Forestry Commission

Research Report

Understanding the carbon and greenhouse gas balance of forests in Britain

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James Morison, Robert Matthews, Gemma Miller,
Mike Perks, Tim Randle, Elena Vanguelova,
Miriam White and Sirwan Yamulki

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Enquiries relating to this publication should be addressed to:

Forestry Commission
231 Corstorphine Road
Edinburgh
Scotland, EH12 7AT

T: 0131 334 0303
E: publications@forestry.gsi.gov.uk

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The authors can be contacted through:

James Morison
Forest Research
Centre for Forestry & Climate Change
Alice Holt Lodge
Farnham
Surrey
GU10 4LH

T: 01420 526175
E: james.morison@forestry.gsi.gov.uk

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Contents

Executive summary

- Forests and woodlands are a key component of the global carbon cycle, and their effective management at global and regional scales is an important mechanism for reducing atmospheric greenhouse gas (GHG) concentrations. Forests constitute a substantial stock of carbon (C) in the trees, other vegetation and in soils; growing more trees will result in more carbon removal from the atmosphere. Forestry products also represent a considerable C stock; moreover, the use of woodfuel can substitute for fossil fuel combustion and wood products can substitute for other more fossil-fuel intensive materials.

- Understanding what determines the size of forestry C stocks and their components, and the processes and controls on the exchanges of carbon dioxide (CO_2) and other GHG in forests and woodland, is critical in assisting the forestry sector to contribute to reducing anthropogenic climate change.

- The objective of this review is to provide that understanding by summarising key information on C stocks and fluxes in UK forests, and the fluxes of other GHGs, and how these are affected by changes as tree stands grow, and by forest management decisions and forestry operations. The review also aims to highlight evidence gaps and limitations in the available information and consequent future research needs and directions.

- The report discusses the processes determining the size and changes in the main 'in-forest' carbon stocks: trees, woody debris, litter and soil, and in the 'out-of-forest' carbon stocks of harvested wood products and their potential GHG emissions abatement benefits through substitution.

- A variety of figures for the C stock in UK forest biomass, litter and soils and in other components of forestry have been quoted in the past; information in this review provides the most accurate figures to date and is important in updating statistics on in-forest C stocks reported by the Forestry Commission.

- Approximately half of the dry mass of vegetation (biomass) is carbon. The main biomass component in woodland is the stemwood, which is well characterised by forestry mensuration measurements. However, the harvestable stem only accounts for approximately 50% of the total tree C. Therefore, characterising the partitioning between components is central to estimation of stand C stocks.

- Various empirical relationships and coefficients are used to calculate total tree C stocks from standard stem volume measurements. Timber C content must be taken into account as it varies by a factor of nearly 2, with broadleaved species typically higher than conifers. Carbon in the branches, non-merchantable stem and foliage may contribute 30–70% of the above-ground biomass (although foliage is a small component). Roots may comprise 20–35% of the total C stocks in trees, with larger proportions in broadleaves than in conifers. The empirical relationships between stem biomass and total tree C stocks are better established for the main commercial conifer species than others. Relationships for broadleaved species, presently 'minor' species, and potential 'new' species are largely unknown. In addition, the influences of different management and silvicultural practices for any species are not well quantified.

- While the main processes in woodland C cycles are believed to be reasonably well understood, details on some components, particularly below ground are lacking. There are very few complete C balances for UK woodlands, where the different C stocks and fluxes between them are well quantified for particular stands. Direct measurement of the CO_2 taken up by UK forests has also been very limited, with just two long-term sets of measurements (one conifer and one broadleaf plantation), and a few shorter records, mostly over conifer plantations, although one recent dataset is for a semi-natural woodland. Measurement and characterisation of C fluxes and stocks in example stands are very important in developing forest C cycle models with sufficient robustness to examine management options and climate change impacts.

- Previous estimates for the total C stock in tree biomass in UK forests range from 469 to 595 $MtCO_2$ (128 to 162 MtC). However, because below-ground tree biomass has been underestimated in many reports, the best estimate is 595 $MtCO_2$ (162 MtC). Of this C stock, 71% is in private ownership, 29% is in broadleaved forest in England, and 32% is in Scotland's spruce and pine forests. The new National Forest Inventory will be key in establishing more reliable biomass C stock figures in the future.

- UK forests absorb a substantial amount of CO_2 from the atmosphere. Net CO_2 uptake by UK forests (after taking account of the removal through harvesting of approximately 6.5 $MtCO_2\ y^{-1}$), has been estimated at between 9 and 15 $MtCO_2$ y^{-1}, and work is under way to improve these important estimates.

- Using the estimates of total UK woodland area and tree total C stocks, provides an average C stock per hectare of approximately 209 tCO_2 ha^{-1} (57 tC ha^{-1}). Clearly, C stocks in tree stands vary with species, age and growth rate, and with management, so the range is as wide as from 0 to 1400 tCO_2 ha^{-1} (0 to 382 tC ha^{-1}).

- Peak CO_2 uptake rates by tree stands (typically between 5 and 20 tCO_2 ha^{-1} y^{-1}, or 1.4–5.5 tC ha^{-1} y^{-1}) occur during peak timber increment (i.e. in the period after canopy closure), and CO_2 uptake rates decline before stand maturity, although C stocks continue to increase. However, stand establishment and early growth up to canopy closure can take many years. Therefore, maintaining high CO_2 uptake rates in the long term in forests depends on rapid establishment of trees, management and harvesting trees at appropriate ages and times.

- The influence of different management regimes on C stocks in trees is illustrated with the BSORT model. Stocks are compared between 'no-thin' and 'standard thinning' regimes, for example typical conifer (Sitka spruce) and broadleaved (oak) stands. More active management of stands result in lower tree C stocks. The management required to achieve peak CO_2 uptake rates is not the same as that for maximising tree C stock.

- The carbon stocks and fluxes related to harvested wood products (HWP) are discussed. Carbon in HWP in use is likely to be a significant 'out-of-forest' C stock, although it is presently not well quantified. In addition, assessing the proportion of the HWP stock originating from UK forests is difficult because of the dominance of imports in many sectors. Increasing wood content in products and particularly in construction offers a way to store C in the long term, but it should be noted that the sequestration is potentially reversible, depending on what happens at the end of the product life.

- The potential of wood substitution for fossil-fuel intensive materials and for energy generation are discussed; this offers substantial additional GHG balance benefits. The important point about substitution is that, unlike C stocks in-forest, which tend towards an upper limit (and are also potentially reversible), the benefits of avoided fossil fuel consumption are perpetual and accrue continually. However, the calculation of the net reduction in GHG emissions through biomass substitution for fossil fuels needs to take into careful account the emissions in harvesting, transport, processing and combustion.

- Litter from foliage, branches and roots is an important component of forest C stocks (typically 30–45% of the C stock in trees alone). Litter is therefore a much more important component than in arable land use, for example. Litter decomposition rates are highly variable because of environmental, soil and tree factors, but they are critical to understanding the effects of trees and their management on soil organic C stocks, and on nutrient cycling and tree growth rate.

- Data derived from the recent extensive BioSoil survey of GB forests (166 plots) show that the organic C stocks in the forest litter (L) and the top organic (F) layer averaged 56.3 and 63.1 tCO_2 ha^{-1} for conifers and broadleaves, respectively (15.4 and 17.2 tC ha^{-1}). These values can be scaled up to an estimated UK forest litter and top organic layer stock of 168 $MtCO_2$ (45.9 MtC). This estimate is approximately 80% higher than values reported previously for national forest C litter stocks derived from models, and is likely to be an improved estimate.

- At most woodland sites, the soil contains more organic C than the trees, particularly sites with organic soils (e.g. deep peats and peaty gleys). Importantly, the BioSoil survey has allowed new, more reliable estimates of forest soil C stocks in GB across the main forest soil groups. The results are a significant improvement because BioSoil used measured bulk density at four different depths to estimate C stocks down to 80 cm. Mineral soil groups had mean total C stocks in the 0–80 cm soil depth between 487 and 570 tCO_2 ha^{-1} (133 and 155 tC ha^{-1}), organo-mineral (peaty gleys, peaty podzols and peaty rankers) approximately double those (1174 tCO_2 ha^{-1} or 320 tC ha^{-1}), and organic soil (deep peats) over three times as much (1644 tCO_2 ha^{-1} or 448 tC ha^{-1}).

- These values of soil organic C for the main soil groups have been scaled up with information on forest cover and soil group areas to provide estimates of total GB forest soil C stock (to 1 m) of 2302 $MtCO_2$ (627 MtC) for a forest area of 2.66 Mha. Most of this is in Scotland (1417 $MtCO_2$, 386 MtC); in England and Wales forest soil C stocks are 693 and 192 $MtCO_2$ (189 and 52 MtC), respectively. These numbers represent soil C stocks to 1 m depth only not total soil C stocks, which for some areas of deep peat soils (particularly in Scotland) will be much higher where organic material extends below 1 m depth.

- Comparisons are made with national scaled-up forestry soil C stocks estimated using other national soil surveys.

The BioSoil-derived estimate of total national soil C stocks down to 1 m depth is substantially larger (GB average of 29%) than previous estimates using National Soil Inventory data, but 12% lower than those used in the FAO Forest Resources Assessment and similar reports. The BioSoil based estimate is likely to be the best current estimate of specifically forest soil C stocks in GB. Furthermore, when the new National Forest Inventory provides updated forest areas, the BioSoil information will enable provision of an accurate figure for UK, country or regional forest soil C stocks.

- Summing C stocks from trees, litter and soil indicates that the mean C stock in UK woods and forests is 1131 $tCO_2\ ha^{-1}$ (308 $tC\ ha^{-1}$). The soil contains (to 1 m depth) approximately three-quarters of the total 'in-forest' stock, which is estimated for the UK at approximately 3223 $MtCO_2$ (878 MtC) for a forest area of 2.85 Mha.

- Organic C can be lost from soils through CO_2 emission during decomposition, through loss of soil particulate organic matter in erosion (usually very small in vegetated soils), and through loss in solution (dissolved organic carbon, DOC). Dissolved C flux from the surface layers can be substantial, although the carbon can be adsorbed and stabilised lower down in the profile, dependent on soil mineral composition. In British conditions, the annual flux of DOC can be important in the overall C balance of forest stands, although it is usually considerably less than 10% of the typical net CO_2 uptake rates by the trees. Such relatively large fluxes are found in organic soils, but on mineral soils typical annual DOC fluxes are usually <1% of likely tree net CO_2 uptake.

- Available information on the fluxes of the three important greenhouse gases carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) from forests is examined. The exchange of CO_2 with the atmosphere occurs for both the trees and the soil but CH_4 and N_2O emissions are largely from the soil alone. The aerobic production of CH_4 in plants is very small, but in flooded or waterlogged forests it is likely that trees form a pathway for CH_4 produced in the soil to be emitted, although the size of this flux is not yet established.

- The review highlights the paucity of comprehensive GHG balances for forest stands in the UK. Soil CH_4 emissions can be important on waterlogged, organic forest soils, but the limited evidence available suggests their contribution to the global warming potential (GWP)[1] is usually small compared to the CO_2 emissions from the soil. Although the evidence base is not large, soil N_2O emissions are usually low from most UK forest soils, although some forest operations may result in temporary increases. The low N_2O emissions are primarily because fertiliser is not used and atmospheric N deposition is relatively small, although it is increasing. Thus the contribution of N_2O fluxes from UK forests to the GWP is small.

- There are presently insufficient measured data from a range of UK climate, land-use and soil type conditions to quantify with confidence soil C changes during afforestation. This is partly because of the difficulties of detecting relatively slow changes in spatially heterogeneous soils. In particular, the effect of soil disturbance during planting is not well characterised, especially for current ground preparation practices, and there is little information on which to base recommendations for different soil types. The more soil disturbance there is during site preparation, the more soil C loss is likely.

- After afforestation it is likely that on mineral soils, particularly those with low C contents due to previous long-term cultivation, there will be an increase of soil C, from soon after planting. Rates of C accumulation reported are in the range 0.2–1.7 $tCO_2\ ha^{-1}\ y^{-1}$ (0.05–0.46 $tC\ ha^{-1}\ y^{-1}$). On high C content soils there is likely to be a period of net C loss following planting, and the duration over which the stand (trees + soil) is a net source of CO_2 before becoming a sink will depend on the initial amount of soil C, the degree of soil disturbance including drainage and consequent aeration, soil nutrient status and the productivity of the trees.

- Stocks of carbon in the biomass of other vegetation types such as permanent grasslands and heather moorland are typically 30 and 40 $tCO_2\ ha^{-1}$ (8–11 $tC\ ha^{-1}$), respectively, although they vary substantially with management and environment. Detailed C balance assessment of C stock changes during afforestation may need to take this into account but, although substantial, such C stocks are typically exceeded by those in even young stands of trees.

- The review compiles available information on the GHG fluxes from fossil fuel combustion during forestry operations (road building, ground preparation, thinning, harvesting and timber haulage). The analysis shows that haulage from forest to primary processor is the largest

1. See Symbols and abbreviations for explanation of global warming potential.

contributor, but all together these GHG emissions in the UK are small, totalling approximately 0.22 $MtCO_2e\ y^{-1}$, which is only 1–2% of the net annual CO_2 uptake by UK forests.

- Information on the GHG emissions due to forestry management and forestry operations has been incorporated into a complex 'C accounting model' (CSORT) that calculates C balances of example forest stands and their components, through growth cycles, taking into account fossil fuel emissions in forest operations and transport, and harvested wood product substitution benefits. Example calculations are shown for Sitka spruce stands with two forest management options (minimal intervention compared to standard thinning and felling). The assumptions and approaches behind the model are described in detail.

- Example results are presented from the CSORT model of C content in the different forest components, and the mitigation benefits through substitution under different management regimes and yield class, for Sitka spruce and oak. Emissions from fossil fuel use during forestry operations have very little impact on the C balance. The modelled contributions from harvested wood products are larger when substituting for fossil fuels than the C sequestration benefit in the products themselves.

- The report concludes by highlighting important gaps in information and noting areas where more evidence is required and tools need to be developed. The main requirements are:

  - more data on soil organic C changes, soil C residence times and GHG fluxes for different soil types during afforestation and under different forest management practices with which to parameterise, develop and validate forest soil C models for UK conditions;

  - data on full C and other GHG balances through the year for contrasting example forest stands on mineral and organic soils, to develop and parameterise models to assess management options, and the impact of climate change;

  - models to quantify the effects of soil disturbance and harvesting forest residues on soil C changes;

  - more information on C stocks and C and GHG balances of semi-natural woodlands, as much of the existing information is derived from intensively managed production forests;

  - better characterisation of C stock changes during early tree growth, both in trees and in other vegetation that is being replaced, to support assessment of afforestation GHG balance benefits, particularly for the new types of woodlands currently being considered;

  - more information on C stocks and their changes in stands after maturity, so that the benefits of extended rotations and non-intervention management options can be better assessed;

  - more information, including information on biomass partitioning and yield time courses, to extend forest C accounting models to new silvicultural systems, a wider species range and new management options;

  - improved quantification of C stocks in HWP and in the different major components with differing lifetimes and end uses;

  - better information on the GHG balance benefits of forest products including woodfuel substituting for energy-intensive materials and for fossil-fuel derived energy;

  - incorporation of fluxes of the other two main GHGs (N_2O and CH_4) in forests into C accounting models;

  - improved approaches to scale-up estimates of forest C stock components for national reporting in order to integrate with forest inventory improvements, and to assess impacts of projected climate changes on C stocks and fluxes.

1. Introduction and objectives

1.1 Background

There is intense, global interest in the role of forests in climate change, because forests have a major influence upon the world's climate (e.g. Bonan, 2008; Nabuurs *et al*., 2008; Nobre *et al*., 2010). Scientists are working on understanding the processes, functions and biodiversity of forests, and on optimising management to provide robust ecosystem services (e.g. carbon (C) sinks, soil stability and water yield); forest managers want to improve performance and deliver sustainable management of the forest resource; policy makers wish to devise guidelines for best practice and instruments for reducing deforestation, while the public voice concern over the fate of favourite woodlands and iconic landscapes. While forests are multifunctional, it is the role of forests as carbon sinks[2] and stores that is of primary importance for climate change mitigation (e.g. Loustau, 2010). In part, this is due to the requirements of the Kyoto Protocol for signatory countries to have effective carbon inventories, in which sinks such as forest growth are set against sources, such as fossil fuel combustion, deforestation and emissions from agriculture and industry.

Carbon science is pivotal for much of forestry. Most obviously, tree growth represents the net accumulation of carbon because approximately 50% of the mass of wood is carbon, so that assessing and predicting forest productivity provides one form of 'carbon accounting'. Carbon cycling also underpins many forest processes and functions: solar energy is stored in biomass by photosynthesis, tree water use is largely because of the requirement for transpiration during photosynthesis, and soil health and fertility are dependent on soil carbon content to a substantial degree.

Clearly, for carbon inventory purposes we need to know the carbon stock in trees, in soil, in timber and in other forest products. We also need to know the consequences of different management options for the carbon balance of forests to maximise the climate change mitigation potential (see summary by Nabuurs *et al*., 2008; Read *et al*., 2009). The four main strategies available are:

1. Increasing forest area.
2. Increasing carbon content per unit area at stand and landscape scale.
3. Expanding use of forest products to substitute for fossil-fuel intensive materials and fossil fuels.
4. Reducing emissions from deforestation and degradation (e.g. Canadell and Raupach, 2008; Nabuurs *et al*., 2008).

However, there are wider links between forestry and climate change, in particular the emissions of other greenhouse gases (GHGs, particularly methane, CH_4, and nitrous oxide, N_2O) through natural processes, largely in the soil and during fossil fuel consumption. Forests also affect the surface energy balance through the partitioning of solar energy, particularly the balance between reflected radiation, heating of the air and evaporation. Forests therefore influence climate directly, and they influence atmospheric chemistry through the emission of volatile organic carbon compounds. These affect the concentration of tropospheric ozone, which is also a GHG (see Jarvis *et al*., 2009 for more detail).

The requirement to minimise GHG emissions may have to be considered in management decisions. However, understanding management-induced changes in GHG budgets is very difficult because of site variation and history, and issues over where to draw the system boundaries (e.g. Lindner *et al*., 2008). Furthermore, wider considerations for an integrated, climate-change-driven land-use policy will require information on forestry's contribution as part of the range of options, particularly on the balance with agricultural food production, but also including activities such as renewable energy production.

1.2 Report origins, objectives and boundaries

This report arises ultimately from the reasons outlined above and the consequent demand for information on carbon and other GHGs in UK forestry. The need arose from several requests from within the Forestry Commission (FC) for figures for carbon stocks and GHG fluxes in forests, timber and soil, and information about the implications for policy (e.g. afforestation targets, open-ground and peat bog restoration) and forest management options. While there is substantial information on carbon in

2. For explanation of terms sink, source etc., see Box 1.1.

UK forests from Forest Research (FR) and FC work on forest mensuration and production forecasts and on carbon accounting, the focus has been on the trees and particularly the timber, and there is little detail on the soil, other components of the forest or on GHGs other than carbon dioxide (CO_2). The information was also rather scattered in different reports and across different programmes. Several previous FR research programmes funded by the FC were working on carbon-related aspects, so FR staff proposed the production of a single, comprehensive, integrated technical report with a particular emphasis on UK conditions. In particular, the report uses new soil carbon information that started to become available in 2008 from the BioSoil programme. The Review had the agreed main objective:

> *To produce a summary of the key information known and not known on the stocks and fluxes of carbon and the fluxes of other greenhouse gases in UK forests, and how they are affected by forest dynamics, management and operations.*

A 'first phase' interim report was circulated for comment to key staff in the FC in August 2008, and the draft final report was circulated in October 2008, for comment and external review. That report helped form the priorities for the present substantial integrated research programme in Forest Research: 'Managing forest carbon and GHG balances (ManForC)', funded by the FC. This report has been substantially updated and expanded to complement the carbon and GHG-related information in the UK Assessment of the Role of Forests (*Combating climate change – a role for UK forests*, Read *et al.*, 2009). In particular it provides more details about soil C stocks and in-forest processes, the effects of forestry management and activities, and the development of the FR forest carbon accounting model CSORT.

The emphasis of the review is on production forests, and it does not consider explicitly semi-natural woodlands, although many of the fundamental processes discussed will apply of course. However, information on these woodlands is more limited. The review primarily considers carbon information in wood and timber production up to the forest roadside, and includes analyses of operational costs and processing of wood products beyond the forest, including 'substitution' benefits. The main subjects that are not covered in the review are:

1. Air quality and air pollution effects on C and GHG balances.
2. Consideration of natural or man-made GHGs other than CO_2, CH_4 and N_2O).
3. Urban woodlands and brownfield restoration by tree planting, and their climate change benefits.
4. The effects on UK forestry C stocks of the international timber trade.
5. Impact of future climate or other environmental changes or atmospheric gas concentration on forest C stocks and GHG fluxes.
6. Detailed consideration of short rotation coppice and short rotation forestry[3]
7. Scenarios of future forest and woodland areas.
8. Economics aspects, including C economics.[4]

Several of these have been covered in *Combating climate change* (Read *et al.*, 2009). Note that given its wide scope, the report does not attempt an exhaustive literature review. While individual sections report key findings from the relevant literature to 2010, in several places for brevity the report points only to accessible recent references, not necessarily to original papers.

This final report is the work of several FR staff identified on the title page. It is not appropriate or possible to identify all the authors for every section. The lead authors on particular components were:

1. James Morison: introduction and background, forest C cycles and components, and C stock estimate comparisons.
2. Elena Vanguelova: soil, litter, and all BioSoil information.
3. Mike Perks: forest operations, forest management impacts.
4. Sirwan Yamulki: forest greenhouse gas fluxes.
5. Robert Matthews: harvested wood products, substitution, stand C dynamics, use of CSORT, management option calculations and country C stocks.
6. Tim Randle: modelling scenarios and description of CSORT.

James Morison coordinated the work; contributed introductory, linking and summary materials; reviewed, edited and finalised the report.

Chapter 2 introduces the forest carbon cycle and the various components, and provides various approximate estimates for the C stocks and flux quantities in forestry in the UK to place the later detailed discussions in context. Chapter 3 describes the main components in

3. A review of Short Rotation Forestry (SRF) information has recently been published by FR (McKay, ed. 2011), as part of the new, extensive Defra and Scottish Government funded trials, see www.forestry.gov.uk/srf.
4. For information on FR's expanding work in this area, see the recent reports by G. Valatin on Forests and carbon: a review of additionality (Valatin, 2011) and on Forests and carbon: valuation, discounting and risk management (Valatin, 2010).

the forestry C balance, and discusses the factors affecting them. Chapter 4 examines the fluxes of C and the other two key GHGs (N_2O and CH_4) in forests and how they are affected by forestry operations. Chapter 5 describes the approach and calculations behind the forest C accounting model CSORT that is used to estimate the effect of different management options on C balance in different woodland types. Chapter 6 summarises key points arising from the review and outlines the major evidence gaps that need to be addressed by continuing research to strengthen our ability to understand and manage forest C and GHG balances for improved emissions mitigation.

1.3 Basic terms and definitions

Key definitions and the meanings of key terms used in this review are given in Box 1.1; there are also lists of symbols and abbreviations, and a glossary, at the end of the report.

Box 1.1 Key definitions and meanings of terms used in forest C and GHG research

Carbon flux: the rate of exchange of carbon between different pools, or in and out of the system. Usually expressed as a mass change per unit time per unit land area (tCO_2 ha^{-1} y^{-1}).

Carbon pools: the different components of the system.

Carbon sinks and sources: a carbon sink is any system which causes a net C transfer from the atmosphere to the system. A growing forest is normally a sink, but there are situations where a forest can beco me a carbon source, transferring C to the atmosphere (e.g. through deforestation or fires).

Carbon sequestration: is said to have occurred if C is removed from the atmosphere and adds to C stock within one or more reservoirs (trees, soil etc). In its legal usage it refers to temporary seizure, thus the C removal can be reversed; in climate science 'sequestration' usually implies periods of years.

Carbon stock: the amount of carbon in the system or its components at a given time. Either expressed as mass per unit land area (e.g. tC ha^{-1}), or as a mass for a defined area (e.g. MtC). In order to compare stocks with CO_2 emissions, they can be expressed as mass CO_2, as used here, by multiplying by the ratio of the molecular masses of CO_2 and C (44/12); thus 1 tC ≅ 3.667 tCO_2.

CO_2e (CO_2 equivalents): To express the emissions of other GHGs as well as CO_2, it is conventional to use 'tonnes CO_2 equivalent' (tCO_2e) which combines the effect of various GHGs with a weighted sum taking into account their differing warming effect, or 'global warming potential' (GWP). For CH_4 (methane) and N_2O (nitrous oxide) these are 25 and 298 times, respectively, the GWP of CO_2 (IPCC, 2007). In this review we use CO_2e **only** where other GHGs have been included.

Forest: a landscape which has a high proportion of woodland, but which may also include other land cover types and uses.

Litter: consists of all debris and material on the ground under woodland that has come from the trees and other vegetation: branches, twigs, leaves, growth and decay of vegetation. In soil science 'litter' on the surface of the soil is part of the organic or O horizon (see Box 3.1).

Soil: consists of the inorganic and organic matter forming the ground under trees (not including litter), above the bedrock or other parent material.

Soil carbon: is an abbreviation for soil organic carbon (SOC) as it should be noted that soil also contains inorganic C in minerals and the soil solution. SOC is the C content of organic matter derived from decomposing plant, microbial and animal material. It does not include live roots, nor the litter which is present on the soil surface.

Stand (forest stand, or stand of trees): a measurable unit of trees with some form of homogeneity, often managed in the same way. For example, a stand of trees may be formed of one species, or several species evenly mixed. A stand may also contain trees all of the same age.

Woodland: an area of any size of land covered by trees.

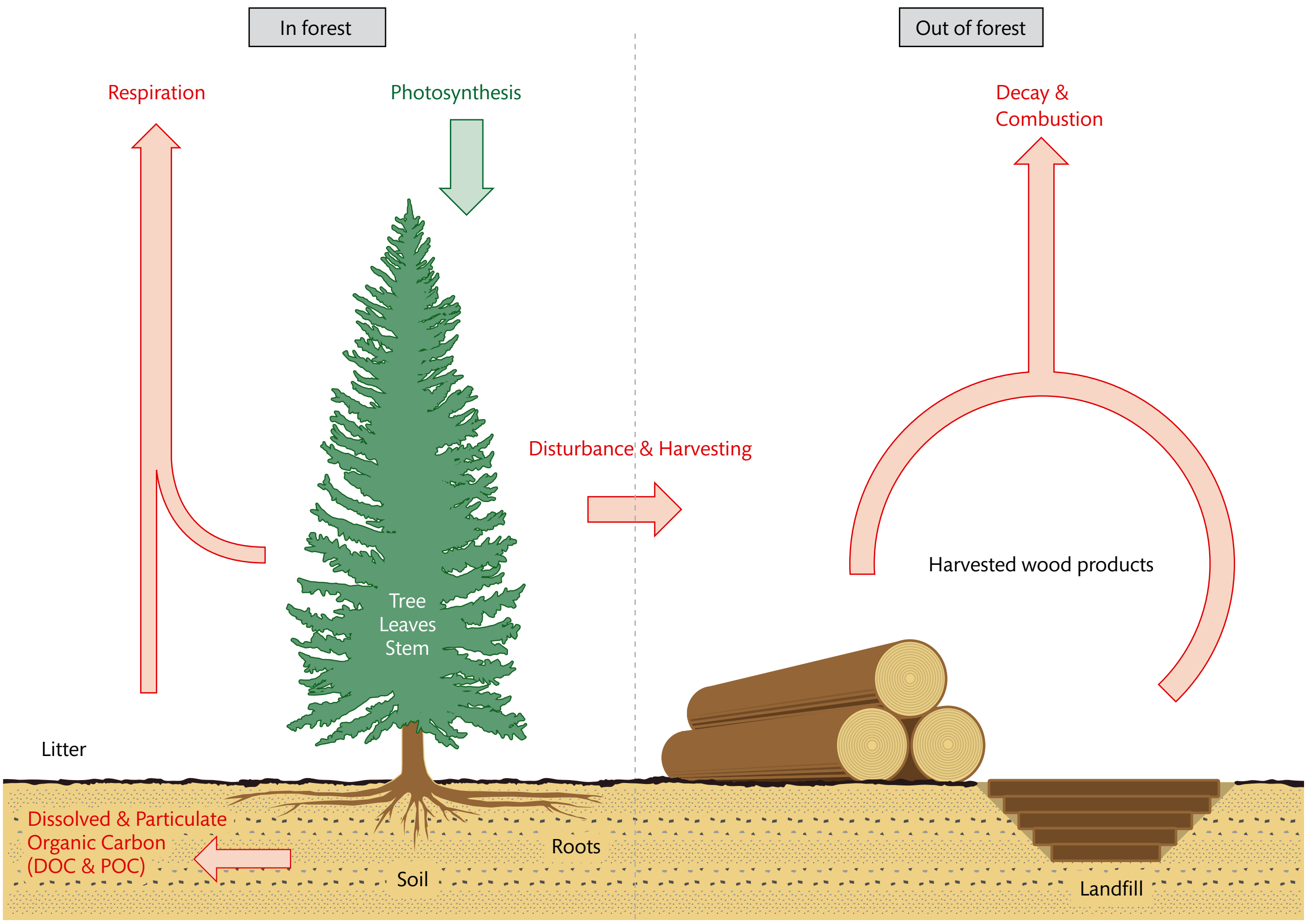
2. Forestry and carbon

2.1 Introduction

There are several main components of the carbon (C) cycle in forestry (Figure 2.1): in the forest the components are trees and other vegetation, litter and soil and outside the forest the components are the harvested wood products (HWP) of woodfuel and timber, both in use and in landfill sites after use. The C stock[5] in forestry at any moment is the sum of the quantities in these components. The various component C stocks can have very different ages and residence times. For example, leaves of some deciduous tree species have lifespans of less than a year (including their breakdown), while obviously the woody frame of some longer-lived species frequently has a lifetime of centuries, as can some timber products. Some soil C components change on millennial time scales. It takes decades to grow trees; therefore C fluxes associated with growth when calculated per year are small compared to stocks. The exceptions are C fluxes due to natural disturbances such as fires and landslides and anthropogenic disturbance such as harvesting, which can all reduce the C stock of an area rapidly. It should be made clear that the C stocks of interest are the organic carbon, not the inorganic C, which is held primarily in minerals in the soil, such as carbonates, that are relatively stable.

Figure 2.1 does not show the fluxes of other GHGs, nor the detail of the natural and forest management processes that determine the fluxes and consequent changes in component stocks shown. It is the purpose of this review to give some of that detail of processes and quantities involved.

Figure 2.1 Main organic C stock components and C fluxes between components in forestry; disturbance includes natural (e.g. fire, landslip, flooding, storm). The green arrow indicates C flux into the forest; red arrows indicate fluxes out of the forest. For simplicity, vegetation other than trees is not shown, nor leachate losses from landfill.



5. See Box 1.1 for explanation of terms used, such as stock, source, sink.

2.2 The woodland C cycle

The contribution made by woodland to the carbon balance at any scale depends on the rate at which CO_2 is removed from the atmosphere and/or the quantity of C retained in the woodland. The C stocks in the forest include the above- and below-ground components of trees, other vegetation and the soil (as shown in Figure 2.2).

Trees (and other woodland vegetation, not shown in Figure 2.2) take up CO_2 from the atmosphere during photosynthesis. Much of the carbon that is taken up by the tree through photosynthesis is released again as CO_2 through above-ground and root respiration, which provides the energy for tissue growth and maintenance. The remaining carbon is partitioned between (or allocated to) leaf, root, seed, stem and branch biomass. Most of the carbon in woodland-grown trees is in the stem (approximately 50%, see examples in Figure 2.3), with substantial amounts in branches and below ground in roots (the latter is approximately 25% across many temperate forest examples, Cairns *et al*., 1997; Mokany, Raison and Prokushkin, 2006). In broadleaved species the branches can comprise 20 to 25%, depending on growth

Figure 2.3 Proportions in different components of the tree biomass for example stands of Sitka spruce and oak. Sitka spruce data from a 30-year-old stand, YC[6] 22, in Dooary, Ireland, total tree C stock = 539 $tCO_2\ ha^{-1}$ (Black *et al*., 2009); oak data from a sample of 20 trees harvested from an 80-year-old stand, YC 6, Straits Enclosure, Hampshire, C stock = 482 $tCO_2\ ha^{-1}$ (E. Casella, FR). Dead above-ground material includes needles and branches in this example for Sitka spruce. Fine root mass was not determined in the oak stand, and coarse root biomass was estimated using an empirical relationship from McKay *et al.* (2003).

