (Mackenzie 1999).

There is some turnover of other woodland cover. The Countryside Survey (Carey *et al.* 2008) showed that felled conifers have been replaced by neutral grassland, acid grassland or bog broad habitats in some places. Woodland removal in Scotland (e.g. for wind turbines) is discouraged by recent policy guidance, and there is an expectation that the rate of such clearance will not exceed the rate of afforestation (Scottish Government 2009) with mechanisms of compensatory planting being introduced. Deforestation in England is permitted in some circumstances, notably where clearance of recent woodland, particularly plantations or scrub, is proposed to restore priority open habitats (Forestry Commission 2010a). More generally, clearance of any woodland is controlled largely through the operation of Felling Licences (though not applicable in Northern Ireland) and Environmental Impact Assessment (Forestry) regulations. Further discussion on drivers of change in woodlands is found in Section 8.2.5. Information on changes in the numbers and extent of small clumps and individual trees is available in Chapters 6, 7 and 10. Non-woodland trees declined in the post-war period (Peterken & Allison 1989) as a consequence of agricultural intensification. Dutch Elm Disease also had a major effect during this time (Burdekin 1983).

There is increasing interest in the extent to which woodlands are functionally connected, and whether new woodland has contributed to, or could make a further contribution to, reducing the isolation of fragments of biodiversity (Bailey 2007). Schemes such as JIGSAW have sought to target new woodland planting to make a

Table 8.4 Woodland area in the UK: changes over 10 centuries. Source: Forestry Commission (2009a).

Year	England		Scotland		Wales		Northern Ireland †		UK	
	Area ('000 ha)	%*	Area ('000 ha)	%*	Area ('000 ha)	%*	Area ('000 ha)	%*	Area ('000 ha)	%*
1086		15 ‡								
c.1350		10 ‡		4 ‡						
17thC		8 ‡		4 ‡				5 ‡		
1905	681	5.2	351	4.5	88	4.2	15	1.1	1,140	4.7
1924	660	5.1	435	5.6	103	5.0	13	1.0	1,211	5.0
1947	755	5.8	513	6.6	128	6.2	23	1.7	1,419	5.9
1965	886	6.8	656	8.4	201	9.7	42	3.1	1,784	7.4
1980	948	7.3	920	11.8	241	11.6	67	4.9	2,175	9.0
1995–1999	1,097	8.4	1,281	16.4	287	13.8	81	6.0	2,746	11.3
2009 ¶‡	1,128	8.7	1,341	17.2	284	13.7	88	6.5	2,841	11.7

* % of the total surface area including inland water. The total surface areas, including inland water are taken from the Annual Abstract of Statistics 2008 (published by the Office for National Statistics).
† For Northern Ireland, the 17th Century figure is an estimate for all of Ireland; the 1905 figure is an estimate for Ulster 1908; the 1947 figure assumes no change from the 1939 to 1940 Census.
‡ An approximation.
¶ The non-Forestry Commission woodland figures for 2008 for England, Scotland and Wales are based on the 1995 to 1999 National Inventory of Woodland and Trees (NIWT) and adjusted for new planting and sales of Forestry Commission woodland, but at present no adjustment is made for woodland converted to another land-use. The NIWT did not include Northern Ireland.

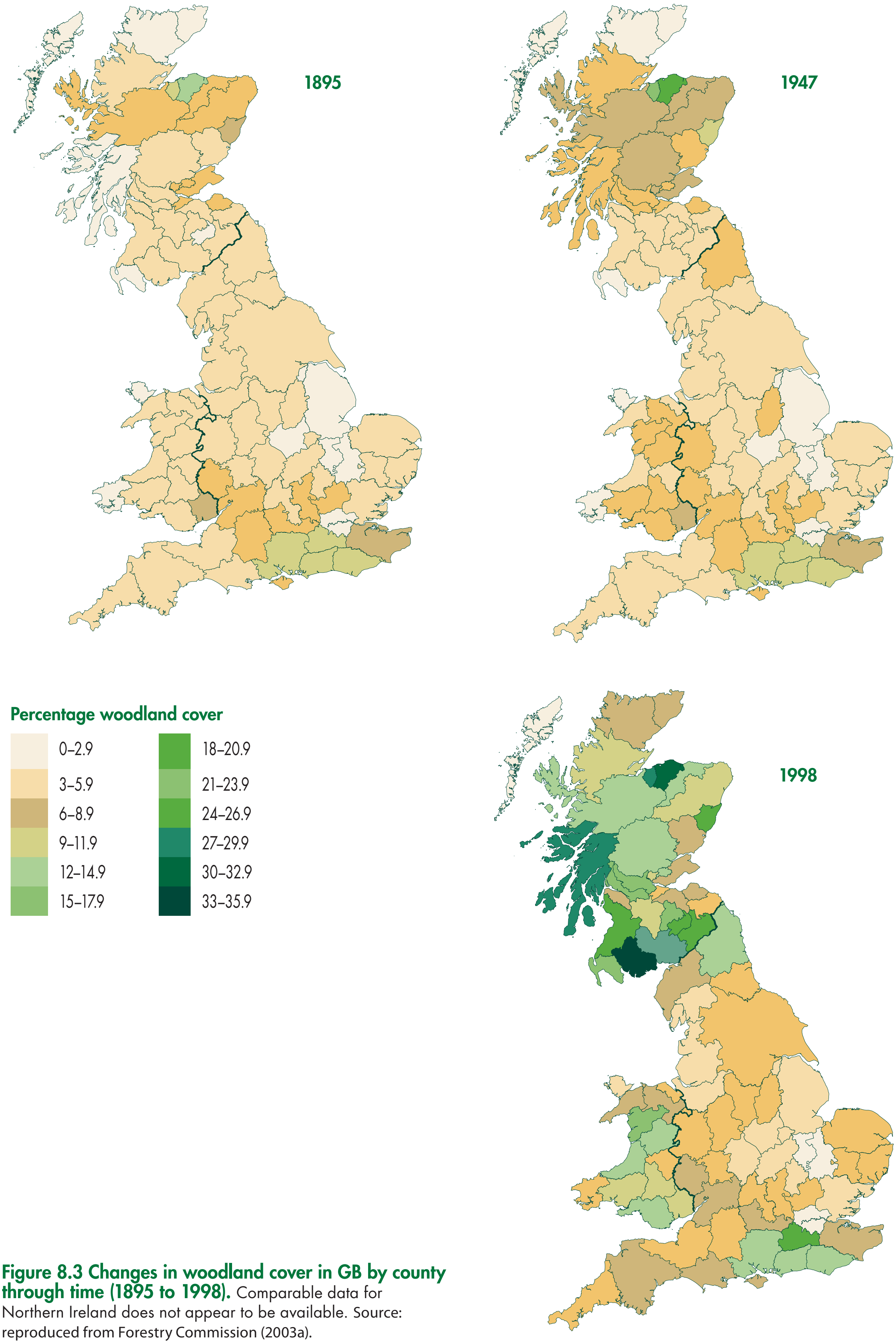

Figure 8.3 Changes in woodland cover in GB by county through time (1895 to 1998). Comparable data for Northern Ireland does not appear to be available. Source: reproduced from Forestry Commission (2003a).

Table 8.5 New woodland creation ('000 ha); five year totals. Source: Forestry Commission (2009a).

		Five year period ending 31 March							
		1976	1981	1986	1991	1996	2001	2006	2009*
England	Conifer	18.3	7.0	5.3	3.9	3.2	3.2	13.0	5.7
	Broadleaves	2.4	1.5	2.3	9.2	21.5	21.2	27.8	11.6
	Total	**20.7**	**8.5**	**7.5**	**13.1**	**24.7**	**24.4**	**40.8**	**17.3**
Scotland	Conifer	148.6	90.9	100.1	94.6	38.3	27.1	11.5	3.9
	Broadleaves	0.6	0.8	0.9	9.2	21.0	28.5	19.7	10.3
	Total	**149.3**	**91.7**	**100.9**	**103.8**	**59.3**	**55.6**	**31.2**	**14.2**
Wales	Conifer	12.9	6.8	5.6	3	0.5	0.7	0.0	0.0
	Broadleaves	0.1	0.2	0.3	1.1	2.0	2.1	1.9	0.7
	Total	**12.9**	**6.9**	**5.9**	**4.1**	**2.5**	**2.7**	**1.9**	**0.7**
Northern Ireland	Conifer	5.0	4.3	3.4	4.4	3.9	2.1	0.5	0.1
	Broadleaves	0.1	0.3	0.4	1.0	1.4	1.5	2.2	1.3
	Total	5.1	4.6	3.8	5.4	5.3	3.6	2.7	1.4
UK	Conifer	184.7	108.9	114.3	105.8	45.9	33.0	25.0	9.7
	Broadleaves	3.2	2.7	3.8	20.4	45.9	53.3	51.6	23.9
	Total	**188.0**	**111.7**	**118.2**	**126.3**	**91.8**	**86.4**	**76.6**	**33.6**

* Three year total to and including 2009.

Table 8.6 Estimates of the woodland habitat area ('000 ha) and percentage of land area in the UK from 1998 to 2007 and in GB from 1984 to 2007. Arrows denote significant change (p<0.05) in the direction shown. Note that because of changes in definitions that have been applied retrospectively, the estimates from 1990 and more especially 1984 are not in all cases directly comparable with later surveys. # denotes data not available. Source: reproduced from Carey *et al.* (2008). Countryside Survey data owned by NERC – Centre for Ecology & Hydrology.

a) Broadleaved, mixed and yew woodland	1984		1990		1998		2007		Direction of significant changes 1998–2007
	Area ('000 ha)	%	Area ('000 ha)	%	Area ('000 ha)	%	Area ('000 ha)	%	
GB	1,317	5.6	1,343	5.8	1,328	5.7	1,406	6.0	↑
England	#	#	887	6.7	927	7.0	981	7.4	↑
Scotland	#	#	284	3.5	229	2.9	251	3.1	↑
Wales	#	#	173	8.2	172	8.1	174	8.2	
Northern Ireland	#	#	#	#	64	4.5	82	5.8	↑
UK	#	#	#	#	1,392	5.6	1,488	6.0	

b) Coniferous woodland	1984		1990		1998		2007		Direction of significant changes 1998–2007
	Area ('000 ha)	%	Area ('000 ha)	%	Area ('000 ha)	%	Area ('000 ha)	%	
GB	1,243	5.3	1,239	5.3	1,386	5.9	1,319	5.7	
England	#	#	241	1.8	260	2.0	257	1.9	
Scotland	#	#	913	11.4	1,030	12.9	956	11.9	↓
Wales	#	#	85	4.0	96	4.5	106	5.0	
Northern Ireland	#	#	#	#	62	4.4	61	4.3	
UK	#	#	#	#	1,448	5.9	1,380	5.6	

Table 8.7 Summary of trends for Priority woodland habitats as reported in the 2008 reporting round.
Source: Biodiversity Action Reporting System (2008).

Priority woodland type	Extent is
Lowland mixed deciduous	Increasing
Lowland beech and yew	Fluctuating, probably increasing
Wet woodland	Report not available
Upland mixed ash	Increasing
Upland oak	Increasing
Upland birch	Report not available
Native pinewood	Increasing

contribution to linking fragments of woodland, and appear to be successful compared to standard planting grants (Quine & Watts 2009). Recently, a landscape connectivity indicator has been developed (Watts & Handley 2010) and applied to Countryside Survey 2007 data for the purposes of biodiversity reporting. Preliminary results indicate regional changes, but overall, still low values in connectivity.

8.2.4.2 Trends in woodland condition

While there is increasing appreciation of the value of woodland for biodiversity, there are concerns as to its condition overall (**Box 8.2**) as reflected in:

- Threats and issues noted in returns under the 2008 reporting round of the Biodiversity Action Plan (UK BAP 2008);
- Threats recorded in the 2005 report on the state of UK protected sites (Williams 2006);
- The UK's submission on the degree to which Annex 1 Habitats are achieving Favourable Conservation Status (JNCC 2007);
- Results from monitoring of various woodland species groups (**Table 8.8**).

The trends identified are likely to continue to be significant in the short- to medium-term. In the longer-term, changes in woodland assemblages as a consequence of climate change will become more important and there is likely to be re-sorting, not just of the plant communities (Keith *et al.* 2009), but of faunal groups as well.

The key threats to semi-natural woodland identified from these activities are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. In addition, more localised pressures include losses to built development (including quarries), inappropriate game management, recreational pressures and drainage or water quality issues. In the long-term, species and assemblages will also be affected by climate change.

Most surveys of woodland condition focus on the semi-natural component of our woodland because, despite its limited extent, it remains one of our richest habitats, with a rich association of rare and priority species. For example, about a quarter of the SSSIs in England, and about one third in Scotland, include woodland; there are 10 Annex 1 types listed under the European Habitats and Species Directive (www.jncc.gov.uk/ProtectedSites/SACselection/SAC_habitats.asp) and eight types included in the UK BAP Priority Habitats List (www.ukbap.org.uk). Woodland Condition Assessment for biodiversity has been developed largely as part of the SSSI Common Standards Monitoring process (Kirby *et al.* 2002; Williams 2006). Woods are assessed in terms of their extent (Section 8.2.4.1) and four attributes: Structure and Natural Processes, Tree and Shrub Composition; Regeneration Potential and Quality Indicators (changes in species). Summary reports for SSSIs in Scotland have been published and will be published soon for England (Kirby *et al.* 2010a; Mackey & Mudge 2010; Natural England in prep, SNH 2010). These show that around 65% of woodland features on Scottish SSSIs are either in Favourable Condition or are Recovering (appropriate management is in place to bring the site into Favourable Condition in future); while in England, the corresponding figure is more than 90%. The positive trend in SSSI condition over the last decade reflects the considerable effort that has been put in to securing positive management, which has not always been the case in non-SSSI woodland, particularly semi-natural woodland. Hence, the indications are that the condition of non-SSSI woodland is generally worse, but surveys currently underway (Scottish Native Woodland Survey; Defra survey of non-SSSI woodland condition) will allow this to be more accurately assessed.

Where non-native coniferous plantations are included in condition surveys, they are often viewed in a negative light because the priority is to restore them to open habitats, such as heathland, or to native tree species. However, there is also increasing interest in the contribution that coniferous plantations created over the last century can make to future woodland biodiversity, not as substitutes for the open ground habitats they replaced or for native woodland, but as new cultural landscapes in their own right (Humphrey *et al.* 2003; Quine & Humphrey 2010). This aspect is not routinely addressed by current monitoring, but could become more important in the future.

8.2.4.3 Changing composition, structure and species

Changes that have been taking place in varying degrees to woodland processes and structure across the UK are discussed here (**Box 8.2**). Overall, while the tendency is towards an increase in structural diversity, there are specific issues in particular areas and woodland types requiring action.

Shifts in the abundance of different tree and shrub species have occurred through the selective clearance of forests, the favouring of particular species, either directly or indirectly, through the management system, and the spread of introduced species both through planting and self-seeding. The abandonment of past forms of management, and responses to climate change (Section 8.2.3), have been associated with changes to the composition of semi-natural woodland, generally towards more mixtures. For example, over the past 40 years, there has been an increase of ash in beech woodland, birch in oak woods, oak in northern birch woods, and the spread of holly in the understorey (Kirby *et al.* 2005; Kirby *et al.* 2009; Amar *et al.* 2010). Grey squirrels (*Sciurus carolinensis*) are likely to reduce the competitive potential of beech in woods where previously it has been dominant.

The effects on woodland biodiversity (i.e. ignoring the declines in open habitat biodiversity through plantations on open ground) from this changing composition can be summarised as follows:

1. Increased abundance of conifer specialists, such as siskin (*Carduelis spinus*), responding to expanding forest area (Baillie *et al.* 2009); and increased woodland generalists (both plants and animals) capable of tolerating deep shade or found in the open habitats in forests (such as rides). The latter tend to already be widespread species, which have seen little change in their range. There are also indications that distinctive assemblages and species are building up in plantations, particularly as many are

now entering their second rotation (Humphrey *et al.* 2003; Quine & Humphrey 2010), and in future, their composition may become more varied (Mason *et al.* 2009). Green and great-spotted woodpeckers (*Picus viridis* and *Dendrocopos major* respectively) have increased rapidly (Baillie *et al.* 2009) and may be benefiting from the maturation of new forests, an increase in standing deadwood due to Dutch Elm Disease and self-shading in unthinned woodlands, and from the increasing provision of winter food in gardens.

2. Loss of habitat area for broadleaved woodland specialists or those of lightly shaded conditions because of the conversion of broadleaved woodland to conifers after 1945 (**Table 8.8**).
3. Since 1986 and the introduction of HAPs, in particular, there has been increasing restoration of plantations to native species on ancient woodland sites (Thompson *et al.* 2003; Goldberg 2003). There are few accounts of the accompanying changes in woodland communities in restored sites (Kirby & May 1989; Harmer & Kiewitt 2007), but a major study carried out for the Woodland Trust is to be published soon (Tim Hodges pers. comm.).

The impact of climate change on regeneration may mean that the classification of 'native', either at the species or provenance level, will need to be reconsidered; species currently restricted to southern Britain may be accepted further north, along with species from the near continent (such as sycamore, *Acer pseudoplatanus*) that are currently often treated as undesirable elements of semi-natural woods from a biodiversity perspective (Wesche *et al.* 2006; Kirby 2009; Kirby *et al.* 2009).

Traditional broadleaved woodland management in England tended to rely on vegetative regrowth from coppice stools and pollard. As these systems have declined, there has been increased interest in natural regeneration of broadleaves from seed (Evans 1988; Harmer *et al.* 2010). Similar concerns have arisen in discussions of the native pinewoods (although their past management was different), with the added factor of the possible role of fire in site preparation (Mason *et al.* 2004; Summers & Wilkinson 2008; Hancock *et al.* 2009). Afforestation in the 20th Century was necessarily based around planting. While the restocking of most of these plantations is likely to continue to be by planting, there is increasing interest in natural regeneration as an alternative (Mason *et al.* 2009).

The major uncertainty with achieving regeneration lies with the levels of wild herbivores, mainly deer, but also grey squirrels and, locally, rabbits (Gill 1992a, 1992b; Section 8.2.5.4). Deer densities have long been seen as a problem for achieving natural regeneration in Scotland (Staines & Welch 1989), but for at least the last 30 years (and probably substantially longer), deer populations and distributions have spread in England and Wales (Ward 2005) and now constitute a major limitation on maintaining or restoring coppice, as well as natural regeneration from seed (Fuller & Gill 2001).

Good practice in deer management across landscapes is promoted via the Deer Commission for Scotland (now part of Scottish Natural Heritage) and the Deer Initiative in England and Wales, but achieving effective collaborative management across multiple ownerships remains challenging (Phillip *et al.* 2009).