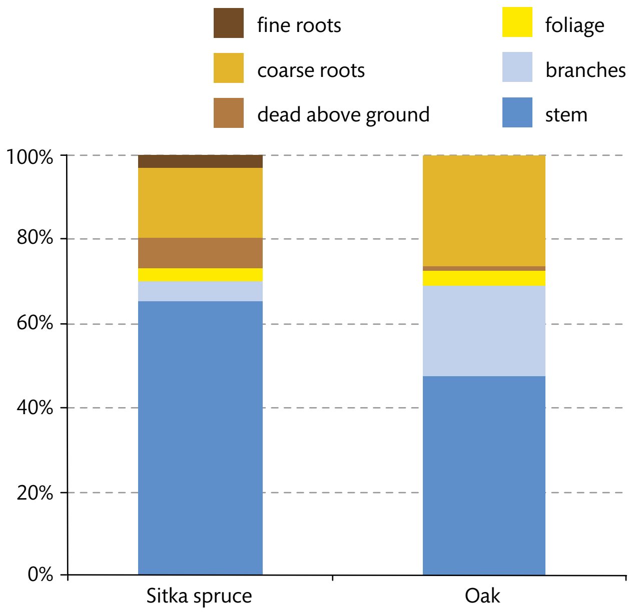
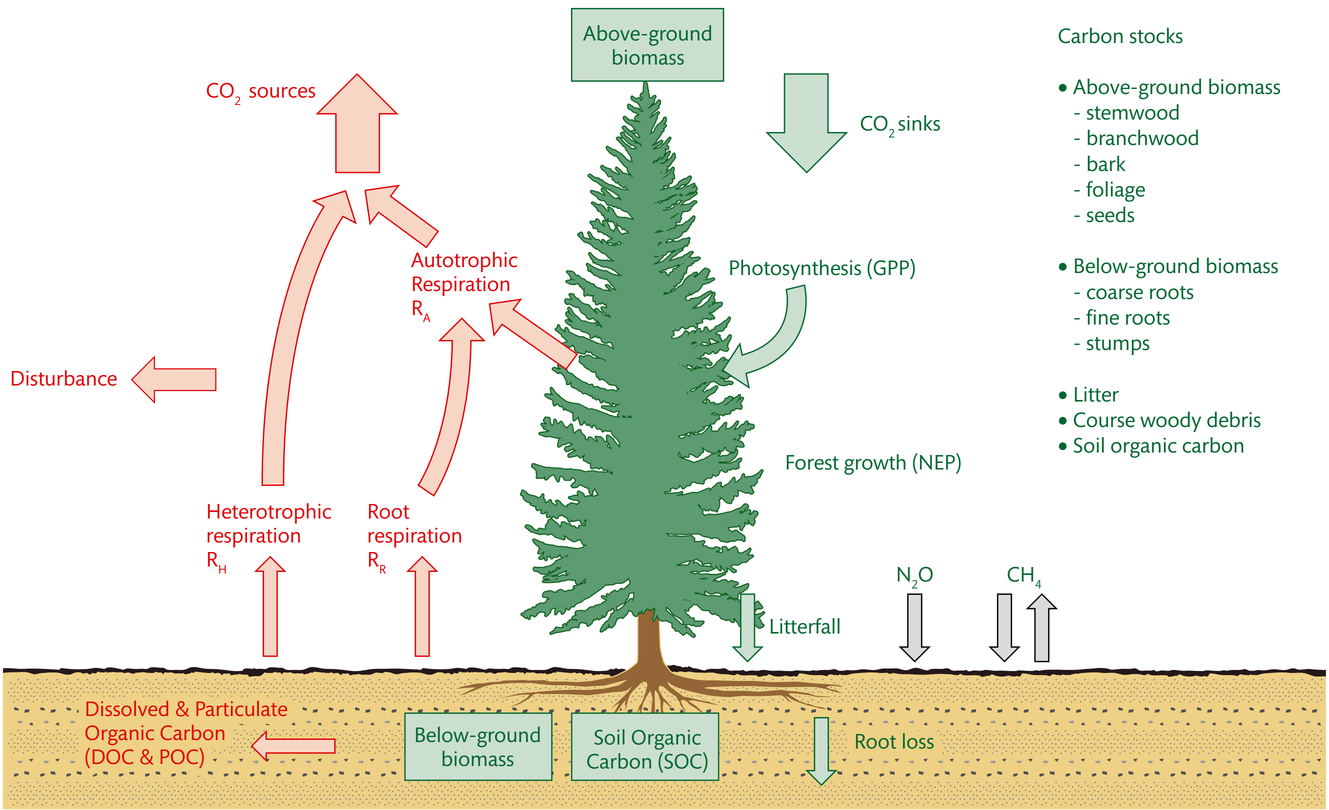


Figure 2.2 Diagram showing the processes and fluxes maintaining or modifying the different C stocks in a forest, and the fluxes of the other key greenhouse gases, CH_4 and N_2O. Green arrows indicate C flux into the forest and between various C stocks; red arrows indicate C fluxes out of the forest.



6. YC is abbreviation for yield class, see Glossary.

form (e.g. Figure 2.3). Live foliage is a relatively small fraction of tree C stock (approximately 5%), but on some conifer species the dead foliage remains on the tree for some time and contributes more to C stock.

During growth and subsequent senescence leaves and other above-ground components such as flowers, fruits and seeds are shed (litterfall), accumulating as litter on or in the soil, the rest being retained in the longer-term pool of woody material. Roots directly add organic material to the soil through exudation (so-called rhizodeposition, which is used by rhizosphere organisms such as mycorrhizae, other fungi, bacteria and invertebrates), through fine root turnover, and through coarse root shedding or death.

Decomposition of these litter and root exudation components releases CO_2. This flux of CO_2 is often referred to as 'soil respiration', although it results from microbial metabolic activity (fungi and bacteria), aided by soil invertebrate activity, which helps to break up and mix material. Note that measurements of 'soil respiration' usually include the respiration of the roots and any soil invertebrates. During decomposition the organic carbon remains in a wide variety of different components of soil organic carbon (SOC), with different lifetimes. In addition, some of this carbon will be dissolved in the soil solution (dissolved organic carbon, DOC). Soil microbial activity can also result in the net release of the greenhouse gases CH_4 and N_2O, although only the former is formally part of the carbon cycle. However, the mass of carbon exchanged via methane fluxes is normally very much smaller than that in CO_2 exchanges (see Section 4.2), so is usually ignored for C balances.

Several terms are conventionally used to represent the different CO_2 fluxes in the system illustrated in Figure 2.2 (see Box 2.1): the net primary productivity (NPP) is the difference between the gross primary productivity (GPP) and the respiration by the trees and other vegetation (autotrophic respiration, R_A), and the net ecosystem productivity (NEP) is the difference between NPP and the respiration by animals and micro-organisms (heterotrophic respiration, R_H). NEP is therefore the rate of carbon accumulation in the system.

Box 2.1 Terms used in ecosystem carbon balance work and abbreviations used in this report

GPP: gross primary productivity (uptake of CO_2 from atmosphere during photosynthesis)

R_A: autotrophic respiration rate (loss of CO_2 by plants, above and below ground)

R_H: heterotrophic respiration rate (loss of CO_2 by animals and microbes, mostly in soil)

R_T: total ecosystem respiration rate (= $R_H + R_A$), (sometimes shown as R_{eco})

R_R: root respiration rate

R_S: soil respiration rate: combined root and soil heterotrophic respiration rates ($\approx R_H + R_R$)

NPP: net primary productivity (= GPP - R_A), see note 1

NEP: net ecosystem productivity (= NPP - R_H, or = GPP - R_A - R_H)

NEE: net ecosystem CO_2 exchange rate with atmosphere, see note 2.

NECB: net ecosystem carbon balance (= NEP - DIC - DOC - VOC - PC – D_i, where D_i represents disturbance such as harvesting, thinning, fire, windthrow etc, and other terms are rates of flux of particular components)

NBP: net biome productivity (= NEP – D_i), applicable at a larger spatial scale, to include removals from forest areas due to natural and management disturbance (D_i)

DIC: dissolved inorganic carbon in soil solution or water bodies

DOC: dissolved organic carbon in soil solution or water bodies

POC: particulate organic carbon in soil solution or water bodies

PC: particulate carbon (e.g. soot from fires, dust, erosion)

Note 1: Some authors suggest that NPP should also be net of volatile organic carbon (VOC) loss from vegetation, i.e. NPP = GPP - R_A - VOC, but the latter is usually a small component.

Note 2: for example, measured by eddy covariance methods. In terrestrial ecosystems where there is little exchange of inorganic C, NEE is a measure of NEP, although NEE is often expressed as negative when CO_2 is being taken up by the vegetation.

References: Chapin *et al.*, 2006; Hyvönen *et al.*, 2007; Lovett, Cole and Pace, 2006.

2.3 Example woodland C balances

There are surprisingly few comprehensive recent measurements of woodland carbon balances in the UK, either for managed or semi-natural woodlands. However, typical woodland carbon balances are given in Figure 2.4, based on measured and modelled elements of (a) an upland Sitka spruce forest in northern England, and (b) a lowland oak woodland in southern England (a managed 80-year-old plantation).

It is not possible to 'close' (or balance) the carbon budget for all the component fluxes of interest for either the conifer or deciduous forest examples, partly because of the difficulties in measurement of several components. For the Sitka spruce stand, the data are compiled from several studies, and there is no complete set of information at one particular site, so the variation in values obtained for component fluxes reflects the impacts of management and the influence of varying soil and biotic factors across the reported studies.

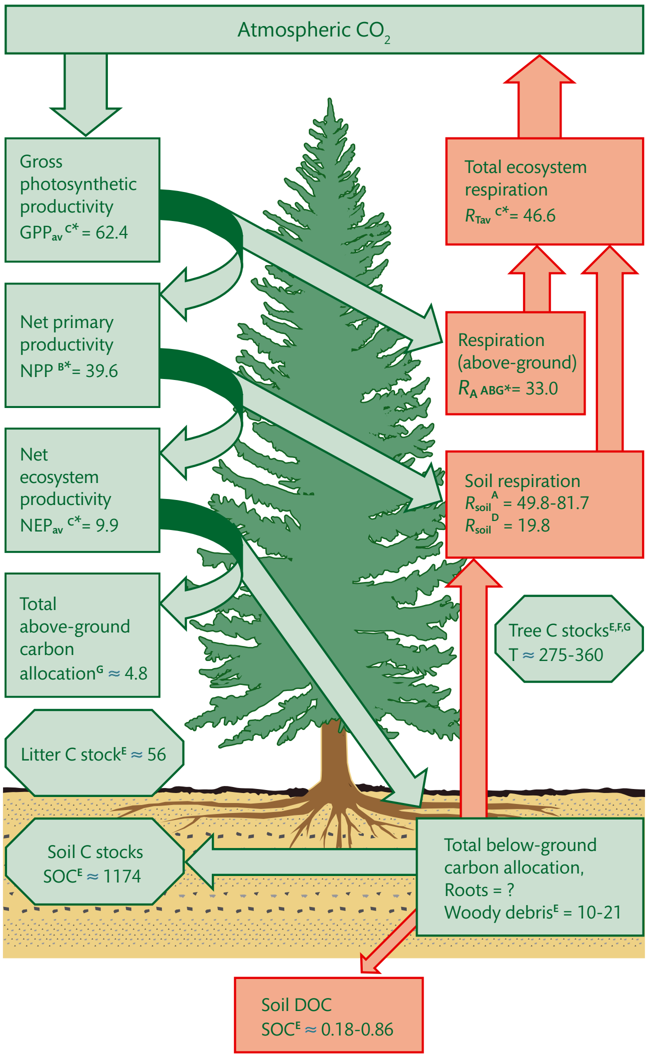
It is evident that for both examples there are large C stocks in the trees (approximately 300 and 500 tCO_2 ha^{-1}, for Sitka spruce and oak stands, respectively). There are larger stocks in the soil, particularly for the Sitka spruce stand, which has 3–4 times the amount of C in the peaty gley soil than in the trees, while for the oak woodland the tree C stock is approximately 85% of that in the soil. Litter stocks are approximately 15% of the stock in the trees in these examples.

The relatively large annual fluxes into and out of both stands are clear (GPP and total respiration, R_T, respectively, see Box 2.1); the net growth (NEP, equivalent to the C sequestration rate) is the result of large photosynthetic CO_2 uptake **less** large respiration and decomposition losses. Respiration by the vegetation (autotrophic respiration, R_A, from above-ground parts and roots) is typically 50–70% of the GPP (see e.g. Litton, Raich and Ryan, 2007; Luyssaert *et al.*, 2008; also DeLucia *et al.*, 2007, for a review of 'carbon use efficiency', the ratio of NPP/GPP). A reasonable assumption is that approximately half the soil respiration is from root activity. For the Sitka spruce example, there is very large variation in estimates of soil respiration, because of differences in soil conditions between the various studies used here. Long-term data that captures both spatial and temporal variation are rare (see Section 4.2).

The net CO_2 uptake by the trees (NPP) is allocated to stem, branches and roots. The calculation of below-ground allocation also includes the litterfall contribution; depending on microbial decay rates (resulting in perhaps half of the soil respiration flux), and dissolved soil organic C (DOC) losses there may be a net accumulation of soil or litter C stocks. In both of these examples, the estimated DOC loss is very small (see Section 4.3 for more information).

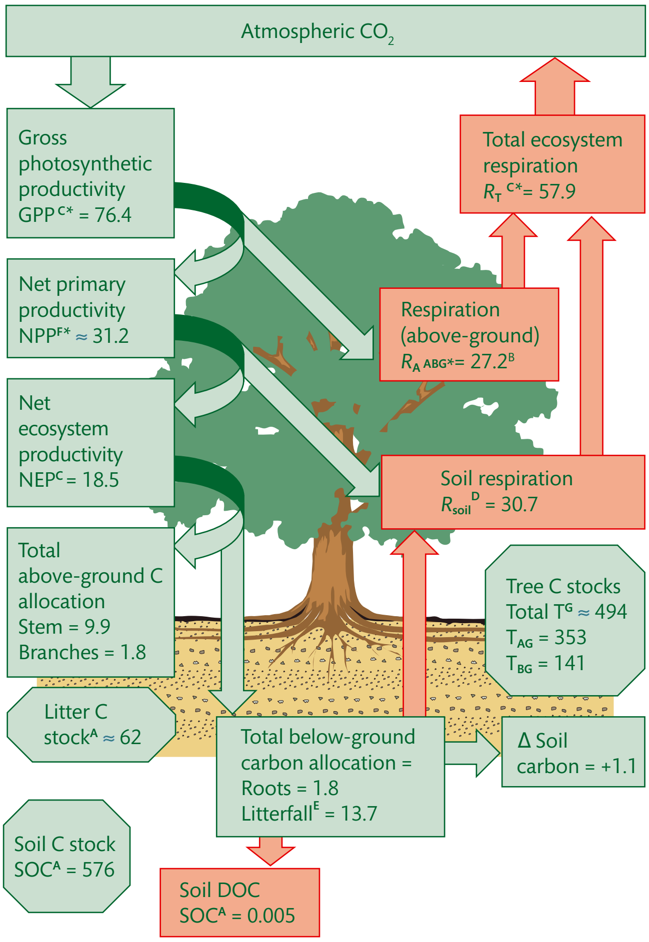
In the oak stand, the above- and below-ground C allocations (including litter) sum to an increment of 27 tCO_2 ha^{-1} y^{-1}, slightly less than the NPP (31 tCO_2 ha^{-1} y^{-1}), probably because the latter includes growth of shrubs and other vegetation, although that is probably a larger component than these sums suggest. In the Sitka spruce studies, with dense canopies, vegetation other than the trees does not contribute significantly to the C balance.

Figure 2.4a A summary of the carbon fluxes (in tCO_2 ha^{-1} y^{-1}) and stocks (in tCO_2 ha^{-1}) associated with the main components of a mature (40-year-old), second rotation Sitka spruce plantation (general yield class 14 m^3 ha^{-1} y^{-1}) on peaty gley soils in northern England. Boxes represent average annual flux, octagons represent carbon stocks.



Data presented are a synthesis of published work by M. Perks, FR, taken from A Ball, Smith and Moncrieff (2007); B Kowalski et al. (2004); C Magnani et al. (2007); D Zerva et al. (2005); E this report, Sections 3.4 and 3.5; F Jarvis and Linder (2007); G Dewar and Cannell (1992). Values with * denote derived estimate. Note: Soil respiration includes root respiration; unknown allocation to roots.

Figure 2.4b A summary of the carbon fluxes (in tCO_2 ha^{-1} y^{-1}) and stocks (in tCO_2 ha^{-1}) associated with the main components of a 70–80-year-old deciduous oak plantation (general yield class 6 m^3 ha^{-1} y^1) on a surface water gley soil at the Straits Enclosure in Hampshire.



Updated from Broadmeadow and Matthews (2003); sources indicated by a superscript: A From this report Sections 3.4 and 3.5; B Estimated as difference $R_{a\ ABG} = R_T - R_{soil}$; C Mean of 11 years eddy covariance data (1999–2009); D Average of 3 years' data, provided by automatic soil chambers, Heinemeyer et al. (2011), note that soil respiration includes root respiration; E Average of 3 years' data (2007–9), R. Pitman, FR; F Estimated using mean ratio NPP/GPP for 3 years', Heinemeyer et al. (2011). G Estimated from plot mensuration and tree sampling in 2009 and standard allometric relationship between diameter at breast height (DBH) and root biomass for oak (see McKay, et al. 2003). Values with * denotes derived estimate.

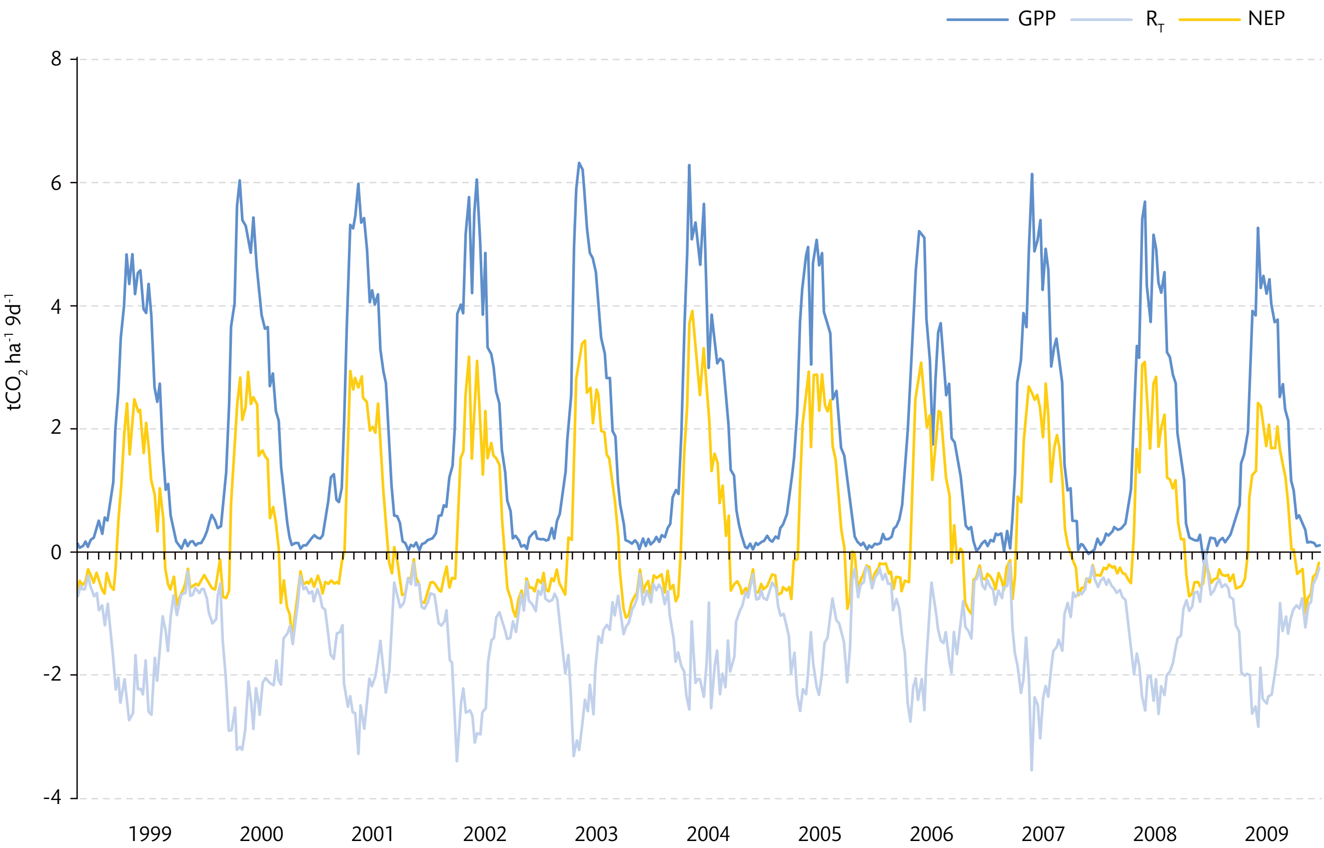
The net ecosystem productivity (NEP) was directly measured in these examples with the so-called eddy covariance method.[7] This provides continuous measurements of net CO_2 flux at such woodland sites, and allows the different components of the C balance to be assessed. An example is shown in Figure 2.5, which shows how NEP is comprised of GPP less R_T. The net annual accumulation of C is the integral of the net CO_2 uptake in the growing season, and the loss during the winter leafless period. The annual variation in the components is striking and shows the effect of weather conditions on woodland CO_2 exchange. The weather acts both directly on the trees and soil in determining photosynthesis, respiration and development rates (e.g. Granier *et al.*, 2007), and indirectly through the influence on the other components of the ecosystem, particularly insect herbivores and fungal pathogens affecting leaf area and thus light interception and photosynthesis (e.g. Pitman, Vanguelova and Benham, 2010). Surprisingly, such long-term, multi-year measurements only exist for two or three woodland sites in the UK at present. Although globally there are many more 'CO_2 flux sites' (see www.fluxnet.ornl.gov/), very few have similar environmental conditions or forests to those in the UK.

The Sitka spruce stands shown (Figure 2.4a) captured about 10 tCO_2 ha^{-1} y^{-1}, but other values for this species in the UK vary from 10 to 26 tCO_2 ha^{-1} y^{-1} depending on stand age and productivity (Magnani *et al.*, 2007). At the long-running Griffin Forest site in Perthshire, the Sitka spruce is approximately 20 years old, growing on a podzolised brown earth soil (Conen *et al.*, 2005). The yield class (YC) is 14–16 m^3 ha^{-1} y^{-1} and the mean NEP is 24 tCO_2 ha^{-1} y^{-1} (Clement, Moncrieff and Jarvis, 2003; Jarvis *et al.*, 2009) compared to 18 tCO_2 ha^{-1} y^{-1} for the deciduous oak woodland (Figure 2.4b). Thus the oak woodland in SE England has a net uptake of CO_2 that is 75% of the evergreen conifer forest in Perthshire (see Jarvis *et al.*, 2009, which also shows graphs of average daily NEP). The relative size of these CO_2 uptakes is smaller than the YC difference of 15 and 6 m^3 ha^{-1} y^{-1} might indicate, but the different density of Sitka spruce and oak timber (density of 0.33 and 0.56, respectively, Appendix A4) should be taken into account, suggesting that the predicted maximum stem biomass yield of the oak woodland would be 68% of the Sitka spruce stand, similar to the ratio of observed NEP values. However, stand net CO_2 uptake contributes to other components as well as stem timber growth, and in the oak woodland measured CO_2 uptake includes that by the substantial shrub understorey.

Recently, Black *et al.* (2009) have published data from a chronosequence of Sitka spruce stands on surface water gley soils in Ireland, varying in age from 9 to 47 years. The productivity was much higher (YC 22–24) and they make an interesting comparison with the above Sitka spruce studies on peaty gley soils. The estimated GPP peaked for 16- and 30-year-old stands at 69 and 51 tCO_2 ha^{-1} y^{-1}, quite similar to those shown in Figure 2.4a, and NPP was also similar. However, net CO_2 uptake (NEP) was higher, with a maximum of 33 tCO_2 ha^{-1} y^{-1} at canopy closure in young stands (16 years old), dropping to approximately 7 tCO_2 ha^{-1} y^{-1} in older, thinned stands. The main reason suggested by Black *et al.* (2009) for these NEP differences is the lower autotrophic and heterotrophic respiration rates at the Irish sites. Other data on the C balance of a mixed deciduous woodland in England have recently been published (Fenn *et al.*, 2010; Thomas *et al.*, 2010), although for only 2 years of measurements. Interestingly, in that

7. Eddy covariance is the standard approach for continuously measuring the exchange of gases and energy between vegetation and the atmosphere (see Jarvis et al., 2009) and Glossary.

Figure 2.5 Measured net ecosystem productivity (NEP, net CO_2 uptake) and its components at the 70–80-year-old oak stand at the Straits Enclosure CO_2 flux measurement site in Hampshire, measured by the eddy covariance method between 1999 and 2009. The gross primary production (GPP) and total ecosystem respiration (R_T) have been derived from the measured NEP using the standard CarboEurope data analysis approach (Reichstein *et al.*, 2005). Data from M. Wilkinson and E. Eaton, FR, updated in 2011. For clarity, R_T is shown as negative.



ancient semi-natural woodland near Oxford, the GPP was very similar to that at the Straits Enclosure (presumably because of similar solar radiation, rainfall and temperature regimes). However, R_T was larger so that net CO_2 uptake (NEP) was much smaller (approximately 4.4 tCO_2 ha^{-1} y^{-1}, compared to 18 tCO_2 ha^{-1} y^{-1}), which the authors attribute to the cessation of active forestry management 40 or more years ago. However, they point out that their measurements (both by eddy covariance and comprehensive inventory methods) show that even this ancient, mixed, semi-natural woodland is a substantial CO_2 sink.

Although Figures 2.1–2.5 add to our understanding of the C components and transfer processes in woodlands, they show only 'snapshots' of the C balance of particular stands at a period, or averaged over a few years. What are required in order to explore the effects of silvicultural practices, afforestation policy or management decisions are accurate quantification of C accumulation and stocks over time, through the life cycle of stands, coupled to quantification of other GHG balances. Furthermore, integrating these different components requires models. These aspects are addressed in Chapters 3–5.

2.4 Estimation of C stocks and components

The estimation of forest C stocks is a large topic and appropriate methods depend on the scale of study – whether estimating stocks in a single stand, or at larger scales, such as national inventories. Key items are the estimation of soil C stock (for more information, see Section 3.5), litter stock (see Section 3.4) and estimation of tree C stocks. Briefly, at the stand scale estimation of tree C stocks requires measurement of standard tree mensuration parameters, such as diameter at breast height (DBH), tree or merchantable stem height and number of trees per area, using appropriate surveying and sampling protocols (e.g. Matthews and Mackie, 2006). These data can then be used to estimate stem volume using mensuration tables or formulae and then appropriate empirical relationships can be used to estimate branch, crown and root biomass (e.g. Figure 2.6a and b) to give whole-tree biomass estimates, and therefore C stocks. Recommended procedures for estimating woodland C stock for different tree species at a range of scales have recently been published in the Forestry Commission *Woodland Carbon Code: Carbon Assessment Protocol* (Jenkins *et al.*, 2011).

The estimation of stem volume is routine and well established in forestry practice, and timber density values are also quite well known, so that stem biomass and C stock is likely to be well estimated for a given stand. For example, mensuration tables and/or formulae for 7 broadleaved species and 12 conifers are described in Matthews and Mackie (2006) and Jenkins *et al.* (2011). Estimating total biomass is less certain, relying on more limited datasets to establish the empirical relationships required between stem volume and biomass of components (McKay *et al.*, 2003; Zianis *et al.*, 2005, Somogyi *et al.*, 2008); and as used in BSORT or C-FLOW models, or to provide 'biomass expansion factors' (BEF, ratio of biomass in a particular component to merchantable timber biomass, Levy, Hale and Nicoll, 2004). For root biomass in particular, there is less information because of the laborious nature of the work; McKay *et al.* (2003) list only empirical relationships for six conifer species (Sitka spruce, Norway spruce, Douglas fir, lodgepole pine, Scots pine, Corsican pine) and one broadleaf (oak). More information is required for a wider range of species, growing in UK woodland conditions, to enable better tree C stock estimates.

The BEF approach can be applied for individual components of C stocks or to total above-ground biomass estimates. The latter approach is included in the IPCC good practice guidance for LULUCF national GHG inventories (Penman *et al.*, 2003). It uses a single BEF (in this case the ratio of total above-ground biomass to merchantable timber biomass) and the density of timber, *D*, to estimate total above-ground biomass from the merchantable stem volume (*V*), and the ratio of the root biomass to above-ground biomass, *R*, to derive total biomass. Using the proportion of C in the dry biomass (F_c, well-determined to be 50%, Matthews, 1993), gives the total tree C stock, C_T (from Levy, Hale and Nicoll, 2004):

$$C_T = [V.D.BEF.F_c] \times (1+R)$$

Levy, Hale and Nicoll (2004) used an extensive dataset derived from FR's tree stability research programme on tree biomass components for 13 conifer species growing across GB to compile estimates of BEF and *R*.

The mean value of BEF for the different species ranged from 1.31 to 1.70 (Figure 2.7), with an average of 1.43 (which is also the value determined for Sitka spruce). However, a key variable determining BEF was tree height (as the proportion of biomass in foliage reduces with tree size) and BEF also varied substantially between sites. For these conifers, the non-merchantable portion of the stem (diameter <7 cm) was almost negligible, and was only significant in very small trees (Levy, Hale and Nicoll, 2004; see also Table 4.2.5 in Jenkins *et al.*, 2011). Relationships for each species between BEF and height are given by Levy, Hale and Nicoll (2004). The mean value of *R* ranged from 0.22 to

Figure 2.6 For Sitka spruce (a) the modelled relationship between total stem biomass (oven dry tonnes), DBH (diameter at breast height) and tree height; (b) modelled relationship between biomass components and DBH (calculated using procedures and empirical relationships for Sitka spruce in Jenkins *et al.*, 2011).

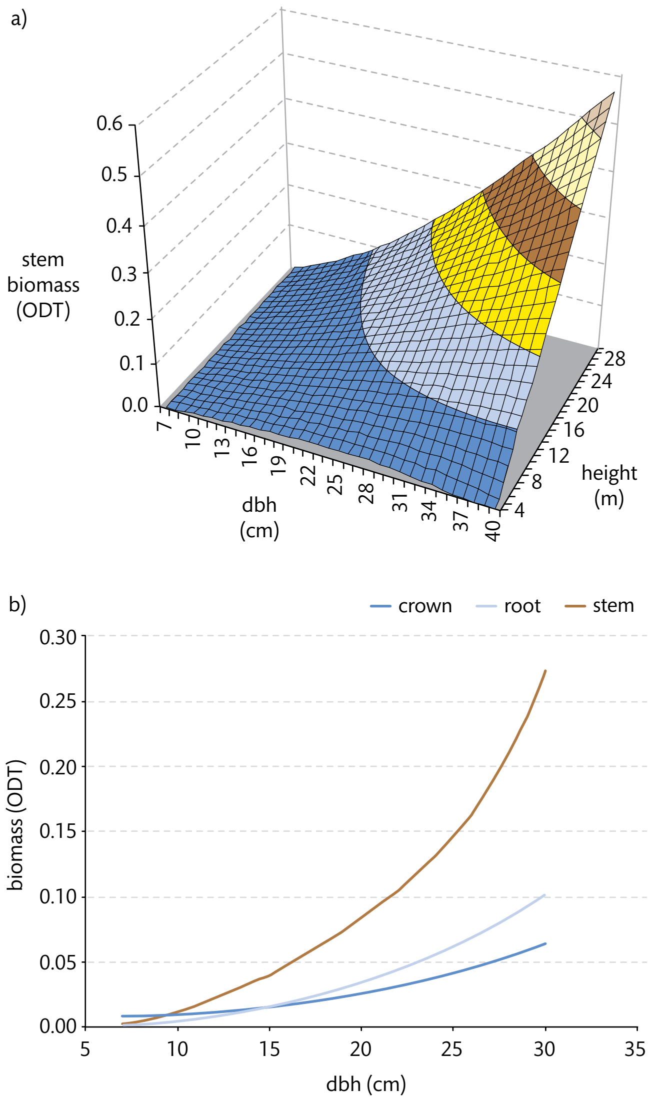
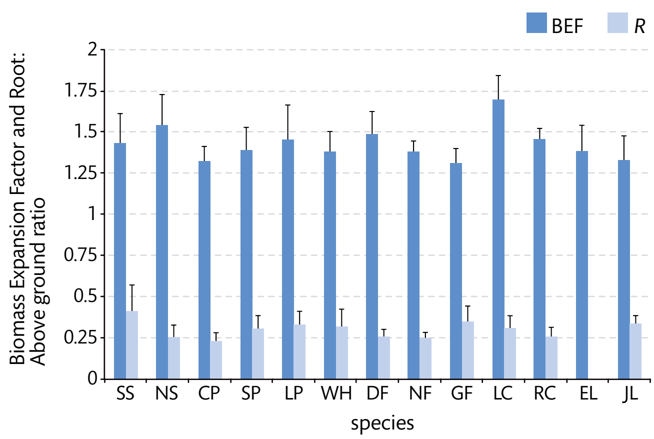


Figure 2.7 Mean biomass expansion factors (BEF) and root to above-ground biomass ratio (*R*) for 13 conifer species growing in GB; data from Levy, Hale and Nicoll (2004). Error bars are s.d. Species are: SS: Sitka spruce; NS: Norway spruce; CP: Corsican pine; LP: Lodgepole pine; SP: Scots pine; WH: Western hemlock; DF: Douglas fir; NF: Noble fir; GF: Grand fir; LC: Lawson cypress; RC: Western red cedar; EL: European larch; JL: Japanese larch.



0.41 between species (Figure 2.7), with an overall mean of 0.36, and Sitka spruce (with the largest sample) showed the highest mean value (0.41), although the median was lower, at 0.37. It is well documented that *R* varies with soil type and site conditions, particularly nutrient status and hydrology. For example, Fraser and Gardiner (1967) found that at one location in the Scottish Borders, Sitka spruce *R* ranged from 0.39 for trees on a brown earth soil, to 0.61 for lower productivity trees on deep peat. There is ongoing discussion about whether BEF and *R* values decline with stand age and/or with productivity (e.g. Tobin and Nieuwenhuis, 2007; Lehtonen *et al.*, 2007; Pitman, 2008). Furthermore, the true size of root biomass stocks and the likelihood of its substantial underestimation for trees have been highlighted (Robinson, 2007). Nevertheless, these various reports of BEF and *R* values illustrate that a substantial proportion of the tree C stock is below ground.

Although focused on the main commercial species, the FR dataset on BEF and *R* for conifer species used by Levy, Hale and Nicoll (2004) is quite comprehensive, but for broadleaved species nothing comparable exists for the UK. This is an important gap that needs to be filled, either from other available information sources or from new measurements.

2.5 UK forest carbon stock and flux figures

In order to understand the potential of forest management to change the contribution that UK forestry can make to reducing net GHG emissions (so-called 'emissions abatement') it is necessary to appreciate the values for key C stocks and GHG flux quantities. This section expands and updates the material in Forestry Commission Information Note 48 (Broadmeadow and Matthews, 2003), which discussed UK forests, carbon and climate change.

2.5.1 Forest carbon stocks

At the ecosystem scale the average total C content (including soil) of the temperate forest biome[8] is approximately 280 tC ha^{-1} (e.g. Saugier, Roy and Mooney, 2001; Grace, 2005) or 1030 tCO_2 ha^{-1} (see footnote[9]). Broadmeadow and Matthews (2003) stated that in managed stands in the UK, averaged over several rotations, the carbon stock can be 'up to' 370 tCO_2 ha^{-1} (Table 2.1). This is a maximum average, not a typical average. A more typical average is 220 to 260 tCO_2 ha^{-1}.

Table 2.1 Approximate values of C stocks in UK woodlands and forests.

C stocks per ha		tC ha^{-1}	tCO_2 ha^{-1}
Maximum in old-growth stands[A]		200	730
Maximum in managed stands averaged over several rotations[A]		100	370
Average in managed stands averaged over several rotations[A]		60–70	220–60
Average in standing trees across all forest and woodland area[B]	*a*	57	209
Average in litter and deadwood[C]	*b*	17	63
Average in soil (to 1 m)[D]	*c*	234	859
Average in trees, litter and soil	*(a + b + c)*	308	1131
Total UK woodland and forest area[E]	*d* 2.85 Mha		
Total UK C stocks in woodlands and forests		MtC	$MtCO_2$
Standing trees[B]	*a x d*	162	595
Litter and deadwood	*b x d*	48	177
Soil (to 1 m)[F]	*c x d*	663	2432
Total in woodlands and forests	*(a + b + c) x d = e*	878	3223
In timber, joinery and paper (excluding landfill) outside forests[G]	*f*	81	297
Total stocks in forestry	*e + f*	**960**	**3520**

Sources: A: Values from Broadmeadow and Matthews (2003), B: Derived in this section from McKay et al. (2003), C: From this report, Section 3.4, deadwood value added in this section, D: From this report, Section 3.5, E: From Forestry Commission (2010a), area at 31 March 2010, F: Note that C stocks larger than in table 3.10 because of larger forest area used, G: Value taken from Alexander (1997), see Section 3.6. Note that b and c were derived from GB data only, and from tree stands, so may result in an overestimate when used with total woodland areas which include some open spaces for rides, roads, unplanted areas and felled areas awaiting replanting.

8. Biomes are the regional-scale terrestrial ecosystems with characteristic vegetation types that have evolved in particular broad climatic zones, see Glossary.
9. See Box 1.1, for explanation of use of tC, tCO_2 and tCO_2e ha^{-1} (CO_2 equivalent).

In order to quantify the GB woodfuel resource McKay *et al*. (2003) carried out a thorough assessment of the standing biomass in GB forests (the *Woodfuel Resource in Britain* report, WRiB). They used sub-compartment database information for the public forests, and the National Inventory of Woodland and Trees (NIWT) data for private woodlands and empirical relationships between stem volume and biomass of components. The data are summarised in Table 2.2 and Figure 2.8, and the key table is reproduced in Appendix 1. While the stem forms the largest single component (average 55% for conifers and 45% for broadleaves), it is clear that any consideration of forest C stocks has to assess the different components of tree biomass (Figures 2.8 and 2.3).

The total C stock in GB woodland standing biomass estimated in WRiB was 584 $MtCO_2$ (159 MtC, excluding fine roots and deciduous broadleaf foliage, Appendix 1). Of this total, 71% is in private ownership, 29% in broadleaved forest in England, and 22 and 29% in Scotland's spruce and pine forests, respectively (Figure 2.8c and WRiB). Adding to this GB total another 11 $MtCO_2$ for the Northern Ireland forest biomass (calculated from FAO, 2010, above-ground biomass figure and an *R* value of 0.3[10]) gives a UK estimate of 595 $MtCO_2$ (162 MtC, Table 2.1). The figure of 162 MtC is only slightly more than the 150 MtC given by Broadmeadow and Matthews (2003) and repeated in Snowdon (2009). However, it is 34 MtC (26%) more than the value of 128 MtC for 2005 given for the UK tree C stock in recent Forestry Commission statistics based upon inventory data and modelled estimates of growing stock. These include *Forestry facts and figures* (Table 4.1, Forestry Commission, 2010a), the report for the FAO Global Forest Resources Assessment (FRA; FAO, 2010) and the report for the State of Europe's Forests 2011 (SoEF, Forestry Commission, 2010b).

A key reason for the difference is that the FRA statistics calculate the [root + below-ground stump biomass (in oven-dried tonnes, odt)] as 10% of [above-ground growing stock (in m^3)]. Although the comparison is not straightforward, because of assumptions in the conversion of growing stock to total above-ground biomass, this is effectively the same as assuming *R* is 0.22 for conifers, and 0.09 for broadleaves. Thus it gives much lower root biomass than the empirical relationships used in WRiB, which give a GB average derived *R* of 0.38. For conifers alone WRiB estimates give a derived *R* = 0.29, which is similar to the measured values reported by Levy, Hale and Nicoll (2004, see Figure 2.7 above) giving confidence in WRiB. Thus the previous Forestry Commission statistics estimate of below-ground biomass is likely to be a considerable underestimate.

Other reasons for the differences between WRiB and FRA may be using different estimates and thresholds for woodland areas. Further work is necessary to reconcile these figures. However, the new National Forest Inventory will substantially improve the accuracy of these estimates for the GB, with C stock estimates in conifers due in 2012, and in broadleaves due in 2013. Furthermore, it will enable uncertainties to be rigorously quantified.

If the UK total woodland C stock in trees is approximately 595 $MtCO_2$ for a woodland extent of 2.85 Mha, the average figure for UK woodland is approximately 209 $tCO_2\ ha^{-1}$ (57 tC ha^{-1}, Table 2.1). Coincidentally, this exactly matches that given for the temperate forest biome by Dixon *et al*. (1994). However, 209 $tCO_2\ ha^{-1}$ is less than the value for the 'average managed stand' given above, because some forest areas are not planted (rides, roads, other open areas and felled areas awaiting replanting), and large areas of woodland are under-managed. Average soil C stocks for woodland soils in the UK vary greatly with soil type (see Section 3.5 and Table 3.8 later in this report), but a GB average value is approximately 859 $tCO_2\ ha^{-1}$ (down to 1 m,

Table 2.2 Summary of C stocks ($MtCO_2$) in standing tree biomass of GB woodlands and forests (from McKay et al., 2003).

Country	Conifer			Broadleaved			Total
	Stems	Other	Total	Stems	Other	Total	
England	54	36	90	76	94	170	260
Wales	23	18	41	14	17	31	72
Scotland	111	98	209	19	25	44	253
GB	189	151	340	109	136	245	584

Data from McKay *et al*., (2003) Table 3; calculated assuming 50% of biomass is C; excludes fine roots and broadleaf foliage (assumed deciduous); excludes privately owned woods <2 ha.

10. This value was chosen because 89% of growing stock in Northern Ireland is conifers, according to the FRA (FC, 2010b).

Figure 2.8 C stocks in the different components of standing tree biomass in GB woodland: (a) as $MtCO_2$, (b) as percentage of total, and (c) for the different countries. Data from Table 3 of McKay et al. (2003, see also Appendix 1); assuming C is 50% of biomass. No broadleaf foliage is included as these species are all assumed deciduous and thus foliage has no permanent contribution.

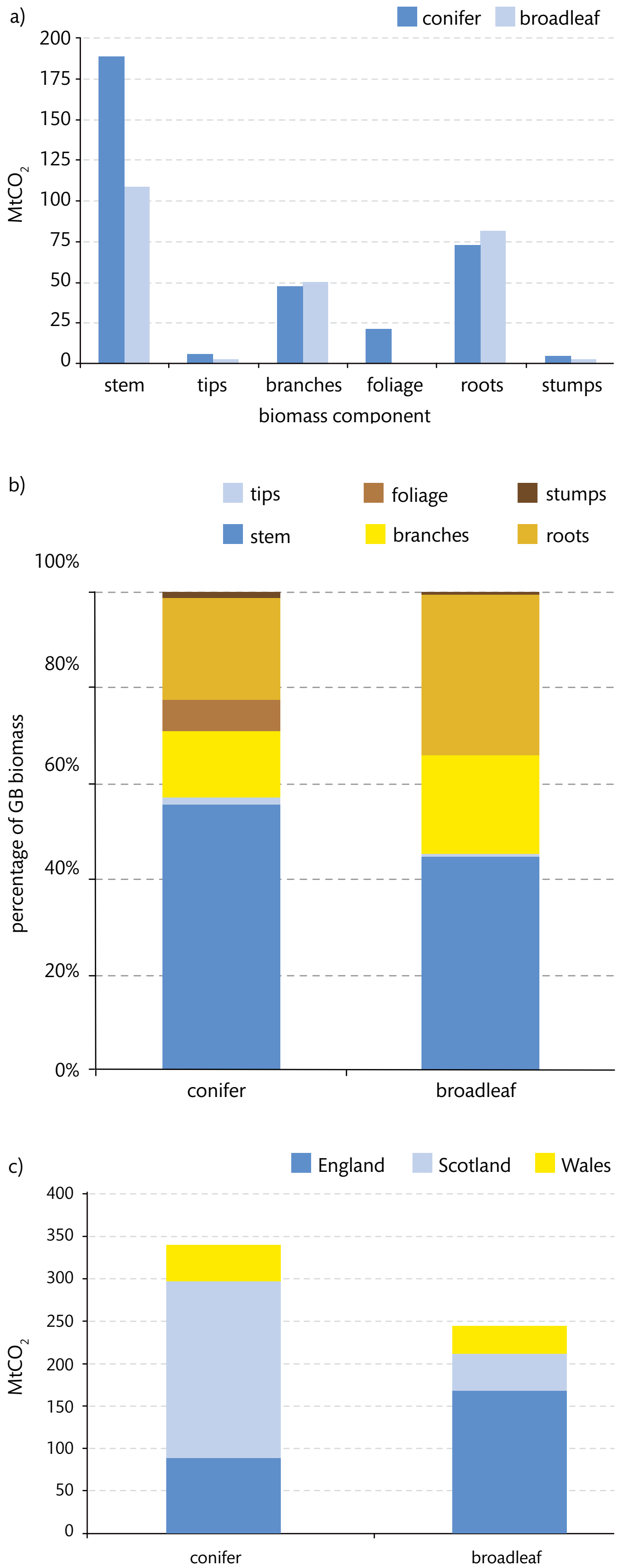


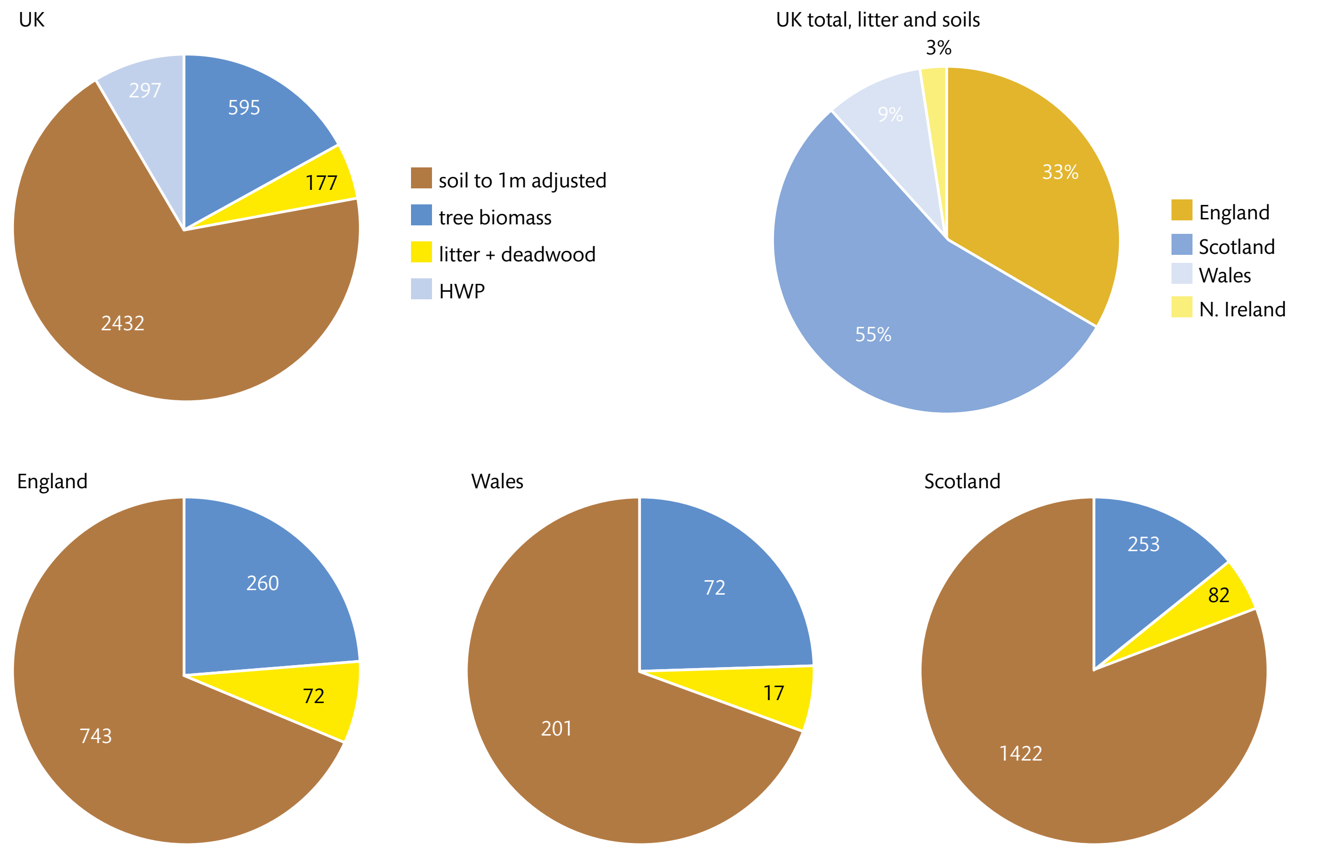
Table 3.10). This is over four times the average figure for C stock in the trees. The carbon in the litter on the soil adds an additional 60 tCO_2 ha^{-1} (see Section 3.4), and to this should be added the deadwood or coarse woody debris component, estimated at 3 tCO_2 ha^{-1} (Gilbert, 2007[11]). The average UK woodland C stock[12] is therefore 1131 tCO_2 ha^{-1}, about 10% more than the reported temperate biome value (1030 tCO_2 ha^{-1}, see beginning of this section). As much of the woodland area in the UK is relatively young, this may be surprising, but it is largely because of the large soil C stock in peatland areas, more typical of the boreal biome, which inflates this crude spatial average. If the deep peat C stocks and areas are excluded, the average soil C stock for GB is 778 tCO_2 ha^{-1}, and the average woodland C stock is then estimated at 1051 tCO_2 ha^{-1}, closer to the temperate biome average.

Scaling up using the UK areas for soil and forest types (see later sections on litter and soil for details), the total woodland C content (soils, litter, deadwood and trees) is approximately 3223 $MtCO_2$ (878 MtC). There is some uncertainty about this number, given the mix of data used, from different dates, and the mix of GB or UK derived figures. Also, it is probable that the C stock in UK forests has been increasing as has occurred across Europe (e.g. Nabuurs *et al*., 2008; Luyssaert *et al*., 2010), because annual growth increment exceeds losses and harvesting (see Table 2.3). Although woodland area might be declining slightly in some areas due to restoration of open-ground habitats, this is a small amount (estimated at 1128 ha y^{-1}, Matthews and Broadmeadow, 2009, <0.05% of total forest area) and much less than present new woodland creation rates (5400 ha in 2009–10, Forestry Commission, 2010a). The results of the new National Forest Inventory will reduce many of these uncertainties considerably. In addition, C stocks in harvested wood products (HWP) in use and in landfill should also be considered; although there is considerable uncertainty about their size, these have probably also been increasing considerably (see Section 3.6).

Clearly, the major component of forestry C stocks is the soil, followed by the trees (Table 2.1, Figure 2.9). For the 'in-forest' C stocks, 55% are in Scotland, 33% England, 9% in Wales and 3% in Northern Ireland. England and Wales have similar proportions of C stocks in soil, trees and litter, but in Scotland, the proportion of the C stocks in soil is considerably larger, because of the extensive peaty soil areas (Figure 2.9).

11. Assuming that the conversion for deadwood volume to mass in odt is 0.5, and to C is also 0.5.
12. This calculation assumes that average soil C stock ha^{-1} in Northern Ireland is the same as that in England.

Figure 2.9 Estimated components of the UK forestry C stocks ($MtCO_2$, data from Table 2.1) and proportions of in-forest stocks for the four countries, and components for England, Wales and Scotland separately. HWP estimates are only known at UK level. Northern Ireland data are not shown separately because the soil C stocks are approximate, not based on soil type, but estimated assuming average soil C ha^{-1} for conifers and broadleaf areas are the same as for England (see FRA section 8.2.3, FAO, 2010).



2.5.2 Forest carbon fluxes

The average temperate forest biome net primary productivity (NPP, see Box 2.1 for explanation of terms) is 28.3 $tCO_2\ ha^{-1}\ y^{-1}$ (7.7 $tC\ ha^{-1}\ y^{-1}$, Saugier, Roy and Mooney, 2001) and the net ecosystem productivity (NEP) (after losses by animal and microbial respiration, see Box 2.1) is approximately 10.3 $tCO_2\ ha^{-1}\ y^{-1}$ (Bonan, 2008). Season-long measurements of whole-stand CO_2 fluxes (net ecosystem exchange, NEE, see Box 2.1) have been made during the last 15–20 or so years using the eddy covariance technique (see examples in Section 2.3). In temperate forests these measurements show a wide range of annual CO_2 uptake values between 7 and 26 $tCO_2\ ha^{-1}\ y^{-1}$ (Baldocchi and Xu, 2005; Hyvönen *et al.*, 2007). This wide range reflects differences in climate, forest type and age, and management history. In the UK the maximum rate of C accumulation in rapidly growing forest is about 36 $tCO_2\ ha^{-1}\ y^{-1}$, but the average is considerably lower, approximately 10 $tCO_2\ ha^{-1}\ y^{-1}$ (Broadmeadow and Matthews, 2003; see Section 2.3). Multiplying the average figure by the total UK woodland area and allowing for 10% of that area that is not planted produces an estimated flux into woody biomass of 26 $MtCO_2\ y^{-1}$ (Table 2.3). Clearly this is a crude approximation,

Table 2.3 Approximate contribution of UK forestry to reducing carbon and GHG in atmosphere.

	$MtCO_2\ y^{-1}$
Total UK net CO_2 flux into woody biomass	+26
Flux out of forest from UK wood production	-6.5
Approximate annual net uptake of atmospheric CO_2 by UK forests[A]	+19
Total UK CO_2 emissions (2009)[D]	481
% of current UK CO_2 emissions sequestered by UK forests	4%
	$MtCO_2e\ y^{-1}$
GHG emissions from forest management activities[B]	0.12
GHG emissions from timber haulage[C]	0.10
Total fossil-fuel derived emissions in forestry	0.22

[A] Assumes that the wood production figure is in equilibrium with losses from decay and combustion of HWP.
[B] This figure includes road building, ground preparation, thinning and harvesting and includes emissions of other GHGs, see Sections 4.4 and 4.6.
[C] Estimated from Whittaker, Mortimer and Matthews (2010) information, see Section 4.7.
[D] DECC provisional figures, released March 2010.

and better estimates can be derived using information on tree species, age profiles and woodland planting and removal rates (Matthews and Broadmeadow, 2009). Work is continuing in this important area, and particularly on projections for future net C uptake values.

The annual UK production of wood (2009 totals = 8.48 green Mt softwood and 0.53 Mt hardwood, *Forestry facts and figures*, Forestry Commission, 2010a) corresponds to a flux from the forest (D, a disturbance flux, see Figure 2.1) of approximately 6.5 $MtCO_2$ y^{-1}. This is approximately 25% of the annual CO_2 uptake. The proportion of this C flux from the forest that becomes an atmospheric emission (and therefore a loss from overall forestry stocks) depends on product use and lifetime. For the simplistic purposes here it is assumed that additions to the HWP pool are balanced by losses annually, so that wood production does not add to stocks, although it is probable that HWP stocks are increasing (see Section 3.6).

Fossil fuel used in establishment, production, harvest and transport of the national wood production may represent an additional emission of approximately 0.22 $MtCO_2e$ y^{-1} (from figures in this report, Section 4); small in comparison to net forest CO_2 uptake rates (<1%).

Therefore the annual **net** CO_2 uptake by UK forests can be estimated as 19 $MtCO_2$ y^{-1}. This very crudely estimated value is surprisingly close to the 15.1 $MtCO_2$ y^{-1} estimated for UK forests for 2008 under the LULUCF reporting process using the C-FLOW model (CEH, LULUCF 2008; Snowdon, 2009) which is estimated using a detailed method with separate conifer and broadleaf forest areas and management assumptions, but only considers forests planted after 1919 (see Matthews and Broadmeadow, 2009, p. 141). Using the CARBINE model, which has more detail on forest types and areas than C-FLOW, Matthews and Broadmeadow (2009, p. 147) estimated a net CO_2 uptake by UK forests and HWP of 9 $MtCO_2$ y^{-1} in 2010, and emphasised that including material and fuel substitution benefits from the wood produced would add an additional 9 $MtCO_2$ y^{-1} net reduction in GHG emissions ('emissions abatement').

The values for annual forest CO_2 fluxes can be put into perspective against the current UK annual total CO_2 emissions, which were 474 $MtCO_2$ in 2009 (DECC final figures[13]). Thus current forest annual CO_2 uptake is approximately 2–4% of national annual emissions. Given that the UK Climate Change Act 2008 set a legally binding target to reduce the UK's emissions of GHGs to at least 80% below 1990 levels by 2050 (i.e. to <155 $MtCO_2e$), forestry policies and practices which increase the net flux of CO_2 into forests could make a significant contribution to the UK GHG balance. If increased forest CO_2 uptake was combined with an increase in the UK's woodfuel production and the substitution by timber for fossil-fuel intensive materials, there would be a considerable contribution to UK emissions abatement, particularly in the medium term as explored by Matthews and Broadmeadow (2009) in the UK Assessment.

However, the figures above for forest C fluxes at UK scale are approximate, and ignore much important detail that will be critical in determining viable management options and appropriate policy. The aim of this review and Forest Research's forest C and GHG-related research is to provide that detail, highlight the gaps in evidence, and seek to fill the key gaps.

13. DECC figures released 1st February 2011; UK emissions of all six GHGs covered by the Kyoto Protocol = 566 $MtCO_2e$.

3. Key components of the forestry C balance

3.1 Introduction

The key components of the forestry C balance are described in detail in each of the sections below: accumulation of C in trees (and the influence of stand management), stocks of C in litter, in soil, and in harvested wood products (HWP) and the C impacts of substitution by wood products for fossil-fuel intensive materials.

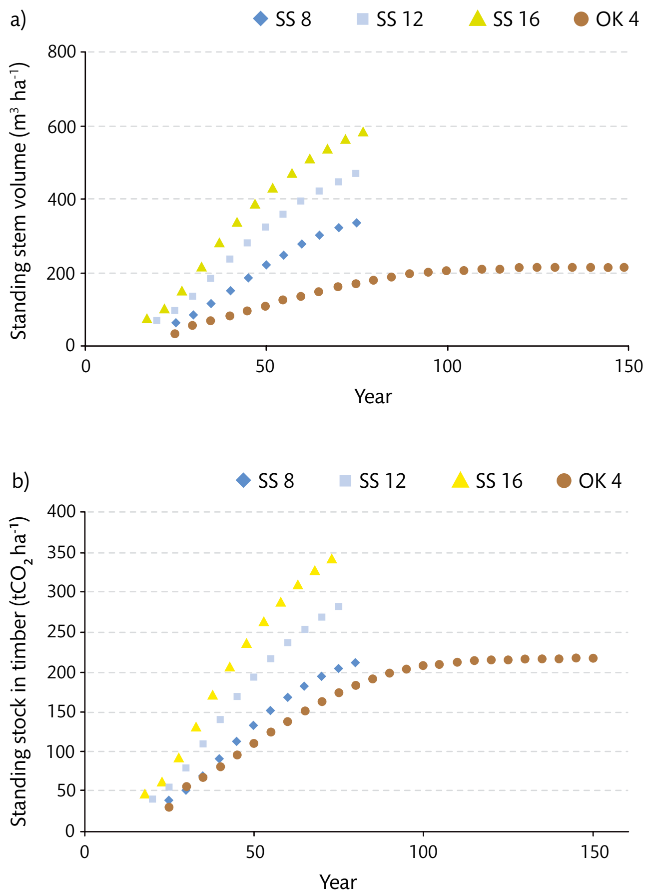
3.2 The dynamics of C accumulation in tree stands

The pattern of C accumulation in a stand of trees over its life cycle follows the growth of timber (the 'increment'), since the dry weight of wood comprises 50% carbon (Matthews, 1993), and stemwood comprises a large part of tree biomass (see Sections 2.2 and 2.4). Therefore, successive forest inventories or models of forecasts of timber growth and yield of forests can be used to estimate C stocks, C accumulation (or sequestration) rates, and how the size of the carbon sink changes with time.

Much quantitative information has been collected on the characteristic time course of stemwood (or sometimes total biomass) accumulation within stands following planting or natural regeneration. Examples of standing volume time courses for Sitka spruce and oak from standard yield tables (Figure 3.1a) show the typical sigmoid ('S'-shaped) pattern of growth in even-aged stands. Stem volume can be converted to mass and thus C stock in the stems, using wood density data (Figure 3.1b). Because of the higher density of oak wood compared to Sitka spruce (0.56 t m^{-3} compared to 0.33 t m^{-3}, Appendix 4) the rate of C accumulation in the standing timber of oak yield class (YC) 4 is similar to Sitka spruce YC 8, for example (Figure 3.1b). In addition, broadleaves typically have a larger non-timber fraction, particularly roots (Section 2.2), so the total C accumulation rate will be comparatively larger in broadleaves than conifers of similar yield class. Note that such yield tables provide no data for early growth (before 20 to 25 years), and may be limited for some species in the data available for later in the growth cycle.

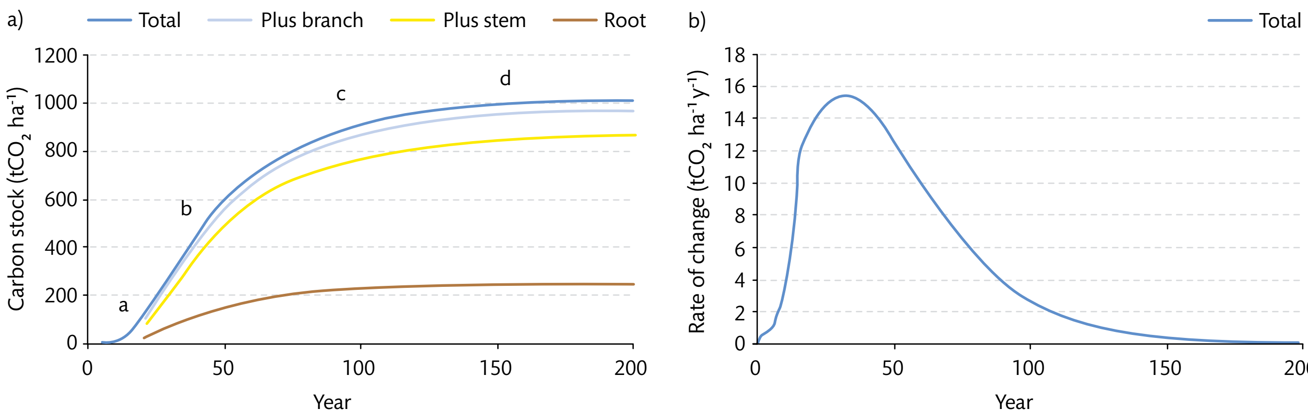
Such yield table data are the basis for the BSORT model of biomass accumulation in forest stands (Matthews and Duckworth, 2005; see Section 5.3.2 later), which can be used to illustrate the different phases of growth (and consequently CO_2 uptake) in even-aged stands (Figure 3.2).

Figure 3.1 Time course of standing timber for three yield classes of thinned Sitka spruce and one of oak, obtained from standard yield tables. Expressed as (a) volume and (b) C stock, calculated from (a) using timber density and C content information.



The hypothetical example in Figure 3.2 is for growth over a 200-year period with no thinning or harvesting – what can be thought of as a model of a stand managed as a 'carbon reserve'. The initial growth rate immediately after planting is relatively slow because of low interception of sunlight until a full canopy develops (the 'establishment phase'), but generally accumulation accelerates until the 'full vigour phase' is reached. During this phase, a high rate of C accumulation is sustained for a number of years, although as a sigmoid time course defines, the rate increases only until the inflexion point on the C stock curve (Figure 3.2, lower graph). The duration and magnitude of the full vigour phase is determined to a large extent by the combination of tree species, site characteristics (e.g. nutrient availability) and climatic conditions. The full vigour phase ends as tree sizes become so large that C losses due to respiration, senescence and death begin to approach the same size as C inputs from

Figure 3.2 (a) Modelled pattern of carbon accumulation in a stand of Sitka spruce (nominally YC 12), left unthinned for 200 years as estimated through the extended yield tables and BSORT model. The stand is assumed to be planted on bare ground with an initial spacing of 2 m. The different components are shown cumulatively: thus the green line shows root + stem + branch. Four phases of growth are shown: (a) establishment phase; (b) full vigour phase; (c) mature phase; (d) old-growth phase. (b) CO_2 uptake rate, calculated by differentiating the total C stock in the upper graph



photosynthesis (the 'mature phase'); i.e. the NPP:GPP ratio declines with age (e.g. Mäkelä and Valentine, 2001; Black *et al.*, 2009). Net growth and C accumulation in the stand slows considerably during this phase. Eventually the tree stand **may** reach a state where C losses more or less balance the gains (the 'old-growth phase'), and the tree stand becomes 'carbon-saturated', with the stand C stock approximately constant, and the net CO_2 uptake close to zero. In the example shown in Figure 3.2 the establishment of a forest stand results in a tree C stock increase on the site from near zero to about 1000 tCO_2 ha^{-1}. The carbon stocks in soil and debris are not included in this example, but C may continue to accumulate in the soil, as the time for soil C to reach equilibrium is typically much longer than that for forest biomass (see Luyssaert *et al.*, 2008).

The discussion of Figure 3.2 leads to two important general points:

1. Planting a stand of trees on an area results in a change in the C stock of that area. Carbon sequestration only occurs while C stocks increase, and the rate is at its maximum in the full vigour phase. Stands of trees alone do not indefinitely sequester C from the atmosphere (but there may be sequestration in soil over a very much longer period – see later).

2. The duration and rate of C accumulation during the full vigour phase of growth are determined by a combination of tree species, site and management characteristics and climate. The size of the average C stock maintained during the old-growth (or 'over-mature') phase is also determined by these factors. If tree species, site types and climatic conditions are selected to maximise the duration and rate of C accumulation during the full vigour phase, this may not necessarily (and indeed is quite unlikely to) result in a high average C stock during the old-growth phase. It is probably more important to select tree species, site types and climatic conditions to maximise the C stock ultimately attained. This is where the sector needs additional information to inform these decisions.

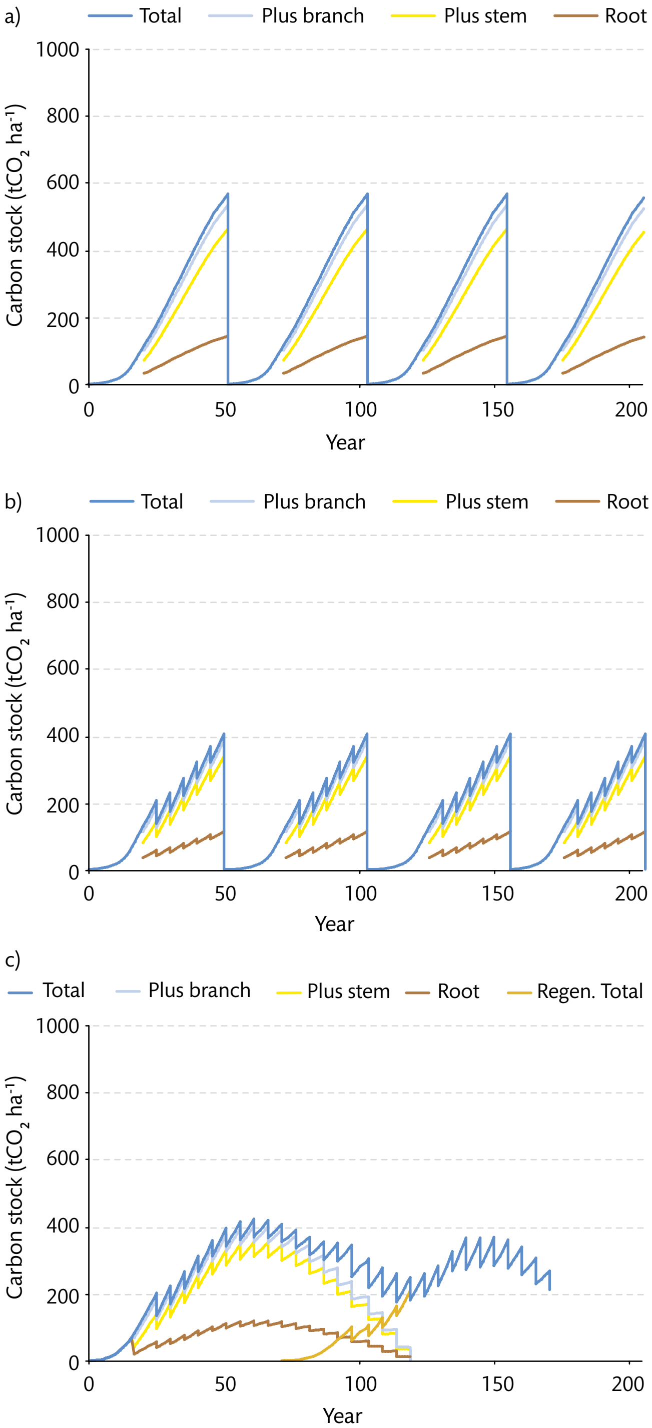
3.3 Impacts of stand management on tree carbon stocks

While site conditions and species are key determinants of C stocks and sequestration potential of woodland, stand management also has profound impacts, particularly decisions about thinning, application of silvicultural systems and rotation length for clearfell systems. Factors such as yield class (which has the largest influence on growth rate, the density of the wood and the risk of disturbance) and site type tend to be secondary influences, particularly as they will often be considered when making decisions about stand management. The potential impacts of stand management on the development of C stocks and sequestration over time can be explored with models such as CSORT (for an outline of the model see Section 5.3). Example estimates are shown in Figures 3.2–3.4 for stands of different species and yield class under a range of management prescriptions:

1. New Sitka spruce woodland, left unthinned and unfelled so managed as a 'carbon reserve' (Figure 3.2).
2. New Sitka spruce woodland managed for timber production as an even-aged stand with no thinning (Figure 3.3a).
3. As 2 but with regular thinning (Figure 3.3b).
4. As 3 but followed by conversion to selection silviculture (Figure 3.3c).
5. As 3 but for oak, not Sitka spruce (Figure 3.4).

Note that these figures give only the tree C stocks, and do not assess where the C 'goes' after for instance thinning or felling. Calculations of this wider C balance are given in Chapter 5.

Figure 3.3 Time course of carbon accumulation in the biomass of an average (YC 12) stand of Sitka spruce in Britain estimated using the BSORT model. The stand is assumed to be planted on bare ground with an initial spacing of 2 m, felled and replanted on a 50-year rotation. (a) stand left unthinned; (b) thinned according to standard management tables (MTs, Edwards and Christie, 1981); (c) thinned then, after an initial even-aged rotation of 50 years, converted to LISS (selection). The different components are shown cumulatively: thus the green line shows root + stem + branch.



For the Sitka spruce stand managed for timber production (Figure 3.3a) calculated C stocks over the rotations follow a cycle between close to zero and 570 tCO_2 ha^{-1}, and the long-term average C stock maintained on the site is approximately 240 tCO_2 ha^{-1} (Table 3.1). Note that the foliage contribution is small (the difference between green line, root + stem + branch and total).