Systematic monitoring of woodland species as quality indicators in recent years is best documented for the six taxonomic groups summarised in **Table 8.8**. In general, woodland birds appear to be declining, together with some mammals such as the red squirrel (*Sciurus vulgaris*). In particular, specialist woodland bird species have shown a decline since 1970, but a modest recovery has been noted in recent years (Defra 2009a; **Figure 8.4**), along with regional variation with, for example, largely positive trends in the index for Scottish woodland birds (Eglington & Noble 2010). For the other groups, there are gains and losses, but there is a lack of information for many taxa. Despite the importance of the relationship between trees and mycorrhizal fungi, there is no routine assessment of their status or condition, although methods are currently being explored (Mueller *et al.* 2004; Feest *et al.* 2010).

Box 8.2 Summary of changes to woodlands.

- The broadleaved resource has aged following the abandonment of coppicing (notably in southern Britain) and limited thinning in the last 60 years (Kirby *et al.* 2005; Amar *et al.* 2006; Hewson *et al.* 2007; Mason 2007). This has contributed to a tendency towards increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby *et al.* 1998).
- Maturing coniferous forests are showing increased structural diversity in, and have also been impacted by the deliberate restructuring of plantations through smaller felling coupes, and the identification of areas to be kept as open or broadleaved along stream corridors (see Kielder case study in Section 8.5).
- There is an increasing interest in alternatives to clear-fell silviculture; for example, a target was set to transform 50% of state forests in Wales from clear felling to continuous cover by 2012 (Mason 2007).
- Windthrow (Quine & Gardiner 2006), localised forest dieback due to disease, and the intrusion of non-crop species (Humphrey *et al.* 1998) have all had a marked effect.
- Changing levels of grazing within woodland are influencing structure. Typically, high-levels of stock- or deer-browsing are leading to a reduced understorey (Fuller & Gill 2001) and, in due course, to limited recruitment of new canopy trees. However, in local areas, the converse also occurs, with reduced grazing and browsing by stock leading to a dense understorey that may shade out ground flora and lichens low-down on tree trunks, and compete with veteran trees (Read 2000).
- There is an increasing 'generation gap', whereby sites with many veteran trees frequently lack mature and younger generations to replace them; in hedges with mature trees, there are seldom younger ones coming along. The apparently increasing threats (Broadmeadow *et al.* 2009) to mature trees from disease (Dutch Elm Disease, Alder dieback, Ash dieback, various syndromes affecting oak, new strains of *Phytophthora* affecting a broad range of trees, etc.) makes a lack of replacement trees even more acute.
- New types of woodland (agro-forestry, short-rotation forestry and energy crops grown as short rotation coppice) may add a new type of structural pattern to rural landscapes.

8.2.4.4 Tree and woodland health

Pests, pathogens, climate and other events which cause tree death play an important role in the dynamics of woodland ecosystems; dead and decaying wood provides important micro-habitats, dying trees allow more light and warmth to reach the forest floor, and nutrient cycles depend upon turnover of biomass (Kirby *et al.* 2010b). However, rapid, widespread tree death can threaten provision of ecosystem services such as production (effects on tree growth), visual amenity (loss of cherished views) and nature conservation (loss of rare species and valued habitats).

There have been issues around woodland health throughout the 20th Century, but in the past decade, new threats have raised the level of concern. During the 1950s, air pollution was a focus for concern regarding tree health, particularly in proximity to large industrial areas in the Midlands and northern England. In the 1980s, attention shifted to the threat of acid rain to trees (as well as water quality) in upland areas (Section 8.2.5). However, annual surveys of crown condition of five forest species, undertaken from 1984 to 2005 (Binns *et al.* 1985; Hendry *et al.* 2005), showed fluctuating canopy condition in response to climate conditions and impact of insects, such as the green spruce aphid (*Elatobium abietinum*), but no major overall deterioration in health.

During the latter half of the 20th Century, the threat to woodland health and the productivity of commercial woodland, in particular, from a range of endemic insects and fungi, resulted in the development of management techniques,

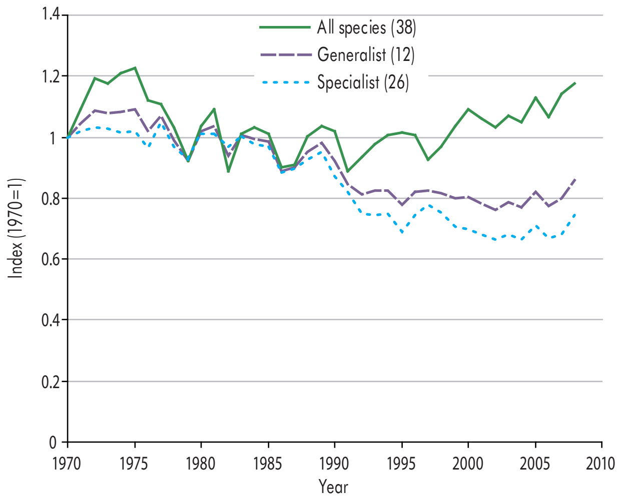

Figure 8.4 UK woodland bird index trends, 1970 to 2008. Figures in brackets show the number of species. Source: data from RSPB/BTO/Defra.

Table 8.8 Examples of changes in status of woodland species.

Vascular plants	Woodland vascular plants have generally shown relatively little change in overall distribution compared with those from other habitats (Preston *et al.* 2002; Braithwaite *et al.* 2006). Within broadleaved woodland, various studies indicate reductions in species-richness and shifts towards more competitive species at different scales (Kirby *et al.* 2005; Keith *et al.* 2009; Carey *et al.* 2008). The following factors appear to be working across a number of landscapes to cause these changes: increasing shade, increasing eutrophication, increasing browsing and grazing.
Lower plants	Relatively little is known about recent trends in lower plants and fungi—apparent changes in distribution and abundance may reflect differences in survey effort rather than real change. However, reductions in air pollution, in particular sulphur, appear to be associated with the recovery and spread of some lichen species (RoTAP 2011).
Woodland birds	There have been increases in birds, such as goshawk (*Accipiter gentilis*) and crossbill (*Loxia curvirostra*), which are associated with large conifer forests in particular. Within broadleaved woodland, eight out of a total of 35 species showed large national declines (>25%) whereas 11 showed large national increases (>25%). All long-distance migrants have declined, whereas the two medium-distance migrants, blackcap (*Sylvia atricapilla*) and chiffchaff (*Phylloscopus collybita*), have increased. Common generalist species, such as blue tit (*Cyanistes caeruleus*) and great spotted woodpecker, have fared better than more specialised and less common species, such as willow tit (*Poecile montanus*) and lesser spotted woodpecker (*Dendrocops minor*). The reasons for the different changes are complex, but in part, related to changing woodland structures (Amar *et al.* 2007; Quine *et al.* 2007; Hewson & Noble 2009).
Lepidoptera	Amongst Lepidoptera there are two clear trends. One is a response to changing woodland structures: of six butterfly species associated with clearings in woodlands, three have shown marked declines, of which, the high brown fritillary (*Argynnis adippe*) showed a 77% decline from 1970 to 1982 (Asher *et al.* 2001). By contrast Hambler and Speight (1995) have argued that leaf-mining Lepidoptera have benefitted from the increase in high forest canopies; lichen-feeding species have also shown increases (Fox *et al.* 2006). However, in practice, many of our moth species are declining, with a decrease of almost a third in the species-richness index for Rothamsted light traps from 1968 to 2002. This decline includes tree-feeding species such as the dusky thorn (*Ennomos fuscantaria*). In parallel, Lepidoptera as well as some other invertebrates may be starting to respond to climate change: there has been considerable northward range expansion by the speckled wood butterfly (*Pararge aegeria*; Hill *et al.* 1999); more localised distribution responses may be shown through species changing their habitat preferences. Thus, butterflies which currently use 'hotspots' in glades may become more common under canopy shade, just as they are in southern Europe (Thomas 1991).
Other invertebrates	The increasing proportion of closed-canopy forests may benefit some canopy species (Hambler & Speight 1995). Deadwood species should benefit from the increases in deadwood reported (Winter 1993; Kirby *et al.* 2005; Amar *et al.* 2006). However, specialist saproxylic species tend to be poor colonists so may be in decline where the veteran trees on which they depend are under threat (Warren & Key 1991).
Mammals	Some mammals have increased over the last 60 years, notably deer in southern Britain (Ward 2005), and grey squirrels and badgers (*Meles meles*); while others have declined, such as red squirrels and dormice (*Muscardinus avellanarius*). For some, such as pine martens (*Martes martes*), yellow-necked mice (*Apodenius flavicollis*) and some bats, there is insufficient evidence to be certain as to the long-term trend (Battersby 2005). While the potential habitat cover for woodland mammals is likely to increase further as forest expansion continues, individual species may be limited by factors such as disease (red squirrel), poor dispersal (dormouse), lack of suitable roost sites (some bats).

such as the treatment of stumps to prevent colonisation by the fungus *Heterobasidion annosum* (Rishbeth 1952; Redfern *et al.* 2001; Redfern *et al.* 2010), and the chemical treatment of planting stock against the weevil *Hylobius* species (Moore *et al.* 2004; Wainhouse *et al.* 2007). Localised outbreaks of particular insect pests were controlled by aerial application of chemicals or the introduction of biocontrol agents; examples include pine beauty moth (*Panolis flammea*) outbreaks on lodgepole pine (*Pinus contorta*) stands in north Scotland (Hicks *et al.* 2001) and the control of the introduced bark beetle (*Dendroctonus micans*) (Gilbert *et al.* 2003) through the introduction of a biological predator (*Rhizophagous grandis*). Some health declines were noted to reflect combined climatic and pest/pathogen stress, or combined insect and pathogen attack (Redfern *et al.* 1987). A number of pests and pathogens present in Europe (including *Ips typographus*) were seen as potential threats to woodland health, and measures such as import controls and port inspections were established to prevent their arrival. The pandemic of Dutch Elm Disease, involving the fungus *Ophiostoma novo-ulmi*, was perhaps unusual because it largely affected non-woodland trees (Chapter 7).

Recently the focus of concern has switched to damaging, or potentially damaging, invasions of newly introduced or newly occurring pests and pathogens. Examples of newly introduced insect pests from Europe include the pine-tree lappet moth (*Dendrolimus pini*), a defoliator of Scots pine, and the oak processionary moth (*Thaumetopoea processionea*) which defoliates oak and can be a threat to human health. Introductions from further afield include the Asian longhorn beetle (*Anoplophora glabripennis*) and emerald ash borer (*Agrilus planipennis*), both of which pose threats to native broadleaved trees in urban and rural settings. A number of serious pathogens have also appeared in the last 20 years, including *Phytophthora ramorum* which has been responsible for widespread oak death in the USA and a bacteria (*Pseudomonas syringae pv. aesculi*), implicated in a disease of horse chestnut (Green *et al.* 2010). A bacteria, together with other factors not yet fully understood, may be responsible for Acute Oak Decline (Denman & Webber 2009; Denman *et al.* 2010). There is evidence of organisms, such as *Phytophthora*, switching host species in an unpredictable way, so that each invasive pathogen represents an uncontrolled, open-ended experiment in evolution (Brasier 2008; Brasier & Webber 2010). Recent widespread death of Japanese larch (*Larix kaempferi*) from *Phytophthora ramorum* (Brasier & Webber 2010) illustrates the dynamic nature of the threat and the scale of possible impact to woodland health, productivity and visual amenity. There is evidence of Red Band Needle Blight (*Dothistroma septosporum)* affecting native Scots pine trees, after initially (from the 1990s) being a problem on Corsican pine and then lodgepole pine. In late 2010, *Phytophtora lateralis* was found in the UK for the first time, killing Lawson's cypress trees in a country park near Glasgow. Understanding of the routes of entry and potential impact is developing rapidly through application of molecular techniques. It is clear that a number of the introductions have pathways which can be traced back to Asia, and there is growing concern about the risks posed by the global trade in large plants and plant material, and the challenges of regulating unknown organisms (Brasier & Webber 2010).

There is also evidence that woodland health does decline, both locally and regionally, through the additional contribution of climatic stress (Tubby & Webber 2010): for example, through drought lowering tree resistance (Green & Ray 2009; Gregory & Redfern 1998) or wetter springs aiding fungal sporulation (Brown & Webber 2008). Such impacts may well increase in future, with climate change predictions suggesting significant drought stress, especially in the south and east (Broadmeadow *et al.* 2009; Ray *et al.* 2010). In addition, the further arrival and establishment of exotic pests and pathogens may well threaten valued woodland habitats, as well as production from managed woodlands. This would lead to woodland health becoming more than a local issue within productive plantations (or an intrinsic part of the stand dynamics of semi-natural woodlands), but one with landscape-scale effects on a range of ecosystem services.

8.2.5 Drivers of Change in Woodlands

A broad definition of 'drivers of change' is: "any natural or human-induced factor that directly or indirectly causes a change in an ecosystem" (MA 2003). The UK NEA has adopted this definition and a modified classification of direct and indirect drivers (Chapter 3). Many drivers act synergistically, and across a range of scales, so that the individual effects can be hard to distinguish. The main drivers affecting forests and woodlands in the UK are climate change, pollution and land-use practice (directly via competition with other land uses, or indirectly through socio-political, demographic and economic drivers). In addition, the particular age structure of woodlands (and the legacy of long-past events, policies and management decisions) is an endogenous driver that is bringing about spontaneous change in provision of goods and services.

Some drivers of change have particular relevance to semi-natural woodland compared with other woodland. **Table 8.9** summarises the direction and magnitude of the main drivers of change in forest and woodland biodiversity. We attempt to summarise the influence of the drivers on extent and on characteristics that govern goods/service provision. This also gives a clear indication of which effects are more complex and where the main knowledge gaps exist.

8.2.5.1 Climate change

Climate has an important influence in shaping the composition and character of woodlands within the UK, with regional forest types reflecting major spatial patterns in climate; palaeoecological studies indicate how such patterns changed as the climate improved after the last glaciations. The main climatic factors affecting tree-growth in forests and woodlands (Pyatt *et al.* 2001) are:

1. Temperature—growing season temperature affects tree-growth; winter/spring temperature affects the degree of frost damage; and range of temperature (continentality) is also influential;
2. Moisture deficit—different tree species differ in their seasonal moisture requirements and drought tolerance; snowfall can physically damage trees or protect them from winter desiccation, especially when small;

3. Wind—can cause physical damage and affect tree-growth form; combinations of wind and temperature also drive tree-line and forest-line limits.

These give rise, in combination with the effects of geology and soil type, to strong spatial patterns in tree suitability and productive potential, as shown for pedunculate oak (*Quercus robur*) and Sitka spruce (**Figure 8.5**).

Change, rather than simply variability, in these factors will be an influential driver of extent and condition. Climate may operate at different stages of the life of trees, and so, particular events may differ in their effects on tree establishment versus mature trees; this is an important distinction to make as it affects the pattern and speed of response to climatic change (Kirby *et al.* 2009).

There is only limited evidence of major climate-related change in the composition of UK forest and woodlands in recent years (Kirby *et al.* 2005). This is partly because most tree species are long-lived, adapted to cope with considerable climate variability and relatively resilient to the small changes in climate that have happened in recent years (Broadmeadow *et al.* 2009). Thus, the total extent and diversity appear to have been little altered by climate drivers. However, recent climate-related changes have been documented in more mobile species, such as insects and birds from woodland and other habitats (Section 8.2), and these variations are predicted to increase, leading to more rapid changes in the complement of species using woodland areas. Increasing summer temperatures have led to faster tree-growth in some areas (Cannell 2002; Broadmeadow *et al.* 2009), but drought has had the opposite effect; increased windiness and storm frequency (Broadmeadow *et al.* 2009), and increased winter wetness, could also affect tree survival, rooting and slope stability (Ray 2008).

Advances in leafing date in response to increasing spring temperatures have been recorded for some tree and ground-layer plant species (Broadmeadow *et al.* 2009), but this has probably also increased the prevalence of late spring frost damage. Changes in phenology may be linked with knock-on effects on associated flora and fauna, and to declines if such species are unable to adapt to changes. In some cases, such adaptation has happened; for example, in Wytham Woods, blue tits and great tits are breeding about two weeks earlier than they did in the 1980s (Perrins & Gosler 2010). However,

Table 8.9 Summary of main driver effects (magnitude and direction) identified as important for forests and woodland. Main effects to date are indicated together with predictions about their future importance. Arrow indicates positive and/or negative effect; multiple arrows indicate strong magnitude and brackets indicate lesser magnitude; = denotes no major effect; ? denotes little is known.

UK NEA Driver		Effect on: total area		Effect on: tree growth and yield		Effect on: service provision and woodland condition (including biodiversity)	
		Past (since 1945)	Future (to 2060)	Past (since 1945)	Future (to 2060)	Past (since 1945)	Future (to 2060)
DIRECT							
Land use / cover	Grazing (livestock)	=	↓↑	↓	↓ =	↓ (↑)	↓↑
	Grazing (wild herbivores)	↓	↓↑	↓	↓ =	↓ (↑)	↓↑
	Afforestation policies	↑↑	↑	↑=	↑ =	↓ (↑)	↑ (↓)
	Biodiversity policies	(↑)	↑	=	=	(↑)	↑
	Agricultural policies	↓ (↓)	↓↑	n/a	n/a	↓	↓↑
Introduced & invasive species	Species introduction / removal	=	↓ =	=	↓	↓	↓↓
Pollution	Nitrogen deposition	=	=	↓↑	?	↓	↓?
	Sulphur deposition	=	=	(↓)	=	?	?
	Ozone	=	=	↓	↓	?	?
Overexploitation	Harvest / resource consumption	↓↓	↓↑	=	=	↓↓	↓↑
Climate change	Temperature	=	↓↑	↑	↑	↓↑	↓↑
	Drought	=	↓↓	↓	↓↓	↓ ?	↓ ?
	Windiness	=	=	Form: ↓	Form: ↓	↑ ?	↑ ?
Other	Age structure of woodlands	=	=	↑	↑	↑	↓↑
INDIRECT							
Demographic	Population growth	↓	↓	=	=	↓	↓↑
	Demographic change	↓↑	↓↑	=	=	↑	↑
	Ethnicity	?	?	?	?	?	?
	Migration	?	?	?	?	?	?
Economic	Market forces	↑	↑	↑	↑	↓↑	↓↑
Socio-political	Government subsidies	↓↑	↑	↓↑	↑	↓↑	↓↑
	Legislation	↓↑	↑	↓↑	↑	↓↑	↓↑
Technology adaptation		↓	=	↑	↑	↓↑	↓

there might come a point when insects may emerge when their food sources are unavailable, vernal plant species may emerge when forest canopies have already closed, and chicks may hatch when the peak invertebrate abundance has passed. These indirect effects of climate change may have substantial effects upon woodland biodiversity as a consequence of reduced habitat specialisation when the temporal synchrony between hosts and herbivores breaks down. If generalist species become more common, there may be increased homogeneity between habitats; there are already indications of this from data on long-term plant species compositional change in UK woodlands and alpine habitats (Keith *et al.* 2009; Britton *et al.* 2009).

It is possible that both semi-natural and planted forests will be able to grow to higher altitudes than they have previously. This is highly dependent on: (i) the balance between increasing temperature and increasing windiness/storminess; and (ii) the degree of invasiveness of the vegetation occupying the areas above the current tree-limit (Hester & Brooker 2007; Broadmeadow *et al.* 2009).