When the effect of regular thinning is included (Figure 3.3b) C stocks follow a cycle between close to zero and 403 tCO_2 ha^{-1} with a long-term average C stock of approximately 160 tCO_2 ha^{-1} (Table 3.1). The maximum sequestration rate is higher for thinned than for unthinned stands, because the remaining trees are more vigorous, with space to grow. However, the total C stock in the trees is greater in the unthinned stands. Conversion of the stand to selection silviculture (LISS[14], Figure 3.3c), obtained by increasing the thinning intensity to allow regeneration at age 50 rather than clearfelling, results in an increased long-term C stock of approximately 260 tCO_2 ha^{-1}. The C stock of the oak stand with a much lower yield class (Figure 3.4) cycles between close to zero and 495 tCO_2 ha^{-1}, with a long-term average C stock of approximately 321 tCO_2 ha^{-1} (Table 3.1).

The different methods of stand management have impacts on C stocks that would be expected intuitively:

1. Leaving trees standing and protecting them leads to larger long-term tree C stocks compared with stands which are periodically felled.
2. By implication from (1), shorter rotations lead to smaller long-term C stocks compared to longer rotations.
3. Thinning reduces long-term tree C stocks compared with unthinned stands.
4. Low intervention (e.g. LISS-type) management leads to greater long-term average C stocks compared to conventional clearfell management prescriptions.

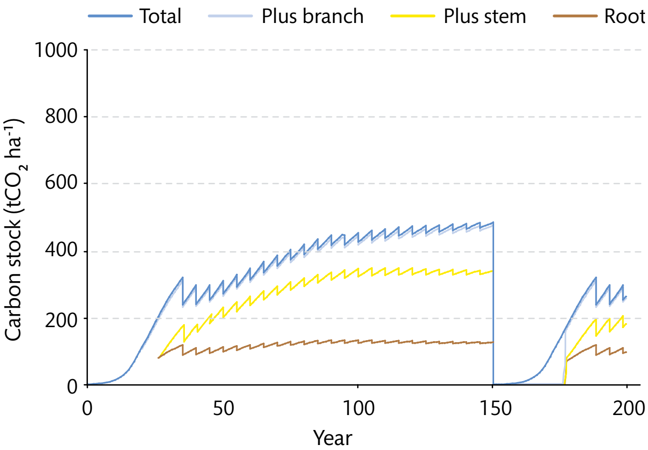
It is also evident from comparison of Figure 3.4 with the others that species and yield class do not necessarily have the impact that might be expected – other than through their influence on decisions about stand management.

Results such as those shown in Figures 3.2–3.4 can be produced using the BSORT model for many combinations of tree species, yield class and management regime. Example key results are the maximum rate of C accumulation (sequestration) the size of the long-term

14. LISS: low impact silvicultural system; see Glossary.

C stock, the period over which this is attained and the consequent average rate of sequestration (Table 3.1). Similar model results reported for British woodlands calculated using the Forest Research CARBINE carbon accounting model (Thompson and Matthews, 1989a; Matthews, 1991) or the Centre for Ecology and Hydrology C-FLOW model (Dewar, 1990, 1991; Dewar and Cannell, 1992; Milne, 2001) are summarised in Table 3.2. Both of these models are recognised as leading examples of mathematical models of forest carbon dynamics. For example, results

Figure 3.4 Time course of C accumulation in an average (YC 4) stand of oak in Britain as estimated using the BSORT model. The stand is assumed to be planted on bare ground with an initial spacing of 1.2 m, thinned according to standard management tables (Edwards and Christie, 1981), then felled and replanted on a 150-year rotation. The different components are shown cumulatively: thus the green line shows root + stem + branch.



from C-FLOW (Figure 3.5a, Table 3.2, Dewar and Cannell, 1992) show that increasing yield class of Sitka spruce results in increased rate of C accumulation in trees (and litter) but shorter rotation, so for example a doubling of YC from 12 to 24 only increased tree C stock by 33% (Figure 3.5a). Thinning is modelled to result in reduced C stocks in both the tree and litter pools. The different models produce broadly comparable tree C stock values for YC 12 Sitka spruce (first three bars in Figure 3.5b) although C-FLOW values are higher than CARBINE as they are when comparing YC 10 Scots pine. As expected the long-term C stock is higher for the higher yield classes (Scots pine examples) and/or longer rotation examples such as Corsican pine (YC 16) and beech and oak examples. Note that the poplar modelled values are for a wide-spaced, unthinned situation.

The reported **maximum** rates of carbon sequestration in Tables 3.1 and 3.2 are consistent with those reported from the few CO_2 flux measurements that have been carried out in stands during their full vigour phase of stand development (see Section 2.3 and Jarvis *et al.*, 2009). However, maximum rates are usually a poor indicator of the long-term rate of C sequestration, which is determined by the combination of long-term C stock and the period over which this is attained. As Tables 3.1 and 3.2 show, the long-term C stocks estimated for different types of forestry systems fall into two distinct groups: smaller values for stands with some form of active management for production and larger values for woodlands with minimal intervention to create 'carbon reserves'. This is reported consistently in the available literature.

Table 3.1 Rates of CO_2 sequestration and long-term C stocks in biomass of trees, for example British stands, modelled using BSORT.

Management prescription[A]	Maximum CO_2 sequestration rate		Long-term CO_2 stock		
	Value $tCO_2\ ha^{-1}\ y^{-1}$	Time years	Value $tCO_2\ ha^{-1}$	Period years	Rate $tCO_2\ ha^{-1}\ y^{-1}$
Sitka spruce, YC 12, 2 m initial spacing					
Unthinned, no fell, low disturbance	15.4	32	960[B]	130	7.4
Unthinned, fell on 50 y rotation	15.4	32	239	50	4.8
MT thin, fell on 50 y rotation	20.5	40	158	50	3.2
MT thin then conversion to LISS	20.5	40	256[C]	180	1.2
Oak, YC 4, 1.2 m initial spacing					
MT thin, fell on 150 y rotation	13.0[D]	55	321	150	2.1

[A] For details, see beginning of Section 3.3. MT = management table.
[B] As no fell is assumed, tree C stock tends towards an asymptote, for this calculation a cut-off increment of <1 $tCO_2\ ha^{-1}\ y^{-1}$ was used.
[C] The establishment of LISS cover is still in an early stage, therefore growth over the whole period is used.
[D] The timing of the maximum CO_2 sequestration rate is taken after the time of first thinning, i.e. when all trees have reached measurable size.

Figure 3.5 (a) Modelled effects of yield class on Sitka spruce long-term tree C stock, thinned and unthinned (T and UT, data from Dewar and Cannell, 1992; Table 5); (b) different modelled estimates of tree C stock, all for 'thinned' situation, except poplar; Sitka spruce examples are YC 12 (see Table 3.2 for details).

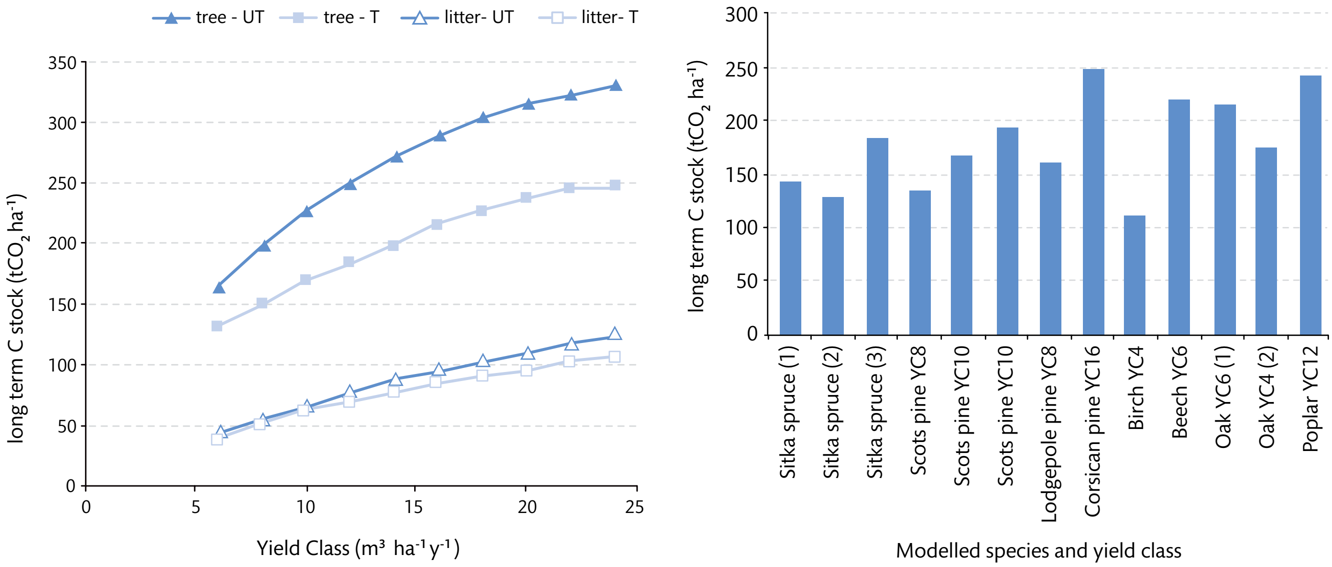


Table 3.2 Previously reported modelled C sequestration rates and long-term C stocks in trees for British woodlands.

Species	Yield class	Initial spacing (m)	Management	Rotation years	Maximum carbon sequestration rate		Long-term carbon stock			Ref
					Value tCO_2 ha^{-1} y^{-1}	Time	Value tCO_2 ha^{-1}	Period years	Rate tCO_2 ha^{-1} y^{-1}	
Scots pine	8	2.0	Thinned	70			136			1
Scots pine	10	2.0	Thinned	65			167			1
Birch	4	1.5	Thinned	45			112			1
Corsican pine	16	2.0	Thinned	50			248			1
Oak (1)*	6	1.2	Thinned	150			216			1
Sitka spruce (1)	12	2.0	Thinned	55			143			1
Sitka spruce	12	2.0	Unthinned – no fell	55	14.7	30	734	90	8.1	2
Sitka spruce	12	2.0	Thinned – no fell	55	9.2	30	120	90	4.8	2
Sitka spruce (2)	12	2.0	Thinned – clearfell	55	8.8	30	128	35	3.7	2
Sitka spruce	24	2.0	Unthinned	47			330			3
Sitka spruce	24	2.0	Thinned	47			246			3
Sitka spruce	12	2.0	Unthinned	59			250			3
Sitka spruce (3)	12	2.0	Thinned	59			183			3
Sitka spruce	6	2.0	Unthinned	68			165			3
Sitka spruce	6	2.0	Thinned	68			132			3
Sitka spruce	12	2.0	Unthinned – no fell	-	18	25	807	90	8.8	4
Sitka spruce	12	2.0	Unthinned – clearfell	50	18	25	257	30	8.4	4
Poplar	12	2.7	Unthinned	26			242			3
Scots pine	10	1.8	Thinned	71			195			3
Lodgepole pine	8	1.8	Thinned	62			161			3
Beech	6	1.2	Thinned	92			220			3
Oak (2)	4	1.2	Thinned	95			176			3

*Numerals refer to different results for similar conditions, and are used to identify values in Figure 3.5.
References: 1. Thompson and Matthews (1989a); 2. Matthews (1991), 3. Dewar and Cannell (1992); 4. Broadmeadow and Matthews (2003).

3.4 Woody debris and litter

3.4.1 Introduction

Litter is an important component of the woodland C balance. The amount of carbon that accumulates and is stored in litter depends on its quality, quantity and decomposition rate, which are all influenced by tree species, tree age, nitrogen deposition, acidity, climatic variables and forest management. The decomposition rate is also intimately linked with soil formation and thus with soil type. In general, litter decomposition is much faster in broadleaved than in coniferous forest, which is due to humus formation, the higher acidity of coniferous litter, and the difference in canopy density and hence the light and water reaching the forest floor (see e.g. Schulp *et al.*, 2008). In this section litter formation, its decomposition and the C stock in litter are reviewed and new figures are produced for typical values in British woodlands.

3.4.2 C stocks in litter

The review of the relevant literature indicates that the litter on the forest floor (L + F layers, see Box 3.1 for explanation) can store on average 66 tCO_2 ha^{-1} (a range from 15 to 150 tCO_2 ha^{-1}, depending on tree species, soil and climate, calculated from figures in Berg *et al.*, 2007). Figures for peaty gley soils in NE England range between 26 and 110 tCO_2 ha^{-1} (Zerva and Mencuccini, 2005a). Estimates of litter C stock from the C-FLOW forest carbon dynamics model for UK conditions for a range of Sitka spruce yield classes and other species (Table 3.3, Figure 3.5a) are broadly consistent with these observed figures. The litter is therefore calculated to represent approximately 23–31% of the total C in trees and litter, or 30–45% of the C in the trees, increasing slightly with thinning, and with increasing yield class. However, at very nutrient poor sites in Sweden, the C accumulation in the forest floor layer reached 235 tCO_2 ha^{-1} (Berg *et al.*, 2007), similar to C stocks in trees (see e.g. Table 2.1).

Estimates of average C stocks in the L and F layers in forest soils from the new BioSoil measurements are shown in Figure 3.6 (details of BioSoil are given in the next section). These were calculated from 166 BioSoil plots using the measured C concentration and density of the L and F layers, sampled from an area of 25 x 25 cm^2.

Across all the soil types and sites, the L layer contained on average 27 tCO_2 ha^{-1} and the F layer contained slightly more, 33 tCO_2 ha^{-1}. It should be noted that at sites on soils with an H layer less than 1.5 cm thick, the F and H layers were sampled together. Taken together, this litter C stock in the L and F layers (which excludes the coarse wood debris) is an important component of the forest C stock, adding an

Figure 3.6 Carbon stocks in the litter (L) and fermentation (F) layers under broadleaves and conifers from 166 BioSoil plots across GB, measured in 2007.

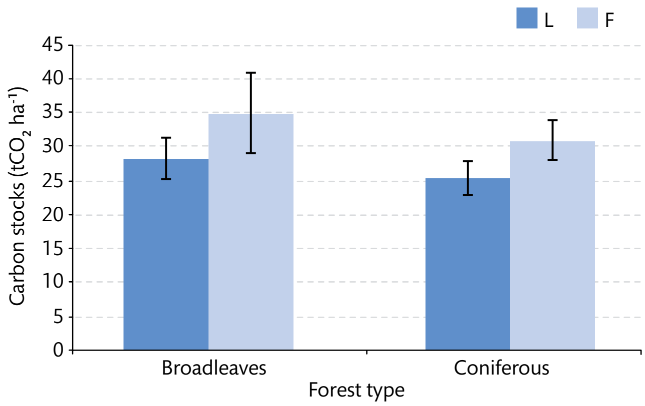


Table 3.3 Modelled estimates of tree and litter C stocks for UK conditions, using the C-FLOW model. Data from Dewar and Cannell (1992); partly from Table 3.2.

Species	Yield class	Initial spacing (m)	Management	Rotation (years)	Long-term carbon stock (tCO_2 ha^{-1})		
					Trees	Litter	Total
Sitka spruce	24	2.0	Unthinned	47	330	125	455
Sitka spruce	24	2.0	Thinned	47	246	106	352
Sitka spruce	12	2.0	Unthinned	59	250	77	326
Sitka spruce	12	2.0	Thinned	59	183	70	253
Sitka spruce	6	2.0	Unthinned	69	165	44	209
Sitka spruce	6	2.0	Thinned	68	132	40	172
Poplar	12	2.7	Unthinned	26	242	84	326
Scots pine	10	1.8	Thinned	71	194	70	264
Lodgepole pine	8	1.8	Thinned	62	161	55	216
Beech	6	1.2	Thinned	92	220	99	319
Oak	4	1.2	Thinned	95	176	73	249

amount that is between 5 and 10% of the soil C stock, and 30% of the average tree C stock (e.g. Table 2.1), similar to the proportions estimated by the C-FLOW model (Table 3.3, Dewar and Cannell, 1992).

BioSoil is the first large European network where density of organic material for each layer was measured and is therefore a valuable resource for model input and calibration. Overall, Level II,[15] BioSoil and additional experimental data from current research under the FR Soil Sustainability Research programme show lower litter C stocks than the C-FLOW model estimates for UK conditions (Table 3.3). The model outputs suggest lower litter C stocks in broadleaves than conifers, as expected due to their higher decomposition rate. Acidic conditions created by growing conifers inhibit microbial activity and reduce decomposition rates and thus the accumulation of litter and C sequestration can be much larger. However, measured data from BioSoil (Figure 3.6, Table 3.4) suggest that C stocks in the L and F layers for broadleaved stands were similar and possibly slightly larger than those under coniferous stands. This is largely because both the L and F layers under broadleaves were much more compacted than under conifers (higher density, Table 3.4), but may also reflect different typical ages of broadleaved and conifer stands. While the L and F layer C concentration and depth were higher in conifers than in broadleaves, the contrasting density had a substantial impact on the overall C stocks and resulted in similar C stocks under both forest types.

The mean litter values (both L and F) can be combined with the total area under broadleaved and conifer forests in the UK to estimate litter stocks. Estimated total L and F stocks are therefore 75.7 and 92.5 $MtCO_2$, respectively (Table 3.4), and the total C stock in UK forest litter estimated from BioSoil = 168 $MtCO_2$ (45.9 MtC). This is likely to be a small overestimate, as the L + F values were determined in tree stands, whereas the forest areas also include open spaces due to rides, roads and other unplanted spaces, and felled areas awaiting restocking (in total approximately 10–15% of forest area). Vegetated areas will have some litter but probably less than tree stands, while felled areas waiting for replanting may have more litter, so it is difficult to refine the estimate further.

Even allowing for a 10% overestimate, the BioSoil derived estimate is much higher (83%) than that reported in recent Forestry Commission statistics (92 $MtCO_2$, Forestry Commission, 2010b; FAO, 2010). This latter value was

Table 3.4 Mean characteristics of the litter (L) and fermentation (F) layers in GB BioSoil plots and scaled-up estimates of C stocks for the countries, GB and UK.

	Broadleaves		Conifers		
No. of plots	77		89		
Layer:	*L*	*F*	*L*	*F*	
Density (kg m^{-3})	64	114	38	59	
C %	46.3	35.7	49.8	44.2	
Mean depth (cm)	8.4	9.8	9.9	11.9	
Mean C stock (tCO_2 ha^{-1})	28.3	34.8	25.4	30.9	
Country C stocks ($MtCO_2$)					Total
England	21.5	26.5	9.3	11.3	68.6
Wales	3.6	4.5	4.0	4.9	16.9
Scotland	8.4	10.3	26.5	32.3	77.6
Northern Ireland	0.6	0.7	1.7	2.0	5.0
GB	33.6	41.3	39.8	48.5	163.1
UK	34.2	42	41.5	50.5	168.2
(MtC)					Total
GB					44.5
UK					45.9

Note: Conifer and broadleaved forest areas from Forestry Commission statistics (Forestry Commission, 2010a).

derived from the litter C stocks for areas afforested since 1920 modelled by the CEH model, increased to cover the total area (FAO, 2010), and was quoted with a range from 92 to 110 $MtCO_2$. The original C-FLOW model (Dewar, 1991) included only two separate components for undecomposed litter and the C in the humus and lower layers, so the current CEH model may be including C that would be measured in the F layer as in the soil organic matter pool, which would underestimate the combined L and F layers measured here (e.g. Table 3.4, Figure 3.6). Clearly, the measured values reported here are likely to be an improvement on these previous modelled values.

3.4.3 Rates of litter C input

Rates of C input to the soil surface from litterfall (including leaves, branches, cones, frass) at the UK Level II[16] intensive monitoring network sites have been measured at between 10 and 26 tCO_2 ha^{-1} y^{-1} (see Table 3.5). Values of C content of litter components showed small variations according to species between 46% in beech foliage and 52 % in Scots pine twig debris.

15. Intensive Forest Monitoring network, see Glossary.
16. See Glossary.

Table 3.5 Annual litterfall values for UK Level II sites – annual averages from three sites per tree species during a 5–6-year monitoring period.

Species	Litterfall range (tCO_2 ha^{-1} y^{-1})
Oak	10–26
Beech	11–19
Scots pine	11–23
Sitka spruce	10–21

Data from: www.forestry.gov.uk/fr/INFD-67MEVC.

Nitrogen deposition can increase the amount of litterfall. For example, total annual litterfall of Corsican pine was 14 tCO_2 ha^{-1} y^{-1} and of beech was 11 tCO_2 ha^{-1} y^{-1} in low N deposition areas compared to 30 and 15 tCO_2 ha^{-1} y^{-1}, respectively, in a high N deposition area (Vanguelova and Pitman, 2009). The litterfall biomass in 30-year-old compared to 50-year-old Corsican pine stands was not significantly different (15 tCO_2 ha^{-1} y^{-1} compared to 16 tCO_2 ha^{-1} y^{-1}). However, the litter quality (fractions and chemical composition) differed significantly, with needles being 88% of total litterfall in the young stand compared to 25% in the older stand where the rest of the litterfall was made of heavier, more slowly decomposable fractions, such as cones and twigs (Vanguelova and Pitman, 2009). The difference in litterfall quality and quantity influences the decomposition rate and breakdown and thus input of C into the soil. For broadleaf species that have sporadic masts (e.g. beech and oak) or in good coning years for pines, the annual litterfall can increase dramatically from dropped seed. For example, under beech in the UK Level II plots, C input changed from 13 tCO_2 ha^{-1} in the 2004 mast year to 4 tCO_2 ha^{-1} in the non-mast year 2005.

Clearly, in establishing forest floor C balances it is critical that decomposition rate is reported alongside litterfall amounts and forest floor C stocks. Leaf litter from broadleaved trees provides organic material that is generally quickly decomposed and incorporated into the upper soil horizon. Differences in litterfall quality and quantity influence the decomposition rate and breakdown and thus the dynamics and incorporation of C into the soil. Litter N and lignin content, C/N ratio, leaf area and calcium content are important litter qualities that are strongly related to litter decomposition rate (Table 3.6). Additional important factors affecting the rate of leaf decomposition are soil pH, soil moisture and temperature, with moist, warmer and base-rich soils providing conditions for the quickest rate of decomposition.

Litter decomposition is frequently negatively correlated with N deposition and can be twice as rapid in areas with lower N deposition compared to rates in areas with very high N deposition (Vanguelova and Pitman, 2009). Factors such as climate, deposition, soil type and litter quality help in the O horizon[17] development of different humus forms or types. These different humus types are of different thickness and density, and can contain different amounts of carbon (Baritz *et al.*, 2010). For example, mull and mull-moder humus forms often develop under broadleaved forests, which have rapidly decomposed litter and are on fairly rich soils. Thus these humus types are thin, but denser and have lower C% (see Table 3.4). Moder and mor-moder humus types usually develop under coniferous forest with slowly decomposing needles and on poor nutrient soils, and are thus thicker, but less dense and have higher C % (see Table 3.4). Mor and peaty mor humus types usually develop under coniferous stands and under very wet soil conditions, so the decomposition is very slow and they are very thick and contain significantly higher C stocks. It is important to note that, in addition to containing different amounts of C, these different humus types have different sensitivities to C loss after disturbance, because of the organic matter residence times of the different materials (see Morison *et al.*, 2010, pp. 14ff).

3.4.4 Root inputs to C cycling

Tree fine roots (<2 mm) are very dynamic and play a key role in forest ecosystem carbon and nutrient cycling and accumulation. Root biomass, distribution and turnover depend on tree species, soil type and climatic conditions. Higher root density is generally found in nutrient-poorer forest ecosystems and root biomass of deciduous trees is higher than that of conifers. For example, Finér *et al.* (2007) analysed variation in fine root biomass of three tree species in many stands across Europe and showed that the fine root biomass of Norway spruce (2.97 odt ha^{-1}) was similar to that of Scots pine (2.77 odt ha^{-1}) but lower than that of beech trees with 3.89 odt ha^{-1}. A strong relationship exists between the fine root biomass and the above-ground biomass, suggesting that trees regulate proportionally their carbon allocation depending on the site conditions (Finér *et al.*, 2007), although estimates of root carbon input are often dependent upon the methodology used (Majdi *et al.*, 2005). In Norway spruce stands in Germany, rates of fine root turnover were higher at more acid soil sites, and root carbon inputs were 0.3 odt ha^{-1} y^{-1} on soil with a high base saturation and 1.5 odt ha^{-1} y^{-1} on acidified soils (Godbold *et al.*, 2003).

17. See Box 4.1.

Table 3.6 Estimated/measured litter parameters of different tree species. (From Vanguelova and Pitman, 2011).

Species	Litter N (%)	Litter SLA [A] (g cm^{-2})	Litter BC content [B]	Litter C %	Rate of decomposition	References
Ash	1.24–2.20	180–300	3.83	31.1	Rapid	Cornelissen, 1996
Alder		210			Rapid	Cornelissen, 1996
Sycamore	0.94	(213 fresh) ~250 litter	3.03	46.2	Rapid	Hobbie *et al.*, 2006; Cornelissen, 1996; Pugh and Buckley, 1971
Hazel	1.34	275–310	2.6	n/a	Rapid	FR
Hornbeam	1.1	210	1.46	46.9	Intermediate–rapid	Hobbie *et al.*, 2006
Birch	1.0–1.4	170–320	1.31–1.41	47.8–52.8	Intermediate	Cornelissen, 1996; Hobbie *et al.*, 2006
Sweet chestnut	0.98–1.30	~150–200	1.19–1.28	49.7–50.7	Intermediate	FR
Southern beech *Nothofagus obliqua*	~ 0.6		~0.97		Intermediate	Wigston, 1990; Adams and Attiwill, 1991
Willow (*Salix alba*, *S. fragilis*)		149/160			Intermediate	FR; Cornelissen, 1996; Withington *et al.*, 2006
Poplar (*Populus trichocarpa*), *P. tremula/P.nigra*		(70–170 fresh) 149–160			Intermediate	FR; Cornelissen, 1996; Withington *et al.*, 2006
Oak	1.0–1.38	165–190	1.83–1.95	36.6–51.1	Intermediate–slow	FR; Hobbie *et al.*, 2006
Eucalyptus (*Eucalyptus nitens*)		61–66			Slow	FR; Wedderburn and Carter, 1999; Lopez, Pardo and Felpeto, 2001
Eucalyptus (*E. gunnii*)		57–67			Slow	FR; Adams and Attiwill, 1991

[A] SLA: Specific Leaf Area, area of leaf per unit mass.
[B] BC: base cation Ca + K + Mg % as defined by Cornelissen and Thompson (1997).

Box 3.1 Soil horizon terms and descriptions (partly from Kennedy, 2002)

Soils are comprised of three main horizons (A, B and C), which may be overlain by a surface organic horizon (O), particularly if the soil is uncultivated:

- O Organic matter including loose, fresh litter, partially decomposed and decomposed litter. Horizons with >30% organic matter are organic horizons under the Forestry Commission classification (Kennedy 2002).
- A Topsoil layer, mainly mineral, which may contain some humified (decomposed sufficiently so as not to be recognisable) organic material; darker horizon.
- (E) As 'A' but oxides and organic matter have been removed and transported downwards; paler horizon.
- B Subsoil layer; may have received material from horizons above, or be weathered parent material from C horizon.
- C (or R) Loose, unconsolidated material (C), usually the 'parent' material for formation of soil; can also be hard bedrock (R).

The O horizon can be separated into three layers, particularly on uncultivated soils:

- L Litter layer formed of recognisable plant and soil animal remains.
- F Fermentation layer, usually consisting of a mixture of organic matter in different stages of decomposition.
- H Humose layer, consisting largely of humified material with little or no plant structure visible.

3.5 Soil C stocks in UK forestry

Soils under forests can contain more organic carbon than the trees, particularly the peat-based soils common in the upland areas of the UK. The literature reviewed so far and the available Soil Sustainability Research programme data show that estimates of C content in forest soils can vary between 90 and 2800 tCO_2 ha^{-1} (25–763 tC ha^{-1}). This variation depends on the soil depth over which these stocks are measured and calculated, the soil type and stand age. The variation between soil types is exemplified by data from the UK Level II sites (Table 3.7). Soil C content estimates in the Level II sites was based on measured soil C concentrations and the pre-set bulk densities of 0.1 g cm^{-3} for F horizons and 0.3 g cm^{-3} for H (peat) organic horizons.

While these examples show the large variation in C stocks between soil types (approximately 10-fold), they are for a very limited set of sites, and do not allow estimation of soil C stocks across UK forests and woodlands. Furthermore, earlier national soil survey figures are dominated by those from soils under agricultural management, due to the small proportion of land area under forestry. However, the recent BioSoil survey has enabled a much better estimate of soil C stocks in UK woodlands and forests, and this is described in the next section.

3.5.1 BioSoil – a new forest soil survey

BioSoil was a large EU soil and biodiversity survey in forestry, part of the programme of the Forest Focus regulation (2003–6). The project was a test for the development of operational soil monitoring at a large scale. BioSoil is the largest single soil monitoring exercise implemented so far at EU scale. It was based on a 16 x 16 km grid under woodland (see BioSoil map right, Figure 3.7).

Figure 3.7 Map showing location of GB BioSoil sampling plots. FE is Forest Enterprise.

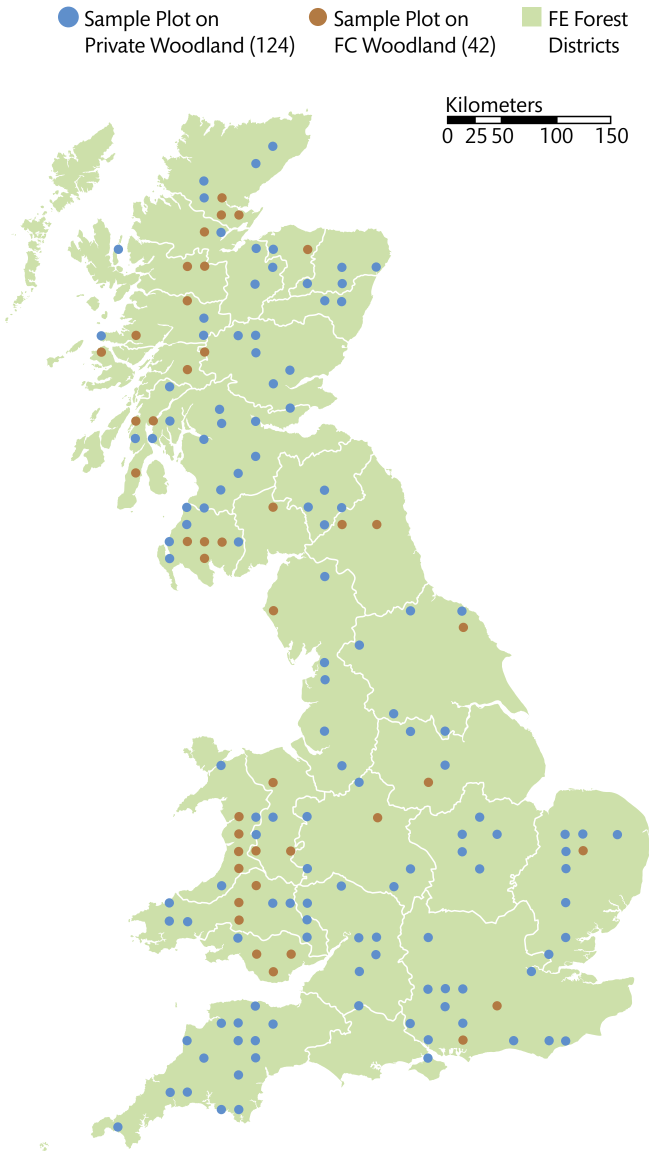


Table 3.7 Soil organic C content at UK Level II sites in 1995, measured to depth of 1 m.

Site	Soil type	tC ha^{-1}	tCO_2 ha^{-1}
Llyn Brianne	Cambic stagnohumic gley soils	753	2761
Rannoch	Humo ferric gley podzol	645	2365
Sherwood	Brown podzolic soil	308	1130
Grizedale	Typical brown podzolic soi	292	1071
Coalburn	Cambic stagnohumic gley soils	290	1064
Savernake	Argillic pelosol	278	1020
Alice Holt	Pelo-stagnogley	157	576
Loch Awe	Stagnogleyic brown podzolic soil	124	455
Tummel	Ferric podzol	106	389
Thetford	Brown calcareous sands	76	279

Three of the eight main specific objectives of the soil module of BioSoil were:

- To establish an improved common European baseline of forest soils for environmental applications, e.g. acidification and/or eutrophication status; C stock assessment, impacts of climatic changes;
- To detect and explain temporal changes in forest soils;
- To improve the existing quality assurance/quality control strategy for European forest soil condition survey.

Measurements and analyses performed on BioSoil plots included soil organic C concentration (C%) and bulk density (BD) for both soil horizons and soil layers 0 to 5, 5 to 10, 10 to 20, 20 to 40 and 40 to 80 cm. This was the first soil survey which measured C in soils as deep as 80 cm with additional measured BD at each soil depth.

Analyses were performed at national level by rigorously inter-compared and benchmarked laboratories. In the UK analyses were performed by the Forest Research laboratory at Alice Holt. In addition to the analysis, a full soil profile description according to the FAO guidelines (1990) and classified according to the World Reference Base (WRB, 2006) was carried out in the field.

BioSoil is the best evaluation of soil conditions in forestry in GB so far, due to its very good spatial coverage, the detailed measurements up to a depth of 80 cm, and the measured BD, all under the best quality assurance and quality control strategy. In GB, there are 167 BioSoil plots, of which 72 are in England, 26 in Wales and 69 in Scotland. Overall, 42 plots are on Forestry Commission land with the remaining 125 plots on private land.

3.5.2 Carbon storage in the main British forest soil types

A detailed split between forest soil C stocks in each main soil group has been carried out for this review. Soil C stocks from 166[18] of the 167 BioSoil plots (see Figure 3.8) were calculated and averaged for each main soil group (for details of the soil groups in the Forestry Commission classification, see Appendix 2, and Kennedy, 2002):

1. Brown earths.
2. Podzols and ironpan soils.
3. Surface water gley soils.
4. Ground water gley soils.
5. Peaty gleys.
6. Deep peats.
7. Rendzinas and rankers.

Samples were assigned into the five soil depths, in each of which C% and BD were measured. Averaged total C stocks for each main soil group and different soil depths are presented in Table 3.8[19]. Litter and F layer data are not included in the soil C stocks in Table 3.8; they are reported separately in the previous Section 3.4.

Table 3.8 Soil carbon stocks (tCO_2 ha^{-1}) at the 166 GB BioSoil sites, measured down to 80 cm. Values are averaged for main soil groups. Values do not include L and F layer C content (Vanguelova *et al.*, in prep.).

Soil group	Number of plots	Soil depth (cm)							
		0–5	5–10	10–20	20–40	40–80	Total (0–15)	Total (0–80)	Total[A] (0–100)
Rankers and rendzinas	3	90	69	113	127	88	215	487	528
Brown earths	74	90	54	91	135	125	189	495	558
Podzols and ironpans	14	79	53	105	142	119	185	499	565
Surface water gleys	13	99	67	99	126	149	215	540	613
Ground water gleys	12	88	60	130	158	134	213	569	635
Peaty gleys / podzols	36	149	131	223	375	297	392	1174	1329
Deep peats	14	115	120	245	497	705	357	1644	1977

[A] Total stocks for 0–1 m soil depth are calculated as the sum of measured 0–80 cm soil C stocks and estimated 80–100 cm soil C stocks.