Impacts of climate change are predicted to increase in future, under all scenarios of change. Some changes are already being made in forest and woodland management plans, such as recommendations on the 'most suitable' species for planting in different areas (Ray, 2008; Broadmeadow *et al.* 2009; Quine & Ray 2010). Forest management requires long-term planning appropriate to the long-lived nature of the main species. Much activity is, therefore, currently focused on the best way for the forest industry to 'prepare' for future change (Broadmeadow & Ray 2005; Freer-Smith *et al.* 2007; Ray *et al.* 2008, 2010). The impact of increased frequency and severity of summer droughts upon forest fires has yet to be firmly established, but past drought years (e.g. 1994–1995) have seen increases in wildfires.

The scope for woodlands to contribute to the mitigation of, and adaptation to, climate change is discussed in Section 8.3.3.

8.2.5.2 Pollution

Aerial deposition of pollutants, originating from fossil fuel combustion and food production, has affected air quality and rainfall chemistry across the UK (NEGTAP 2001; RoTAP 2011). There is some evidence of air pollution impacts on growth and composition of UK forests and woodlands, although reports from surveys of commercial forests

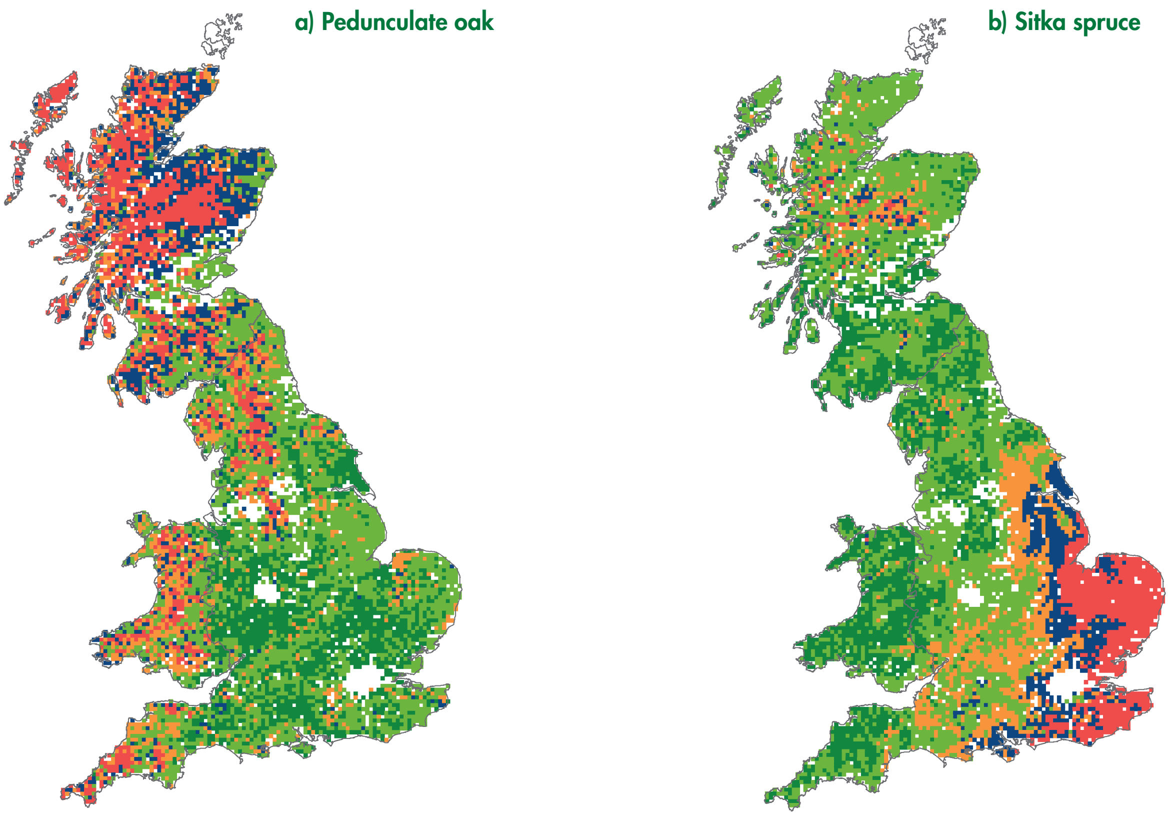

Figure 8.5 Spatial variability in productivity of pedunculate oak and Sitka spruce. The 'suitability' (defined as productivity relative to maximum productivity achievable by that species under current climatic conditions) for a) pedunculate oak, and b) Sitka spruce under Baseline (1961–1990) scenarios. The results are based upon Ecological Site Classification. Dark green = very suitable (>70% of current maximum productivity); light green = suitable (50–70% of maximum productivity); orange = marginal (40–50% of maximum productivity); blue = poor (30–40% of maximum productivity); red = unsuitable (<30% of current maximum productivity). Source: reproduced from Read *et al.* (2009).

indicate 'no widespread forest damage' overall (e.g. Defra-funded UKREATE programme: www.bangor.ceh.ac.uk/terrestrial-umbrella/). Recent policy-driven reductions in nitrogen oxides and sulphur have been recorded, and these have had knock-on effects on soils, waters and foliar sulphur concentrations. However, nitrogen deposition, and ozone levels in particular, are still above 'critical loads' for some habitats—for example, it has been estimated the critical loads for ground flora and epiphytic lichens is exceeded in almost 100% of UK Atlantic oak woodlands (RoTAP 2011). In a few woodlands, however, there is some evidence of recovery in woodland lichens (Section 8.2 & 8.3). Past concerns about acidification of afforested catchments led to the development of the Forests and Water Guidelines (Forestry Commission 2003b).

Nitrogen deposition increases the growth rates of some forest trees, but is known to have strong detrimental effects on many lower plants. Nitrogen deposition is also cited as a major factor in the overall decline in fungal diversity (especially mycorrhizal diversity) in Europe. Many species show a very rapid response to nitrogen deposition—some decline and others increase—but there is no uniform response and the reasons for the different responses are not yet fully understood (Nygren *et al.* 2008). Nitrogen deposition has been implicated in recent increases in cover of nitrogen-demanding ground-layer species and reductions in overall species diversity in, for example, UK broadleaved woodlands (Haines-Young *et al.* 2003; Kirby *et al.* 2005; RoTAP 2011; **Table 8.8**). So far, increases in carbon dioxide concentrations are predicted to have had relatively little effect on forests as compared to nitrogen deposition (van Oijen *et al.* 2008; RoTAP 2011); future increases in carbon dioxide levels are predicted to interact with nitrogen deposition to favour faster-growing species (Jarvis *et al.* 2009). Conversely, cumulative ozone concentrations are apparently reducing both forest growth and carbon sequestration across Europe (RoTAP 2011). Furthermore, faster growth following nitrogen addition is likely to increase shoot vulnerability to winter damage or other stressors, as found for some woody shrubs (Power *et al.* 1998; Power *et al.* 2004). Therefore, predicting the nature and importance of both current and future changes in pollutant deposition is complex, difficult and highly dependent on changes in other drivers, particularly climate (RoTAP 2011). Aerial pollutant experimentation is difficult in forest habitats, so most current pollutant data from forests are derived from spatial and temporal correlations between forest survey and deposition data (which makes direct allocation of causality impossible), with limited data from semi-natural woodlands (Kennedy & Pitman 2004; Vanguelova *et al.* 2007). Most manipulative experiments on pollutant effects in the UK have been carried out in open-ground habitats and so, are not directly applicable to forested habitats (RoTAP 2011).

There is currently limited evidence of the impact of point source pollution from smelters and livestock facilities. However, interest in the latter is increasing because changes in the regulatory regime now require this impact to be assessed.

The scope for woodlands to contribute to the capture and immobilisation of pollutants is discussed in Section 8.3.3.3.

8.2.5.3 Land Use

Land use change has undoubtedly caused substantial changes in forests and woodlands in the UK, but it involves many different 'drivers', both direct and indirect, including human behaviour and values, policy and economic drivers, and climate. The major expansion of woodland area since 1945 was driven by a perceived need to increase strategic timber reserves (Section 8.1) and domestic timber supply, and was achieved with government support in the form of grants and favourable tax regimes, and direct investment in public forests. Policy priorities have evolved over time to include rural employment, support for a domestic wood-using industry and, most recently, a broad mix of 'public good' targeting environmental quality and social value (Rollinson 2003). Longer-term drivers, influential in the decline of UK semi-natural woodland both in terms of area and composition, include felling and free-range livestock grazing (Birks 1988; Mitchell & Kirby 1990; Hester *et al.* 1998).

Economic forces partially underlie several of the relevant land use drivers. In particular, the domestic markets for wood products are affected by factors such as global forestry demands, forest cover in other countries, import/export costs and exchange rates, and food/fibre 'security' issues (Thomson 2004). UK- and European-level policies directly and indirectly affect many of these factors—some of the most important indirect effects act by driving other 'competing' land uses, such as livestock management (and associated subsidies) and EU targets for using biomass as a source of renewable energy. Road-building and expansion of settlements have also contributed to woodland loss and fragmentation across the UK, although there are measures in place to reduce the rate of such loss; for example, Environmental Impact Assessments for new construction projects and recent deforestation policies requiring compensatory planting (Section 8.2). Technological adaptation has included the adoption of chainsaws, the mechanisation of harvesting and extraction operations, and the development of highly mechanised sawmills. These have influenced the costs of production and the demands from the domestic market; this, in turn, influences the objectives for the management and profitability of woodland enterprises, and the market for woodlands. Harvesting from the public forest estate is required to maintain the viability of the domestic industry, while harvesting from private woods is more responsive to market trends. The extent of coppicing has reduced as labour costs have increased and traditional niche markets declined. However, a trend towards 'getting back to nature' is increasing interest in small-scale workings and the use of natural regeneration, rather than planting, in woodland re-establishment. There are also changes in the ownership of woodlands with a decline in public ownership (FAO 2010) and an increase in private ownership. There have been increases in the amount owned by environmental charities, such as the Woodland Trust, the National Trust and The Wildlife Trusts, and by individual owners for whom production is not a primary concern.

Woodland fragmentation and loss are regarded as having long-term detrimental effects upon biodiversity and other woodland values. This has led to direct action to improve the condition and extent of semi-natural woodland across the UK, particularly for 'biodiversity aims', with national and international obligations to protect and expand a number of woodland habitats and species through HAPs and BAPs (Section 8.2.4). Direct policy drivers causing increases in semi-natural woodland area and condition mainly involve grant-aid—until recently, 'nested' within a range of land use policies such as Environmentally Sensitive Areas (ESAs) and farm woodland schemes. In Scotland, the grant-aid for native woodland expansion and increases in connectivity has recently been brought under the Scotland Rural Development Programme (SRDP). Changes in area of commercial forestry (both public and private) are still mostly driven by economics, although 'multifunctionality' benefits are increasingly being recognised, and the balance of importance is changing towards biodiversity and landscape issues in these forests. Carbon sequestration may become a major driver of UK forest policy in the future (Section 8.3.3), but there is much debate about the costs/benefits of temperate forests versus open-ground habitats for carbon sequestration terms, particularly on highly organic soils (Broadmeadow & Matthews 2003; RoTAP 2011; Lindsay 2010).

Other increasingly important social and demographic drivers of land use change affecting the composition and extent of both woodland and forest include recreation (responding to increased leisure time and mobility, and a greater range of activities, e.g. mountain biking) (Section 8.3.4) and construction activities. There have been substantial developments of facilities in coniferous woodlands to accommodate walks and cycle tracks, and to enhance visual appearance; much of this has been grant-aided for private owners. Use of volunteers to undertake traditional management, such as coppicing, has also increased in popularity, but the benefits to semi-natural woodland structure and composition in some areas may still be hampered by high densities of deer (Fuller & Gill 2001). These activities have significantly changed the diversity and form of forested areas but have not led to major changes in total area across the UK as a whole.

8.2.5.4 Biotic pressures from herbivores, pests, pathogens and invasive species

Despite the recent large-scale reductions in livestock in some areas of the UK, grazing-related impacts have led to significant net losses in overall area and diversity of semi-natural woodland (Section 8.2.4.2) to date, and significant costs in terms of tree/seedling damage in commercial forests. The main exceptions are in the lowlands where areas traditionally managed by livestock, such as common land, are no longer grazed, and scrub encroachment is occurring. There has been a gradual increase in wild herbivores across the UK in recent decades, particularly deer of several species (Staines *et al.* 1995; Ward 2005; Gill 2006). Smaller herbivores like hares, rabbits, small mammals, can also reduce regeneration, but their impacts are usually more local. However, past experience shows that they can become national pests, for example, rabbits prior to 1955.

Grazing and browsing by both wild and domestic herbivores (primarily sheep and deer) is arguably the primary driver of biodiversity change in UK woodlands and forest. These herbivores remove tree regeneration (ultimately leading to loss of total area and fragmentation) and affect both species and structural diversity (Gill 1992a, 1992b, 2006; Broadmeadow *et al.* 2009). Most semi-natural woodlands are not fenced, so have no protection against deer and other wild herbivores, and even commercial plantations, which are normally fenced, have resident populations of these species.

The biggest driver of changes in free-ranging livestock numbers is agricultural policy, but disease can also cause rapid and dramatic variations, as recently illustrated by the response to the Foot and Mouth outbreak of 2001. At a national level, policy drivers include stocking level requirements within ESAs. At a European level (with local modification), the Common Agricultural Policy (CAP) has been a major driver of livestock numbers for some time. The recent CAP reform has led to massive reductions in free-ranging sheep in the uplands of the UK (SAC 2008), which could start to reverse the long-term decline in semi-natural woodland cover and diversity. However, the benefits may be restricted if wild herbivore numbers increase in response to the reduction of stock. Changes in wild herbivore numbers, particularly in deer, are partly human-driven, such as winter-feeding, and partly climate-driven in terms of greater over-winter survival in warmer winters. Policies and legislation related to wild herbivore control (for example, the Wild Deer Act) have had relatively few effects to date (Phillip *et al.* 2009). There are moves, however, to provide greater powers to government agencies (particularly the Deer Commission in Scotland and, to a lesser degree, the Deer Initiative in England and Wales) to cull deer.

Increasing outbreaks of certain insect pests, particularly in commercial plantations, are compounded by winter-warming, but also driven by the increasing volume of global trade in plant products (Broadmeadow *et al.*, 2009); this is also the case for a number of pathogens. For example, some *Phytophthora* species are thought to have been introduced in nursery stock and now represent a major threat to many woody species in the UK (Chavarriaga *et al.* 2007).

The spread of invasive species, such as rhododendron, insect pests and pathogens, has greatly reduced forest and woodland diversity in many parts of western Britain in particular. Grey squirrels, for example, are limiting the growing of quality broadleaved timber (Mayle *et al.* 2009), the survival of the red squirrel, and may affect canopy composition by impacting upon thin-barked tree species such as beech. Feral boar are increasing in number and range, with possible effects on woodland ecology and with local impacts on recreation (e.g. in the Forest of Dean).

8.2.5.5 Ageing of woodland stock

An endogenous driver of change is the ageing of woodlands within the UK. Age determines many characteristics of woodlands, including size and structural diversity (see Section 8.2.2). Very large areas of woodland were cleared in the First and Second World Wars; combined with the regeneration of felled woodland and the new planting triggered by concerns over timber supply, this has produced a particular, unbalanced, age class distribution (Mason 2007). The ageing of these cohorts changes the delivery of, and potential for, goods and services arising from the woodlands. Such changes also impact upon habitat quality (Quine *et al.* 2007) and are thought to be influential in the changes to the woodland bird indicator (Hewson *et al.* 2007; **Table 8.8, Figure 8.4**).

8.3 Ecosystem Goods and Services Provided by Woodlands for Human Well-being

"such forests...are of considerable service to neighbourhoods that verge upon them by furnishing them with peat and turf for their firing; with fuel for the burning of their lime; and with ashes for their grasses; and by maintaining their geese and their stock of young cattle at little or no expense"
(Gilbert White 1789)

The concept of ecosystem services is based on an understanding that sustainable human life depends not just on the raw products that different types of ecosystem produce (such as food and timber), but on a much wider range of goods and services. Many of these are unseen, many go unnoticed, and many are unrewarded.

An early modern reflection on the concept of ecosystem services came from Krutilla (1967) who considered them as: "present and future amenities associated with unspoiled natural environments, for which the market fails to make adequate provision". Other definitions have since been used, including that from the Millennium Ecosystem Assessment (MA): "those processes of ecosystems that support (directly and indirectly) human wellbeing" (MA 2003; Patterson & Coelho 2009). Several typologies of ecosystem services have been put forward such as those of de Groot *et al.* (2002) and Campos *et al.* (2005); but Costanza (2008) argues that a pluralism of typologies is needed. In the UK NEA, the definition of ecosystem services identifies 'benefits' rather than 'processes' following Defra (2007), and the typology of services (provisioning, regulating, cultural and supporting) is fully described in Chapter 2.

8.3.1 Multifunctional Forestry and Ecosystem Services

There is evidence that the multiple use of forests was a goal of Anglo-Norman landowners in the 12th Century (Wilson 2004); for instance, the combination of hunting deer and managing trees by pollarding is just one example of the combination of uses parkland offered our ancestors. The evolution of modern-day multi-functional forestry to encompass a range of ecosystem services can be traced through successive editions of the manual "Forestry Practice", published by the Forestry Commission from 1933 onwards. In the first edition, there was no reference to any goods and services other than timber production (Forestry Commission 1933), and this position was sustained up to, and including, the eighth edition (Edlin 1964). In the ninth edition (Blatchford 1978), a chapter was provided on recreation as an objective for woodland management, reflecting a government instruction in 1973 for the Forestry Commission to "give still further emphasis to recreational provision". A chapter on wildlife management acknowledged the amenity, scientific, economic and educational services that woodland fauna provide. Conservation of ecosystems, landscape design and consideration of water and soil were introduced in the tenth edition (Hibberd 1991). Gradually, forestry policy started to shift from multiple-use (e.g. a forest used for production and recreation) to multifunctionality, i.e. management to bring the forest to a state that several services can be delivered and possibilities are kept open to fulfil new services in the future, without endangering or impoverishing the ecosystem.

Modern multifunctional forestry policy in GB stems from the principles of Sustainable Forest Management (SFM) first enunciated at the United Nations Conference on Environment and Development—the Rio Earth Summit—in 1992 (Rollinson 2003; Forestry Commission 2004; Section 8.5). The concept of ecosystem services, including the development of methodologies for monetising some of the most important services, provides the potential for optimising decision-making with regards to habitat management.

A classification of the principal types of services considered important in UK woodlands/forestry is presented in **Table 8.10** using the UK NEA typology, and developed in the following sections.

8.3.2 Provisioning Services

For millennia, wood has been a key raw material for society, providing fuel, construction materials, chemicals (from tan bark), utensils and many other necessities. Traditional forest products, such as timber, are included as goods derived from provisioning services (i.e. trees) (Chapter 15). Arguably, these could also appear in regulating services as options for reducing carbon dioxide emissions into the atmosphere through the substitution of wood for building materials such as steel or concrete (with higher embedded carbon), and the substitution of woody biomass to generate heat and/or power instead of fossil fuels (Chapter 14). Perhaps to a lesser extent in the UK than in some other cultures (Shvidenco *et al.* 2003), forests and trees have also been the source of non-wood products such as food (fruit, fungi), meat (domestic stock grazing, wild game), animal bedding (litter and bracken), foliage, medicines and cosmetics.

8.3.2.1 Trees for timber, fibre, fuel

Timber remains a major component of various industries, especially construction and pulp- and paper-making. Currently, about 80% of the UK's wood and wood-product needs are met by imports (Forestry Commission 2009a); this compares with approximately 96% in the 1940s (Taylor 1946). Most of this demand is for coniferous wood, but there are imports of hardwoods as well. There are complex pathways of products from forest to processing to end use (Scottish Forest Industries Cluster 2004).

From 1999 to 2007, the total consumption of wood-products tended to increase, but dropped in 2008 (Forestry

Table 8.10 Types of ecosystem service provided by woodlands.

Ecosystem service provided by woodlands	Examples of goods and benefits in the UK	Key references
Provisioning services		
Crops, **livestock** and **fisheries**	Little tradition of agro-forestry other than grazing particularly as part of wood-pasture systems; non-timber forest products (NTFPs) for commercial and domestic use, e.g. meat (including from culled deer), berries, honey, fungi, medicinal derivatives and drugs.	Martin *et al.* (2006); Emery *et al.* (2006); Kirby *et al.* (1995)
Trees for timber	Provision of raw timber materials for use in commercial and domestic enterprises; provision of wood chips for boards and pulp for paper. Use of timber as an alternative for other building materials such as steel and concrete in order to reduce use of fossil fuels and enhance building standards.	Forestry Commission (2003a) Suttie *et al.* (2009)
Trees for bio/woodfuel	Timber products (e.g. harvesting residues, stumps and roots, recycled wood) as fuel for heat and power plants, as domestic firewood, for biochar and as raw material for processed hydrocarbon fuels.	Chapter 14 Ireland *et al.* (2004)
Woodlands and **water supply**	Wooded catchments especially in the uplands provide important water supplies for major urban areas (e.g. Thirlmere and Manchester).	Ritvo (2009)
Regulating services		
Climate	Avoidance of climate stress. Tree cover can help dampen the climatic effects experienced in the open, thus protecting soils, animals and humans from extremes of temperature, strong winds and UV light.	Mason *et al.* (2009)
	Carbon sequestration. Woodlands and their soils are important reserves of terrestrial carbon, and timber products can also be considered.	Morison *et al.* (2009); Lorenz & Lal (2010)
Hazard	Soil protection. Tree cover can offer protection from soil erosion and slope failure. Forest management will reduce exposure to chemicals and pesticides and likelihood of soil compaction compared to agriculture.	Moffat (1991); Nisbet *et al.* (2008)
	Flood and water protection. Woodlands moderate rainfall events and river and stream hydrographs, delaying and reducing flood events.	Nisbet *et al.* (in press)
Disease and pests	Woodland dwelling organisms can help in regulating the incidence and spread of insect pests of crops and pathogens of importance to humans, livestock, crops and ecosystems.	Chapter 14
Detoxification and Purification	Water quality. Because of minimal use of pesticides and fertilisers, woodlands managed under sustainable principles also offer benefits of water quality.	Nisbet *et al.* (in press)
	Soil quality. Woodland cover can stabilise contaminated brownfield land and hinder the pathways between source and receptors.	Moffat & Hutchings (2007)
	Air quality. Capture of atmospheric pollutants in tree canopies can lead to consequent reduced exposure for humans, crops, buildings etc.	NEGTAP (2001)
	Noise reduction. Belts of trees between residences and transport routes can absorb sound.	Huddart (1990)
Pollination	Woodlands likely provide habitat for diverse wild pollinator communities of importance to trees, crops and other plants.	Devoto *et al.* (2011)
Cultural services		Edwards *et al.* (2009)
Wild species diversity	Biodiversity. UK forests, including plantations, provide habitat for a wide range of fauna and flora but a limited genetic resource (e.g. compared to tropical forests).	Humphrey *et al.* (2003)
Environmental settings	Trees and woodlands are valuable for personal enlightenment and as places or catalysts for social activity and cohesion.	O'Brien (2006); Lawrence *et al.* (2009)
	Forests are increasingly acknowledged for their educational value.	O'Brien & Murray (2007)
	Trees have been perpetual motifs in fine art, and influenced many other art forms.	Phythian (1907); Hohl (1998)
	Many forests are open to the public for the enjoyment of outdoor pursuits and recreational activities. Their access facilitates exercise and benefits human health and longevity.	Woodland Trust (2004); O'Brien & Morris (2009)
	Trees and woodlands increase the diversity of landscape character; their existence provides a link with the past when man's existence was more closely linked to woodlands and their products; woodlands reduce the rate of, or eliminate the need for, cultivation, a significant cause of archaeological destruction.	Rackham (1976); Smout (2002); Crow (2004)
Supporting services		
Soil formation, nutrient cycling, water cycling, oxygen production	Forests facilitate soil formation and other biogeochemical processes essential to life.	Fisher & Binkley (2000)
Biodiversity	Little in way of unique species (endemism) at least amongst the well-know groups, but locally adapted provenances and distinctive assemblages associated with some species being at the edge of their range in Britain; a distinctive maritime climate; and historical differences. These include 'Atlantic' elements such as the abundance of bluebells, rich bryophyte communities in western oak woods, ash-hazel dominated woods (beyond range of beech), abundance of veteran trees with associated lichen and saproxylic associated species.	Rodwell (1991); Peterken (1996); Kirby *et al.* (2005)

Commission 2009a). Use of timber in construction—both home-produced and imported—could increase in future because of its potential to substitute for products such as concrete and brick, the use and production of which involves higher emissions (Suttie *et al.* 2009).

A total of 8.4 million green tonnes of softwood was produced in the UK in 2008 (Forestry Commission 2009a). This level of production has grown slightly over the last ten years and is substantially greater than that of 50 years ago. Over the same period, hardwood production has declined and currently stands at 0.4 million green tonnes (69% of which was used as fuel; Section 8.3.6).

About 60% of the gross annual increment of conifers is harvested (Forestry Commission 2002). The age structure and changing composition of softwood plantations have led to the forecast that production will to rise to 11–12 million tonnes in the 2020s, but will decline thereafter. Any subsequent increase will depend on future rates of afforestation and forms of restocking.

Only about 20% of the gross annual increment of broadleaves is currently harvested, so that production could increase substantially from the existing woods (Forestry Commission 2002). There is considerable interest in encouraging further use of this resource, but there are thought to be operational, market and attitudinal barriers to harvesting a greater proportion of the increment; the fragmented ownership and typical location of small woodlands within an agricultural landscape are particular barriers. The only market that seems likely to be able to expand sufficiently to take advantage of this potential resource is wood-fuel. The markets for traditional coppice products remain steady, or are increasing slightly, and seem unlikely to increase much (Sanderson & Prendergast 2002).

The gross value added in forestry and primary wood-processing (Forestry Commission 2009a) had been fairly stable from 2003 to 2006 at about £1.7 billion, but increased to £2.05 billion in 2007 (note that this includes processing of imported timber). About half of this is in the pulp and paper sector, with the other half split between the panel, sawmilling, and forestry/logging/related-services sectors.

Across much of country, wood was the major source of fuel until the widespread availability of cheap coal (and later oil, gas and electricity) during the 19th Century (Rackham 2003). It was not just used for domestic purposes, but also for industrial functions, such as iron smelting which contributed to the maintenance and expansion of broadleaved woodland in some regions at particular times. Industrial uses of wood-fuel had effectively disappeared by 1950, and its use as a domestic fuel continued to decline, in part because of Clean Air Acts and the greater convenience of coal, oil and gas.

The oil crisis in 1974, and interest in 'self-sufficiency', sparked a limited revival of wood-fuel through the use of wood-burning stoves. Throughout the 1990s, much work was done on wood-chip as a potential fuel based on short-rotation coppice (Tubby & Armstrong 2002), but little came of this in Britain; interest has been more sustained in Northern Ireland (McCracken 2007).

More recently, the general firewood market has been very buoyant, as has the demand for wood-burning stoves, due to factors such as increases in gas and electricity prices, and interest in reducing carbon footprints. The Stove Industry Alliance estimates 186,000 stoves were sold in 2008 alone, largely as secondary heating sources (Angela Duignan pers. comm.). Modern stoves can be more than 90% efficient, whereas logs burnt on an open fire are only about 30% efficient. A well-stocked, mixed broadleaved coppice woodland might produce about 3 tonnes of air-dried wood/ha/yr; 7–9 tonnes per year are needed to heat an average three-bedroom house. The popularity of barbecues has helped to increase interest in the charcoal market—much of the charcoal (about 75%) is still imported (Sanderson & Prendergast 2002), but there are initiatives such as BioRegional that seek to coordinate and increase local supply to the major charcoal markets (www.bioregional.com/what-we-do/our-work/bioregional-charcoal/).

Government subsidies allied to more consistent fuel type, quantity and quality (e.g. www.southwestwoodshed.co.uk/static/wp-content/uploads/woodfuel-standards.pdf) have encouraged active interest in wood-chip for small to medium-sized combined heat and power units. Wood-use is also being considered as a fuel for major power stations through co-firing (e.g. www.power-technology.com/projects/drax) and the dedicated wood-burning E.ON plant at Lockerbie. In England, a wood-fuel strategy envisages increasing the harvest of wood-fuel from existing woods to 2 million tonnes per annum (Forestry Commission 2007a).

There is likely to be an increase in fuel demand and provision from UK woodland. There are some suggestions that demand might outstrip domestic supply, leading to an increase in imports.

8.3.2.2 Wild food, medicines and ornamental products

Non-timber Forest Products (NTFPs), such as fruit, fungi, moss and foliage, are harvested every year from forests on both a commercial and non-commercial basis (FAO 2010). Across the country as a whole, the amounts are relatively small (Sanderson & Prendergast 2002), but these products can be important in supporting local industries and preserving traditions and skills. In Scotland, 18–24% of people regularly access forests for wild fruit, fungi and other NTFPs, such as lichens for dyes and foliage for floristry. In 2005, the total commercial value of these NTFPs was estimated at greater than £9.2 million (Chapter 15). The harvest and trade of Scottish wild moss alone is worth approximately £0.5 million a year and supports 125 jobs (Staddon 2006). Information on species and products, as well as guidance on gathering, trading and managing woodlands for NTFPs, is provided by the Forest Harvest Project (www.forestharvest.org.uk/). However, there is a need to improve knowledge of the reproductive systems, ecology and population structure of NTFP species (especially fungi) to be sure about sustainable harvesting levels.

Woods and forests are critical to the success of many game shoots (Gray 1986; Robertson 1992) and organised pheasant and partridge shooting has increased substantially over the last 20 years (Chapter 15). Shooting can make significant contributions to the income of individual estates and their local communities (PACEC 2006). According to data presented in PACEC (2006), of the 1.9 million ha of shooting land in the UK, approximately 40% is woodland. This equates to an approximate value of £640 million to the UK economy and the support of around 28,000 jobs. The long-term sustainability of intensively run shoots may be questioned, however, because they depend on breeding and releasing large numbers of birds each year. Release pens are present in around 5% of woods (Sage *et al.* 2005) and there can be conflicts with biodiversity conservation, although other management activities associated with shooting may benefit biodiversity (Draycott *et al.* 2008). Small-scale shoots or those involving wild birds are likely to be more compatible with other woodland uses (Chesterton 2009).

The increase in deer populations in recent years requires management to reduce their impact on woodland composition, structure and regeneration, as well as on the prevalence of road-traffic accidents, crop damage and other, non-woodland impacts. There is some potential income from the venison market, although this will rarely be enough to cover the costs (Chapter 22). In England, most deer stalking for sport takes place within woodland; in Scotland, hill stalking is more typical, but woodlands are vital to provide food and shelter for deer, especially in winter. Unless carefully managed, wintering deer cause woodland to deteriorate, as they eat young seedlings, preventing them from developing into a new generation of trees. The need for deer management is likely to increase further, so there could be benefits from developing these markets further (www.macaulay.ac.uk/RELU/ ; also www.scottish-venison.info/).

Woodlands also provide valuable shelter for livestock production, as shown in studies at Kirkton Glen SAC farm (Pollock 2005).

8.3.2.3 Genetic resources

There is little in way of unique species (endemism) in UK woodlands, at least amongst the well-known groups of organisms. There are locally adapted provenances and distinctive assemblages, however, which have developed through a combination of: some species that are common on the continent being absent or at the edge of their range in Britain; the distinctive maritime climate; and historical differences. These result in a good representation of ash-hazel dominated woods (beyond the range of beech), 'Atlantic' elements, such as the abundance of bluebells and rich bryophyte communities in western oak woods, and the abundance of veteran trees with associated lichen and saproxylic species. There are likely to be genetic differences associated with such distinctiveness, but these are largely unknown.

The area of woodland for *in situ* genetic conservation is nearly 18,000 ha, for *ex situ* genetic conservation is 250 ha, and for seed production is nearly 2,250 ha.

8.3.2.4 Biodiversity

As well as being a key supporting service, biodiversity, can also be considered as a provisioning service because resources are invested in forest management (e.g. through the BAP process, agri-environment schemes and support for SSSI management) to generate particular types of diversity and species assemblages. These assemblages can have value as goods and services in their own right. Both the cost of providing biodiversity and benefits to people of this provision can be monetised. The costs of delivering biodiversity for the first set of BAPs produced in 1994 were estimated (Shepherd *et al.* 2002), but since then, both the targets and the range of habitats covered have changed. A study commissioned by Defra to estimate the economic value of biodiversity benefits delivered by the BAPs is currently being finalised.

In a recent survey, people recognised the provision of 'places for wildlife to live' and, hence, biodiversity, as one of the main benefits of forests (70–80% of respondents) (Forestry Commission 2009b). Biodiversity has also been included as part of the estimation of non-timber values associated with woodland. In one study, for example, the marginal benefits of woodland were estimated to be 35p per household/year for enhanced biodiversity in 12,000 ha (1%) of commercial Sitka spruce forest; 84p per household/year for a 12,000 ha increase in Lowland New Broadleaved Native forest; and £1.13 per household/year for a similar increase in Ancient Semi-natural Woodland (Willis *et al.* 2003). Several studies provide estimates of values for protecting habitats, increasing populations, or reintroducing particular species of woodland animals. These include estimates of median annual values for increasing red squirrel populations under the BAPs by 25–50% of £2.67 per person in North Yorkshire (White *et al.* 2001), mean annual values for protecting red squirrel habitat in Kielder Forest by the Northumberland Wildlife Trust of £2.94 per member (Garrod & Willis 1994), and of around £28 per household/year in the Aberdeen area for increasing capercallie populations in Scotland over a 10-year period (Philip & Macmillan 2005). For reintroductions, estimated values include £22–£24 per household/year in the Aberdeen area over a 10-year period for a pilot project to reintroduce beavers in Argyll, Scotland (Philip & Macmillan 2005), and reported lump sum values of £8–£10 for the reintroduction of pine marten populations in England (Bright & Helliwell 1999; White *et al.* 2001). Values associated with the conservation and extension of woodland habitats are also partly reflected in charitable giving to bodies such as the Woodland Trust (which received total legacy and membership income of £12 million in 2009).

8.3.3 Regulating Services

Across the globe, forests are one of the main habitats providing regulating services for the environment. Their current and future role in UK conditions has often been under-estimated because of the limited extent of tree cover, and because of the negative impacts that have arisen in some conditions such as reduced water quantity or poorer water quality (acidification) in upland catchments, and carbon losses through the drainage of peatlands for planting. There are still risks from inappropriate woodland creation or management, but these can be dealt with by following good practice set out in UK Forest Standard and associated guidelines (Section 8.5). The role of trees and forests in helping to regulate our environment is likely to increase in future under climate change scenarios (Handley & Gill 2009). While only limited work has been done on monetising this role (CJC Consulting 2005), it is likely to be significant.

8.3.3.1 Avoidance of climate stress

Timber products can have an important role in substitution as alternatives for other building materials, such as steel and concrete, which have higher embedded energy (ECCM 2006).

One of the most important regulating services that woodlands provide is their capacity to sequester carbon. The total carbon stock in UK forests (including their soils) is around 800 megatonnes of carbon (Mt C) (approximately 2,900 Mt of carbon dioxide (CO_2) equivalent) and the stock in timber and wood products outside forests is estimated to be a further 80 Mt C. Broadleaved woodland in southern England is currently the most important vegetation carbon store (Milne & Brown 1997), though this is dwarfed by the carbon stored in soils, particularly in heathlands and blanket bogs. It has been estimated that the average carbon content across non-organic forest soils in GB is 288 t CO_2 equivalent/ha, while on peaty soils and deep peats, carbon stocks of 160–700 t CO_2 equivalent/ha are found depending on peat-layer depth. The choice of forest management systems influences the rate of sequestration and the amount of carbon stored on site, and forests can capture up to 800 t CO_2 equivalent/ha in tree components (Mason *et al.* 2009; Morison *et al.* 2009). Measurements suggest that annual removal from the atmosphere of around 24 t CO_2/ha/yr can be achieved in a coniferous forest at peak growth, with a net long-term average for productive coniferous crops of around 14 t CO_2/ha/yr (Jarvis *et al.* 2009). Rates of around 15 t CO_2/ha/yr have been measured in oak forests at peak growth, with a net long-term average likely to be around 7 t CO_2/ha/yr. Further studies are required to fully determine long-term rates of carbon storage, taking into account the type of use of timber products which influences the amount of carbon stored away from woodlands.

The strength of the UK forest carbon sink increased from 1990 to 2004, but may now start to decline as a result of the uneven age class distribution, the rotational harvesting of mature trees and the fall off in planting rates over the last 20 years. There is renewed policy interest in forest expansion in England, Scotland and Wales (Read *et al.* 2009; Moffat *et al.* 2010). Woodland planted since 1990, coupled with an enhanced woodland creation programme of 23,000 ha/yr over the next 40 years, could be delivering emissions abatement equivalent to 10% of the total greenhouse gas emissions at that time. Woodland creation is judged to be a highly cost-effective and achievable form of emission abatement at less than £100/t CO_2 equivalent, and while conifer plantations and energy crops were, not surprisingly, judged the most cost-effective options for carbon sequestration, mixed and broadleaved woodland that deliver a wider range of other benefits were still only about £41/t CO_2 equivalent (Matthews & Broadmeadow 2009; Section 8.3.6). Tree-planting on high carbon soils can lead to emissions of carbon dioxide, though these are progressively counterbalanced by uptake in the growing trees. Other types of land-cover can also support carbon sequestration, such as peatland (Lindsay 2010), but forests are less limited in where they can be grown, have a greater potential to generate income as a land use (through timber, etc.), and have potentially high value for other services.