18. One of the selected sites had been damaged by windthrow and was considered too hazardous to sample from.
19. Note: figures revised from those presented in Mason et al. (2009), the UK Assessment, p.105.

The variability in C stocks for each soil group can be seen in the error bars in Figure 3.8, where soils were ranked by total C stocks. The variability is due to differences in soil properties of each soil type within the main soil groups. For example, the 'brown earths' soil group contained (a) typical brown earths, (b) basic brown earths, (c) podzolic brown earths and (d) brown alluvial soils, and the 'peaty gley' soil group include (a) peaty gleys, (b) peaty podzols and (c) peaty rankers, which can also be shallow or deep peaty gley soils. The peaty gleys/podzols soils have the highest variability despite the large number of plots (36). Generally, carbon-rich soils will always require larger numbers of samples to provide a precise estimate of the mean due to their high variability in large C stocks. The depth of peat soils has a very large influence on total stocks (Figure 3.9), so variation in soil depth contributes to the overall variation in C stocks.

Previous research has indicated that the organic layer itself in organo-mineral soils (such as in peaty gleys and peaty podzols), contains between 235 and 418 tCO_2 ha^{-1} in the horizons between 5 and 20 cm depth. However, this total can reach a C stock of between 620 and 1400 tCO_2 ha^{-1} depending on the age of the stand and the depth of the organic horizon. Peat soils have been shown to contain about 1600 tCO_2 ha^{-1} between 0 and 40–50 cm. Detailed examination of GB BioSoil forest soil data from 36 plots on peaty soils and 14 plots on deep peats (peat layer >40 cm) suggest that C stocks in the peat layer are between 160 and 1700 tCO_2 ha^{-1} depending on peat layer depth (Figure 3.9). These measured C stocks are in line with others reported for UK peaty soils with shallower depth of peat layer (Zerva and Mencuccini, 2005a; Zerva *et al.*, 2005).

Not surprisingly, the order of soil group according to their C storage is deep peats > peaty gleys > ground water gleys > surface water gleys > podzols > brown earths > rendzinas and rankers. The value for rendzinas and rankers soil C stocks are from only three BioSoil sites, so they have high variability indicated by the standard error of the mean, and the average C stock should be treated with caution when soil C stocks of rendzinas are needed. More soil samples from rendzina soil types are required, particularly for estimating soil C stocks in England where these soils have a large extent under forestry (see Table 3.9).

For all mineral soil groups, C content is highest near the surface (0–5 cm, Table 3.8, Figure 3.10), with a decline in the 5–10 cm layer, and then increasing to 40 cm with a higher C content for the bulk of the profile (40–80 cm). For organic soils ('PGP', peaty gleys and peaty podzols and 'DP', deep peat) C content increases substantially below the 5–10 cm depth (Figures 3.9 and 3.10). These soil C profiles show the necessity of assessing more than the top 15 or 30 cm to quantify soil C stocks.

Figure 3.8 Total soil carbon stocks (tCO_2 ha^{-1}) for each main soil group measured to a depth of 80 cm in the GB BioSoil plots. Bars show averages from the number of plots per soil group shown in Table 3.8. Error bars represent the standard error of the mean (s.e.m.) (Vanguelova *et al.*, in prep.).

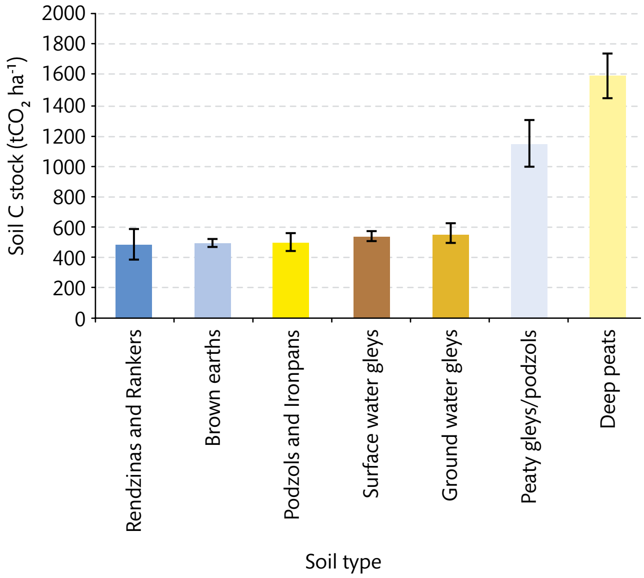
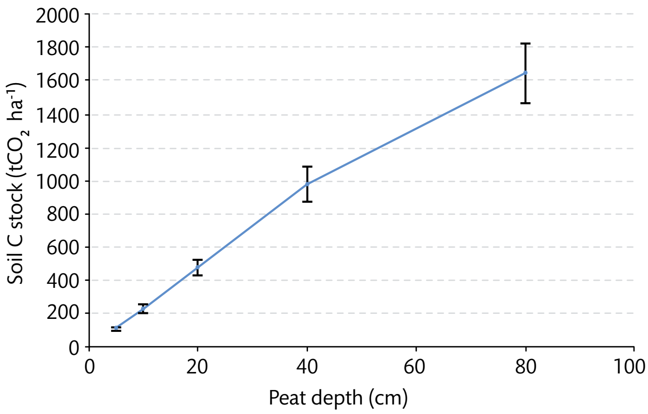
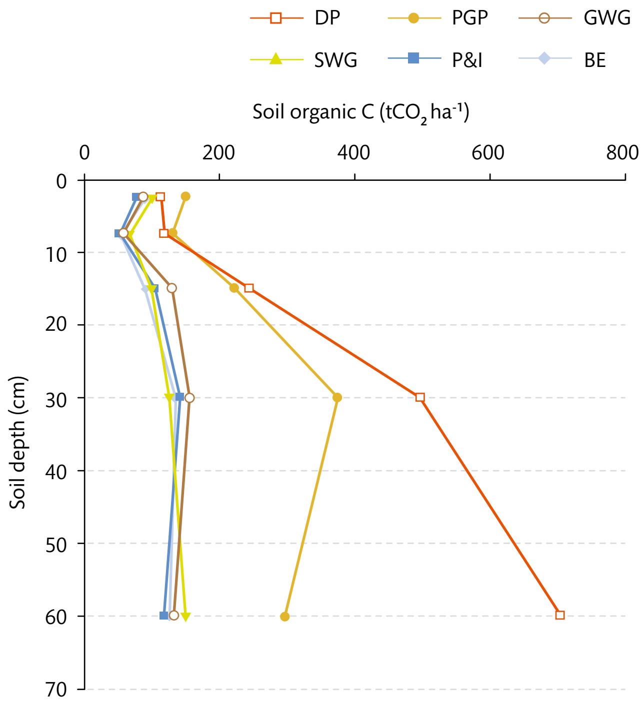


Figure 3.9 Soil organic carbon stocks (tCO_2 ha^{-1}) related to peat layer depth in the GB BioSoil plots. Vertical bars show s.e.m. (Vanguelova *et al.*, in prep.).



The measured soil C stocks from the BioSoil network are comparable to those estimated in the ECOSSE 2008 (Estimating Carbon in Organic Soils Sequestration and Emissions) report (Smith *et al.*, 2009). For example, average measured total C stock (0–80 cm) in peaty gley soils averaged from 36 BioSoil plots is 1174 tCO_2 ha^{-1} (minimum of 250 tCO_2 ha^{-1} and maximum of 3650 tCO_2 ha^{-1}) in comparison to an average of 822 tCO_2 ha^{-1} in the ECOSSE report (calculated from total land area for peaty gleys in Scotland reported in Table 1.2 on page 5 and total C stock for peaty gleys reported in Table 1.12 on page 22). The difference between these stock estimates may be due to the different methodology used for estimating BD, as in ECOSSE the soil C stock estimation used a standard regression equation while in the BioSoil survey BD was measured in triplicate for each soil depth.

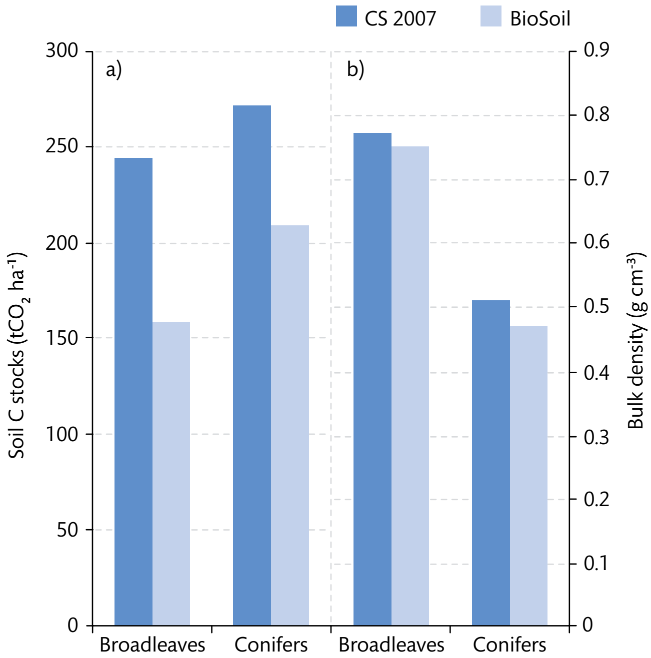
Figure 3.10 Change of soil C stocks with depth averaged across the main soil groups measured in BioSoil. Data replotted from Table 3.8. BE: brown earth; P&I: podzols and ironpans; SWG and GWG: surface and ground water gleys, respectively; PGP: peaty gleys and peaty podzols and DP: deep peats. Note: Values are plotted against the mid-point for each depth layer; i.e. 0–5 cm value is shown at 2.5 cm. A profile for the 'rendzinas and rankers' group is not shown as it is a small sample.



Soil C stocks from the topsoil (0–15 cm) from BioSoil for both broadleaved and coniferous forests are lower than those measured in the 2007 Countryside Survey (CS2007, Chamberlain *et al.*, 2010; Figure 3.11a) probably because of the lower BD measured in BioSoil compared with CS2007. Both BioSoil and CS2007 estimates were calculated from measured C% and BD, but the BD in the BioSoil survey was measured for each different soil depth (e.g. 0–10, 10–20 cm) compared to one BD measurement for the whole 0–15 cm depth in CS2007. Measured bulk density was always lower in the top 0–10 cm compared to 10–20 cm for all soil types which explains the lower BD and thus lower soil C stocks in both broadleaves and conifers from the BioSoil survey compared with CS2007.

Carbon stock values between 238 tCO_2 ha^{-1} for a first rotation 40-year-old stand and 642 tCO_2 ha^{-1} (to a depth of 45 cm) in a 30-year-old second rotation stand were reported by Zerva and Mencuccini (2005a) in a chronosequence of Sitka spruce on peaty gley soils in northern England. Of the total 36 BioSoil plots on peaty gleys, 25 are under Sitka spruce with an average soil C stock of 880 tCO_2 ha^{-1} to 40 cm depth. However, they cover different stages of their life cycle and are either in first or second rotation, which accounts for the high variability of C stocks with values from a minimum of

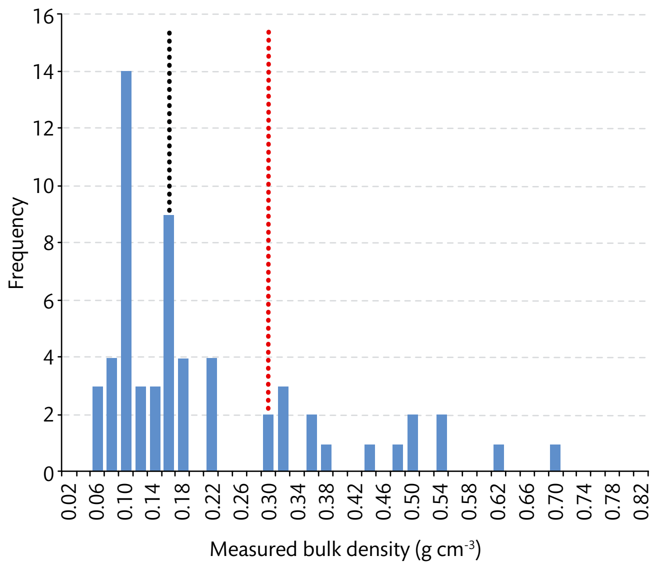
Figure 3.11 Comparison between Countryside Survey 2007 (Chamberlain *et al.*, 2010) and BioSoil values for topsoil (0–15 cm). (a) C stocks, (b) bulk density under broadleaved and coniferous stands.



140 tCO_2 ha^{-1} to a maximum of 4500 tCO_2 ha^{-1}, spanning the values reported by Zerva and Mencuccini (2005a).

The error associated with the uncertainties in estimation of BD and the depth of organic layer were highlighted by Dawson and Smith (2006), who compared C estimated from the same areas where National Soil Inventory (NSI) data were available and a second, small-scale, detailed study which used true surface areas. The detailed study estimated that the C storage was a third of that from NSI data. The situation with peat soils is especially difficult. First, the peat BD is quite variable and, secondly, as peat shrinks BD increases so general relationships between BD and C% are unlikely to be accurate. This is highlighted by the lack of relationship between C% and measured BD in 55 BioSoil samples that had C% >25% (figure not shown, R^2=0.01, ns). Bulk density is critical in C stock calculation for all soils and if it is not measured this inevitably introduces uncertainty into the calculations. The variability in BD of the organic samples (with C% >25%) from BioSoil peaty gleys is shown in Figure 3.12. The average BD is 0.12, with a minimum of 0.05 and a maximum of 0.7 (Vanguelova *et al.*, in prep.). Bulk density decreases with depth, and varies depending whether the peat is shallow or deep (Smith *et al.*, 2007). Detailed investigation and measurements of BD of deep peats at one Welsh site, Plynlimon, have shown that it decreased significantly from 0.20 g cm^{-3} at a depth of 0–15 cm to 0.12 g cm^{-3} at 50–65 cm; this is less than half the 0–15 cm depth BD in shallow peats (0.31 g cm^{-3}). The use of average BD for

Figure 3.12 Distribution of measured bulk density in soil samples from peaty gley soils from BioSoil with soil C >25%. The black dotted line represents the average measured bulk density and the red line represents the bulk density used in Level II soil C calculations (shown in Table 3.7) as well as in the C stock calculation for Wales based on National Soils Inventory (NSI) survey datasets (Alton *et al.*, 2007).

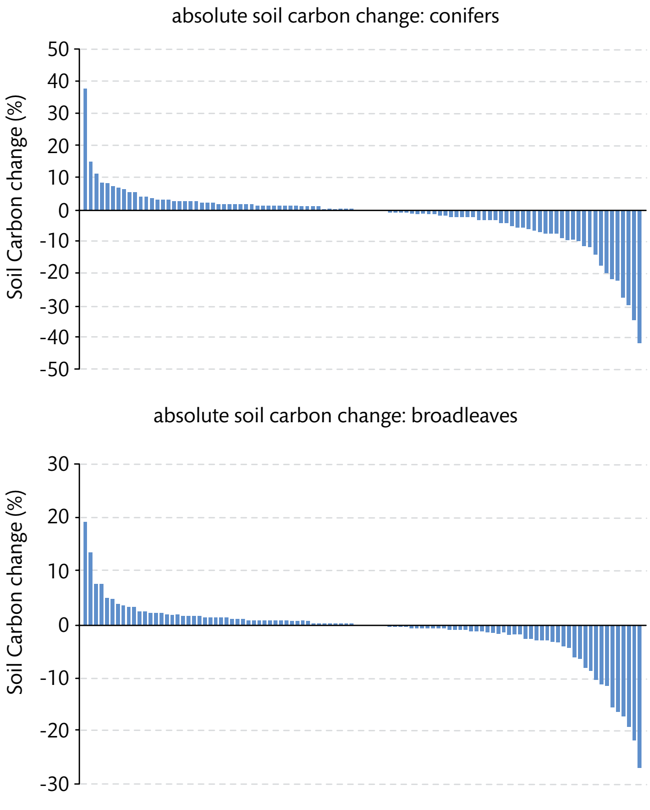


both 0–15 cm and 50–65 cm depth produced differences of around 40% in total C stocks, compared to those determined using the measured BD at each depth. That ECOSSE analysis emphasises that the BD figure used should be representative, particularly for organic soils, otherwise it can produce erroneous results in C stock estimates in soils (Smith *et al.*, 2007). Measured bulk density values of topsoil (0–20 cm, or 0–15 cm) under both broadleaves and conifers from the BioSoil and CS2007 surveys were very similar (Figure 3.11b).

3.5.3 Changes in soil carbon under forestry

The majority of organic carbon in terrestrial ecosystems is stored below ground (Janzen, 2004). Thus, even small rates of C loss from soils can have significant consequences for the overall C budget and atmospheric CO_2 concentrations (Dawson and Smith, 2006). The paper by Bellamy *et al.* (2005) on C losses from all soils across England and Wales reported changes in soil organic C contents from the original NSI dataset (5662 sites sampled from 1978 to 1983) and re-samplings of about 2000 sites after 12–25 years. The paper reported significant loss of soil C under all land uses including the forestry sector. Data from the plots under forestry, either broadleaf or conifer, are shown in Figure 3.13. Original mean C content was 6.78% under broadleaves and 10.99% under conifers and these decreased to 5.46% and 8.74% C, respectively after 25 years. The absolute changes of % C (Figure 3.13) suggest that, overall, the split between sites with increased and decreased C soil content was 50:50 for both forest types. However, the survey had some limitations. One of the more significant was that no land management information from the second sampling event was collected, so possible impacts of changes in management in the 12–25 year period cannot be assessed. In addition, some of the changes in soil C between samplings were implausibly large. For example, some forest sites surveyed had lost up to 40% of their C concentration (see Figure 3.13). The Powlson (2006) report on the NSRI survey highlighted some sites with estimated rates of change of 0.1% C per year, which is extraordinarily rapid for temperate climate regions in the absence of major management or land-use change. In forestry (assuming there are no disturbances), soil C generally accumulates during the forest stand development due to the input of woody and other debris. On the other hand, forest management could promote loss of soil C due to clearfelling and ground preparation for the second rotation (e.g. Zerva and Mencuccini, 2005b; see Sections 4.5 and 4.6). The timescale of this loss is significantly shorter than the timescale of the subsequent gain of soil C due to new plantation growth. Therefore, assessment of changes in forest C soil stocks must account for forest management interventions. In addition, comparing % C change rather than C stock is rather questionable, especially in highly organic soils, where changes may result in mass loss but not concentration and a record of organic layer depth and density is particularly important.

Figure 3.13 Distribution of values of absolute soil carbon changes under woodland (conifer and broadleaf) from the NSI soil resurvey datasets (1970–2003).

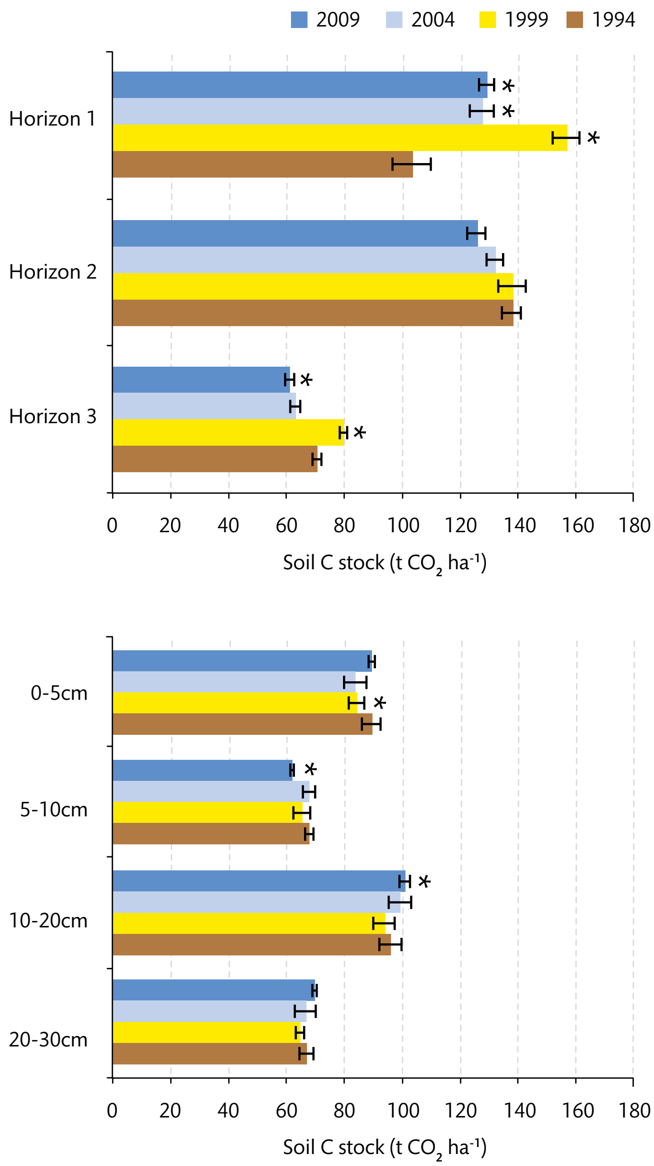


No significant change over the last 30 years was reported in soil C stocks under broadleaved/mixed and coniferous woodland in Wales based on NSI data from the repeated soil survey in England and Wales (Alton *et al.*, 2007). The survey by Kirby *et al.* (2005) specifically examined changes in woodlands between 1971 and 2000–3 and found no significant change in topsoil C concentration. In contrast, the Countryside Survey (Emmett *et al.*, 2010) estimated increases in woodland topsoil C densities of between 0.4 and 1 tCO_2 ha^{-1} y^{-1}, values which are similar to model estimates of European forest soils by Liski, Perruchoud and Karjalainen (2002) and Nabuurs *et al.* (2003), which suggest net sequestration of 0.7 and 0.4 tCO_2 ha^{-1} y^{-1}, respectively.

Soil properties in the Environmental Change Network (ECN) plots have been extensively assessed and characterised every 5 years. At the Alice Holt Oak ECN site (ECN-AH) the first soil sampling was in 1994, repeated in 1999, 2004 and most recently in 2009. Soils have been sampled per depth and per horizon, as shown in Figure 3.14. Results from the four sampling programmes at ECN-AH suggest that soil C stocks in horizons 1 and 2 have increased significantly between 1994 and 1999 and subsequent years 2004 and 2009. The overall increase in C stock is about 25 tCO_2 ha^{-1} between 1994 and 2009, which is an average rate of 1.7 tCO_2 ha^{-1} y^{-1}. The soil C accumulation at Alice Holt is at the top of the range estimated by other studies (e.g. 0.4 to 1.1 tCO_2 ha^{-1} y^{-1}), which confirms the high decomposition rate of litter and organic material and high soil fauna activity observed at this site, which help the bioturbation[20] and C accumulation within the mineral soil. In a separate study of a chronosequence of oak stands at Alice Holt a rate of 0.7–1.1 tCO_2 ha^{-1} y^{-1} increase was calculated comparing soil C stocks from eight young (<25 years old), six mid-rotation (30–75 years old) and ten old (120+ years old) stands (Benham, Pitman and Poole, in prep.) which confirms the rate calculated from the ECN site which is a mid-rotation age stand. The rate of soil C increase was higher at an early stage of stand development and slowed down when the stand was reaching the mature stage, presumably reflecting the reduced rate of net CO_2 uptake by the trees (see Section 3.2).

It should be noted that the significant increase in C is not observed when soil was sampled by depth compared to horizon (Figure 3.14). Where soil sampling is carried out by horizon, the accumulation of all components in the soil is accounted for by the increase in the thickness of that horizon (by input of organic material and litter). However, if soil is sampled by depth this accumulation is not accounted for, and components are measured in the same amount of soil, which results in a misleading comparison of soil properties over time. This is an important observation from the evaluation of the ECN soil data, highlighting the need to account for the accumulation of organic material in the organic layer, which is considerably higher under forestry than under other land use. Thus, using soil sampling by horizon rather than by depth in forests may have implications when monitoring temporal changes in soil properties in response to changes in climate and air pollution or forest management practices.

Figure 3.14 Soil C stocks in tCO_2 ha^{-1} measured in (a) soil horizons and (b) soil depths in the ECN oak site (surface water gley soil, Wickham series) at Alice Holt in 1994, 1999, 2004 and 2009. Significant differences at 5% level between 1994 and subsequent sampling years are shown as *. (From Benham, Vanguelova and Pitman, 2012).



20. Bioturbation: physical mixing of soil and other substrates through biological activity.

The same issue arises when comparing soil C stocks from other soil national surveys, such as the NSI survey, which sampled only the top 15 cm of soil without accounting for any other layers and their change in depth over time.

3.5.4 Distribution of organic carbon in GB forest soils

The total land occupied by each soil group and the land occupied by forests (split between broadleaves and conifers) on each soil group in each country is shown in Table 3.9. The National Soil Map and Soil Classification for England and Wales (NATMAP) vector dataset prepared by the NSRI, Cranfield University (copyrights dated 2004) was used. The soil associations are mapped at a scale of 1:250 000. These were split according to the main soil groups used in Table 3.8 based on the soil classification/map legend. For Scotland the Macaulay Land Use Research Institute digital soil map of Scotland was used, which has a similar but slightly different legend. The Forestry Commission woodland cover map data were used, which are based on the NIWT (National Inventory of Woodland and Trees) dated 31 March 2002. Woodland areas >2 ha are classed as either broadleaf or conifer according to the IFT (Interpreted Forest Type). The 'broadleaf' type includes broadleaved shrubs, mixed, coppice and coppice with standards trees. 'Conifer' includes felled conifer, ground prepared for planting and young trees. The NIWT data were then divided into smaller regions, Wales, Scotland, Northern England, SE and SW England, using Forestry Commission Conservancy boundaries. Finally, the soil data were clipped on to the woodland data and the area of each soil group for each woodland cover type was calculated.

In England, the largest area occupied by both conifers and broadleaves is on brown earth soils, followed by surface water gley soils. In Wales, the majority of soils under conifers are brown earths, podzols and the peaty gley soil groups. Broadleaves in Wales cover mostly brown earth soils and surface water gleys. In Scotland, the dominant soils under conifers are peaty gleys, followed by podzols and deep peats. Most of the broadleaves in Scotland are also found on brown earths followed by podzols and peaty gley soils.

Based on the area occupied by each soil type (Table 3.9) and their estimated C content to 1 m depth (Table 3.8), the overall total C stocks for each soil type, country and forest type were calculated (see Table 3.10). Overall, forest soils in Scotland under conifers contain the most carbon, followed by England and Wales, with total values of 1271, 305 and 132 $MtCO_2$, respectively. Forest soil C stocks under broadleaves are highest in England followed by Scotland and Wales, with total values of 388, 146 and 60 $MtCO_2$, respectively. The peaty gleys/podzols and deep peats contribute most to the total soil C stocks in Scotland, but brown earths and peaty gleys have the largest contribution in Wales. In England, brown earths and surface water gleys contain the most carbon. Total C stocks in forest soils in England, Wales and Scotland under both broadleaves and conifers are summarised in Figure 3.15.

Table 3.9 Land area (km^2) occupied by different soil groups in England, Wales and Scotland and land area for each soil group under both coniferous (C) and broadleaved (B) woodland (Vanguelova *et al.*, in prep.).

Soil group	Land area (km^2)								
	England total	C	B	Wales total	C	B	Scotland total	C	B
Brown earths	46537	1446	2727	10987	790	692	13385	1425	740
Podzols and ironpans	3840	555	398	2013	434	6	8495	1668	496
Surface water gleys	30975	745	1703	3476	92	183	10096	1090	309
Ground water gleys	11273	109	273	605	8	21	41	0	0
Peaty gleys/podzols	4208	550	81	1624	228	41	30094	5540	369
Deep peats	3942	239	79	697	118	5	8818	1452	40
Rankers and rendzinas	7811	379	1228	21	70	39	4989	155	84
Other	21215			1195			2861		
Sum	**129803**	**4023**	**6490**	**20618**	**1741**	**986**	**78779**	**11332**	**2038**
Total C + B		**10513**			**2727**			**13370**	

Total forest area = 2.66 Mha[21].

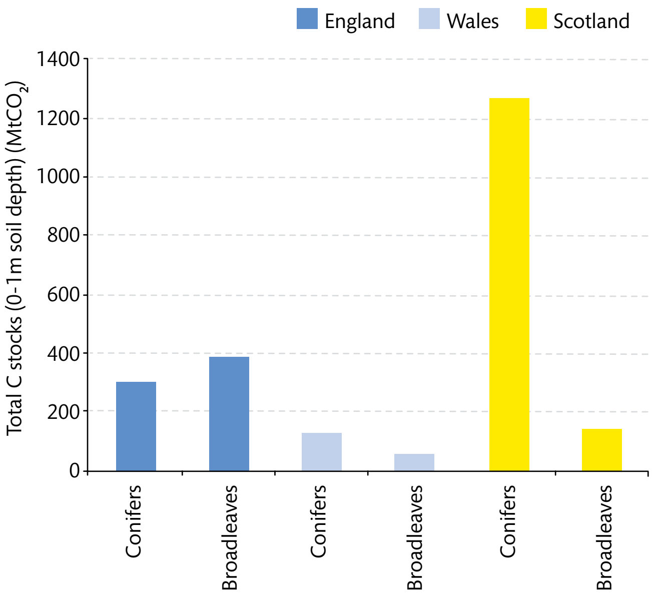
21. Note: the areas in Table 3.9 are derived from soil maps and the GB total of 2.66 Mha is slightly lower than that reported by the Forestry Commission (2.75 Mha).

Table 3.10 Soil organic carbon stocks ($MtCO_2$) to a depth of 1 m, averaged for soil group, country and forest type by combining BioSoil measurements with national soil maps (Vanguelova *et al.*, in prep.).

Soil group	England		Wales		Scotland	
	Conifer	Broadleaf	Conifer	Broadleaf	Conifer	Broadleaf
Rankers and rendzinas	20	65	4	2	8	1
Brown earths	81	152	44	39	79	41
Podzols and ironpans	31	22	25	0	94	28
Surface water gleys	46	104	6	11	67	19
Ground water gleys	7	17	1	1	0	0
Peaty gleys/podzols	73	11	30	5	735	49
Deep peats	47	16	23	1	287	8
Total	**305**	**388**	**132**	**60**	**1271**	**146**
Total conifer + b'lf	**693**		**192**		**1417**	

Total forest soil C for GB = 2302 $MtCO_2$ (628 MtC)

Figure 3.15 Total C stocks in forest soils under broadleaves and conifers in England, Wales and Scotland estimated by up-scaling GB BioSoil results (Vanguelova *et al.*, in prep.).

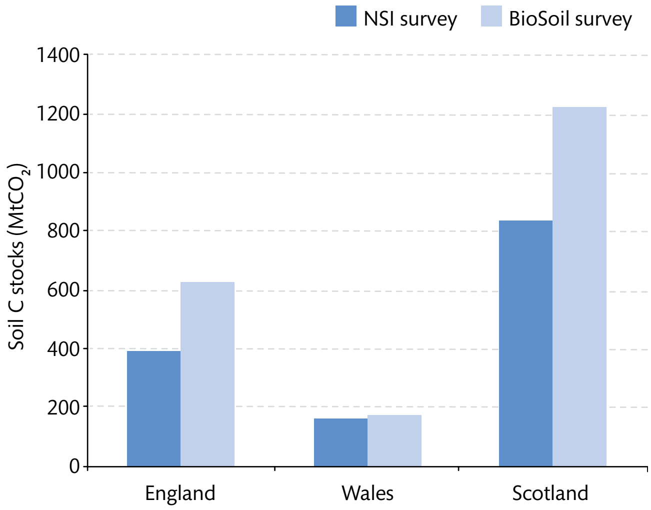


These estimates indicate that England and Wales have 30 and 8% of the total forest soil C stocks, respectively, and Scotland has 62% of the total (although only 50% of the forest area, Table 3.9). This is because of the large areas of forested organic soils (peaty gleys and deep peats) in Scotland. However, it should be noted that these values will be underestimates of the total stocks, as they do not include the soil C held below 1 m, which will be important in deep peat soils, which cover a significant area (approximately 0.19 Mha, 7%). Because the threshold depth for 'deep peats' in the above soil group categorisation is 50 cm, while BioSoil measured to 80 cm, and mean peat depth in forest soils is not known, it is not possible to assess the size of the underestimate. The difficulties in estimating C stocks in organic soils has been discussed in detail in the ECOSSE reports (Smith *et al.*, 2007, 2009) and the implications for forestry in Morison *et al.* (2010).

Total soil C stocks down to 1 m soil depth have been calculated from land cover and soil C content for England, Scotland and Wales from National Soil Inventory (NSI) datasets and reported by Bradley *et al.* (2005) and for Wales only by Alton *et al.* (2007). Under woodland, total soil C stocks for Wales from the BioSoil survey agree closely with those from Bradley *et al.* (2005) but are 43% larger for England and 41% higher for Scotland (Figure 3.16). For Wales, BioSoil estimates in the topsoil (0–15 cm depth) were 20 $MtCO_2$ and 40 $MtCO_2$ under broadleaves and conifers, respectively, very similar to the 21 $MtCO_2$ in the 0–15 cm layer reported for broadleaves, but substantially lower than the 62–65 $MtCO_2$ estimated for conifers by Alton *et al.* (2007). The differences between these survey estimates could be due to: (a) differences in measurement methods (e.g. for BD, fragment content, C%); (b) use of measured against modelled soil C stocks with soil depth; (c) difference in land cover estimation. The BioSoil survey used measured values throughout for soil C stock calculation, while the NSI survey used measured values for the topsoil C calculations and modelled the soil C stocks with depth (Bradley *et al.*, 2005). Alton *et al.* (2007) reported nearly twice as much area for broadleaves, 1615 km^2 compared to the Forestry Commission NIWT estimate of 986 km^2 (Table 3.9), but a conifer area of 1436 km^2, 18% lower than the Forestry Commission estimate of 1741 km^2. Thus Alton *et al.* (2007) estimated a much larger broadleaved woodland cover than coniferous cover in Wales, which is the opposite of that calculated in Table 3.9.