Forests can also reduce some of the effects of climate change, notably in dampening temperatures in the soil and beneath the canopy, and in providing shade and shelter for animals and human visitors. Woodland cover can provide shade, reducing overheating and the need for air conditioning, and shelter from strong winds, reducing heat loss and soil erosion (Gardiner *et al.* 2006). Increasing temperatures will increase the shade and shelter value of trees in towns (Handley & Gill 2009), and also for livestock in the country. Shading of streams can aid thermal regulation and fish survival.

8.3.3.2 Avoidance of hazard

Tree cover can offer protection from soil erosion and slope failure. Recent soil strategies, such as those from Defra (2009b) have focused attention on soil loss and degradation, although mainly in the context of agricultural land use (POST 2006). There has been little attempt to estimate the contribution of British forests to soil protection, yet ancient woodland has a value in protecting relatively undisturbed soils (Ball & Stevens 1981).

Erosion in the forest itself can occur following large-scale and badly implemented forest operations, particularly large clear-fells. Good practice guidelines have, therefore, been developed (Forestry Commission 2003b; Forestry Commission in press a). In erosion prone sites, there is likely to be greater use of low-impact silvicultural systems, and continuous cover forestry may become more favoured. Increasingly dry summers and heavier winter rainfall (particularly extreme events) will increase the importance of this service, highlighting the need for more research in this area.

Forests moderate rainfall, delaying and reducing flood events. Forest and tree cover can help to regulate flows in streams and rivers, and also affects the quality of that water (Calder *et al.* 2008; Nisbet *et al.* in press). The effects can be either positive or negative, depending on the context, and are localised. In general, benefits from increased cover are likely in the upland and upland fringes, and may not require very large changes to the land cover of whole catchments. There is likely to be some benefit from increasing tree cover in floodplains (Thomas & Nisbet 2007). Any increase in cover will need to be carefully planned, identifying where a slowing of water movement may be desirable, and ascertaining the possible implications of trees caught in floods blocking bridges downstream (Nisbet *et al.* in press).

8.3.3.3 Detoxification and purification of water, air and soil

Forest cover of catchments may reduce yield, but has been used as a way of minimising the need for water treatment by excluding livestock from watercourses and their immediate catchments, therefore reducing the risk of potential water contamination. Forestry operations have the potential to cause problems for water quality, but these can be addressed through application of the Forestry and Water Guidelines (Forestry Commission 2003b). The presence of trees can also contribute to water quality by maintaining cool temperatures for fish, intercepting pollution from point sources and capturing diffuse pollution (Nisbet *et* *al.* in press). Trees and woodland can contribute to water management, for example, more sustainable surface drainage in urban areas (Handley & Gill 2009; Chapter 10).

Trees are effective scavengers of pollutants from the atmosphere both through internal absorption of pollutants, and external adsorption on to leaf and bark surfaces; hence, problems arise when the acidity scavenged finds its way into watercourses. While some atmospheric pollutants, such as sulphur dioxide, have reduced in concentration, some remain of concern, and climate change will exacerbate others—there is the potential for increased ozone, for instance (Section 8.2.5.2). Targeted tree and woodland development around intensive livestock units (Pitcairn *et al.* 1998) and alongside roads (Bignal *et al.* 2004) can limit the spread of pollutants on to more sensitive habitats such as heathland.

Net pollution absorption by trees was considered to reduce the number of deaths brought forward by air pollution by 5–7 per year, and to reduce hospital admissions by about 4–6 per year. With a discounted value of life and cost of hospital admission, this suggests a benefit of £0.9 million per year for Britain (Powe & Willis 2004). This is small compared to some other non-market benefits, but in urban areas, the relative benefit of small woods (high edge-ratio) will be comparatively high in this respect.

Forest cover may have a remedial role on post-industrial and contaminated soils as an alternative, potentially productive, land use (Lynch & Moffat 2005). This may be considered a viable low-cost alternative to more expensive engineering solutions (Duggan 2005; Pulford & Watson 2003).

Belts of trees and shrubs can be effective at reducing noise pollution—a 33 m-wide tree buffer may reduce noise levels by 6-8 dB (Leonard & Parr 1970). While a relatively minor effect in most situations, this could provide an additional argument for trees and shrubs, rather than other forms of greenspace, in some urban situations, on roadsides and adjacent to industry such as quarries.

8.3.4 Cultural Services

The cultural services offered by woodlands and forests must not be underestimated (Edwards *et al.* 2009; Chapter 16). Their importance for recreation and informal leisure activities is hugely significant, with between 250–300 million visitors to British forests each year. Most visitors undertake various forms of physical activity, such as horse riding, cycling, walking or jogging, and thus, enhance their general health. A range of pilot initiatives have begun where members of the public with certain ailments will be referred to local woods or forests in order to take physical exercise; an NHS Forest Project has also begun (**Box 8.3**). Research has shown both the physical and mental health benefits that woodlands can facilitate, even to those who simply live amongst them. Increasingly, community activities are being held in or around woodlands, in order to promote healthy outdoor pursuits, but also to develop community cohesion. In the public forest estate, Forest Rangers are employed to provide an informal educational role. In addition, there are nearly 150 Forest Schools in GB set up to promote outdoor play and learning (Knight 2009). Such opportunities can improve self-confidence and self-esteem, especially for those who find indoor classrooms less conducive to learning (O'Brien & Murray 2007; Lovell 2009).

Depending on their character and location, forests and woodlands can have significant aesthetic appeal, and can enhance landscape character. These services are appreciated by those who live amongst, or visit, such places. In urban areas, even small woodland blocks can improve the visual appearance and, therefore, the 'feel' of a neighbourhood, and property values often correlate with the degree of trees present (O'Brien *et al.* 2007). The tree and woodland motifs are used extensively in the arts, and form the inspiration for fine art, poetry and music. Woodlands also provide a 'sense of place', a community focus and a spiritual resource. Such services are probably taken for granted by most, but are immeasurably life-enhancing.

Trees and woods are highly valued by people for their historic and cultural values, and as places for quiet (and not so quiet) recreation. It is increasingly acknowledged that UK woodlands contain a diverse array of historic environment assets which are often well-preserved when compared to those in cultivated landscapes, and which provide links to past woodland management or an earlier, pre-wooded landscape. They offer not only a valuable educational resource, but help to create a sense of place and contribute to cultural identity. Historic buildings and monuments in England received an estimated 51 million visitors during 2009, 71% of which expressed an interest in their local history (English Heritage 2010). But while this can offer some indication of heritage value at monitored sites, the contribution of heritage in woodland remains unknown and unvalued.

8.3.4.1 Recreation and tourism, health and well-being

Woodlands provide a setting for a wide range of activities from dog walking to mountain biking, for the short and longer-stay visitor.

Box 8.3 The NHS forest

The NHS has realised that sustainability is part of the core business of the health service, rather than a green add-on. It is one of the biggest resource-users and carbon-generators in the UK, and there are many efforts underway both nationally and locally to reduce its carbon footprint. The creation of an NHS forest is one response. There are 1.3 million employees in the NHS, and the NHS Forest aims to have 1.3 million mature trees in 20 years' time. It is hoped that at the centenary of the birth of the NHS, the forest will play a significant part in offsetting the carbon footprint of the health service by as much as 10%.

The NHS Forest will consist of trees on every NHS campus, and in the local surrounding area, giving opportunities for involvement from the wider community and contributing to a 'Natural Health Service' through the benefits of the greenspace. The NHS Forest will also be available for commemoration and celebration by patients, relatives and professionals. The Campaign for Greener Healthcare is working with the Forestry Commission, the Woodland Trust, Natural England, the Sustainable Development Unit and others to develop partnerships to identify suitable land for planting and existing woodlands close to NHS sites which could be 'adopted' as part of the NHS forest.

Woodland is listed as one of the most popular destinations for countryside visits (around 250 million day visits per year (Forestry Commission 2009a). Three-quarters of respondents in the Public Opinion of Forestry Survey (Forestry Commission 2009b) had visited forests or woodland in the last three years for walks, picnics or other recreational activities. This was an increase over the results in 2005, but similar to the results in 2007. An annual aggregate value for recreation in GB forests has been estimated as £393 million (Willis *et al.* 2003). National Nature Reserves containing woodland provide an example of popular, individual, woodland locations, with Burnham Beeches (220 ha) attracting an estimated 750,000 visitors and Hatfield Forest (392 ha) attracting 250,000 visitors in 1997–98 (English Nature 2002).

Access to woodland has been promoted, particularly by the Woodland Trust (Woodland Trust 2004). In Scotland, there is a general right of responsible access to all land and water, including woodland. In England, over half the area of woodland has public access, some of which is secured under the Countryside and Rights of Way Act, mainly through the dedication of the Forestry Commission's public estate, and voluntary and public bodies. Much of the population, however, still does not have the opportunity to experience woodland in their local area; only 55% of the population have access to woods greater than 20 ha within 4 km, and 10% have access to woods greater than 2 ha within 500 m of their home (Woodland Trust 2004).

More specifically walking in the outdoors is increasingly promoted as part of encouraging healthy life-styles (www.whi.org.uk). The benefits of being in woodland are not just a physical effect: trees and woods are seen as affecting our spiritual and emotional sides as well, as illustrated in the modern writings of Roger Deakin (2007). In the Judeao-Christian faith tradition, trees and woodlands are often seen as reflections of the strength, majesty and creative skill of a transcendent God. In other religious traditions, such as animism, trees and woodlands themselves are imbued with a spiritual presence and power.

Woods may also be the venue for more organised commercial events such as orienteering and paintball games, venues for pop concerts and by the adventure company GoApe which provides a mixture of canopy walkways and zip-wires. Across all its UK sites, GoApe have over 30,000 users per annum (www.countrysiderecreation.org.uk/events/Activity%20Tourism/Go%20Ape!.pdf). In Scotland, the Enchanted Forest (www.enchantedforest.org.uk/) sound and light show at Pitlochry is a popular event every autumn. Forestry Commission Scotland's 7stanes mountain biking network throughout south Scotland has a range of tracks, including one suitable for disabled riders. Developments of facilities at Bedgebury Forest have resulted in a four-fold increase in visitors in recent years. There are also substantial facilities in North Wales (Coed y Brenin) and the Lake District (Grizedale and Whinlatter).

The value of forests for recreation is likely to increase as part of the transition to low-carbon living, through greater interest in holidays and recreational activities in Britain and better local access to greenspace. There is also remote appreciation of woodland through membership of bodies, such as the Woodland Trust, or books such as *Meetings with Remarkable Trees* (Pakenham 1996) and the related television programmes.

Woodlands can have substantial biodiversity, valued for its existence by many different people from casual walkers to wildlife enthusiasts; the opportunities that woodlands provide for viewing wildlife is a major motivation for woodland visits.

8.3.4.2 Heritage goods—citizenship and other cultural services including historical and landscape values

Trees and woodlands are valued for personal enlightenment: they provide special moments, places of sanctuary and even burial areas. They also provide a link with the past, contributing to cultural memory and development. Woodlands can be places of learning, providing evidence of the workings of the natural world. They can be a focus for community development around both their formation and management—there are a substantial number of volunteer groups utilising woodlands.

Ancient woodland and veteran trees are historic features in their own right and provide a link to past society and culture (Rackham 2003). Many 'Royal Forests' have hundreds of years of history, tradition, myth and legend associated with them, helping to create important historic landscapes. Ancient woodland is also increasingly appreciated for its archaeological content. This includes the archaeology of the woods—banks, sawpits, old coppice stools and other features that relate to the history of the land as woodland—and also the archaeology in the woods: the traces of earlier land uses that have survived because the woodland soil surface has often been less disturbed than surrounding land (Rotherham *et al.* 2008). There is increasing interest in these services: amongst the public with regards to local history, as evidenced by TV programmes such as 'Who do you think you are' and 'Time Team'; amongst policy makers with regards to the added protection given to ancient woods and veteran trees in England through Planning and Policy Statement 9 (ODPM 2005); and amongst organisations and initiatives such as the Ancient Tree Hunt (www.ancient-tree-hunt.org.uk/). There are now many community-based heritage projects where local interest groups and volunteers are helping to record veteran trees and archaeological features within some of Britain's forests. Woodlands can also contribute to the protection of soil profiles, cultural artefacts and archaeological remains beneath them—providing a link with the past. British forests include nearly 5,000 Scheduled Ancient Monuments and a much larger, but unknown, number of sites of archaeological interest. Forests can help protect such evidence from disturbance, unless events such as catastrophic windthrow occur. Forest operations have also contributed to access for education and research into geological sites of interest (**Box 8.4**).

Trees and woodlands increase the diversity of landscape character. They provide a sense of place in key locations and form the major components of many landscapes, from the pinewoods of Glen Affric and the hanging oak woods of North Wales, to the beech woods of the Chilterns. The

Box 8.4 Geodiversity and woodlands

Britain is arguably the most geologically diverse landmass of its size in the world with a sequence of rocks representing every major Period of geological history, permitting a variety of sediments, fossils, igneous and metamorphic rocks and structures to be seen. Many of the pioneers of the science of geology were British, and the names of several geological Periods, Series and Stages (e.g. Cambrian, Devonian, Wenlock, Tremadoc, Bathonian) refer to areas or places in the UK.

Woodland character responds to the geological diversity and its influence on soil type and wetness (Pyatt *et al.* 2000; Section 8.2.5). In addition, woodlands cover some key sites of geological interest, which may be of value for research and education. For example, in England, approximately 50 of the 1,200 nationally important geological SSSIs are found on Forestry Commission land, along with an unknown number of Regionally Important Geological and Geomorphological Sites (RIGS). These include sites where active geomorphological processes are taking place, as well as sites that provide evidence of Britain's geological history and may take a variety of forms from quarries and road cuttings, to stream sections and natural rock exposures, amongst others. In some cases, the activities required for the development and management of commercial or sustainable forest have exposed the now-designated features. Features of geological or geomorphological interest may require various forms of management in order to maintain access to the features, or to maintain the geomorphological processes related to the features (Prosser *et al.* 2006).

Two examples provide an indication of the variety of interests that occur on these sites:

Geological history—Mortimer Forest SSSI, near Ludlow, on the Herefordshire/Shropshire border

This SSSI consists of a series of exposures formed by disused quarries, extensive forest road-cuttings and stream sections. Together, these quarries, cuttings and stream sections provide extensive exposures through a succession of rocks belonging to the Wenlock and Ludlow Series of the Silurian, and contain the internationally recognised reference section for the boundary between the Wenlock and Ludlow Series and the boundary between the Gorstian and Ludfordian Stages of the Ludlow Series. The term Ludlow Series was first proposed by the pioneering geologist Sir Roderic Impey Murchison in 1833 based on the rocks exposed around Ludlow, Mortimer Forest and adjacent areas. Since that time, there has been an almost continuous history of research on the rocks, fossils and ancient environments represented in Mortimer Forest, with the investigation of the ages of thin bands of volcanic ashes within the succession being one of the most recent. The importance of the rocks themselves, and their fossiliferous nature, has meant that Mortimer Forest is frequently used for teaching by schools and universities, and is also visited by geology societies and interested individuals. Much of the research of the past 50 years has relied on the presence of the cuttings and disused quarries in the forest. During the late 1970s, the then Nature Conservancy Council (NCC), together with the Forestry Commission, cleared a series of 13 roadside exposures along the Wigmore Road, providing representative sections in all the rocks comprising the Ludlow Series to form the basis of a geological trail. This was accompanied by a trail guide initially published by the NCC and later by the Forestry Commission. Today, the exposures within Mortimer Forest and the geological trail are managed by the Forestry Commission with advice from Natural England, so that they remain visible and accessible.

Active geomorphology—Slade Brook SSSI in Gloucestershire

This (illustrated below) contains a long series (about 700 m) of spectacular, actively forming tufa dams. Although the management of this site relies, in part, on the hydrogeological conditions of the surrounding area, the degree of shade surrounding the stream, the humidity and temperature, the availability of dead vegetation, such as twigs, and the presence of certain algae and mosses are all factors controlling the precipitation and formation of the tufa.

Tufa dam. *Photo courtesy of Natural England.*

contribution may be solely aesthetic, or linked to how that landscape has developed historically and culturally, such as the Binsey poplars that were celebrated in the poem of Gerald Manley Hopkins. Individual trees, like the Birnam and Major Oaks, are strongly associated with the character of particular sites. Trees contribute to amenity values and their presence can even increase property values. The UK adoption of the European Landscape Convention is likely to mean that the significance of landscape issues can be expected to increase.

There is some association between perceptions of landscape value and woodland characteristics: for example, woodland type (broadleaves tend to be more favoured than conifers), tree age (large, old trees tend to be favoured over young ones), openness (valued more than dense, closed areas) and diversity (mixtures and variation valued over uniformity) (Willis *et al.* 2003). Willis and others have explored expressing these preferences in value terms, via willingness-to-pay or hedonic pricing methods. For instance, they have estimated a marginal value of £269 per annum/household for those households on the urban fringe with a woodland landscape view, and £1,500–£2,000 (present value per hectare) for the contribution of new planting to the landscape in Central Scotland (CJC Consulting 2005). Overall, Willis *et al.* (2003) give annual values attributable to landscape values of £150 million for GB forests (**Table 8.11**).

Large-scale afforestation, almost from its inception, has been criticised as damaging valued open landscapes (Symonds 1936; Tompkins 1989; Smout 2000). Pioneering work was done by Sylvia Crowe in the 1960s to improve the fit of new forests into the landscape and this has been followed with research and guidance (Bell 2003; Forestry Commission in press b). Despite this, objections are often raised to the clearance of trees and woods in order to restore heathland and other open habitats. People do not like sudden change: communities may object to the planting of new woodland but, when it is mature, object to its felling. In turn, this may be reflected in the use of Tree Preservation Orders which, while more usually applied to individual trees, have been applied to areas of woodland as well. Environmental Impact Assessments (Forestry Commission

2007b) are now required for any large-scale schemes, both woodland creation and clearance; and Landscape Character Assessments can contribute to identifying how and where changes in tree and woodland cover may best be achieved.

The heritage associated with this widespread landscape change has achieved some recognition. There are a number of initiatives to capture the oral history of the large-scale afforestation, and its impacts on the landscape, from forest workers and surrounding communities (Smout & Tittensor 2008), and to summarise the evolution of organisations and objectives (Tsouvalis 2000; Foot 2010).

8.3.5 Supporting Services

Underpinning the three groups of ecosystem services discussed above are the supporting services of soil formation, nutrient cycling, water regulation and oxygen production. These are common to almost all terrestrial ecosystems which support vegetation (Chapter 13). Woodland biodiversity, including genetic diversity, can be regarded as another supporting service. In particular, below-ground fauna and flora (including mycorrhizal relationships) promote essential biogeochemical processes that, in turn, lead to the renewal of soil, plant nutrients and fertility. Above-ground fauna both help and hinder woodland dynamics and natural woodland regeneration. But biodiversity is an important management objective for many woods and forests in Britain in its own right—in other words, in order to promote a rich fauna and flora because these land uses are seen as having the capacity to do so. In this respect, one can view biodiversity as a provisioning service, but for some it may even take on the role of a cultural one.