The land cover data used in Alton's report are from the Countryside Survey Land Cover 2000 map (CS2000, CEH, 2000). The CS2000 land cover is a raster (pixel based)

Figure 3.16 Soil carbon stocks (0–1 m depth) calculated from National Soil Inventory (NSI, Bradley *et al.*, 2005) and BioSoil soil survey (Vanguelova *et al.*, in prep.).



satellite image dataset, while the Forestry Commission NIWT woodland cover 2002 dataset is a vector aerial photography dataset, so the boundaries are more accurate in the latter. The large difference between the 'Broadleaved Woodland' habitat area from CS2000 and the Forestry Commission NIWT value was highlighted by the Haines-Young *et al.* (2000) report assessing habitats in the UK countryside. In addition, the figure of new broadleaf planting reported by the Forestry Commission statistics (75 000 ha for GB between 1991 and 1998) is about half that reported by CS2000. The differences between estimates is possibly because CS2000 records all types of woodland development, not just those which can be attributed to formal planting schemes, and also includes natural regeneration and conversion from conifer plantation. Haines-Young *et al.* (2000) concluded that further work is required to determine the reasons for the difference, which may reflect the survey methodologies. CS2000 has since been updated and land cover is now estimated by a different methodology closer to that for the Forestry Commission NIWT, so that comparison between the different surveys of soil C stocks at national level may be possible when the new woodland cover map is available.

Average forest soil C stocks per country can be calculated from Tables 3.9 and 3.10, although it should be noted that the soil C stocks measured in BioSoil were those in tree stands, and may not represent soil stocks across all forest areas that include some open spaces for rides, roads and other unplanted areas. These may have lower C stocks than tree stands, although it is very difficult to assess the degree of inaccuracy. These average stocks (Table 3.11) can be compared with those reported in the recent UK report for the Forest Resources Assessment 2010 (FRA, FAO, 2010), which is based on average soil C contents by country from Milne *et al.* (2004). These FRA values are much lower than earlier values used for LULUCF reporting, such as those quoted by Broadmeadow and Matthews (2003): 796, 836 and 2126 tCO_2 ha^{-1} for England, Wales and Scotland, respectively. Values estimated from BioSoil (Table 3.11) are substantially lower than the FRA values for Scotland and Wales, and as forested areas are slightly lower in the BioSoil estimates (Table 3.11), the FRA estimates of total soil C stocks are approximately 15% higher for Wales and Scotland than the BioSoil values.

Part of this difference is because the FRA and BioSoil soil C stocks are calculated and modelled differently and extrapolated down to 1 m (for BioSoil data the extrapolation is only between 80 and 100 cm). Unfortunately, it is difficult to extrapolate figures for one depth of profile to another deeper profile (e.g. from 80 cm to 1 m), because for some soil types the depth to the parent material will be less than 1 m. Nevertheless, this depth difference would particularly affect deeper organic soils, and thus may explain the larger differences in Wales and Scotland, with more peat soils under forestry. The BioSoil data suggest that peat soils have an additional 14% C stock between 80 and 100 cm (see Morison *et al.*, 2010, p. 8). However, most of the difference will be due to the use of estimated bulk density in the original datasets on which the average soil C values are based for the FRA (see above discussion about the resulting inaccuracies). Further analysis is required to resolve the differences and provide the best estimates for national reporting requirements, although it is likely that the detail of C stocks per soil group provided by BioSoil (Figure 3.8) is an important improvement.

Table 3.11 Comparison of country soil C stocks under forestry estimated from BioSoil data and reported in the UK Forest Resources Assessment 2010 (FAO, 2010).

Title	England	Wales	Scotland
Average C stock per area, tCO_2 ha^{-1}			
BioSoil	659	704	1060
FRA	643	762	1222
Difference	-2%	8%	15%
Total forest area, km^2			
BioSoil	10 513	2727	13 370
FRA	11 290	2910	13 580
Difference	7%	7%	2%
Forest soil C stock, $MtCO_2$			
BioSoil	693	192	1417
FRA	732	225	1624
Difference	6%	17%	15%

Note: FRA values are those given for 2005. % difference is calculated as 100 x (FRA – BioSoil)/BioSoil.

3.6 Harvested wood products

A proportion of the C accumulated in the woody tissue in trees can be harvested and turned into useful products (harvested wood products, HWP). The C remains 'fixed' (sequestered) within these products throughout their useful lifespan and is only released back to the atmosphere if the wood is oxidised as a result of combustion or decomposition. Human activity affects the C amounts contained in forests and wood products by:

- removing C from the forest to make wood products;
- redistributing C within the forest (e.g. leaving parts of harvested trees on the forest floor);
- altering the dynamics of C exchange between the atmosphere and the forest (e.g. the trees and other vegetation left standing after harvest may grow differently because competing trees have been removed);
- disposing of wood products at the end of their useful lives.

3.6.1 HWP stocks in the UK

The size of the HWP pool in the UK in 2000 (excluding that in landfill) was estimated at 293 $MtCO_2$ (80 MtC) by Broadmeadow and Matthews (2003), and increasing at a rate of 1.6 $MtCO_2\ y^{-1}$, based upon the work of Alexander (1997). This is substantial (e.g. about half the estimated stock in trees, 595 $MtCO_2$, Table 2.1), but it should be noted that about 85% comes from imported timber (Broadmeadow and Matthews, 2003). Recently, Suttie *et al.* (2009, Table 7.3) provided estimates of the amount of C in HWP in housing and construction (69.2 $MtCO_2$), other products such as furniture, fencing and transmission poles (15.2 $MtCO_2$) and packaging (20.4 $MtCO_2$), totalling just 105 $MtCO_2$. This is only about a third of the value proposed by Broadmeadow and Matthews (2003), although the proportion in building construction is approximately the same (65%). Suttie *et al.* (2009) also quote an estimate of the average annual increase in the UK HWP stock of 11.9 $MtCO_2\ y^{-1}$ between 1990 and 1999 (Hashimoto *et al.*, 2002), nearly an order of magnitude higher than the previous estimate, but in the same range as their estimates for several other similar EU countries (Hashimoto *et al.*, 2002).

Alexander (1997) also estimated that the C stock in HWP in landfill in 2000 would be 792 Mt (216 MtC), much larger than that in HWP in use, although considerable uncertainties in the size of this stock need to be noted. HWP stocks in landfills are large and are likely to be increasing globally (e.g. Skog, 2008). HWP material in landfill changes only very slowly, with some estimates that only a maximum of 30% of the carbon in paper and 0–3% of the carbon in wood are ever emitted (Micales and Skog, 1997). Recent information from landfill site investigations in Australia show that woody material and even paper can remain relatively unchanged for up to 30 years, depending on the characteristics and conditions of the landfill), and for wood <20% of the original C content had been lost after 46 years in a Sydney landfill (Ximenes, Gardner and Cowie, 2008). Decomposition rates for wood in landfills are consequently being revised downwards (Ximenes, Gardner and Cowie, 2008); moreover, information from past landfill sites recently reopened is not necessarily a good indicator of HWP stocks and longevity now, as sorting and disposal procedures are very different. More analysis of available information to establish robust quantification of C stocks in HWP both in use and in landfill in the UK is clearly necessary.

3.6.2 The dynamics of carbon in harvested wood

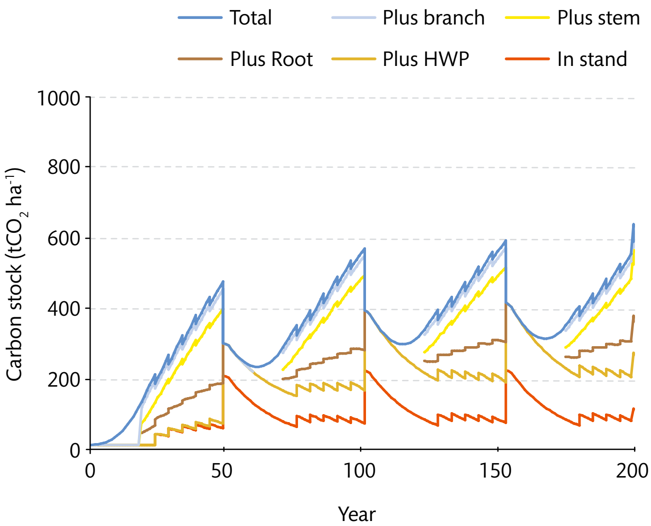
Harvested wood is used to make primary products, and at the end of their useful lives, the wood may be reused in secondary products. Both primary and secondary wood products make a contribution to carbon dynamics.

Carbon in primary wood products

Carbon stocks and flows associated with wood products depend first and foremost on socio-economic forces, not the biophysical processes that dominate in forests. As explained in Appendix 3, the size of a particular wood product pool is a direct consequence of the number of units of the product in use at a given time and the average amount of wood contained per unit. The patterns in utilisation of wood are thus the main driver of wood product C dynamics although, ultimately, this is limited by the potential for forest areas to produce timber to meet requirements. In principle, measures that encourage use of more wood products should result in HWP stocks increasing, so that C is sequestered in products. However, it is also important to understand the impacts of increased wood production on the amount of C retained in the forests.

Forest carbon accounting models such as BSORT have been used to estimate the potential for sequestering C in wood products compared to maximising C stocks in woodland by avoiding harvesting. An example of such a comparison based on a hypothetical stand of YC 12 Sitka spruce is given by comparing Figures 3.2 (no harvesting, Section 3.2 above) and 3.17 (wood production, below).

Figure 3.17 Time course of C accumulation in an average (YC 12) stand of Sitka spruce in Britain estimated using the BSORT model, showing the contributions due to tree components and harvested wood products (HWP). Soil C has not been included. The stand is assumed to be planted on bare ground with an initial spacing of 2 m, thinned according to the standard management tables (Edwards and Christie, 1981), then felled and replanted on a 50-year rotation. Total C stocks accumulate, and the long-term average total carbon stock maintained on the site is approximately 422 tCO_2 ha^{-1}. The contribution due to harvested wood products is approximately 143 tCO_2 ha^{-1}.



The result of the periodic thinning and felling for HWP for standing trees has already been presented in Figure 3.3 but in Figure 3.17 the contribution made by C in HWP is also shown. Each harvesting event adds to the C quantity retained in primary wood products. However, quite a large proportion of the harvested wood has a relatively short lifespan; for example, wood processing factories often burn a significant fraction of off-cut wood to provide heat and electricity for the manufacturing process. The remaining harvested wood goes to make longer-lived primary wood products, with service lifespans ranging from 1 or 2 years up to 40 years. Exceptionally, primary wood products may remain in service for 100 years or more.

The size of the C stock in primary wood products generated by a particular area of woodland is determined by the balance between the amount of wood that can be harvested sustainably from the woodland and the average lifespan of the products. In the example model simulation in Figure 3.17, this C stock is estimated to be around 143 tCO_2 ha^{-1} (39 tC ha^{-1}). As discussed earlier and also evident from a comparison of Figures 3.2 and 3.17, in general woodland management with timber harvesting results in lower average tree C stocks when compared with management without disturbance. In Figure 3.2, the average C stock retained by the woodland after 100 or more years is about 960 tCO_2 ha^{-1} (262 tC ha^{-1}), while with harvesting (Figure 3.17) the average C stock in trees, coarse woody debris and HWP is about 420 tCO_2 ha^{-1} (115 tC ha^{-1}), less than half the undisturbed value.

The dynamics of HWP carbon suggest that the potential for increased sequestration is limited – because people can only find use for so many products. However, there may be considerable scope for increasing the quantity of wood in individual units of a product. For example, modern house designs often involve relatively small amounts of structural wood, so by changing designs, the quantity of wood contained in a house could be increased (see Suttie *et al.*, 2009 for recent discussion of the amount of wood products used in construction).

It should also be noted that, as with trees, C sequestration in HWP is potentially reversible. If existing or new wood products are replaced with non-wood products at some point in the future, C stocks in wood products will decrease, with implied C emissions if wood products are taken out of use and are burned or decay.

Carbon in secondary wood: bury, recycle or burn?

When people finish with a wood product, it can be buried in a landfill, recycled into a secondary product, or burned. These three options have different positive and negative C balance impacts that are summarised in Table 3.12.

When these are considered alongside other factors, it is often difficult to identify the option that is going to have the most favourable GHG balance. Of the three options, the C dynamics are most simple for burning wood: C fixed in wood is released back to the atmosphere immediately. The mix of carbon-based gases released depends on how efficiently the wood is burned. If this is efficient, most of the C returns as CO_2. For less efficient cases (e.g. poorly tended open log fires), a proportion is returned as more complex hydrocarbons. The C dynamics of recycled wood products are similar to primary products – the main determining factor is the requirement for the particular product being manufactured. However, there is no clear picture about the interactions in the consumption of virgin and recycled wood. Quite high uncertainty also surrounds the C dynamics of landfilled wood. The quantity of C in wood in landfill could be significant (see beginning of this section), but new estimates are needed of C stocks and rates and mechanisms of their change.

Results such as those in Figure 3.17 can be generated using models such as BSORT, C-FLOW or CARBINE for many combinations of tree species, yield class and management regime (Thompson and Matthews,

Table 3.12 Carbon impacts of landfilling, recycling and incinerating wood. All three options require transport of wood, which requires energy.

Landfilling	Recycling	Incineration
Positive impacts		
• C in wood is stockpiled underground, rather than released to the atmosphere at once. The time taken for landfill wood and paper to decompose can be very long. • Methane released by decaying landfill material could be trapped and used for energy.	• The time for which C is retained out of the atmosphere is extended through reuse of wood in secondary products. • Recycling may reduce the requirement for virgin wood. • Secondary products could be used in place of non-renewable materials.	• Burning wood products at the end of their service life could be used to provide heat or generate electricity. • Wood ash could be returned to forests as a source of nutrients, although there are issues with high pH.
Negative impacts		
• Decaying wood may release methane to the atmosphere, which is a strong greenhouse gas.	• Recycling processes require energy and sometimes use of additional chemicals.	• Materials need to be sorted carefully to avoid contamination. Energy may be needed for this. • Burning needs to be efficient to ensure that C is released as CO_2, not as more complex carbon compounds. • C locked in wood is released back to the atmosphere.

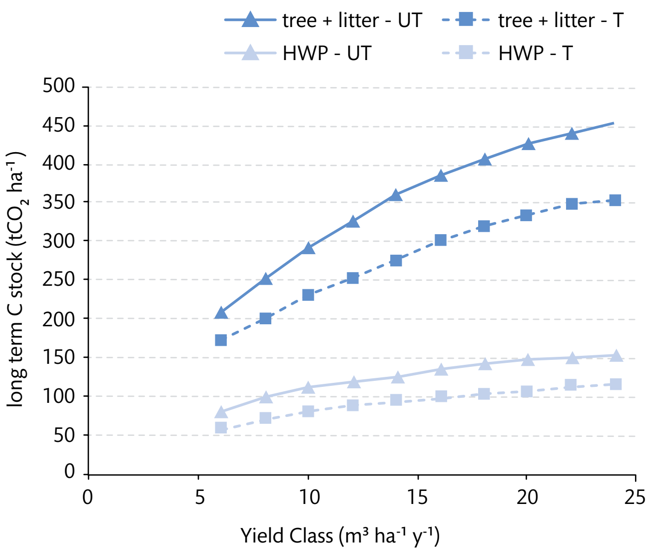
Table 3.13 Modelled estimates of long-term C stocks in trees, litter and wood products for British woodlands. Data from Dewar and Cannell (1992), partly shown in Tables 3.2 and 3.3.

Species	Yield class	Initial spacing (m)	Management	Rotation (years)	Long-term carbon stock value (tCO_2 ha^{-1})		
					Trees and litter	Wood products	Total
Sitka spruce	24	2.0	Unthinned	47	455	154	609
Sitka spruce	24	2.0	Thinned	47	352	114	466
Sitka spruce	20	2.0	Unthinned	51	425	147	572
Sitka spruce	20	2.0	Thinned	51	334	106	440
Sitka spruce	12	2.0	Unthinned	59	326	117	444
Sitka spruce	12	1.8-2.0	Thinned	55-9	253-82	88-143	341-425
Sitka spruce	6	2.0	Unthinned	68	209	81	290
Sitka spruce	6	2.0	Thinned	68	172	59	231
Poplar	12	2.7	Unthinned	26	326	132	458
Scots pine	10	1.8-2.0	Thinned	65-71	238-64	95	334-59
Lodgepole pine	8	1.8	Thinned	62	216	70	286
Beech	6	1.2	Thinned	92	319	95	414
Oak	4	1.2	Thinned	95	249	70	319

1989a; Dewar and Cannell, 1992). Table 3.13 reports the modelled long-term C stock in trees and coarse woody debris and the additional contribution due to HWP estimated by C-FLOW (Dewar and Cannell, 1992). The results can be compared to those in Tables 3.2 and 3.3 to see the contribution made to the C balance by HWP when establishing new woodland or changing management regimes. Figure 3.18 shows that C-FLOW estimates that the C stock in HWP increases less with increasing yield class than the in-forest stock (trees + litter); that is, the ratio of HWP/(trees + litter) declines slightly.

The contribution of HWP to overall C stocks for the forest sector is relatively small, although not insignificant. Uncertainties and knowledge gaps regarding the production and ultimate fate of harvested wood are unlikely to change such a conclusion. There are, however, a number of areas in the modelling of HWP carbon where improvement is desirable and in some cases really needed (see Section 6).

Figure 3.18 Modelled effects of yield class on Sitka spruce long-term tree C stock, thinned and unthinned (T and UT, data from Dewar and Cannell, 1992; see Table 3.13 for details).



3.7 Substitution by woodfuel and wood products

Harvesting trees reduces the C stock in the forest and thus reduces C sequestration, dependent on the use of the harvested tree (see previous section), and the regrowth at the site. As explored elsewhere in this chapter, the net change in forest C stock is affected by many environmental, biological, ecological and management variables. However, even if there is a net C removal from the forest, it may help reduce C emissions from fossil fuel use through substitution, either **directly** (use as an energy source) or **indirectly** (through use in place of other, more energy-intensive materials such as steel, bricks or concrete). In many instances, wood products may deliver multiple substitution benefits. For example, waste wood created when machining logs into products may be used as energy to drive the manufacturing process, or an article of furniture may be disposed of by burning and use of the energy released.

3.7.1 Direct substitution: woodfuel

Several points about woodfuel use and its potential for substitution need to be made:

- If woodfuel is used to substitute for fossil fuel, then obviously GHG emissions from fossil fuel combustion are avoided. The net effect of woodfuel use on the atmospheric GHG balance depends on the rate of uptake of CO_2 by forests, and the rate of GHG emissions during woodfuel production and combustion per unit energy released.
- **If forests regrow after harvest** and re-fix CO_2 lost during woodfuel production, combustion and energy generation, a sustainable cycle of woodfuel harvesting and forest regrowth continues to avoid fossil fuel CO_2 emissions, so the GHG 'benefits' continue to accrue. This is unlike measures to cause additional C sequestration in forests, which come to a saturation, determined by the environmental and forest characteristics at a site (see Section 3.2).
- Bioenergy forests usually have lower C stocks per area than forests managed less intensively, although if bioenergy plantations are established on, for example, farmland the C stock per area is likely to be increased.
- The potential for woodfuel to produce direct substitution benefits is determined by the biomass productivity and the fuel conversion process.
- The calculation of the net reduction in fossil fuel consumption avoided by bioenergy obviously needs to take into careful account the fossil fuel use in harvesting, processing and transport of woodfuel.

Woodfuel emission factors range between 4 and 8 g CO_2 MJ^{-1} (Matthews and Robertson, 2003) if transport is excluded. For woodchip production, typical figures are 3–5 g CO_2 MJ^{-1}, and by comparison with coal, combustion of woodchip produces a saving of about 100 g CO_2 MJ^{-1}.

The woodfuel resource in Britain was assessed in the WRiB study (McKay *et al.*, 2003, Table 3.14). Most of the estimated fuel resource is in small round-wood (36%), with saw-mill co-product contributing 26% and arboricultural arisings about 15%. However, there are some markets for the products already, so a more conservative estimate of 'new' resource of 1.31 Mt wood was also made. Assuming that the C content of this oven dry wood is 50% gives a potential resource in GB of 0.65 MtC y^{-1} (no data were available for Northern Ireland).

Table 3.14 Current potential woodfuel resource expressed as annual production of oven-dried biomass (M odt y^{-1}; from McKay *et al.*, 2003).

	England	*Scotland*	*Wales*	*Total*
Without competing markets	1.45	1.38	0.49	3.31
With competing markets	0.74	0.38	0.20	1.31

3.7.2 Indirect substitution: use of wood products

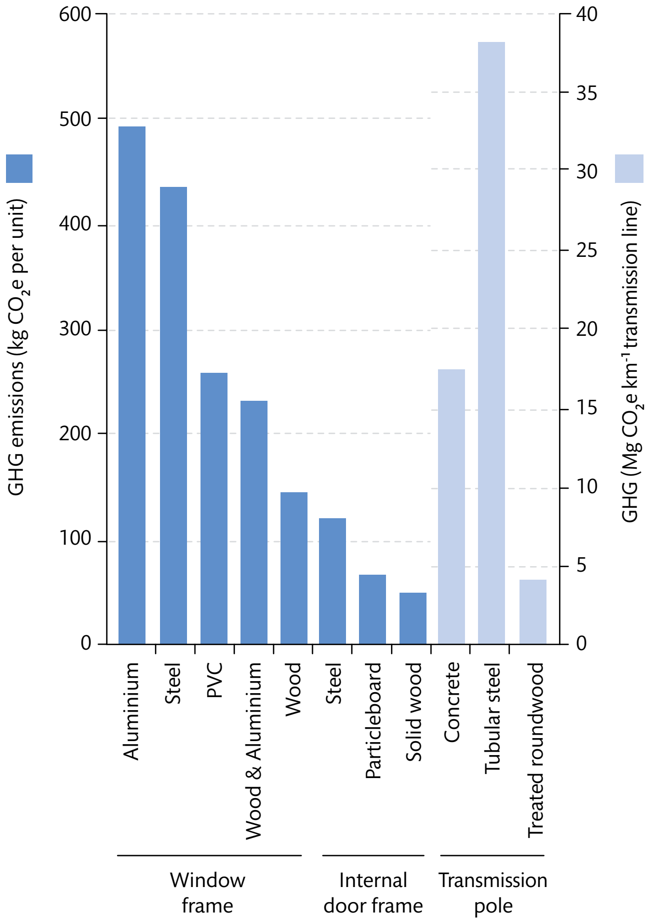
GHG emissions may also be reduced by using wood in place of other more energy-expensive (and therefore fossil-fuel intensive[22]) materials. This will only contribute to emission reduction if:

- wood use patterns are actively modified to achieve effective substitution;
- existing energy-intensive products are not replaced with wood products until end of life;
- opportunity for substitution of materials for particular products or uses is real, and product lifetimes are taken into account in the assessment;
- material substitution does not lead to creation of alternative markets for the materials not being used (i.e. 'leakage').

Suttie *et al.* (2009) presented values for the net CO_2 emissions during the manufacture of the major products used in the construction industry, and emphasise that all non-woody components are net CO_2 sources, while woody products are net sinks. For example, emissions for steel products are between 4.0 and 5.3 $tCO_2\ m^{-3}$, bricks 0.2–0.4 $tCO_2\ m^{-3}$, whereas sawn timber represents a sink of approximately 0.9 $tCO_2\ m^{-3}$ if the carbon sequestered in the wood is accounted for (Suttie *et al.*, 2009, Figure 6.4). There has been considerable work on the opportunities for materials substitution in the building industry, including life-cycle analysis (LCA), which attempts to account for the specific mix of GHG emissions arising from the use of different materials and methods of manufacture, the maintenance and disposal and the service life. Figure 3.19 gives an example of the potential emissions reduction for the manufacture of window frames, door frames and transmission poles from different materials (from Richter, 1998). The window frame calculations also include varying rates of heat loss during service life, which illustrates the complexity of LCA.

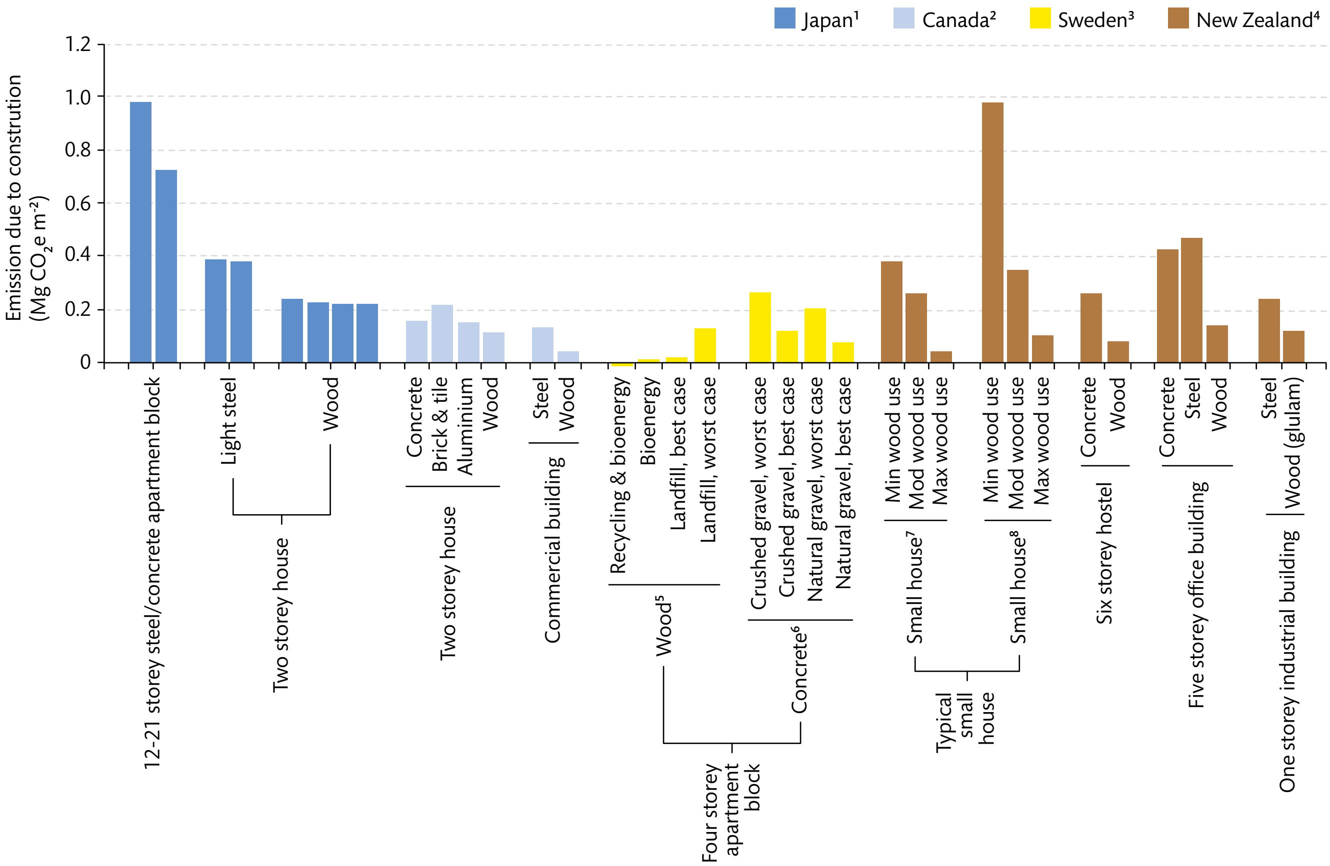
Results from a number of other studies of the C emissions arising from construction and maintenance of buildings are summarised in Figure 3.20. Although the estimates show considerable variation, depending on the types of buildings assessed and the methodologies used, lower C emissions are associated with wood-based construction in all cases. A recent study by the Edinburgh Centre for Carbon Management has provided a 'carbon calculator' for the building industry (the 'Buildings Material Carbon Indicator', ECCM, 2006), using various emission factors for materials. The Building Research Establishment (BRE) has also produced the Green Guide for Specification (available on-line, BRE, 2006) which compares the environmental impacts of different building elements based on full LCA (Suttie *et al.*, 2009). Furthermore, a large meta-analysis of the GHG 'displacement factors' due to wood product substitution has recently been published (Sathre and O'Connor, 2010), and it estimated a mean value of 2.1 tC (3.9 tCO_2e), that is, 'for each tC in wood product substituted in place of non-wood products, there occurs an average GHG [emission] reduction of approximately 2.1 tC' (Sathre and O'Connor, 2010). It is clear that the substitution benefits of woodproducts are significant.

Figure 3.19 Estimated GHG emissions (CO_2e) resulting from manufacture of window frames, door frames and transmission poles (from Richter, 1998).



22. Note: production of 'energy-intensive' materials does not necessarily result in more GHG emissions, as some energy sources are not fossil fuel derived. However, the present low proportion of energy consumption from renewable and nuclear sources in the UK (3% and 7%, respectively) means that in general material substitution by wood as described above will reduce fossil fuel combustion.

Figure 3.20 Summary of research results on CO_2e emissions arising from construction of buildings with different materials. Emissions calculated per unit total floor area. Taken from Matthews and Robertson (2003).



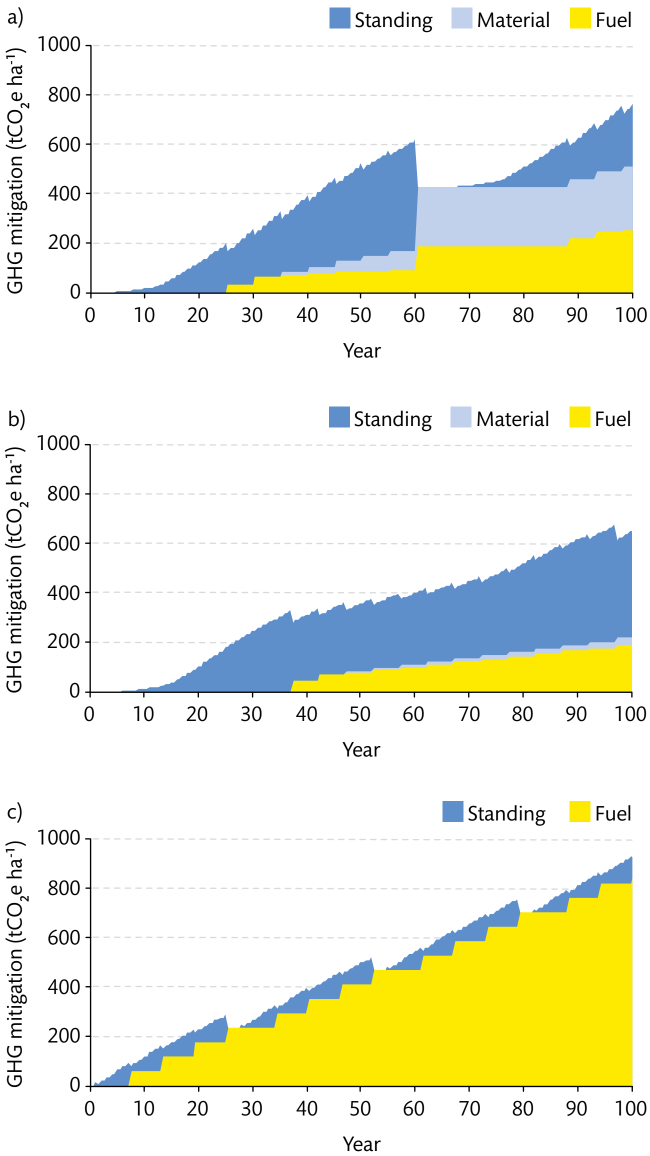
(1) After Suzuki, Oka and Okada (1995 based on construction only. (2) After Marcea and Lau (1992) based on construction and building maintenance. (3) After Börjesson and Gustavsson (2000) based on building life cycle including disposal of materials. (4) After Buchanan and Honey (1994) based on construction only. (5–6) Alternative estimates of emissions for typical small house (New Zealand): (5) based on construction only; (6) includes allowance for energy use to heat building over 25 years. (7–8) Alternative estimates of emissions for four-storey apartment block (Sweden) based on different assumptions about either disposal of wood at end of building life or different assumptions about concrete used in construction.

3.7.3 The dynamics of emissions reductions through substitution

Carbon accounting models such as CARBINE and CSORT can be used to estimate the impacts of the utilisation of wood harvested from forest stands in Britain. Examples shown in Figure 3.21 dramatically illustrate the potential size of emissions reductions due to substitution. In all cases the graphs show the total cumulative GHG emission mitigation achieved by establishing the example woodlands, comprising the contributions from standing material, from fuel substitution and from the use of larger timber material in building construction substitution. In two cases (Figure 3.21a, conventionally thinned Sitka spruce on a 60-year rotation, and 3.21c, short rotation coppice, SRC) the largest contributions to the overall emissions reduction by the stand and its products are estimated to be those due to fuel and/or material utilisation, which continue to increase over time (because the amount of fossil fuels that have not been consumed continues to grow). The contribution from the tree standing stock varies over the thinning cycles and the rotations, as seen previously (see Section 3.3, Figure 3.3). In the third case (oak under LISS management option, Figure 3.21b), mitigation due to substitution increase gradually over time, while the contribution from C in standing trees remains roughly constant. For the Sitka spruce YC 12 example (Figure 3.21a), the substitution benefits accumulate over time at average rates of approximately 2.5 $tCO_2e\ ha^{-1}\ y^{-1}$ for both fuel and material, respectively. For the oak woodland example, managed under LISS (Figure 3.21b), the contributions are estimated to accumulate at average rates of 2.0 and 0.3 $tCO_2e\ ha^{-1}\ y^{-1}$, for fuel and material, respectively. The smaller contribution of timber material is due largely to the type and size of material generally extracted during thinnings (a removal across all size classes to maintain a constant woodland). Once established, the C stock in the stand fluctuates around approximately 400 $tCO_2e\ ha^{-1}$. For the SRC example (Figure 3.21c), the mitigation due to bioenergy use accumulates at an average rate of approximately 9 $tCO_2e\ ha^{-1}\ y^{-1}$, while the average C stock in the coppice is approximately 50 $tCO_2e\ ha^{-1}$, much lower than the forest examples. In the SRC example no material is assumed to be used in building material substitution.

Similar results have been reported previously, for example by Marland and Marland (1992), Matthews (1994) and Nabuurs (1996). However, results for substitution impacts tend to represent either 'generic' wood products or very specific cases of production and utilisation. Current understanding of the emissions reduction contribution from substitution relies on a relatively small information and research base and further work is needed for precise quantification.

Figure 3.21 GHG emissions offset by growing example stands of trees estimated from the CSORT model, showing the contributions due to C stocks in trees ('standing') and emissions reductions achieved through utilisation of harvested wood for fuel and building materials ('materials').



(a) Average (YC 12) stand of Sitka spruce assumed to be planted on bare ground with an initial spacing of 2 m, thinned according to the standard management tables (Edwards and Christie, 1981), then felled and replanted on a 60-year rotation.
(b) Average (YC 4) stand of oak established and managed under a LISS (selection) plan.
(c) Short rotation coppice with an assumed productivity of 6.5 odt $ha^{-1}\ y^{-1}$, planted on bare ground with a stocking density of approximately 10 000 stools per hectare, harvested on a 7-year cycle, and replanted after four harvests.