8.3.6 Valuation of Services Over Time

As has been illustrated in respect of individual services (see Section 8.3 above) there have been some previous attempts to assess the value of the services provided by woodlands and forests. One of the most comprehensive (Willis *et al.* 2003), is summarised in **Table 8.11**.

The UK NEA valuation exercise has focused on assessing changes both in past benefits associated with ecosystem service provision, and in potential future benefits (Chapter 22). Woodlands provide a wide range of ecosystem services, of which seven provisioning services, 12 regulating services, four supporting services and 10 cultural services were initially considered as potential candidates for valuation (Chapter 22). The extent to which the service could be attributed exclusively to woodlands, and the availability of time-series data, led to the following being selected for description here:

i) Wood production (timber and fuel): an example of a provisioning service;

ii) Carbon sequestration: an example of a regulating service.

8.3.6.1 Provisioning services: wood production values

Wood production. Overall, total wood production in the UK has risen substantially since the mid-1970s as forests planted earlier in the 20th Century have matured. For softwood, production by the Forestry Commission is the major element, while for hardwood most production comes from the private sector (Chapter 22). The quantity of wood produced from Forestry Commission land tends to be more stable between years compared with wood harvested from private woodlands; the former reflects commitments to support domestic wood-using industries, while the latter is influenced to a greater extent by prevailing price levels.

Softwood production has increased substantially since 1945, (Chapter 15) but detailed annual data are only available since 1976. Much of the increase in softwood production since the mid-1990s has been in Scotland (**Figure 8.6**).

By contrast, hardwood production has declined since the mid-1970s, and is largely concentrated in England (**Figure 8.7**). High volumes of production during the period 1988 to 1990 were probably partly due to clearance after the storms of 16 October 1987 in south-east England (resulting in windthrow of 3.9 million m^3 or 13–24% of the growing stock), and of 25th January 1990 in south and west Britain

Table 8.11 Annual and capitalised social and environmental benefits of forests in GB (at 2010 prices).

Environmental benefit	Annual value (£ millions)	Capitalised value (£ millions)
Recreation	484	13,825
Landscape	185	5,290
Biodiversity	476	13,592
Carbon sequestration*	115	2,676
Air pollution absorption*	0.5	14
Total	**1,261**	**36,019**

* An approximation, since carbon sequestration, and probability of death and illness due to air pollution, varies over time. More carbon is sequestrated in early rotations than in later rotations, resulting in an annuity stream that is inconsistent over multiple rotations. Similarly for air pollution, that results in an individual's life being shortened by a few days or weeks at the end of the individual's life at some point in the future. More recent work puts a much higher value on the carbon sequestration benefits (Read *et al.* 2009).

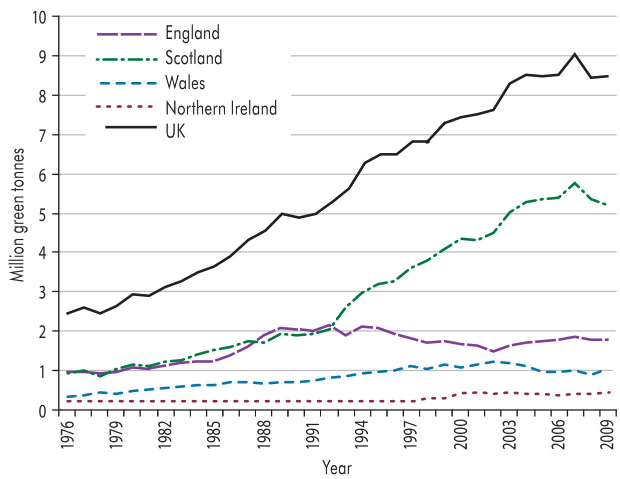

Figure 8.6 UK softwood production by country (million green tonnes). Source: Forestry Commission (2009e).

(resulting in windthrow of 1.3 million m³ or 1–3% of the growing stock) (Quine 1991).

Price. Forest management decisions concerning rotation length can affect the prices of wood. Prices tend to be higher for larger (i.e. generally older) trees and lower for small trees as they cannot be used for producing sawlogs (which tend to be relatively high-value). Relationships between tree size and price are traditionally modelled as price-size curves (Mitlin 1987; Whiteman 1990; Whiteman *et al.* 1991; Sinclair & Whiteman 1992; Pryor & Jackson 2002; Bateman *et al.* 2003). However, the largest trees are not invariably the most valuable (Chapter 22), and a range of other factors including species mix, quantity sold, timber quality, site conditions, and strength of local demand, also influence prices.

Over the period 1971/72 to 2009/10 mean softwood standing sales prices for the Forestry Commission estate fell in real terms by about two thirds from around £30/m³ overbark (£35 per green tonne) to £10/m³ overbark (£12 per green tonne) at 2010/11 prices (**Figure 8.8**). This appears to follow a longer-term downward trend: the British softwood standing sales price index reported by the Forestry Commission in 1977 indicated a fall of around one quarter in real terms over the period 1958 to 1972 (Mitlin 1987; Insley *et al.* 1987).

Time series data on hardwood standing sales prices are not available. The mean price for all hardwood sales in 2007/08 by the Forestry Commission in England (which accounted for over 90% of hardwood sales by the Forestry Commission that year, although less than 10% of total UK hardwood production) was around £15/m³ (equivalent to about £18 per green tonne at 2010/11 prices). Comparison of recent Forestry Commission hardwood price data with prices for all British woodlands reported in Whiteman *et al.* (1991) suggests that, in real terms, hardwood prices probably fell by at least a third from around £90 per green tonne in 1989, to somewhere in the range £18–£61 per green tonne in 2007/08 at 2010/11 prices (Chapter 22). A drop in average hardwood prices in real terms appears consistent with the apparent decline in demand for UK-grown hardwood by some upstream sectors. The past decade has been characterised by a collapse in purchases of British hardwood by UK sawmills, and by pulp- and paper-mills, which fell from 227,000 and 191,000 green tonnes respectively in 1999, to 67,000 green tonnes and zero green tonnes in 2008—the numbers of UK sawmills processing hardwoods roughly halving (Forestry Commission 2009a). Overcapacity and investment in Eastern European mills are also considered important factors in this decline (Lawson & Hemery 2007).

With domestically grown wood accounting for less than a fifth of the total wood used in the UK (Forestry Commission 2009a), this country is generally assumed to be a price-taker with respect to global prices (McGregor & McNicoll 1992; Thomson & Psaltopoulos 2005; Lawson & Hemery 2007), and the supply of imported timber perfectly elastic at the world price (Bateman & Mellor 1990). To the extent that UK standing sales prices simply reflect those on the world market, the marginal value of wood produced by UK woodlands can be considered independent of the quantity produced.

Value. Wood production can be valued at standing sales prices, at roadside prices, or using an average of these. Standing sales prices appear to provide the better basis for estimating the contribution of woodland habitats to ecosystem services.

Valued at standing sales prices, the gross value of UK softwood production has shown little trend from 1976 to the present. Falling real prices for softwood were largely offset by increasing volumes as coniferous plantations matured, with removals of roundwood apparently more than trebling from 2.4 million green tonnes of softwood in 1976 to 8.5 million green tonnes in 2009 (Forestry Commission 2010b). However, despite little overall trend, gross values varied considerably in real terms over this period, with peaks in the

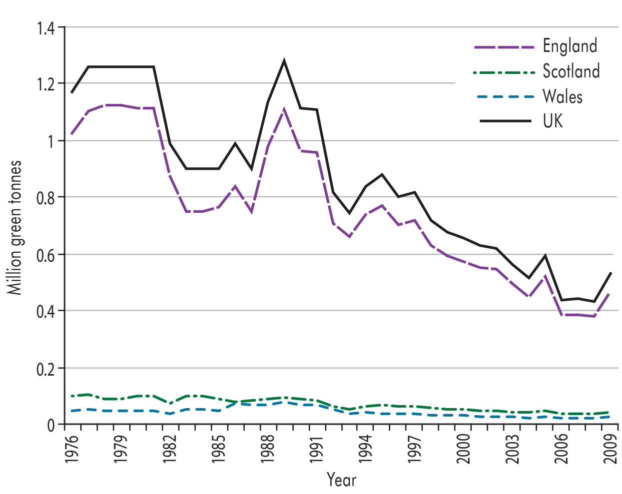

Figure 8.7 UK hardwood production by country (million green tonnes). Hardwood (broadleaved) production in Northern Ireland is estimated to be negligible. Source: Forestry Commission (2009e).

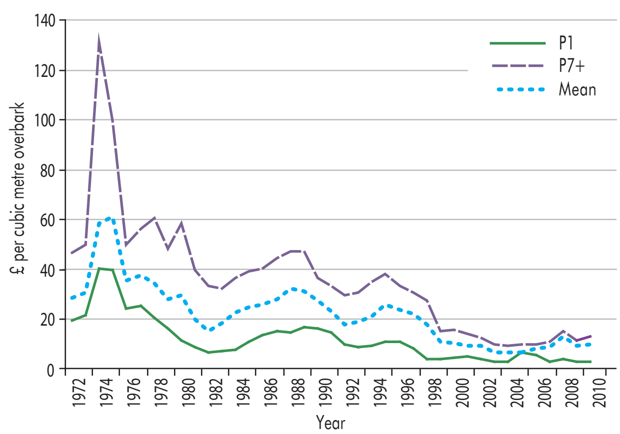

Figure 8.8 Softwood standing sales prices in Britain by tree size (at 2010/11 prices)*. Source: Forestry Commission (2010c). * Data for financial years ending in March of year shown; tree sizes: P1: ≤0.074m³; P7+: ≥0.425m³; mean: all size categories (P1–P7+).

late 1980s and mid 1990s, including a maximum of over £190 million in 1995 at 2010 prices, and troughs in the early 1980s and early 2000s, including a minimum of under £60 million in 1981 at 2010 prices. The total area of predominantly coniferous woodland in the UK has remained at around 1.5 million ha over the past two decades, while there has been an expansion in predominantly broadleaved woodland in the UK from 0.9 million ha in 1990 to 1.1 million ha in 2010 (Forestry Commission 2010b). Indicative per hectare values for softwood production, derived by dividing total production by the total area of coniferous woodlands, suggest gross production values fell from £95/ha/yr in 1990 to around £66/ha/yr in 2009 at 2010 prices.

Information to construct trends in hardwood values is sparse. However, the reduction in the total volume of UK hardwood sold (which more than halved from over 1.2 million in 1989 to 0.4 million green tonnes in 2007–08), combined with an apparent fall in hardwood prices, suggests total gross values of UK hardwood sold fell sharply over this period. Gross values for hardwood production were around £120/ha in 1989. Valued using the above lower- and upper-bound mean price estimates (of £18 and £61 per green tonne at 2010/11 prices), gross values in 2007/08 would have been £7–£25/ha at 2010/11 prices. The decline is consistent, with reports that it is often less expensive at present to buy imported hardwood than to process UK-grown wood; this could change in the future.

How forestry costs are apportioned between wood production and other ecosystem services (such as carbon sequestration), combined with the range of costs covered, and the assumed shadow price of labour, may all affect estimates of marginal values of wood production net of capital and labour costs and of subsidies. More fundamentally, whether expenditures and revenues are simply compared for individual years, or annual equivalents computed over a single or series of rotations, also affects estimated values, with annualised estimates sensitive to the discount rate adopted in comparing costs and revenues over time.

Regular surveys of forestry costs, incomes and expenditures were undertaken from the 1960s until the early 1990s for private woodlands and typically showed expenditure exceeding income (Todd *et al.* 1988; Dolan & Russell 1988; Chapter 22). Reflecting a market perspective, no account is taken of shadow values for labour, non-market values of wood production (e.g. carbon substitution benefits associated with using wood instead of fossil fuels or more fossil fuel-intensive materials) or non-market ecosystem services (to the extent that these costs are partly associated with provision of these wider benefits). Therefore, they can be considered to significantly under-estimate marginal social values of wood production. However, in the absence of mechanisms that value such wider, non-market ecosystem services, profitability has tended to be low, with woodland grants and tax incentives being major influences on woodland planting rates. Forestry costs per hectare of woodland vary between sites, depending upon characteristics such as slope, road access, size of woodland and species planted. Establishment costs tend to be higher for broadleaves than conifers.

Net values vary considerably between sites, and depend upon species and rotation length choices; for example, Pryor and Jackson (2002) report mean net annual income estimates (that take no account of changing values over time and discounting) for 11 species/yield class (YC)/rotation length combinations considered typical of plantations on ancient woodland sites. These range from -£2/ha for birch YC6 to £262/ha for ash YC10 (both over a 70-year rotation) for broadleaves, and from £77/ha for larch YC12 to £247/ha for Douglas fir YC18 (both over a 45-year rotation) for conifers.

8.3.6.2 Regulating services: carbon sequestration

Estimates of the net amount of carbon (in the form of carbon dioxide) sequestered by woodland, and of changes in carbon stored in Harvested Wood Products (HWP), are available from the Centre for Ecology and Hydrology (CEH) (Dyson *et al.* 2009). Based upon a carbon accounting model (C-Flow) for woodlands planted after 1921 that includes transfers of carbon from living biomass (roots, trunk, branches, foliage) to litter and soils, estimates are produced on behalf of the government as part of the UK's commitments under the Kyoto Protocol (Chapter 22). The CEH estimates show net carbon sequestration in UK woodlands rising from 2.4 Mt of carbon dioxide in 1945 to a peak of 16.3 Mt of carbon dioxide in 2004, and then falling back to 12.9 Mt of carbon dioxide in 2009. Net sequestration rates for the period 2001 to 2009 have been estimated at around 5.2 t CO_2/ha across all UK woodlands (and 0.3 t CO_2/ha for HWP). An increase in net carbon sequestration by UK woodlands since 1945 is consistent with increased afforestation rates from the 1950s to 1980s, although exclusion of forests planted before 1921 probably results in lower estimates for the early part of the period than would otherwise have been the case. Carbon accounting methodology is still evolving and it is probable that current estimates from existing models underestimate carbon sequestration rates (Robertson *et al.* 2003; Dyson *et al.* 2009; Matthews & Broadmeadow 2009).

Permanence can be a key influence on valuation of carbon benefits, with the present value of the carbon at the point at which it is re-released to the atmosphere generally needing to be subtracted in valuing current gross sequestration. Where future carbon values are expected to rise, the present value of future emissions can even exceed the value of the original sequestration in some cases (Valatin 2010). However, permanence issues can be ignored if the total carbon stock in UK woodlands and HWP combined is expected to remain at least at the current level in perpetuity once account has been made for carbon substitution benefits (associated with using wood instead of fossil fuels or more fossil fuel-intensive materials).

A wide range of values have been used to value carbon. UK government guidance on valuing carbon (DECC 2010) is currently based upon a target-consistent approach which, at 2010 prices, includes using a central value of around £53/t CO_2 (equivalent to £193/t C) for 2009 for the 'non-traded' sector (i.e. that not covered by the EU emissions trading scheme). As forestry carbon sequestration is not covered by the ETS at present, values for the 'non-traded' (rather than the 'traded') sector are relevant, and in the absence of a consistent time-series of values back to 1945, the 2010 value is applied to value sequestration in previous years.

Valued at the DECC central estimate of £53/t CO_2 in 2009, the CEH estimates suggest that the gross value of net carbon sequestration by UK woodlands increased five-fold, from £124 million in 1945, to £680 million in 2009 at 2010 prices, with Scotland currently accounting for around two thirds of the total (**Figure 8.9**).

Values per hectare at the DECC central social value of carbon of £53/t CO_2 rise from £270/ha in 2001 to a peak of £304/ha in 2004, before falling back to a minimum of £239/ha in 2009 (Chapter 22); a mean of £276/ha is seen over this period. For woodland planted since 1921, per hectare values are significantly higher. Dividing by CEH estimates for these woodland areas suggests values of net carbon sequestration at 2010 prices of around £590/ha in 1945, falling to about £400/ha in 2009, valued at the DECC central social value of carbon of £53/t CO_2; a zero value is assumed for net carbon sequestration by woodlands planted prior to 1921.

The mean value of net carbon sequestration by woodlands and in HWP combined over the period 2001 to 2009 was £286/ha at 2010 prices valued at the central estimate of the social value of carbon (with £143/ha and £429/ha as the low and high social value estimates). The combined value remained fairly stable, rising from £279/ha in 2001 to £293/ha in 2004, before falling to £281/ha in 2009.

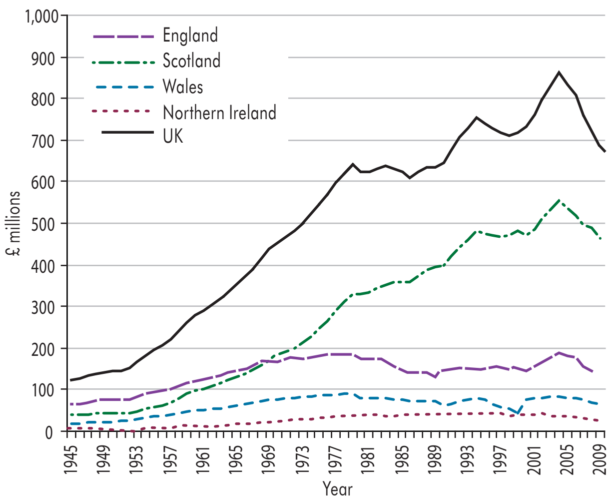

Figure 8.9 Value of annual carbon sequestration by UK woodlands (at 2010 prices). Based upon CEH estimates and the DECC (2010) central estimate of £53/tCO_2.

In addition to net sequestration (i.e. change in carbon stored), existing carbon storage in woodlands can also be valued relative to its release to the atmosphere; forest carbon stock estimates have been recently published (Forestry Commission 2010b). Valuing the total stored (803 Mt C) in 1990 at the central DECC estimate of £193/t C (£53/t CO_2) would imply a total value of £155 billion and an average gross value of around £59,000/ha at 2010 prices, increasing slightly over the past decade consistent with a positive per hectare net carbon sequestration rate*. However, over four fifths of this total is soil carbon which may vary relatively little with land use change in the short-run. Valuing carbon storage in above- and below-ground biomass, dead wood and litter alone (which may be more sensitive to land use change) at the central DECC estimate of £193/t C would imply a total value of £28 billion, and a per hectare value of around £11,000/ha at 2010 prices (equivalent to an annuity value of £380/ha/yr at a 3.5% discount rate). In contrast to the net sequestration values, carbon storage values could be expected to be highest in woodlands planted prior to 1921, other factors being equal.

8.3.6.3 Valuation: summary and concluding remarks

Paucity of data and different reporting periods precludes identification and comparison of trends in values for each ecosystem service provided by UK woodlands. Nonetheless, the review of changes in values of two ecosystem services has highlighted some clear differences in magnitude and trend (**Table 8.12**). The sharp decline in hardwood prices is consistent both with reports of low import price and with a shift in woodland management objectives away from timber production towards provision of multiple ecosystem services. In the case of the per hectare hardwood values, a decline is also consistent with an expansion in the total area covered by broadleaves and the associated transition to a lower average age of stands.