4 Fluxes of C and other GHG in forestry

4.1 Introduction

Chapter 3 described the key C stocks in forests in trees, soils and litter and stocks outside the forest in harvested wood products and substitution. This chapter discusses the net emissions of CO_2 and the two other key greenhouse gases CH_4 and N_2O from forests, and the loss of C from forests as dissolved organic carbon (DOC). It considers both the GHG fluxes during natural processes in forests and the effect of forest management on those fluxes, and also the GHG emissions produced during forest operations through the use of fossil-fuel powered machinery. The anthropogenic emissions of CH_4 and N_2O contribute approximately 16%, and 5%, respectively, to the positive radiative forcing underlying global warming (IPCC, 2007), so they must be considered in estimating the GHG balance of forests, as well as CO_2.

The exchange of CO_2 with the atmosphere occurs for both the trees and the soil during photosynthesis and respiration (see Figure 2.2), but CH_4 and N_2O emissions are largely from the soil alone. Microbial production of CH_4 is strictly anaerobic, production of CO_2 aerobic and N_2O can be produced under both aerobic and anaerobic conditions, and it may be consumed in wet, nitrogen-poor soils (Chapuis-Lardy *et al.*, 2007). There have been measurements of CH_4 emission from plants (e.g. Keppler *et al.*, 2006), but as the mechanism has become clearer the current view is that the aerobic production of CH_4 in plants is very small (Nisbet *et al.*, 2009; Bloom *et al.*, 2010). However, it is well established that methane produced anaerobically by soil microbes can be transported through wetland plants and released into the atmosphere. Therefore, in flooded or waterlogged forests it is likely that trees form a pathway for CH_4 emission, either diffusing through air spaces in the stem, or being transported dissolved in the transpiration stream (e.g. Gauci *et al.*, 2010; Terazawa *et al.*, 2007) and being emitted from foliage and other above-ground parts. However, the size of this flux has still to be established, and only preliminary data has been published. Therefore in this section the focus is on soil exchanges of CO_2, CH_4 and N_2O, and whole-stand-scale net exchanges of CO_2 (i.e. tree and soil). In practice, measurement of 'soil emissions' usually also includes the fluxes from the litter layer and the CO_2 produced by root respiration, although there have been some measurements where the litter was removed (e.g. Fenn *et al.*, 2010).

4.2 GHG fluxes from forests

4.2.1 Review of available information

The fluxes of GHGs are influenced by many natural factors, particularly soil moisture content, water depth, aeration, soil type, soil temperature and soil pH, and man-made factors such as land use and management (which determine the disturbance history), soil nutrition, forest type and physiological differences associated with age. There are also strong interactions between these factors, for example through N input, soil C:N ratio and water table. While C:N ratio is the most important factor affecting N_2O emissions, the groundwater table depth is a key factor determining the size of soil CO_2 and CH_4 fluxes (von Arnold *et al.*, 2005a).

This section reviews the available information on emissions from, or uptake by, forests for the key long-lived GHG (CO_2, CH_4 and N_2O). Tables A5.1 and 5.2 in Appendix 5 summarise results from more than 60 papers published between 1987 and 2010 where measurements of the flux of these gases from forest soils (mainly in the temperate region) were made for at least a one-year period. They can therefore be regarded as reliable estimates of GHG fluxes.

There are many ways in which the soil GHG flux results could be categorised and presented. They could be analysed by soil type, tree species, tree age, management, climatic regions, methodology or land use, and ideally the categorisation would identify the uncertainties involved including those due to methodology, monitoring and integration period for up-scaling. However, because of (a) lack of all year-round flux measurements, (b) lack of simultaneous measurement of all GHGs under the same environmental conditions, (c) large variations and uncertainties in the fluxes for each category and (d) large differences in the impact of environmental conditions on each gas, the results were simply categorised as:

- standing forest on mineral soils (e.g. brown earths, surface water gleys);
- standing forest on organo-mineral soils (e.g. peaty gleys);
- standing forests on deep peat soils;
- clearfelled sites;
- unforested deep peat and wetlands;
  fluxes from other vegetation on peatland sites.

(see detailed tabulation in Appendix 5, Table A5.1). Some indications are also given about the soil types, species, peat environment (viz. minerotrophic,[23] oligotrophic or ombrotrophic), land use (viz. forested peat drained, restored peat undrained, pristine, cutaway/harvested) and peat vegetation type (e.g. fen, mire, agriculture).

Because of the limited amount of available soil flux data for all GHGs for forests and woodlands in the British Isles, this review was extended to include results for other European and worldwide forest soils (but mainly temperate region) for comparison, categorised into organo-mineral and organic soils, mainly peat (Appendix 5, Table A5.2).

The available data show that there are considerable uncertainties in soil GHG fluxes in forests that arise because:

1. There are very few measurements of simultaneous CO_2, CH_4 and N_2O fluxes either outside the UK (von Arnold *et al.*, 2005a, b, c, Minkkinen *et al.*, 2002) or within the UK (Freeman, Lock and Reynolds, 1993; Zerva and Mencuccini, 2005a, b). Those were only measured by chamber methods and therefore have large uncertainties due to spatial variations. Simultaneous measurements of all GHG are necessary to calculate the CO_2-equivalent flux in order to estimate the total global warming potential (GWP[24]) of forestry, forestry practices and any land-use change activities.

2. GHG fluxes are very variable because of large temporal variations. Fluxes can vary on timescales of minutes and hours with weather conditions, seasonally and between years. For example, the annual sum of N_2O emissions at the Höglwald Forest in Germany varied from 0.18 to 1.5 $tCO_2e\ ha^{-1}\ y^{-1}$) within a 4-year observation period (Butterbach-Bahl *et al.*, 2002a). Similarly, Chen *et al.* (2004) measured stand-scale C fluxes in a 450-year Douglas-fir dominated forest in Washington, USA. Large inter-annual differences for the summer months were apparent for cumulative CO_2 exchange, with a 1.7 $tCO_2e\ ha^{-1}$ loss in 1998 but 4.2 $tCO_2e\ ha^{-1}$ gain in 1999. For methane, Ball, Smith and Moncrieff (2007) measured annual CH_4 emissions from a peaty gley clearfelled forest area in Harwood Forest, northern England, and found large inter-annual variations with fluxes of 0.17 ±0.03 and 0.45 ±0.03 $tCO_2e\ ha^{-1}\ y^{-1}$ in 2001 and 2002, respectively. Diurnal variation in soil CO_2 efflux, however, seems to be less important in forested areas compared to agricultural land presumably because of the reduced temperature variation under the shade and shelter provided by trees.

3. GHG fluxes are very variable because of large spatial variations. For example, Saiz *et al.* (2006b) investigated the seasonal and spatial variations of soil respiration (CO_2 emission) in a first rotation Sitka spruce stand in central Ireland and found significantly higher respiration rates from furrows and close to tree stems compared with ridges and undisturbed ground.

Therefore, high uncertainties originate from (a) the lack of all year-round GHG flux data from forest soils and (b) the lack of measurements that integrate the large spatial variation across forest stands. While the latter can be provided by measurements of GHG exchange above the forest or other vegetation (i.e. net ecosystem exchange, see Box 3.1), using micrometeorological techniques these are presently very limited for CH_4 and N_2O. Furthermore, where year-round data do exist these typically have been collected for less than 5 years and so the inter-annual variation may not be adequately sampled (Byrne *et al.*, 2004).

Available information on how the main site and management-related factors influence GHG fluxes is discussed briefly in the sections below.

N input and C:N ratio

Pilegaard *et al.* (2006) reported the results of continuous measurements of nitric oxide (NO) and N_2O emissions at 15 different forest sites during the NOFRETETE programme to study factors controlling regional differences in soil emission of nitrogen oxides. They concluded that although many studies have shown that the temporal variation on a specific site is related to soil moisture and soil temperature, when comparing annual emissions on a regional scale factors such as N deposition and forest and soil type are much more important.

According to Kesik *et al.* (2005) emissions of N_2O from forest soils have most probably increased in recent decades and will probably increase in the future due to the anthropogenic perturbation of the global N cycle (Galloway *et al.*, 2004) and consequent high rates of atmospheric N deposition to many forest ecosystems in Europe, North America and Asia (Aber *et al.*, 1989; Bowden *et al.*, 1991; Hyvönen *et al.*, 2007). Recent publications provide evidence that N deposition to forest

23. See Glossary for explanation of terms minerotrophic, oligotrophic and ombrotrophic.
24. GWP – see list of abbreviations for explanation.

ecosystems is positively correlated with N_2O emissions (e.g. Ambus, Zechmeister-Boltenstern and Butterbach-Bahl, 2006). Kesik *et al.* (2005) reported N_2O fluxes from different forest sites between 1989 and 2003 that ranged between 0.02 and 3.1 $tCO_2e\ ha^{-1}\ y^{-1}$, although it should be noted that the measurement period for the sites referenced varied from 10 to 359 days.

The soil C:N ratio can affect soil microbial processes such as mineralisation, nitrification and denitrification and can therefore have a significant effect on GHG production and consumption. Rates of mineralisation and nitrification increase in forest soils with a reduction in the C:N ratio of organic matter (e.g. Ollinger *et al.*, 2002; Klemedtsson *et al.*, 2005; Ambus, Zechmeister-Boltenstern and Butterbach-Bahl, 2006); that is, N_2O emissions are inversely correlated with soil C:N ratio. Based on the relevant European literature, Jarvis *et al.* (2009) have recently summarised the interacting effect of soil temperature, C:N ratio and N deposition on N_2O emissions from UK afforestation. They indicated that most of the soils used hitherto for afforestation in the UK have low soil temperatures (because of their upland locations), fairly high C:N ratios and lower atmospheric N deposition compared with, for example, Central Europe, and these conditions therefore explain the comparatively modest N_2O emissions that have been measured.

Species

Most of the recent studies indicate a tendency towards higher N_2O emissions from deciduous than coniferous forest soils (Skiba *et al.*, 1996; Ambus, Zechmeister-Boltenstern and Butterbach-Bahl, 2006; Pilegaard *et al.*, 2006) due to tree species-related differences in litter quality (and litter layer compaction, see Table 3.4) and soil moisture (Butterbach-Bahl, Rothe and Papen, 2002b). However, McNamara *et al.* (2008) found no differences in soil CH_4 fluxes from plots of different tree species (Norway spruce, Scots pine, oak and alder) at the Gisburn Forest experiment (East Lancashire). Similarly, the literature on stand-scale CO_2 fluxes (net ecosystem exchange, NEE,[25] see Box 2.1) suggested no clear difference in flux magnitudes measured from broadleaf forests and coniferous forests (e.g. Table 4.1, Law *et al.*, 2002). However, such aggregation of diverse datasets results in difficulty in detecting the effect of species alone, because of the confounding effects of stand ages, soil types and environmental conditions.

Table 4.1 Summary of annual net ecosystem exchange (NEE) rates for broadleaved and coniferous forest for temperate, boreal and mediterranean regions (negative CO_2 flux values are sinks, positive are sources, n is number of studies); data reviewed by Law *et al.* (2002).

Forest type	NEE ($tCO_2\ ha^{-1}\ y^{-1}$)	n	Minimum	Maximum
Broadleaved	-7.11	22	-31.9	2.31
Coniferous	-6.82	28	-25.9	3.85

Thinning

It is generally expected that soil respiration and CO_2 efflux decreases with increasing stand density or reduced thinning activity since soil temperature often decreases with increasing stand density (Nilsen and Strand, 2008). Therefore, reduced thinning activity may lead to a build-up of soil C due both to reduced soil temperatures, and thus reduced soil respiration, and to increased litter fall. However, the disappearance of ground vegetation due to high tree densities could reduce this effect (Nilsen and Strand, 2008). The long-term effect of early thinning (measured 32 to 33 years after thinning) on C storage and fluxes in litterfall and soil respiration was investigated by Nilsen and Strand (2008) in a Norway spruce stand reduced at 22 years old from 3190 trees per hectare to 2070, 1100 and 820 trees per hectare. Their results showed that increased thinning intensity at an early stage reduced the above-ground C storage, but increased the soil respiration rate. Nilsen and Strand (2008) concluded that, since no significant differences were observed in soil temperature, soil moisture, litterfall chemistry or humus and mineral soil C storage, the time period of 32 years may have been too short to detect effects of thinning on overall soil C.

Soil cores (to 25 cm) taken from thinned and clearfelled sites have shown significant differences in CH_4 oxidation (Bradford *et al.*, 2000). Soils under beech, Japanese larch and oak showed net CH_4 oxidation but this was significantly reduced in clearfelled plots and increased by thinning. These effects were maintained after soil moisture was standardised, so factors other than soil water content were responsible for the different rates of CH_4 oxidation. In the same study N_2O emissions increased after clearfelling and decreased after thinning. High soil nitrate concentrations after clearfelling (resulting from nitrogen release from brash) have been shown to reduce soil CH_4 uptake (Bradford *et al.*, 2000; Wang and Ineson, 2003). Reduced soil pH and increased compaction as a result of machinery traffic can also contribute to reduced CH_4 uptake (Dedysh, Panikov and Tiedje, 1998; Hansen, Mæhlum and Bakken, 1993).

25. Net ecosystem exchange of CO_2 (NEE) is conventionally expressed as a negative flux when towards the surface (i.e. surface is a sink) and positive away from the surface. Thus it is numerically the same as net ecosystem productivity (NEP), but with the opposite sign; see Box 2.1.

Stand age

As discussed in Section 3.2, the CO_2 uptake rate by trees varies during the growth cycle, peaking in the full vigour phase, but the net CO_2 uptake of the stand also depends on the changing soil respiration as the stand ages, and litter inputs and root activity change. This is exemplified by results from Chen *et al.* (2004), who measured NEE during the summer from three different aged forest stands (Douglas fir dominated) in Washington State, and observed much higher net CO_2 uptake by the 40 year stand (-101 kg CO_2 ha^{-1} d^{-1}) than 20 and 450-year-old stands (-1.1 and -1.4 kg CO_2 ha^{-1} d^{-1}), and in two consecutive years the 450-year-old stand varied from a small source to a small sink. However, while it is generally thought that ageing forests cease to accumulate C the review by Luyssaert *et al.* (2008) of forest C balance data reported that old-growth forests continue to accumulate carbon, so the assumption of old-age C sink 'saturation' needs re-evaluation.

Ball, Smith and Moncrieff (2007) measured the annual soil fluxes of CO_2, CH_4 and N_2O from a Sitka spruce forest chronosequence on a peaty gley soil in Harwood Forest, NE England, and observed lower annual soil CO_2 effluxes in a 20-year-old stand than in 30-year-old and clearfelled stands. CH_4 efflux was highest at the clearfelled site and N_2O efflux was highest at the 30-year-old site (see Table 4.13 in Section 4.6 on harvesting). Saiz *et al.* (2006a) also studied the effect of stand age on soil respiration and its components (i.e. heterotrophic vs autotrophic, Table 4.2) in a first rotation Sitka spruce chronosequence composed of four age classes established on a surface water gley soil in central Ireland. The youngest stands had significantly higher respiration rates than more mature sites due to a decrease in fine root biomass over the chronosequence (Saiz *et al.*, 2006a).

Table 4.2 Mean soil respiration rates (tCO_2 ha^{-1} y^{-1}) and standard error for stands of different age in a Sitka spruce chronosequence in Ireland (Saiz *et al.*, 2006a).

Stand age	10 years	15 years	31 years	47 years
Total, R_S	35.7 ± 5.3	26.8 ± 4.2	22.1 ± 3.3	22.1 ± 3.6
Heterotrophic, R_H	15.6 ± 1.7	11.8 ± 11	9.7 ± 1.8	11.3 ± 1.7
Autotrophic, R_R	20.1 ± 3.6	15.0 ± 2.5	12.4 ± 1.9	10.8 ± 1.6

See Box 2.1 for explanation of respiration components.

Soil compaction

Compaction of soils decreases the total pore volume, especially that of large pores. This decreases aeration, and in wet soils results in more anaerobic sites where denitrification can occur and consequently increases the emission of N_2O and CH_4 from the soil. Conversely, in drier soils compaction can reduce oxygen diffusion and reduce the CH_4 oxidation rate, leading to reduced CH_4 uptake. Teepe *et al*. (2004) measured N_2O and CH_4 fluxes across a skid trail (established by applying two passes with a forwarder) at three beech forest sites with soils of different texture. Soil compaction in the middle of the wheel track caused a considerable increase of N_2O emissions, with values elevated by up to 40 times the uncompacted ones. Compaction reduced the CH_4 uptake at all sites by up to 90%, and at the silty clay loam site its effect was such that CH_4 was even released. Teepe *et al.* (2004) indicated that despite the significant changes in fluxes on the skid trails, the combined effect of both gases was small compared to total emissions (across the area of the stand). However, if soil trafficking is not restricted to the established skid trail system, the area of compaction and consequently the GHG emission may increase with every harvesting operation.

4.2.2 Peatland GHG exchange

Peatlands cover approximately 15% (2.46 x 10^6 ha) of the total UK land area and comprise a C stock estimated to be approximately 2302 MtC (Billett *et al.*, 2010), although there is some uncertainty in these values (see e.g. Lindsay, 2010). The C in peat is therefore >50 % of the total UK soil C pool with the majority found in Scottish peat soils (Dawson and Smith, 2006; Billett *et al.*, 2010). Pristine peatlands are considered to be CO_2 sinks and CH_4 sources, while disturbed peatlands under different land uses can become substantial CO_2 sources (e.g. Lindsay, 2010; Morison *et al.*, 2010). Blodau (2002) reviewed the literature on carbon cycling in peatlands and the importance of water table depth on CO_2, CH_4 and N_2O release. He indicated that methane fluxes between peatland and the atmosphere may range from small uptake rates to emissions of more than 91 tCO_2e ha^{-1} y^{-1} (i.e. nearly 10 x the typical rate of uptake of CO_2 by a forest, e.g. Table 2.3). Methane fluxes are temporally and spatially highly variable. Average emissions of 0.5 to 7.3 tCO_2e ha^{-1} y^{-1} (5.5 to 80 mg CH_4 m^{-2} d^{-1}), are typical of northern peatlands. The methane flux distributions are skewed to larger values; for example, Meyer (1999) measured emission up to 28 tCO_2e ha^{-1} y^{-1} from restored drained peatland, but according to Blodau (2002) values over 9.1–18.3 tCO_2e ha^{-1} y^{-1} are unusual and restricted to ponds. Fens generally are stronger emitters than bogs because the anaerobic zone is on average closer to the peatland surface and methane emissions up to 46 tCO_2e ha^{-1} y^{-1} were measured from continuously water-covered ditches in a drained fen (Minkkinen and Laine, 2006). Within bogs, significant CH_4 emissions are restricted to lawns and hollows.

The effect of vegetation heterogeneity and drainage ditches on CH_4 variability has been studied by Minkkinen and Laine (2006) from peatlands drained for forestry in Finland. Fluxes were found to be related to peatland site type, plant community, water table position and soil temperature. Little variation between plant communities was observed at the nutrient-rich sites with predominantly CH_4 sinks (-2.1 to -0.7 kg CH_4 ha^{-1} y^{-1}). At the nutrient-poor bog sites the highest emissions were measured from a *Eriophorum vaginatum* community (67.4 kg CH_4 ha^{-1} y^{-1}), with decreasing trend to sphagnum (19.1 kg CH_4 ha^{-1} y^{-1}) and forest moss communities (4.8 kg CH_4 ha^{-1} y^{-1}). Their measurements also showed much higher methane emission rates and variation from fen sites (drained, undrained, ditch) compared with corresponding bog sites (Table 4.3). These variations were mainly due to differences in the origin and movement of water in the ditches, as well as differences in vegetation communities in the ditches.

Table 4.3 Estimated annual CH_4 emissions by site (kg CH_4 ha^{-1} y^{-1}); data from Minkkinen and Laine (2006).

Site	Ditch diffusive	Ditch ebullition	Drained	Undrained
Fen	420 to 1640	22 to 84	17 to 45	21 to 131
Bog	70	0	34	65

Note: 'Diffusive' is continuous release of gas by diffusion from surface; compared to 'ebullition', which is sporadic release of bubbles from the surface.

A significant contribution of gullies to landscape-scale GHG fluxes has been reported by McNamara *et al.* (2008) from an upland blanket peat site at Moor House NNR, in the English North Pennines. They showed that although gullies occupied only 9.3% of the total land surface, they accounted for 95.8% and 21.6% of peatland net CH_4 and CO_2 respiratory fluxes, respectively.

The recent study of Dinsmore *et al.* (2009) from the Auchencorth Moss, Scotland (~85% of area peat covered), highlighted the underlying environmental and vegetation characteristics which lead to within-site variability in both CH_4 and N_2O emissions and the importance of such variability in up-scaling. The study was conducted within a catchment with (i) patches of *Calluna vulgaris* with better drainage, (ii) an even mix of hummocks dominated by grasses and sedges, and (iii) a riparian zone dominated by *Juncus effusus*. They indicated that CH_4 emissions varied considerably across the catchment, and appeared to be linked to areas with consistently near-surface water tables with the riparian zone representing a significant hotspot. Contrary to many previous studies, the presence of either sedges or rushes containing aerenchymatous[26] tissue decreased net CH_4 emissions during the two growing seasons. Up-scaling the calculated fluxes using vegetation cover estimates from a satellite image, gave annual mean (and coefficient of variation) catchment CH_4 and N_2O emissions of 1.06 kg ha^{-1} y^{-1} (300%) and 0.02 kg ha^{-1} y^{-1} (410%), respectively, although these values were very sensitive to the vegetation cover estimates.

Effect of land use and restoration on GHG fluxes from peatlands

On a European level, a comprehensive review of carbon and GHG fluxes from peatlands has been provided by Byrne *et al.* (2004). Table 4.4 shows the GHG emission rates from European undisturbed mire based on median values of all the reviewed data for ombrotrophic bog and minerotrophic fen peatland management types. The study concluded that when emission rates are summed as CO_2 equivalents per hectare, these types of peatland are generally sources of GHGs with emission intensities increasing in order:

- bog: forestry < mire < restoration < new drainage for forest/peat cut < peat cut < abandoned after harvest = grass < crop;
- fen: (restoration <) forestry <= mire < new drainage for forest < grass < crop.

However, they noted that the observations that the forested peatlands emit less GHGs than undisturbed mire has to be viewed with caution because the studies cited deployed mild drainage only, in which CH_4 emissions are reduced but peat formation still goes on. Furthermore, they indicated that emissions vary with stand age as transpiration by trees affects the water table of the peat in a way that can result in high water tables and high CH_4 emissions under young stands changing to significant lowering of water table and increased CO_2 emissions under older stands.

26. Aerenchymatous tissues are plant tissues with substantial connecting air spaces that enable long distance gas diffusion.

Table 4.4 Stand-scale emission rates (median values) based on measured fluxes from different bog and fen management types in European peatlands (recalculated from Byrne *et al.*, 2004). Negative values indicates uptake. Note the differences in units for CO_2.

Peatland type	CO_2 $tCO_2\ ha^{-1}\ y^{-1}$	CH_4 kg $CH_4\ ha^{-1}\ y^{-1}$	N_2O kg $N_2O\ ha^{-1}\ y^{-1}$	CO_2e $tCO_2e\ ha^{-1}\ y^{-1}$
Bog (ombrotrophic)				
Afforested (drained)	-0.7	14.9	0.06	-0.31
Drained (for forest and peat cut)	4.03	26.7	0.04	4.72
Grassland	8.62	2.7	0.02	8.69
Arable	16.1	0	0	16.1
Extracted (peat cut)	6.42	23.0	0.63	7.18
Restored	2.27	20.0	0.03	2.78
Pristine (temperate)	-2.6	232	-0.02	3.19
Pristine (boreal/sub-arctic)	-0.73	50.0	0	0.52
Fen (minerotrophic)				
Afforested (drained)	-0.73	-0.07	2.88	0.12
Drained (for forest)	1.47	1.33	1.65	1.99
Grassland	15.1	0.53	7.94	17.5
Arable	15.0	-0.27	18.2	20.4
Restored	No data	16.5	1.01	0.71
Pristine (temperate)	-1.47	189	No data	3.27
Pristine (boreal/sub-arctic)	-1.80	160	0.18	2.26

The impact of peat bog restoration on total GHG emission or sink strengths is currently not clear and in need of more evidence. This is due to the contrasting effect of soil water level, aeration and temperature on the production and consumption of CO_2, CH_4 and N_2O. A rise in the water level (e.g. from seasonal variation, after clearfelling or after drain blocking for peatland restoration) can increase CH_4 emission (e.g. Funk *et al.*, 1994; Aerts and Ludwig, 1997) but peat temperature may also increase, particularly in colder climates (e.g. Prevost, Plamondon and Belleau, 1999; Huttunen *et al.*, 2003) and thus it may cause higher CO_2 emissions. In contrast, Van den Bos (2003) indicated that wetland restoration of reclaimed peat areas in the western Netherlands leads to a reduction of GHG emission because the expected increase in anaerobically generated CH_4 release is much smaller than the decrease in aerobically produced CO_2. Also, according to Tuilttila *et al.* (2000) drainage decreases CH_4 release but rewetting does not necessarily lead to an immediate rise in CH_4 release.

4.2.3 Summary of GHG emissions from forest soils

Table 4.5 summarises the results from this review of available soil GHG flux measurements from forests and peatlands over a large range of climatic, soil types and forest management factors. The large variation in soil fluxes of CO_2, CH_4 and N_2O is evident; they range from 3.7 to 70, -0.26 to 45 and -0.21 to 15.2 $tCO_2e\ ha^{-1}\ y^{-1}$, respectively. In order to reduce the influence caused by outliers, the median values are also shown with the mean. The mean values and ranges are also shown in Figure 4.1.

CO_2 fluxes

Within the British Isles studies, CO_2 emissions declined in the order mineral soils with standing forest > organo-mineral soils with standing forest > peatland sites. The higher emissions from mineral soils may be due to the temperature differences between lowland mineral soils and upland organo-mineral soils. Similar trends in CO_2 fluxes were observed from the clearfelled, other vegetation and non-British sites, with higher CO_2 emissions from organo-mineral sites compared with peatland or organic soil sites (mainly fen or bog). For organo-mineral and peat soils CO_2 emissions were lower on clearfelled sites. Emissions of CO_2 from peatland/wetland sites were lower compared with emissions from peat soils with other vegetation cover (e.g. grasslands and heathlands). There are no literature values on CO_2 and CH_4 emissions from deep peat clearfelled sites but a new experiment is scheduled by Forest Research to quantify soil CO_2, CH_4 and N_2O emissions directly after clearfelling of a deep peat bog site at Flanders Moss during 2012.

CH_4 fluxes

For CH_4, emissions at the standing forest sites showed increases as expected in the order of: mineral soils with CH_4 oxidation < organo-mineral soils < peatland. Emissions of CH_4 were higher from clearfelled sites compared with standing forest, but maximum CH_4 emissions were observed from the unforested deep peatland/wetland sites.

N_2O fluxes

Emissions of N_2O showed a different trend to those for CO_2 and CH_4 with emissions reducing in the order: organo-mineral soils with standing forest > mineral soils standing forest > peatland sites. However, N_2O emissions are at their maximum on clearfelled sites. The lowest N_2O emission values have been observed from unforested or other vegetation peatland/ wetland sites.

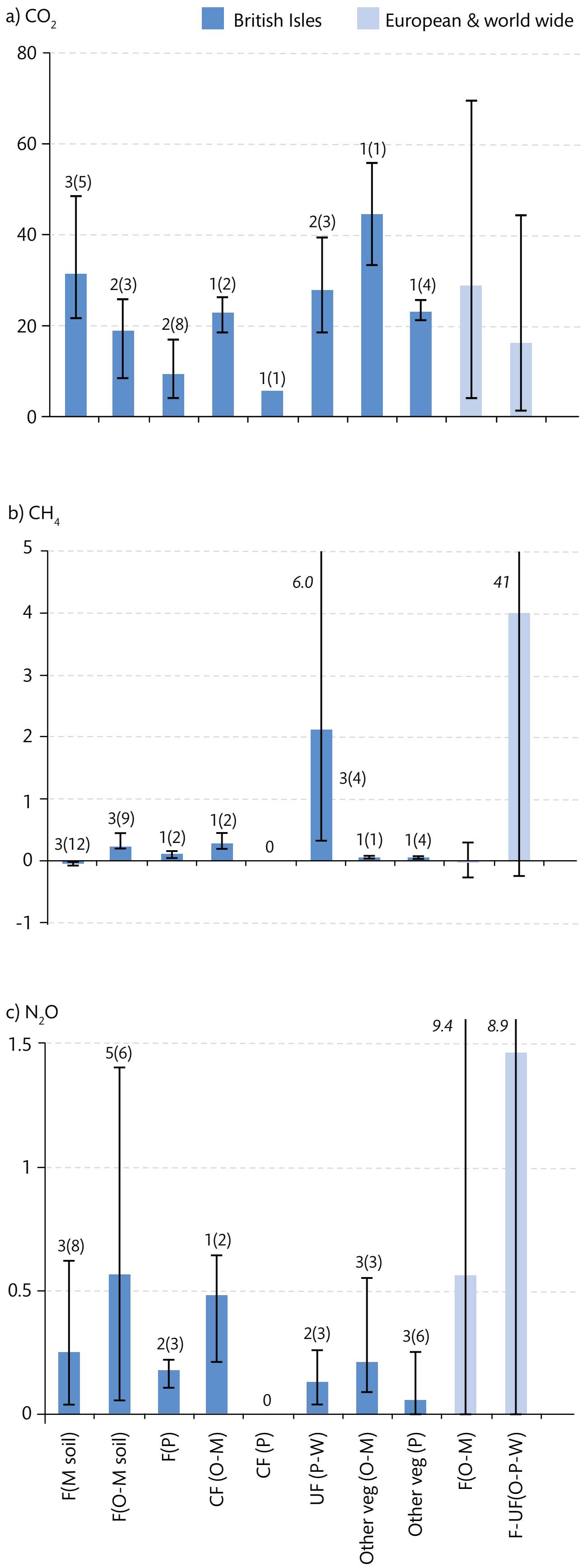
Looking at all the remaining reviewed studies ('European and worldwide', mainly from the temperate zone) showed similar trends to those from the British Isles alone, with higher CO_2 emissions from organo-mineral compared with organic soils and, in contrast, higher CH_4 and N_2O emissions from organic compared with organo-mineral soils.

Table 4.5 Summary of available data on GHG fluxes ($tCO_2e\ ha^{-1}\ y^{-1}$) from forest soils, separated into mineral, organo-mineral (peaty gley, peaty podzol) and deep peat soils (bog and fen). Columns show: means with median in parentheses; number of cited references (n) and sites or treatments within each reference in parentheses; and ranges. Full details and references are given in Tables A5.1 and A5.2 in Appendix 5.

Soil type	CO_2			CH_4			N_2O			Total GHG
	Mean (median)	n	Range	Mean (median)	n	Range	Mean (median)	n	Range	Mean (median)
British Isles										
Standing forest mineral soil	31.2 (29.9)	3 (5)	21.3 to 48.5	-0.03 (-0.02)	3 (12)	-0.08 to -0.01	0.25 (0.17)	3 (8)	0.04 to 0.62	31.4 (30.1)
Standing forest organo-mineral soil	18.5 (22.8)	2 (3)	7.8 to 25.5	0.21 (0.04)	3 (9)	0.01 to 0.44	0.56 (0.24)	5 (6)	0.06 to 1.4	19.3 (22.8)
Standing forest deep peat	8.9 (7.9)	2 (8)	3.7 to 16.6	0.1 (0.1)	1 (2)	0.04 to 0.16	0.18 (0.2)	2 (3)	0.11 to 0.22	9.2 (8.2)
Clearfelled sites, organo-mineral	22.6 (23.7)	1 (2)	18.2 to 26.0	0.27 (0.20)	1 (2)	0.17 to 0.45	0.48 (0.60)	1 (2)	0.21 to 0.64	23.4 (24.5)
Clearfelled sites, deep peat	5.5	1 (1)								
Deep peatland/ wetlands (unafforested)	27.7 (25.5)	2 (3)	18.2 to 39.4	2.1 (1.33)	3 (4)	0.33 to 5.66	0.13 (0.10)	2 (3)	0.04 to 0.26	30.0 (27.0)
Other vegetation sites, organo-mineral	44.5	1 (1)	33.1 and 55.8	0.05	1 (1)	0.03 to 0.07	0.21 (0.11)	3 (3)	0.09 to 0.55	44.7 (44.6)
Other vegetation sites, peatland	22.7 (22.2)	1 (4)	21.0 to 25.4	0.02 (0.01)	1 (4)	0.01 to 0.05	0.06 (0.04)	3 (6)	-0.03 to 0.25	22.8 (22.3)
European and worldwide										
Organo-mineral	28.6 (25.7)		3.7 to 69.7	-0.02 (-0.04)		-0.26 to 0.30	0.56 (0.24)		0.0 to 9.37	29.1 (25.9)
Organic	15.9 (15.0)		0.8 to 44.4	3.99 (1.43)		-0.24 to 41.0	1.46 (0.27)		-0.21 to 8.94	21.3 (16.7)

Notes: CO_2 fluxes denote soil effluxes (soil respiration or forest floor respiration, usually including microbial respiration from litter decomposition and from root respiration). Negative values are fluxes from air to the soil surface.

Figure 4.1 Summary of forest soil GHG flux data from Table 4.5. All values in $tCO_2e\ ha^{-1}\ y^{-1}$. Mean values shown, with ranges indicated by the error bars. Refer to Table 4.5 for category labels. Numbers are the studies included, parentheses indicate number of sites, italic numbers show upper values for error bars that fall outside range of y axis.



4.2.4 GHG emissions from forest stands

There are even less data on GHG balances at the whole-stand scale than for soil-level emissions. This is mainly because of the technical challenges – the static chamber method used for soil emissions of GHG is relatively straightforward and very well established. While eddy covariance for CO_2 is now almost routine (for suitable sites, see Section 2.3), eddy covariance studies of other GHG fluxes are still very limited, although suitable field instruments are now becoming available. The summary of available measurements of the CO_2 net ecosystem exchange (Table 4.6) for similar climatic regions showed NEE values ranging from -32 (strong CO_2 uptake) to 3.2 $tCO_2\ ha^{-1}\ y^{-1}$ (small CO_2 emission) and median values of -9.1 and -5.4 $tCO_2\ ha^{-1}\ y^{-1}$ for British Isles studies and all studies, respectively. The wide range of site and environmental factors affecting CO_2 exchange, and the comparatively limited sets of data mean that it is difficult to identify clear differences between forest or soil types. Given that soil is a net source of CO_2, net CO_2 uptake values for NEE (negative values) derived at the whole-tree plus soil scale demonstrate the high rates of CO_2 uptake by the tree canopy.

Table 4.6 Summary of available data on CO_2 net ecosystem exchange (NEE) from temperate forests: median values and ranges observed in the literature.