The estimates of gross values take no account of capital and labour costs, but surveys of private woodland owners

* Based instead upon the carbon stock estimate of 790 Mt C cited in the Read Report (Read *et al.* 2009) and elsewhere in this chapter, the estimates are £125 billion and £58,000/ha respectively.

Table 8.12 Changes in values of ecosystem services from UK woodlands (at 2010 prices).

	Annual value (£ millions)				**Annual value per ha** (£/ha)		
Ecosystem service	**1945**	**1976**	**1990**	**2009**	**1990**	**2001**	**2009**
Softwood production *		112	145	105	95	55	68
Hardwood production †			110	8–26	120		7–25
Net carbon sequestration by woodlands ‡	124	563	642	680	246	270	239
Net carbon sequestration in harvested wood products ‡	0	13	77	119	29	8	42

* Estimates based upon prices for standing sales from Forestry Commission land.
† Dates differ: the first year is 1989 and second 2007/08. Estimates based upon average prices for both standing sales and direct production (data comparability unclear).
‡ Based upon the central estimate of the social value of carbon for 2009 recommended for non-ETS sectors (DECC 2010) at 2010 prices; per ha estimate for 1990 derived by dividing by UK forest area from Forestry Commission (2010a).

suggest these averaged around £200/ha–£300/ha during 1961 to 1986 at 2010/11 prices (recent information on total Forestry Commission expenditure in England of £275/ha in 2007/08 suggests these costs are currently of a similar magnitude). To the extent that these costs are common to the provision of each ecosystem service provided by woodlands (including those considered in other chapters), gross estimates provide a useful rough indication of relative social value.

Although sensitive to the social values of carbon used and to assumptions of permanence, at present, the highest values are for carbon sequestration by woodlands. Remaining largely a non-market value, however, there is little incentive currently for landowners to increase provision of this ecosystem service or to maintain existing carbon storage.

The extent both of ecosystem service provision, and to which trade-offs or synergies exist in their provision, is sensitive to land management approaches and objectives (Chapter 5). For example, stalking and venison production are complementary activities to wood production and habitat conservation in many cases, but where the primary focus of land management may largely be substitutes.

Significant data gaps include the lack of consistent time-series data on hardwood standing sales prices, on woodfuel, on venison and stalking revenues, and on forestry costs. Given this lack of data, knowledge gaps remain concerning spatial differences in marginal values between regions, different woodlands types and forestry management approaches. More fundamentally, the marginal value of individual ecosystem services net of capital and labour costs remains indeterminate in the absence of an accepted approach to apportioning forestry establishment and management costs between different ecosystem services. Under these circumstances, any analysis of issues of trade-offs and synergies, and of optimal land use, will have to rely on comparisons of differences in the total net value (including non-monetised contribution to human well-being) of ecosystem services provided. However, in view of existing data and knowledge gaps, including the need still to develop a suitable methodology to quantify some ecosystem services (e.g. spiritual values), robust estimates of total net values to facilitate comparisons of alternative land uses seem elusive at present.

Figure 8.10 Carbon sequestration is one of the most important ecosystem services of woodlands; at peak growth, coniferous forest can sequester around 24 tonnes of carbon dioxide per hectare per year, with a net long-term average of around 14 t CO_2/ha/yr. Fresh needles signal spring growth in Sitka spruce. *Photo courtesy of: FC Picture Library / Isobel Cameron.*

8.4 Trade-offs and Synergies Among Woodland Goods and Ecosystem Services

"Newly planted forests may, at times, offend the aesthetic conscience, and feelings are stirred by some aspects of their early growth and by fears regarding their future development." (W.L. Taylor 1946)

"All these examples, from Burns onwards, demonstrate the post-Romantic, post-Enlightenment conflict between use and delight which is the constant theme in the environmental history of the last two centuries. After all, the various landowners....were only trying to realise a timber crop, or to plant efficiently to suit the tree species to the soil. In the eyes of their critics, however, they were destroying the natural and the beautiful which, by being in the public eye, belongs to us all." (Christopher Smout 2000)

8.4.1 Introduction

Previous sections have highlighted the trends in policy and management of woodland since 1945. Some of these changes have reflected the shift in emphasis of woodland expansion and management from a dominant production focus, to one targeting a wider set of values (as espoused in the terminology of 'multipurpose forestry' and 'sustainable forest management'). Implicit in these shifting objectives is the requirement to make choices about the combination of goods to be sought, and which methods are to be deployed to achieve them. These decisions and techniques are further elaborated in Section 8.5. In this section, examples are provided of particular trade-offs and synergies (**Table 8.13**); note that interactions between UK NEA Broad Habitats are covered in Section 8.1.3.

8.4.2 Synergies in the Provision of Services

The following examples illustrate where focusing on the delivery of one group of services can provide the simultaneous supply of other services.

8.4.2.1 Achieving provisioning and regulating services

New forests have been identified as a means of increasing carbon storage and achieving some climate change mitigation through the expansion phase (i.e. before the forests reach a steady state). Faster growing species or the use of better quality sites will not only fix carbon quicker, but also produce timber of an utilisable size quicker (**Figure 8.10**). Felling may take place sooner and, provided that the carbon is then stored (e.g. through embedding in construction), another rotation can be established to further sequester carbon. Such synergies have attracted policy support, with new woodland expansion targets for Scotland, Wales and England.

8.4.2.2 Achieving provisioning and cultural services

The character of many of the UK's woodlands reflects the interaction between past policies, past management and site quality. Many lowland woodlands still show the legacy of wartime felling and subsequent regeneration, or planting, leading to a cohort of woodlands that have now reached a particular stand stage (Section 8.2.2; **Figure 8.2a,b**). This stage is known in ecological literature as 'stem exclusion' and is characterised by dense canopies, the shading of the understorey and the restriction of physical access. Natural stand dynamics, driven by storms, insect infestation or disease, will eventually lead to the opening up of such woods, but the length of time this will take is at odds with the demands of today's society. Stand management, in the form of thinning, can harvest some of the production (provisioning). In doing so, it will increase the light levels within the stand, leading to improved conditions for ground and shrub layers (and those organisms, such as woodland birds, that depend upon them), and improved access for recreational activities (including the gathering of non-timber forest products). Reinstatement of a coppice regime could foster crafts and traditional skills, maintain a supply of niche products and woodfuel, and bring biodiversity benefits.

8.4.2.3 Achieving regulating and cultural services

Natural flood management is being increasingly promoted as a way of reducing the costs of flooding to society, coping with intense rainfall events as a result of climate change, and reducing the cost of hard-engineering solutions. Restoration of riparian/floodplain woodland may delay and reduce flood peaks. Such restoration would also lead to the reinstatement of habitats that have been lost over large parts of Europe, favouring a range of aquatic and terrestrial organisms, providing wildlife corridors and making a significant contribution to landscape quality and diversity.

8.4.3 Trade-offs in the Provision of Services

The following examples illustrate where focusing on the delivery of one group of services can negatively impact on the supply of other services.

8.4.3.1 Trade-off between provisioning and regulating services

Trees must be felled in order to make fibre available for use in timber products, construction, paper-making, or fuel. However, this felling removes the carbon store in the above-ground parts of the tree, and may lead to increased rates of soil carbon loss as a result of changes in the microclimate and soil. The magnitude of this trade-off depends upon the end use of the tree products (including the extent to which these products become an off-site store), and the rate of restoration of forest conditions after felling. The former depends upon markets, the latter on management options, including the scale of the felling intervention, and the manner by which a successor tree crop is established. Similarly, water quantity may be increased, but quality decreased, by felling, and regulation is diminished until vegetation cover re-establishes.

Table 8.13 Six examples of trade-offs and synergies in provision of ecosystem services and human well-being provided by woodlands.

	Provisioning services	Regulating services	Cultural services
Provisioning services		**Synergy 1:** Increased growth and production can increase carbon storage.	**Synergy 2:** Thinning of neglected woodland can open access for visitors and improve habitat for wildlife (enhancing opportunities to observe); opportunities may be provided for employment, volunteering and craft development.
Regulating services	**Trade-off 1:** Increased harvest reduces carbon sink/store unless products are long-life.		**Synergy 3:** Restoration of riparian woodland to aid flood regulation may enhance landscape and opportunities for recreation (including fishing).
Cultural services	**Trade-off 2:** Increased production may reduce quality of woodland environment for recreation (e.g. increased traffic and machinery), and reduce visual quality (e.g. clearfells).	**Trade-off 3:** Most efficient carbon capture may be with novel crops/species that are not familiar or liked.	

8.4.3.2 Trade-off between provisioning and cultural services

Removal of trees, as part of the supply of a provisioning service, is increasingly mechanised in developed countries such as the UK both for cost reasons and because of the unattractiveness of the manual forms of harvesting to the labour pool. Large machines bring a form of industrialisation to what is regarded as a natural habitat by many, and may require a large-scale of working. There are potential conflicts between harvesting operations and recreational use as access may be restricted to prevent accident, and the resulting change of character may influence the perceived attractiveness of the area. Heavy machinery may also disturb wildlife and reduce opportunities to view charismatic species, at least in the short-term (in the medium-term, the habitat manipulation may be beneficial). Clear-felling silvicultural systems are an efficient way of growing fibre, particularly when using pioneer, light-demanding species, and are well-suited to mechanisation. But clear-felling creates rapid and large-scale change in the landscape that attracts unfavourable comment and dramatically affects visual appearance.

8.4.3.3 Trade-off between regulating and cultural services

Trials of eucalyptus and other fast-growing species are under way as a source of biomass for biofuel. Their very rapid growth rates (which result in swift carbon capture), are expected to be many times greater than those of native broadleaved tree species on the same site type (Matthews & Broadmeadow 2009). The rate of carbon sequestration and renewable fuel production will have to be considered in the light of obvious trade-offs. These include:

- concerns over eucalyptus and its high water-usage;
- potential invasiveness into other ecosystems (something associated with use of the genus in other countries such as Spain and Portugal);
- poor quality habitat due to leaf-fall and slow leaf decomposition leading to suppression of the ground flora;
- uncertainties over impact on landscape aesthetics, with conflicting public opinion (distinguishing eucalypts from other species may be difficult for many, especially at a distance, but large-scale harvesting is unlikely to be appreciated: Section 8.4.3.2);
- extensive recreational use is unlikely to be compatible with the optimal stand structure for fuel production; and
- the increased fire hazard of using eucalypts, particularly in light of predictions of increased summer drought.

8.4.4 Choices in Trade-offs and Synergies

Section 8.5 provides a review of the policy and management instruments that govern the specification of woodland management, and how a balance is struck between competing or synergistic demands. Choices are governed by planning tools such as cost-benefit analysis, standards, consultation and Environmental Impact Assessment.

8.5 Options for Sustainable Management

"A culture is no better than its woods." (W.H. Auden 1955)

8.5.1 Introduction

The concept of sustainability in woodland management has a long history. Osmaston (1968) describes the development of forest management for sustained yields of timber in Continental Europe over 700 years ago. A 14th Century French ordinance required management such that "forests can perpetually sustain a good state" (Huffel cited in Osmaston 1968), and by the mid-19th Century, the principle of sustained yield was understood and applied in France, Germany and Austria. During the latter part of the 20th Century, the concept of sustainable forest management evolved to encompass the wider ecological and social functions of forests (FAO 1993).

Today, UK forestry is making substantial progress in the elaboration of approaches to sustainability and active land use, and in developing the concepts of management to provide for multiple goods and services. A number of recent developments exemplify the options available to enhance ecosystem services. An increasing area of both state and privately owned woodland has gained certification status under the United Kingdom Woodland Assurance Standard (UKWAS 2008); this is recognised as the standard for sustainable forest management in the UK by the Forest Stewardship Council (FSC). As a consequence, there has been increased interest in lower-impact silvicultural systems used in continuous cover forestry (*sensu* Mason). Many of the ancient woodlands planted with non-native conifers are being restored to native woodland communities (Thompson *et al.* 2003). There is developing interest in adaptation to climate change (Ray 2008) and to the opportunities for woodlands to play a role in climate change mitigation through sequestration and substitution (Broadmeadow & Matthews 2003; Read *et al.* 2009). The broader context and specific options are now examined in more depth. Note that much of the literature on this subject uses the term "sustainable forest management" (rather than sustainable woodland management); this and the word 'forest' are retained where they are used in the source literature.

8.5.2 Sustainable Forest Management in Europe

The Ministerial Conference on the Protection of Forests in Europe (MCPFE) was founded in 1990 to develop common strategies for its 46 member countries and the EU on the protection and management of forests. In 1993, the Second Ministerial Conference produced general guidelines for the sustainable management of forests in Europe, defining sustainable forest management as:

> "the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economic and social functions at local, national and

Table 8.14 MCPFE criteria and quantitative indicators for sustainable forest management at the national level.

Criteria	No.	Indicator
Maintenance and appropriate enhancement of forest resources and their contribution to global carbon cycles	1.1	Forest area
	1.2	Growing stock
	1.3	Age structure and / or diameter distribution
	1.4	Carbon stock
Maintenance of forest ecosystem health and vitality	2.1	Deposition of air pollutants
	2.2	Soil conditions
	2.3	Defoliation
	2.4	Forest damage
Maintenance and encouragement of productive functions of forests (wood and non-wood)	3.1	Increment and fellings
	3.2	Roundwood
	3.3	Non-wood goods
	3.4	Services
	3.5	Forests under management plans
Maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems	4.1	Tree species composition
	4.2	Regeneration
	4.3	Naturalness
	4.4	Introduced tree species
	4.5	Deadwood
	4.6	Genetic resources
	4.7	Landscape pattern
	4.8	Threatened forest species
	4.9	Protected forests
Maintenance, conservation and appropriate enhancement of protective functions in forest management (notably soil and water)	5.1	Protective forests—soil, water and other ecosystem functions
	5.2	Protective forests—infrastructure and managed natural resources
Maintenance of other socio-economic functions and conditions	6.1	Forest holdings
	6.2	Contribution of forest sector to GDP
	6.3	Net revenue
	6.4	Expenditure for services
	6.5	Forest sector workforce
	6.6	Occupational health and safety
	6.7	Wood consumption
	6.8	Trade in wood
	6.9	Energy from wood resources
	6.10	Accessibility for recreation
	6.11	Cultural and spiritual values

global levels, and that does not cause damage to other ecosystems" (MPCFE 1993)

Pan-European criteria and indicators for sustainable forest management were agreed at the Third Ministerial Conference (MPCFE 1998) and revised in 2002 (MCPFE 2002). The current set of quantitative indicators is summarised in **Table 8.14** (there are also 17 qualitative indicators which are not shown). Progress towards sustainable forest management at the national level in the 46 MPCFE member countries, assessed using these criteria and indicators, is described in The State of Europe's Forests 2007 (MPCFE 2007). Forests in the UK, like most of those elsewhere in Europe, appear to be in "a comparatively good state" and delivering a range of ecosystem services.

The criteria and indicators used for monitoring and reporting at a national level are of little practical help to owners and managers trying to practise sustainable management at the level of an individual forest or a forested landscape (Forestry Commission 2004; Bell & Apostol 2008). Pan-European operational-level guidelines for sustainable forest management (MCPFE 1998) go some way to addressing this problem, but need to be modified for local economic, environmental, social and cultural conditions.

8.5.3 Sustainable Forest Management in the UK: the UK Forestry Standard

In GB, the publication of Forestry Commission guidelines on environmental and cultural aspects of forest management started in the early 1990s, before the appearance of the MPCFE guidelines for the sustainable management of forests in Europe. The Forestry Commission guidelines do not use criteria and indicators, but take a practical approach to management of a particular component of the forest ecosystem, or of particular forest features that have cultural or social value. The current guidelines cover nature conservation, recreation, landscape design, historic environment, soil conservation and water, and revised guidelines have recently been prepared (Forestry Commission in press a,b&c).

Starting in 1995, the existing guidelines and developing ideas on criteria, indicators and standards were brought together into a single document, the UK Forestry Standard. Consultations took place in 1996 and 1997, and the first edition was published in 1998; a second edition was produced in 2004 to reflect devolution in the UK (Forestry Commission 2004), and the third edition is due to be published in 2011 following consultation (Forestry Commission 2009c; R. Howe pers. comm.). In the international arena, the UK Forestry Standard shows how international protocols on sustainable forest management are implemented here. It applies to all forests and woodlands in the UK and provides a performance standard for sustainable management at the level of the forest management unit, normally the area subject to a forest management plan or proposal (Forestry Commission 2009c).

The third edition of the UK Forestry Standard identifies eight elements of sustainable forest management: general forestry and legal conformity; forest planning and general forestry practice; forests and landscape; forests and biodiversity; forests and water; forests and climate change; forests and soils; and forests and people. For each element, legal requirements and good forestry practice requirements are identified, and sustainable forest management is demonstrated by full compliance with both sets of requirements. There are about 90 requirements in total, of which about a third are legal requirements. Many of the legal requirements relate to legislation that implements EU Directives and is not specific to forestry (e.g. the EU Habitats Directive and the EU WFD). A few, such as the requirement to obtain a licence for felling trees, relate to GB legislation under the Forestry Act (and its amendments). A Woodland Carbon Code is also being developed (www.forestry.gov.uk/website/forestry.nsf/byunique/infd-863ffl) to set standards for voluntary carbon sequestration projects that incorporate core principles of good carbon management as part of modern sustainable forest management.

8.5.4 Forest Certification: the UK Woodland Assurance Standard (UKWAS)

Increasingly, buyers of wood and wood products wish to be assured that these products have been sourced from sustainably managed forests. Forest certification schemes provide independent verification of sustainable forest management, allowing forest-managers to claim that they are meeting specified standards of management and, therefore, to market their wood more effectively. Two certification schemes operate in the UK (Bills 2001)—the Forest Stewardship Council (FSC) and the Programme for Endorsement of Forest Certification (PEFC)—and 45% of the UK's forests are certified (Forestry Commission 2009d). In addition, all wood currently produced from Forestry Commission land in Britain, and from Forestry Service land in Northern Ireland, as well as around 70% of wood from non-Forestry Commission/Forestry Service land, is certified as meeting sustainable forest management criteria.

The UKWAS is a voluntary certification standard that can be used by independent certification organisations, such as FSC and PEFC, for the certification of UK woodlands. The certification standard has eight sections that deal with different aspects of woodland management: compliance with the law and conformance with the requirements of the certification standard; management planning; woodland design (creation, felling and replanting); operations; protection and maintenance; conservation and enhancement of biodiversity; the community; and forestry workforce. The standard sets out a number of requirements (criteria) in each of the eight aspects of woodland management, and suggests means of verifying (indicators) that the requirements have been met.

Although its eight sections do not map exactly on to the nine elements identified in the UK Forestry Standard, UKWAS is "designed to ensure that it reflects the requirements of the Government's UK Forestry Standard" (UKWAS 2008). UKWAS emphasises the operational aspects of sustainable woodland management, which is helpful for woodland owners and managers. Thus, rather than identifying landscape, water, climate change and soils as distinct elements of sustainable woodland management (as is done in the UK Forestry Standard), UKWAS includes them in guidance on management planning, woodland design and operations. UKWAS claims that its process attracts international interest from countries setting up their own national processes, and that this is a measure of its success. But it is not clear if the process has resulted in improved woodland management, nor whether certified woodlands are any better managed than the 55% of UK woodlands that are not currently certified.

8.5.5 Implementing Sustainable Forest Management

Innes *et al.* (2009) discuss the scale at which planning for sustainable forest management should take place. Strategic plans should be used for the long-term management of all the forest resources in a large area; this is planning at the catchment- or landscape-scale. Tactical plans, equivalent to forest management plans in the UK, cover a shorter period and focus on the implementation of strategic plans. Operational plans describe what is to be done on small areas in the short-term (one to five years), and are equivalent to the work programmes that form part of many forest management plans in the UK. It is important that the different levels of plan are consistent with each other, and problems can arise if they are drawn up by different individuals or organisations. In the UK, tactical plans (forest management plans) and operational plans (work programmes) are usually prepared by the same person (the forest manager or planner), while strategic plans are normally the responsibility of organisations such as local authorities or national park authorities.

Efforts have been made to translate higher-level policies into operational guidelines. But however willing woodland owners and managers may be to deliver the ecosystem services demanded by society, their ability to do so depends on the size and location of their woodlands, and the cost of the management required to provide particular goods and services.

8.5.5.1 Management objectives for woodlands

Owners have different objectives for their woodlands, and different ways of achieving them. The UK Woodland Assurance Standard recognises the diversity of woodland management in the UK, considers this to be valuable in its own right, and acknowledges that owners and managers will decide how best to meet the requirements of the certification standard. Many management objectives are directly related to ecosystem services (e.g. timber production is a provisioning service, recreation is a cultural service and biodiversity is a supporting service), but few, if any, woodland owners and managers think in terms of ecosystem services when setting objectives.

Many authors, such as Bell and Apostol (2008), have suggested that the mix of goods and services that a forest provides should be a matter of negotiation between the forest owner and the local community, and should also take account of wider interests in forests. But who pays the management costs of delivering such negotiated goods and services? Owners can receive revenue from the sale of goods (provisioning services) and sometimes from recreation (cultural services), and these may offset or exceed management costs. At the present time, payments are not made to owners who are providing supporting or regulating ecosystem services from their forests (Valatin 2010).

8.5.5.2 Size of woodland management units

Small (less than 10 ha), individual woodlands can make an important contribution to the delivery of ecosystem services at the larger, landscape-scale. Yet, the owners of small woodlands will often only have a very limited number of objectives.

Larger woodlands (several hundred hectares) can deliver more services, and the owners of larger woodlands will normally have several objectives. The current edition of the UK Forestry Standard warns that "[while] compromise is accepted as necessary in these circumstances, loss of potential benefits through neglect or mismanagement must be avoided" (Forestry Commission 2004). More than half (55%) of the UK's forests are multipurpose and managed for more than one objective, but how do owners ensure that there is no "loss of potential benefits" from their forests? If the forest is in single ownership, it may be possible to divide it into a number of zones, concentrating different activities (e.g. recreation, intensive timber production) in different zones. This is a straightforward way of ensuring that multiple objectives are achieved, but it may not optimise the delivery of ecosystem services, and may not be a practical option for every woodland.

At the landscape-scale, woodlands may cover several thousand hectares and could, in theory, deliver the full range of ecosystem services. However, the problem of optimising the delivery of ecosystem services is the same as for a large woodland block, and is compounded by the fact that the woodlands are likely to be in different ownerships, with owners who may have different objectives.

8.5.5.3 Management plans

The importance of management plans is recognised in both the UK Forestry Standard and the UKWAS. Only half of UK woodlands currently have a management plan; increasing the coverage of management plans is central to sustainable forest management and the delivery of ecosystem services.

In Wales, the Better Woodlands for Wales scheme paid grants for activities against an approved 20-year management plan that identified the main features of the woodland; the features were chosen from a list, some of which were clearly ecosystem services. In the plan, the desired characteristics were listed, current and target levels of each desired characteristic were stated, and the monitoring of progress towards target levels was explained. This approach allowed managers to identify management conflicts and complementarities, and consider ways of resolving them or arriving at compromises. The scheme closed at the end of 2010, after only five years of operation. This illustrates the wider problem of forestry policies changing more rapidly than management plans can be implemented, resulting in a lack of continuity and a potential waste of public funds.

8.5.5.4 Silviculture

Achieving most (though not all) of the objectives of forest management requires the use of silviculture. Indeed, silviculture can be defined as the manipulation of forest stands to accomplish a specified set of objectives (Lorimer 1982). Operational forest plans (work programmes) are largely concerned with silviculture and the application of silvicultural treatments such as thinning, pruning, site preparation, planting/direct-seeding, establishment and protection. Silvicultural treatments can be organised into planned, long-term programmes for tending, protecting, harvesting and regenerating forest stands; these are 'silvicultural systems' (Matthews 1991; Smith *et al.* 1997). In some respects these systems are stylised ideals, but the concept provides a useful framework for thinking about long-term management at the level of the individual stand.

The silvicultural system used for a stand determines its age structure and (through the methods of harvesting and regeneration) species composition, which both influence the ecosystem services that can be provided from the stand. In the last decade, there has been considerable interest in moving away from clear-cutting, the most common silvicultural system in the UK, to alternatives such as shelter-wood and selection systems. This is expected to have a positive impact on the delivery of ecosystem services such as soil protection, flood and water protection, carbon sequestration and the character of the wider landscape, although internal landscapes may be negatively affected. There are particular challenges in developing systems that permit the transformation of forests from plantations to more diverse structures (Mason 2006).

8.5.6 Future Developments to Enhance the Provision of Ecosystem Services

The MCPFE concept of sustainable forest management is useful at a national level, but cannot be achieved at the woodland level in the UK. Here, as elsewhere in the world, the majority of individual woodlands fulfil only a proportion of the requirements of sustainable forest management (Innes *et al.* 2009). Bell and Apostol (2008) question whether criteria, indicators, standards and guidelines really help responsible forest owners and managers to do what they are trying to do: sustain the forest's capacity to deliver goods and services, and meet the needs of future generations. What else can be done to help these people enhance the ecosystem services provided by the UK's forests and woodlands?

Multi-criteria decision analysis is a technique used by decision-makers for evaluating alternative courses of action in situations where there are conflicts. Because it incorporates the preferences of the decision-maker, it is a subjective method, but it has potential for optimising the delivery of ecosystem services from woodlands at national or regional level.

At woodland level, owners and managers need *decision support tools* to help them choose between alternative, and perhaps conflicting, management options. Several decision support tools are already available such as the wind risk model ForestGALES (Gardiner *et al.* 2004), the Ecological Site Classification (Pyatt *et al.* 2001) and a knowledge management system for habitats and species HARPPS (Ray & Broome 2007). So far, these are mainly implemented at site or stand level, rather than woodland level, not least because of the lack of appropriate digital soil data.

Payments for ecosystem services may encourage and enable owners to manage their woodlands to deliver services that currently have little or no financial value. Public payment schemes and both regulatory and voluntary markets for ecosystem services exist in a number of countries and natural resource sectors (Perrot-Maître 2006; Richards & Jenkins 2007; Jack *et al.* 2008; Turpie *et al.* 2008), and these should be explored for the forestry sector in the UK.

Adaptive management, in which the effects of different management options are tested either experimentally or by modelling, can be particularly valuable at times of change or uncertainty. Experimental testing of options is problematic in forest management, given the timescale on which forestry operates, but there is scope for learning from implementation. However, in addition, a modelling approach has considerable potential for examining the effects of management on the flows of ecosystem services from forests, and for ensuring that these flows are maintained under future climates (Innes *et al.* 2009).

8.5.7 Kielder Forest: a Case Study of Changing Objectives and Adaptive Management

The parallel evolution of growing forests, changing policies, developing markets, and shifting societal demands, has been an underlying theme of this chapter. The integration of these to shape the development of a forest and its services is exemplified by this case study of Kielder Forest.

Covering some 50,000 ha, Kielder Forest in Northumberland in northern England is one of the largest man-made forests in Europe (McIntosh 1995). Over the past 80 years, Kielder has been at the forefront of delivering forest ecosystem services in response to changing economic demands, public aspirations and government policies relating to the environment. A summary of the changes in importance over time of the various ecosystem services is shown in **Table 8.15**.

Afforestation began in 1926 and continued into the 1980s with a focus on raw timber as the primary provisioning services (**Table 8.15**). At first, a mix of non-native conifer species were planted, including Sitka spruce, Norway spruce, and Scots and lodgepole pine, but over time the superior growth rate of Sitka spruce was recognised and used to a such a degree that it currently comprises 72% of the forest (McIntosh 1995).

In the early part of the 20th Century, state forestry policy was largely focused on creating a strategic timber reserve and providing rural employment (Edlin 1952; McIntosh 1995). In Kielder, this led to the establishment of a village to house forestry workers (with pre-chainsaw predictions of the community developing from tens to hundreds of workers) and a large-scale planting programme. Timber production remains one of the key ecosystem services provided by

Table 8.15 Changes over time in the delivery of different ecosystem services over time at Kielder Forest.
- no delivery; -/+ some delivery but not significant; + delivery; ++ significant delivery; +++ very significant delivery.

Ecosystem service category	Main services	Time periods			
		1920–1960	1960–1980	1980–1995	1995–2010
Provisioning services	Food	-	-	-/+	-/+
	Freshwater	-	-/+	+++	++
	Wood, fibre, employment	+++	+++	++	++
	Fuel	-/+	-/+	-/+	+
Regulating services	Climate regulation	-	-	-	++
	Flood regulation	-	-	+	+
	Disease regulation	-	-	-	-
	Water purification	-	-	+	+
Cultural services	Aesthetic	-	++	+++	+++
	Spiritual	-	-/+	++	++
	Educational	+	+	++	++
	Recreational	-	++	+++	+++
Supporting services	Biodiversity	-	-/+	++	+++
	Soil formation and nutrient cycling	+	+	+	+

Figure 8.11 Kielder Forest showing its patchwork of forest stands of different ages, rides, open habitats, and felled areas.
Source: Forestry Commission Kielder Forest District derived from IKONOS 2004 imagery.

Kielder Forest, yielding approximately 1,400 tonnes of timber a day on a sustainable basis and providing the UK with 5% of its softwood requirements (GPFLR 2001).

By the 1960s, public dislike of large, even-aged blocks of conifer woodland initiated action to visually 'soften' forest edges, ensure that areas of new planting were designed to be in-keeping with the landscape, and create a more aesthetically pleasing experience for the visitor (**Table 8.15**). From 1974 to 1980, over 400 ha of forest were cleared to make way for the establishment of the Kielder Water reservoir. Together with an improved road network, the creation of this resource had a significant impact on recreational use of the forest; today, over half a million visitors make use of the forest for sight-seeing, cycling, horse riding and other outdoor pursuits (GPFLR 2001). The forest also continues to have a role, albeit a minor one, in regulating, water quality and quantity in the Kielder Water catchment area (Dunn & Mackay 1995).

In the early 1980s, it became clear that biophysical constraints of the Kielder Forest area placed a significant restriction on tree growth. In particular, shallow, wet soils and high exposure combine to increase the risk of windthrow, so that stands rarely survive beyond 50 years of age (McIntosh 1995; Mason & Quine 1995). A programme of restructuring to create a more diverse and mixed-aged forest was initiated, not only to address the problem of windthrow, but to ensure a more sustainable timber supply over the longer-term and to enhance recreation, visual impact and biodiversity (Hibberd 1985). Today, the forest landscape is made up of a patchwork of different types and age of forest stand, open space and rides (**Figure 8.11**). By the 1990s, the value of the forest for wildlife conservation and biodiversity was being increasingly recognised (Petty *et al.* 1995). A wide range of habitats are provided for species groups such as raptors (Petty *et al.* 1995), songbirds (Patterson *et al.* 1995; Fuller & Browne 2003), plants, fungi and invertebrates (Butterfield *et al.* 1995; Humphrey *et al.* 2003). Today, amongst general objectives for increasing habitat and species diversity, there is a special focus on the restoration and management of mire habitats (Smith *et al.* 1995) and the protection and enhancement of red squirrel populations (Lurz *et al.* 1995). In the Strategy for England's Woods, Trees and Forests (Forestry Commission 2007c) there is specific mention of the forests of the north-east as having key strategic importance in sustaining England's red squirrel population. The strategy also reflects a sharpening of government forestry policy with respect to ecosystem services. In the 1990s, the Forestry Commission was charged with managing the public forest estate for 'public benefits' (Forestry Commission 1993), and now those benefits are more specifically articulated and costed (Forestry Commission 2007c).

Three recent developments in government policy have also initiated a shift in emphasis for forest management: The Climate Change Act; the Renewable Energy Strategy and the implementation of the EU WFD. Not only do forests need to protect and enhance environmental resources of water, soil, air, biodiversity and landscapes, they also have a role in mitigating the impact of climate change, and promoting new and improved markets for sustainable wood products such as wood fuel (**Table 8.11**). Translation of policy objectives into practice is achieved through forest design planning. The forest area is divided into management units ranging from 1,000–10,000 ha and felling and restocking plans drawn-up in relation to strategic objectives (Graham Gill pers. comm.). The planning process allows a range of environmental, social, and economic priorities to be considered at the appropriate spatial scale, and a number of computer-based decision support tools are available to help determine management priorities. These include site evaluation tools, such as Ecological Site Classification (Pyatt *et al.* 2001), and ForestGALES (Gardiner *et al.* 2004), as well as tools for modelling ecological connectivity and biodiversity enhancement, such as BEETLE (Watts *et al.* 2007). Research is also beginning to make available tools for assessing the sustainability of whole forest-wood-chains ([FWC] i.e. from forest to wood product), which allows modelling and evaluation of the impacts of different management options on the provision of ecosystem services throughout the chain (Lindner *et al.* 2009). In the future, Kielder Forest is likely to provide an important test area for how effectively the forest industry can continue to deliver a mix of ecosystem services in response to evolving environmental and policy drivers.

8.6 Future Research and Monitoring Gaps

Identification of gaps in data, knowledge and understanding is a benefit of the integration of knowledge stimulated by the UK NEA. The following are amongst the key areas that require further investigation as shown by the material drawn together for this chapter.

8.6.1 Climate Change and Other Threats

- Development and monitoring of approaches to adaptation to climate change in forest management;
- Understanding the interaction of climate change and some forms of service delivery; meeting the challenges of perpetuation of cherished species and habitats;
- Effective management and control of new/emerging pests and diseases, and established (expanding) populations of deer and grey squirrels.

8.6.2 Valuation

- Methods of monetising or finding ways of comparing the value of different services;
- Comparison of marginal versus absolute benefits of changing extent or management of woods;
- Improved understanding of differences in marginal values between regions, between different woodlands types and forestry management approaches;
- An accepted approach to apportioning forestry establishment and management costs between the provision of different ecosystem services.

8.6.3 Condition

- Comprehensive data on the extent and condition (including components of supporting services) of the broad habitat, in particular, native woodland/semi-natural woodland;
- The status and trends of novel woodland habitats and improved biodiversity monitoring of plantation forests.

8.6.4 Integrated Land Use and Landscape-scale Action

- Planning methods and incentivisation of owners to achieve landscape-scale action across large spatial scales (e.g. catchments);
- Promotion and monitoring of changing, landscape-scale impact of new woodland and tree cover.

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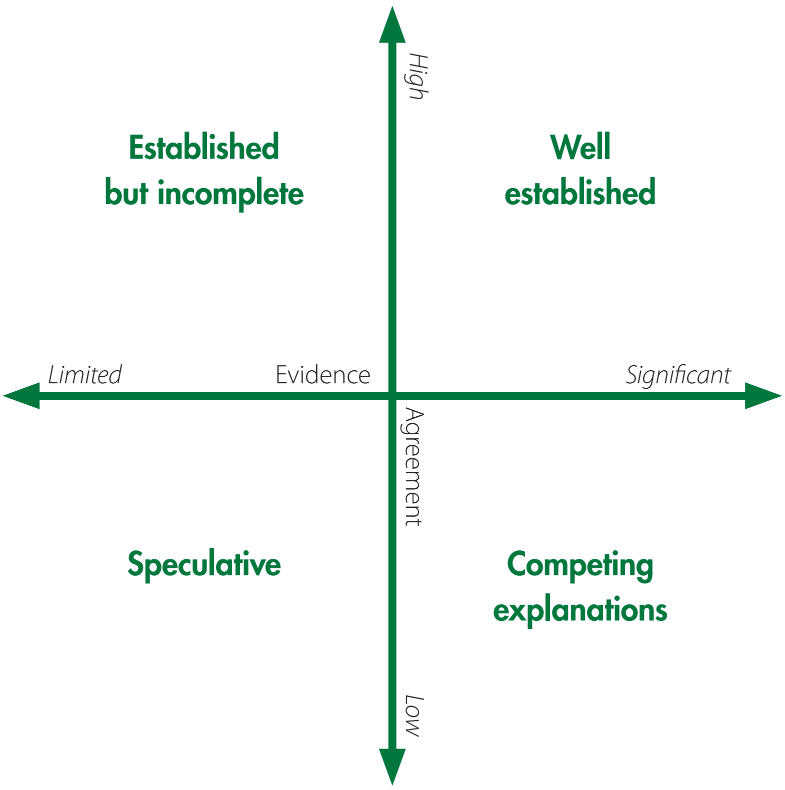
Appendix 8.1 Approach Used to Assign Certainty Terms to Chapter Key Findings

This chapter began with a set of Key Findings. Adopting the approach and terminology used by the Intergovernmental Panel on Climate Change (IPCC) and the Millennium Assessment (MA), these Key Findings also include an indication of the level of scientific certainty. The 'uncertainty approach' of the UK NEA consists of a set of qualitative uncertainty terms derived from a 4-box model and complemented, where possible, with a likelihood scale (see below). Estimates of certainty are derived from the collective judgement of authors, observational evidence, modelling results and/or theory examined for this assessment.

Throughout the Key Findings presented at the start of this chapter, superscript numbers and letters indicate the estimated level of certainty for a particular key finding:

1. *Well established:*	high agreement based on significant evidence
2. *Established but incomplete evidence:*	high agreement based on limited evidence
3. *Competing explanations:*	low agreement, albeit with significant evidence
4. *Speculative:*	low agreement based on limited evidence



a. *Virtually certain:*	>99% probability of occurrence
b. *Very likely:*	>90% probability
c. *Likely:*	>66% probability
d. *About as likely as not:*	>33–66% probability
e. *Unlikely:*	<33% probability
f. *Very unlikely:*	<10% probability
g. *Exceptionally unlikely:*	<1% probability

Certainty terms 1 to 4 constitute the 4-box model, while *a* to *g* constitute the likelihood scale.