Region	NEE ($tCO_2\ ha^{-1}\ y^{-1}$)		
	Median	n	Range
British Isles	-9.1	6	-32.6 to 3.7
All	-5.4	22	-31.9 to 4.2

However, as explained earlier, N_2O and CH_4 sources and sinks are from the soil and litter with no interaction with the above-ground parts of trees, so it is possible to use the stand-scale CO_2 fluxes with information on soil-level N_2O and CH_4 to estimate total stand GHG balances.

The combined total soil GHG flux expressed in CO_2 equivalents (CO_2e) in Table 4.5 was calculated by multiplying the flux of each gas by its global warming potential (GWP) and summing. Using the median values to assess the percentage contribution of soil CH_4 plus N_2O to the total soil GHG flux (CO_2e) shows that CO_2 effluxes dominate (~98%) the total GHG balance. Jungkunst and Fiedler (2007) also reviewed the available annual data on GHG soil for different climates and land uses and showed that despite the fact that CH_4 has a higher GWP than CO_2 it did not outweigh the much larger soil CO_2 losses from soil organic matter (SOM) decomposition in temperate and tropical regions. Based on Martikainen *et al.* (1993) they also indicated that although lowered water tables in the boreal zone may eventually increase N_2O emissions, which would influence the total GHG flux pattern substantially, this is of minor importance at present.

However, the contribution of CH_4 and N_2O to the total GHG budget could be significant in some conditions depending on the forest and soil type and on management practice. High contributions could be expected from N_2O emissions from soils with high mineral N input through either N deposition, fertiliser application or grazing, or after clearfelling. European forest soils have been identified as significant sources of atmospheric N_2O emitting approximately 82 kt N_2O per year, which represents about 15% of N_2O emissions of agricultural soils (Kesik *et al.*, 2005). Furthermore, Kesik *et al.* (2006) indicated that this relatively high contribution of forest soils to total N_2O emissions is at least partly a result of chronically high rates of atmospheric N deposition to formerly N-limited forest ecosystems in Europe.

Methane emissions could make a significant contribution to the GHG balance in wet forested organic soils including deep peatland or wetland areas. Values between 15 and 58% have been reported for CH_4 in peatlands and values between 17 and 20% for N_2O from organic soils (e.g. Meyer, 1999; von Arnold *et al.*, 2005a, b, c). However, Zerva and Mencuccini (2005b) (see later Section 4.6 on forest harvest), showed that the N_2O contribution was only 5.8% of the total GHG budget (7.5% total for N_2O and CH_4) after clearfelling a Sitka spruce plantation on a peaty gley soil in NE England. Similarly, Ball, Smith and Moncrieff (2007) showed that the contributions of non-CO_2 gases to the total GHG budget across a Sitka spruce chronosequence on a peaty gley soil in Harwood Forest was 5.2%. Significantly higher contributions of N_2O and CH_4 to the total GHG emissions have been reported over a range of organic soil coniferous and deciduous forests in Sweden and from restored or drained peatlands in Germany (Jungkunst and Fiedler, 2007), with values ranging from 7.2 to 58.2% for CH_4 (from six study sites) and 8.7 to 23.9% for N_2O (from seven study sites).

It is important to notice that these estimates of total GHG balance are based on soil effluxes only. At the stand scale net CO_2 emissions will be offset by the photosynthetic uptake by trees and other vegetation, so that the contribution of non-CO_2 gases to the net GHG flux will be significantly larger. Friborg *et al.* (2003) indicated that the impact of CH_4 effluxes on the total GHG flux (as CO_2e) increases with increasing latitude and may dominate if plant CO_2 uptake is included. Therefore, as climate, soil and forest management factors have different effects on each GHG of interest the only accurate way of quantifying the contribution of each gas to the total GHG budget would be by simultaneous monitoring of stand-level CO_2, CH_4 and N_2O from a site and for at least one complete year. Currently there are no such data available but some should soon emerge from the EC 'NitroEurope' programmes, where all gases will be simultaneously monitored from 'super sites' across Europe with the main aim of developing and verifying processed-based models for up-scaling and for country and regional inventories.

A comparison, however, can be drawn from the preliminary results (unpublished) of Forest Research's current GHG measurement at the Straits Enclosure oak forest site at Alice Holt. While the mean annual soil CO_2 emissions measured with chambers was approximately +31 $tCO_2\ ha^{-1}\ y^{-1}$, the mean annual net CO_2 exchange for the stand from eddy covariance was -18 $tCO_2\ ha^{-1}\ y^{-1}$ (i.e. a net sink, 11-year average, see Figure 2.4b). The N_2O emissions measured from soil chambers through the year were small in comparison, about 0.9 $tCO_2e\ ha^{-1}\ y^{-1}$, and there was a negligible CH_4 oxidation rate of about -0.07 $tCO_2e\ ha^{-1}\ y^{-1}$.

Globally, the importance of natural ecosystems and CH_4 and N_2O emissions in the balance of GHG and the consequent GWP was highlighted by Dalal and Allen (2008). They estimated that although tropical forests provide a large carbon sink, they have limited capacity to reduce total GWP because of high N_2O emissions, possibly from rapid N mineralisation under their warm and moist conditions. They indicated that most natural ecosystems decrease net GWP from -0.03 ± 0.35 $tCO_2e\ ha^{-1}\ y^{-1}$ (tropical forests) to -0.90 ± 0.42 $tCO_2e\ ha^{-1}\ y^{-1}$ (temperate forests) and -1.18 ± 0.44 $tCO_2e\ ha^{-1}\ y^{-1}$ (boreal forests), mostly by being CO_2 sinks in plant biomass, microbial biomass and soil C. However, the net GWP contributions from wetlands are very large, which is primarily due to CH_4 emissions. The review of Brumme *et al.* (2005) indicated that the net atmospheric impact of exchanges of CH_4 and N_2O by forests (i.e. their contribution to GWP) reduced from tropical to temperate and then boreal forests (491, 142, and 43 kg $CO_2e\ ha^{-1}\ y^{-1}$). They indicated that, although for boreal and temperate forest biomes the combined contribution of N_2O and CH_4 to the total GHG balance (as CO_2e) is small (0–1.9 % and 1–3.61%, respectively), in the tropics the net CO_2 exchange was lower (2.2 $tCO_2\ ha^{-1}\ y^{-1}$), so the net atmospheric impact of N_2O and CH_4 emissions may significantly reduce (by 38%) the potential cooling effect of the CO_2 sink.

4.2.5 National forest GHG budget verification

In order to obtain a robust estimate of the national GHG budget for inventory purposes it is essential that estimates based on the different GHG and C accounting methodologies or modelling for forestry are verified against experimental emission data. For example, von Arnold *et al.* (2005a) showed that using default emission factors provided by the *Good practice guideline for land use, land use change and forestry* (Penman *et al.*, 2003) resulted in a net GHG sink of -5.0 $MtCO_2e$ y^{-1} (range -12 to 1.2) for the national GHG budget of drained organic forestlands in Sweden. However, an estimated net source of 0.8 $MtCO_2e$ y^{-1} (range -6.7 to 5.1) was calculated when available emission data for the climatic zones spanned by Sweden were used. The discrepancy was found to be mainly due to differences in the emission rates for heterotrophic respiration and the main uncertainties were related to carbon changes in litter pools and releases of soil CO_2 and N_2O.

Comparisons between methods can provide valuable information for quantifying uncertainties associated with different carbon accounting methodologies for forestry practice. For example, Black *et al.* (2007) made a comparison between annual NEP of a Sitka spruce forest over 2 years, using either eddy covariance assessements or stand-level C stock change inventory estimated using either ecological or mass-balance approaches (Curtis *et al.*, 2002; Kolari *et al.*, 2004). Estimates for annual net C uptake by the forest varied between 26.8 and 42.0 tCO_2 ha^{-1} y^{-1} using ecological inventory methods and 28.2 to 34.6 tCO_2 ha^{-1} y^{-1} using eddy covariance-based assessments. These differences were not significant. However, the mass-balance inventory values were significantly higher (27–32%), when compared to the other methods. Black *et al.* (2007) suggested that there was a systematic overestimation of the C stock change by mass-balance assessments due to unaccounted decomposition processes and uncertainties in the estimation of soil C stock changes. Fenn *et al.* (2010) found a very good match between a 'bottom-up' approach of measuring plant and soil components ('ecological inventory') and the 'top-down' eddy covariance-derived NPP estimates at Wytham Wood, Oxford, over 2 years.

Overall, although there is some information on GHG fluxes from temperate forests appropriate to UK conditions, there is still considerable uncertainty over the relative contribution they make to the GHG balance and GWP in different situations. In particular, there is a lack of simultaneous measurements of CO_2, CH_4 and N_2O fluxes necessary at appropriate scales to quantify reliably the GWP associated with particular forest management activities and thus assess the impact and potential of particular forest management options for climate change mitigation. Key gaps in evidence are listed in Chapter 6.

4.3 Fluxes of soil DOC in UK forestry

As described in Chapter 2, carbon can be lost from soils through CO_2 emission during decomposition (so-called 'soil respiration'), through loss of soil particulate organic matter in erosion (sometimes termed particulate organic carbon, POC), and also through loss in solution (dissolved organic carbon, DOC). Michalzik *et al.* (2000) reviewed measurements of DOC fluxes in forest soils, and reported fluxes from the O horizon[27] to range from 367 to 1470 kg CO_2 ha^{-1} y^{-1}. Neff and Asner (2001) reviewed further data, and reported a range of 740 to 3000 kg CO_2 ha^{-1} y^{-1} for DOC fluxes in soils up to 20 cm deep, under a variety of vegetation covers, although mostly eucalyptus, coniferous and deciduous forest. If DOC fluxes were commonly at the high end of this range in British forests, then they would certainly need to be taken into account in forest C balances as they would be of the same order of magnitude as annual CO_2 uptake fluxes by trees (see Section 2.2).

In their review of carbon loss from UK soil, Dawson and Smith (2007) reported DOC fluxes in the range 290 to 950 kg CO_2 ha^{-1} y^{-1} for catchments of 24 upland streams and rivers. The mean value of approximately 367 kg CO_2 ha^{-1} y^{-1} is at the low end of the ranges quoted above. However, many of the locations in the Dawson and Smith (2007) compilation contain substantial areas of peatland, and so the quoted surface water fluxes are probably at the high end of the likely range. Against this, stream fluxes are somewhat lower than the corresponding topsoil values, due to the sorption and mineralisation of DOC during transport through deeper soil (Kalbitz *et al.*, 2000).

Annual DOC fluxes from shallow and deeper soil layers in nine Level II forest monitoring sites have been measured (Table 4.7, ordered by flux magnitude). The mean value of 340 kg CO_2 ha^{-1} y^{-1} is close to the lower end of the ranges quoted in the reviews and close to that estimated by Dawson and Smith (2007), but it obviously varies greatly with soil type (range from 4 to 857 kg CO_2 ha^{-1} y^{-1}). Generally, the sites on soils with higher soil C content (e.g. Llyn Brianne, Coalburn, and Grizedale; from Table 3.7

27. See Box 3.1 for description of soil horizons.

in Section 3.5) have higher DOC fluxes at shallow depth of soil than the sites with lower soil C (e.g. Alice Holt, Savernake, Thetford). The shallow depth of sampling was at 10 cm, which for the sites with high soil C stock is within the H layer. DOC is released into soil water moving through the upper organic horizons from the organic matter decomposition and mineralisation in the soil promoted by microbial and fungal activity. It has been shown that hydrologic variability in soil horizons can be as influential as biological activity in determining DOC concentrations in soil solution (Kalbitz *et al.*, 2000). However, removal (adsorption) of DOC from the soil solution occurs via organo-mineral interactions on the surfaces of aluminium (Al) and iron (Fe) oxides and hydroxides in the mineral horizons as part of the podzolisation process. An example is the averaged DOC flux of 425 kg CO_2 ha^{-1} y^{-1} from shallow depths at a podzolic site (Sherwood) which is seven-fold higher than the DOC flux of 58 kg CO_2 ha^{-1} y^{-1} lower down in the soil (50 cm). This deeper soil C can be relatively stable, remaining stored for several hundreds of years (McDowell and Likens, 1988). However, processes that disturb organic matter stability, such as site preparation for planting, clearfelling, destumping etc, will probably reduce the turnover time of this stored carbon (see subsequent Sections 4.4 and 4.6).

Other factors governing organic matter and the retention of DOC in mineral horizons include pH and texture, particularly the proportion of silt and clay particles and the formation of soil aggregates, which protects organic matter from decomposition (Jones and Donnelly, 2004). Different soil types have different sensitivities to C release and different capacity to retain carbon in lower depths. In their comprehensive review Dawson and Smith (2007) reported a study of soil solution DOC retention from a topographic sequence of upland soils in northeast Scotland, and showed that compared to other upland soils, peaty podzols had the greatest potential for DOC retention, mainly by dissolution/precipitation reactions. Shorter contact time between the DOC and amorphous Al and Fe in mineral particles, an increase in soil solution pH and increased frequency of wetting–drying and freeze–thaw cycles, all caused decreases in DOC retention in the Fe-rich horizons of podzols. In highly organic soils the reduction in DOC fluxes that occurs with depth is smaller, due to the lack of available mineral binding capacity. An example is the Llyn Brianne site with only a five-fold reduction in soil solution DOC at depth (e.g. from 857 to 155 tCO_2 ha^{-1} y^{-1}).

Soil DOC fluxes measured at 50 cm may not be the actual fluxes that reflect losses to groundwater and potential drainage, which will depend on the soil depth, soil type and rainfall.

Table 4.7 Soil solution DOC fluxes at UK Level II sites, 1995, measured in triplicates by lysimeters at 10 and 50 cm depths in the soil. The DOC fluxes were calculated with a water balance model using rainfall, through-fall and evapotranspiration, estimated by the Penman-Monteith semi-empirical model (Vanguelova *et al.*, 2010).

Site	Soil type	Annual DOC flux kg CO_2 ha^{-1} y^{-1} (kg C ha^{-1} y^{-1})	
		Shallow soil (10 cm depth)	Deep soil (50 cm depth)
Llyn Brianne	Cambic stagnohumic gley soils	857 (233)	155 (42)
Tummel	Ferric podzol	605 (165)	243 (66)
Sherwood	Brown podzolic soil	425 (116)	58 (16)
Grizedale	Typical brown podzolic soil	403 (110)	92 (25)
Coalburn	Cambic stagnohumic gley soils	221 (60)	not measured
Thetford	Brown calcareous sands	211 (57)	135 (37)
Rannoch	Humo ferric gley podzol	161 (44)	not measured
Savernake	Argillic pelosol	153 (42)	55 (15)
Alice Holt	Pelo-stagnogley	4 (1)	5 (1)

Whether increased forest C stock as forest stands mature causes greater production of DOC is uncertain. A chronosequence study of Norway spruce in Norway showed no significant difference between stands of 10, 30, 60 and 120 years old for O horizon soil water DOC concentrations (Clarke, Wu and Strand, 2007), but this did not evaluate the actual fluxes and C losses. Stand age and canopy changes will undoubtedly influence the interception and the amount of water reaching the soil, and flowing through it. The chronosequence study of 20 Welsh Sitka spruce plantations showed a wide range of DOC fluxes (50–190 kg CO_2 ha^{-1} y^{-1}) at catchment level with no consistent relationship between DOC flux and stand age (Reynolds, 2007). Estimated soil DOC water fluxes beneath the rooting zone in the podzolic B horizon of these forest stands showed a decrease with age (CEH unpublished data reported by Reynolds, 2007). In the same study stand age and surface (O horizon) DOC fluxes were not related, agreeing with the indications from the Clarke, Wu and Strand (2007) study that the stage of forest development has little effect on DOC production in surface organic horizons. However, the B horizon of older forest soils may have a greater capacity to adsorb carbon (Reynolds, 2007).

In UK conditions, annual DOC flux was strongly dependent upon annual run-off, estimated from rainfall and evaporation data, varying by 3.5-fold between years when averaged over all sites (Buckingham, Tipping and Hamilton-Taylor, 2008). On average, 75 % of the DOC was exported during the winter period (October to March).

Forest soils are frequently subjected to drying and wetting cycles but little is known about the effects of repeated drying and wetting and wetting intensity on DOC, which is important when evaluating the possible effects of forest management on soil DOC fluxes. A controlled study from Germany by Hentschel, Borken and Matzner (2007) with soils collected from under Norway spruce stands, reported increases of 27–42% in DOC concentration from the organic soil only and of 43–52% from the organic and mineral soil subjected to repeated drying and wetting treatments. The enhanced release of DOC in the initial wetting phases was from easily decomposable substrates and in the final wetting phases was from decomposition-resistant substrates. Different wetting intensities of 8, 20 and 50 mm d^{-1} did not alter DOC concentrations and fluxes. The authors' overall conclusions were that drying and wetting would cause a small additional DOC input from organic horizons to the mineral soil.

Overall, it is evident that in British conditions, the annual flux of DOC can be important in the overall C balance of forest stands, although it is usually considerably less than 10% of the typical net CO_2 uptake rates by the trees. Such relatively large fluxes are found in organic soils, and on mineral soils typical annual DOC fluxes are usually <1% of likely tree net CO_2 uptake.

4.4 Site preparation

4.4.1 Soil disturbance during preparation

There are currently a variety of treatments which can be performed before sites are planted, whether for afforestation or reforestation. Any disturbance may potentially have an impact on the C or GHG balance of the forest. The disturbance of the soil aims to improve the effectiveness of establishment and subsequent maintenance, through increases in plant survival and enhanced growth of young trees. The optimal treatment for a given site will vary depending upon the soil type, topography and climatic factors, and recommendations for this are provided in FR Information Note on *Forest Ground Preparation*, (Forest Research, 2002) and also Forestry Commission Bulletin 119: *Cultivation of soils for forestry* (Patterson and Mason 1999).

Johnson (1992) reviewed the literature on the effects of various forestry practices upon soil C content. He found that site preparation in general led to C losses, which varied with the severity of the disturbance. The ground area disturbed by common site preparation treatments has been quantified by Worrell (1996, Table 4.8). As expected, mechanised preparation disturbs much more soil than hand treatments. Therefore it is logical to assume that ploughing is likely to cause the greatest C losses, whereas hand screefing is likely to cause the smallest C losses. However, this is likely to be modified by the amount of organic material left on the site following preceding harvesting, as Johnson (1992) noted that less C is lost when brash is incorporated into the soil. Indeed, the overall C content of the soil is likely to be important in determining the C loss following site preparation.

Table 4.8 The soil disturbance of site preparation treatments typical in upland UK forestry (after Worrell, 1996).

Treatment	% area affected	Soil volume (m^3 ha^{-1})
Ploughing	44–60	540
Dolloping	29–31	300
Disc trench scarifying	20–32	290
Disc patch scarifying	14	275
Hand turfing	4–7	60
Hand screefing	negligible	negligible

Johnson (1992) identified a trend that soils with higher C contents lose more carbon following site preparation disturbance than sites with low C content, which may even show an increase in C content. As the majority of production forestry occurs on upland, relatively C-rich soils, it is likely that all preparation will cause some loss of soil carbon.

In Canada Schmidt, Macdonald and Rothwell (1996) and Mallik and Hu (1997) both concluded that mechanical site preparation could cause a significant loss of soil organic carbon (although one site in the investigation of Schmidt, Macdonald and Rothwell (1996) showed no change). In contrast, a literature review and meta-analysis by Paul *et al.* (2002) concluded that there was no significant effect of disturbance level, and that instead the decrease in soil C was attributable to the reduced plant matter input to the soil in the early years of tree growth (<10 years).

One of the largest potential C losses is due to the drainage of high C content soils (e.g. peat) for afforestation. This may cause a shift from a carbon sink to a carbon source (Cannell, Dewar and Pyatt, 1993). Harrison *et al.* (1995) quantified values of potential losses that may occur due to drainage in relevant literature from around the world. Values ranged from 1.5 to 18.3 tCO_2 ha^{-1} y^{-1}, depending on either the site's mean temperature or a combination of its temperature and annual precipitation. Any drainage-induced increase in soil C efflux would result in a concomitant reduction in methane emissions from the soil (Cannell, Dewar and Pyatt, 1993). Although there are less emissions of methane from undrained peat than CO_2 emissions from drained peat, methane has a higher greenhouse effect than CO_2 (see Section 4.1). A recent study by Mojeremane (2009) provides UK data of net GHG fluxes from site preparation of a peaty gley soil, and showed a net GHG loss from drainage but a gain from cultivation, with the balance being a net loss of 10.9 tCO_2e ha^{-1} y^{-1}. Over the longer term, Hargreaves, Milne and Cannell (2003) concluded that restocking of previously disturbed high C content soils (deep peats) with a new rotation will provide a net GHG balance benefit for between 90 and 190 years, dependent on the rate of peat loss under forestry and the productivity of the forest crop. However, it should be noted that the afforestation of deep peat (>1 m) soils is no longer common practice in UK forestry, and Forestry Commission Scotland has issued interim guidance with a presumption against planting on peat deeper than 50 cm (see Morison *et al.*, 2010, for more detail on the possible GHG consequences of forestry on peat soils).

The ECOSSE report (Smith *et al.*, 2007) gives insight into the impacts of site preparation for forestry on organic and organo-mineral soils in the UK, and reiterates the observation that increased disturbance increases C losses through accelerated decomposition rates (see also Smith *et al.*, 2010a). Changes in soil C storage have been reported from a number of studies based on a chronosequence of stands, paired plots and repeated sampling (Jandl *et al.*, 2007), with reported changes in soil C following afforestation varying markedly (from positive to negative). Carbon loss can occur in a brief period following afforestation, when there is an imbalance between C loss by soil microbial respiration and C gain by litterfall (Vesterdal, Ritter and Gundersen, 2002). Planting leads to soil disturbance and can stimulate the mineralisation of soil organic matter, and the low C input by litterfall in a young plantation does not necessarily offset these losses. The ECOSSE report (Smith *et al.*, 2007, 2010b) also gives preliminary management recommendations to reduce C losses associated with site preparation for forestry: soil disturbance should be minimised and as much vegetation cover as possible should be maintained.

4.4.2 Fossil fuel use

The majority of current forestry operations are now mechanised; harvesting, forwarding, site preparation (drainage, mounding) and even planting can now be carried out mechanically. Therefore this uses fossil fuel, usually diesel, which results in emissions of CO_2 and other GHG, in addition to any related to disturbance of soil, for example in ground preparation. For a given fuel or energy use, emission figures can be derived, using conversion factors such as those listed by Defra (2010) of 3.179 kg CO_2e l^{-1} (0.320 kg CO_2e kWh^{-1}) for diesel. These values include both direct emissions (emissions at the point of use of a fuel) and indirect emissions (emissions prior to the use of a fuel, i.e. as a result of extracting and transforming the primary energy source, but not accounting for vehicle construction; so-called 'well-to-tank' emission factors, Defra 2010). For diesel, the CO_2 emissions account for 99% of the total direct emissions, and the indirect emissions are 16% of the total.

Figures derived from UK forestry contractor fuel use information during ground preparation, working predominantly on peaty gley soil types in southern Scotland, are given in Table 4.9.

Table 4.9 UK operational diesel fuel use figures for standard establishment procedures.

Operation	Machinery	Fuel GHG emissions tCO_2e ha^{-1}	
		Direct	Total
Site preparation	Excavator	0.588	0.699
Site preparation	Scarifier	0.214	0.254
Agricultural conversion	Agricultural plough	0.065	0.078

Sources: FR Technical Development; FR Internal Project Information Note 'Fuel Usage Assessment in Forest Operations' and Defra (2010) GHG fuel conversion factors.

These figures are higher than those from other reported studies. For site preparation (establishment) Karjalainen and Asikainen (1996) reported an average GHG emission of 0.35 tCO_2e ha^{-1} for scarification, ditch clearing and remedial drainage in Finland. Higher figures were obtained in Sweden where standard site preparation practice had a net GHG emission equating to approximately 0.6 tCO_2e ha^{-1} (Berg and Lindholm, 2005). Whittaker, Mortimer and Matthews (2010) report a value of 0.367 tCO_2e ha^{-1} from a Forestry Commission study of mounding with an excavator in Scotland, and estimated that typical fencing requirements result in additional emissions that were approximately double those in ground preparation, largely due to the wire-netting manufacture. However, even with the above uncertainty about the emissions from fossil fuel use, it is clear that they are very small compared to the likely CO_2 emissions from the soil itself (e.g. range of approximately 10–45 tCO_2e ha^{-1}, Table 4.5). In addition, it should be noted that site preparation is a single event for a 50-year or longer rotation length, whereas soil emissions are continual, even if temporarily increased by ground disturbance.

Overall, it is clear that there is limited information available on the impacts of normal afforestation and restocking practice on GHG balances, with little understanding of the influence of soil type on site C and GHG fluxes during the establishment phase.

Scaled-up estimates can be made of GHG emissions from fuel use in ground preparation at country and UK scales using a few assumptions (Table 4.10). Taking soil type area estimates for each country (Table 3.9), and assuming new planting or restock soil types are proportional to those in existing forested land area, an area estimate for each soil type that is planted or restocked can be calculated, using Forestry Commission statistics (Forestry Commission, 2010a), split by country and into either broadleaf or conifer establishment. For new planting the soil area for peat types is assigned to peaty gley, as no deep peats are considered for planting. The soil type is then assigned an appropriate cultivation type and fuel use emissions are allocated from Table 4.9. An error is introduced in that areas established by natural regeneration are included in the statistics and thus have cultivation assigned here, although they will not in practice, but this is a very small proportion (e.g. natural regeneration accounted for 0.2% of private sector new planting in Scotland in 2010).

The UK total of 9069 tCO_2e y^{-1} (Table 4.10) for 20 400 ha of planting in 2010, is about a third of the value estimated by Mason, Nicoll and Perks (2009, Figure 6.3, 24,044 tCO_2e y^{-1}), using a different approach based on a categorisation into 'Forest Management Alternatives' and estimated areas and rotation lengths. Here current planting area statistics are used, with assumptions about the type of preparation.

The total of 0.009 $MtCO_2e$ y^{-1} also implies that at current rates of establishment emissions from fuel use for ground preparation are only approximately 0.06–0.1% of annual forest CO_2 sequestration (9–15 $MtCO_2e$ y^{-1}, see Table 2.3). Even with an aggressive woodland expansion the emissions from fuel use for new planting would have negligible impact on the overall GHG balance. The much more important unknown is the impact of soil disturbance during ground preparation on soil CO_2 emissions, which could be much more significant, as suggested by the figures in Table 4.5.

Table 4.10 Estimates of the GHG emissions from fossil fuel use in ground preparation activities for the UK countries.

Country	Woodland type	Establishment fuel GHG emissions (2010) tCO_2e	
		New planting	Restocking
England	Conifer	0	716
	Broadleaf	920	630
Wales	Conifer	0	501
	Broadleaf	25	208
Scotland	Conifer	248	4235
	Broadleaf	664	484
Northern Ireland	Conifer	0	270
	Broadleaf	52	52
UK	Conifer	265	5723
	Broadleaf	1708	1374
UK total	**9069**		

Note: As areas of soil types in NI were not known, the NI planting areas were used with average preparation emissions factors derived for Wales.

4.5 Stand establishment

4.5.1 Plant production

The provision of seedling tree planting stock is an area of forest C and GHG budgeting that has received scant attention. Two studies have investigated the carbon costs of seedling production: Schlosser *et al.* (2003) reported a C balance assessment for containerised seedlings in Russia, and Aldentun (2002) reported a life-cycle inventory for containerised seedlings across Sweden. These studies aid in identifying the components required for consideration of a (near) complete life-cycle assessment (LCA) of UK tree seedling production. The study of Schlosser *et al.* (2003) includes the facilities' production costs (i.e. including infrastructure construction costs) whereas the Aldetun study considers only seedling production costs. In order to calculate the C emissions in seedling tree production the volumes of materials used, seedling size and species, numbers and transportation need to be quantified alongside site energy usage. The Swedish study gives a figure of approximately 126 kg CO_2e per 1000 seedlings. A preliminary analysis by Elsayed, Matthews and Mortimer (2003) suggested for Delamere nursery (Cheshire) that a value of 68 kg CO_2e per 1000 seedlings was appropriate; this is about half the Swedish estimate, but within reasonable agreement given the uncertainties. For reference, this CO_2e emission (per seedling) is about 2.5-fold larger than the CO_2 contained in the seedling (approximately 28 g CO_2, see Section 5.4). More importantly, it is a very small net GHG emission compared to the typical uptake by a stand of trees over their life cycle (see e.g. Section 2.3). Calculation of delivery distance to the plantation site will allow estimation of the GHG emissions to the forest gate; this is likely to be more significant than the GHG emissions during production for most transfers from nurseries given that road haulage emissions are estimated at 77 kg CO_2e per 100 km for a 17-tonne vehicle (Defra, 2010).

In both the nursery and at the planting site the operational application of pesticides and herbicides incurs a GHG emission. Volume (per hectare) application data for effective control of pests and weeds exist and the CO_2 equivalence of application/release needs to be determined for common products utilised in forestry operations. Assumptions regarding the carbon costs of production might then allow calculation of the impact of this input to the forest management chain, as has been attempted for large-scale agricultural land units, for example in Canada (Coxworth, 1997). For the UK Elsayed, Matthews and Mortimer (2003) suggested that the total GHG emissions due to herbicide use in forestry are less than 0.1 tCO_2e ha^{-1}.

Following the planting of a site, seedling size and thus C contribution per unit area is very small until canopy closure. Therefore there will be relatively little C input into the soil in the form of litter or transfer to the rhizosphere from root C exudation into the soil profile. The effect of the soil disturbances caused by standard forest site manipulations to aid seedling establishment will be the dominant influence on soil C changes (see previous section). These changes are dependent upon a number of site factors, including site preparation, species, rotation length and soil type, as described above.

4.5.2 Restocking

There are some data available on the effect of restocking on soil and stand GHG balances. Restocking a peaty gley site (i.e. the second forest rotation) resulted in an increase in soil C at a rate of ≈14.6 tCO_2e ha^{-1} y^{-1}, assessed over the first 12 years (Zerva *et al.*, 2005), although these estimates have a large degree of error associated with them due to the large spatial heterogeneity of soil (see Conen *et al.*, 2005. Other reports from UK studies show that second rotation Sitka spruce net CO_2 uptake rates can range from 25.6 tCO_2 ha^{-1} y^{-1} at canopy closure to 11.0 tCO_2 ha^{-1} y^{-1} in older stands (Kowalski *et al.*, 2004; Magnani *et al.*, 2007). A substantial contribution to the respiratory loss of CO_2 (R_T, see Box 2.1) comes from ectomycorrhizal fungal activity, which is often 'missed' by static chamber assessments, thus underestimating R_T. Heinemeyer *et al.* (2007) suggested this contribution could be around 25% in a growing pine forest, and contributions up to 40% have been measured. Recent results from the oak stand at the Straits Enclosure, Alice Holt, suggest a mean mycorrhizal contribution of 18% (over 3 years, Heinemeyer *et al.*, 2011). These fungi will presumably die after tree harvest, which may produce a relatively large, albeit temporary, soil CO_2 efflux. Thus the total below-ground C allocation appears to change with stand age and rotation, and is an important component of total site C flux which requires further quantification.

Restocking impacts over the rotation

Overall, it is important to acknowledge that these changes in C flux and storage are from a limited number of investigations, and also only provide values for the early part of the rotation. Later changes during the rotation should also be considered so that an integrated, long-term forest C balance accounting for various management options, rotation lengths and rotation cycles can be considered. The application of a gap-type model simulating the dynamics of forest ecosystems in Finland (Pussinen *et al.*, 2002) suggested that the soil C stock in the organic layers did not increase as the length of the rotation period increased from 40 to 110 years. The amount of soil organic matter (SOM) was at its maximum about 5 years after the clearfell, at the time when the input from litter was high due to the decomposition phase of the logging residues. Therefore, the amount of SOM was highest when the rotation length was relatively short (Pussinen *et al.*, 2002).

The recovery of soil C after disturbances will be dependent, over the short term, on the growth rate of the tree species, site nutritional status and climatic characteristics. Nitrogen fertilisation may increase the decomposition rate of SOM in the long term, leading to an increased C stock deeper in the soil profile. Planting more mixed species stands, or deciduous trees, may increase C sequestration in soils. On the other hand, planting faster growing coniferous species at high density in many cases sequesters more C overall and stores C longer than ecosystems dominated by deciduous trees when product substitution is accounted for (see Section 5.5).

There is obviously limited information available on the C emissions in the UK associated with existing seedling production systems, although in the overall C budget of forests this is a relatively small component. More important are the impacts of site maintenance and seedling growth after afforestation/restocking, and little is known about how these differ with species, soil type and establishment practices.

4.6 Forest harvest

4.6.1 Thinning and harvesting operations

To date, there has been little investigation into fuel use and resultant GHG emissions of forestry operations in the UK. What information there is about site operations mostly examines C emissions, and not production of other GHGs, although a recent study assessing the carbon footprint of UK timber transportation and associated infrastructure and operations addresses this deficiency (Whittaker, Mortimer and Matthews, 2010; see Section 4.7). Some measurements of 'on-site' fuel use from conifer forest harvesting operations, predominantly on peaty gley soil types in southern Scotland, are available from FR Technical Development Services. The calculated emissions from a full LCA are shown in Table 4.11. Note that, while it is possible for some operations (e.g. harvesting) to estimate fuel-derived emissions as C or CO_2e per timber volume, for other operations (e.g. woody biomass provision from stumps) the appropriate measure is CO_2e emitted per oven-dried tonne (odt). Figures for fossil fuel use during stump harvesting have been revised from those in Mason, Nicoll and Perks (2009), in the light of additional information from the forestry industry.

Currently the most directly comparable published data on forest operational emissions comes from a Scandinavian study (Berg and Lindholm, 2005). They calculated that harvesting incurred a cost of 4.4 kg CO_2e m^{-3} and forwarding 3.6 kg CO_2e m^{-3}, similar to those in Table 4.11.

Karjalainen and Asikainen (1996) performed a comprehensive assessment of the energy use and resultant GHG emissions in Finnish forestry operations for 1993, covering silviculture, forest improvement, harvesting, forwarding and also the transportation of heavy machinery to sites. That assessment calculated that harvesting incurred a cost of 3.9 kg CO_2e m^{-3}, thinning 8.2 kg CO_2e m^{-3} and forwarding 4.1 kg CO_2e m^{-3}. Given the differences in stand structure and operational efficiency likely between the UK and Scandinavia, and changes in machinery efficiency since 1993, these figures are also similar to those in Table 4.11.

Harvesting emissions 'to the forest gate' can be estimated from UK forestry statistics for softwood (conifer) timber production (Forestry Commission, 2010a). Over the last decade (2000–9) average total softwood production has been 8.23 million green tonnes per year. One green tonne is equivalent to 1.22 m^3 overbark standing timber, so assuming a harvesting operational fuel use of 6.68 kg CO_2e m^{-3} (i.e. summing harvesting and forwarding) provides an estimate of UK average annual softwood harvesting machinery emissions of 0.0683 $MtCO_2e$ y^{-1}. Figures for fuel use and emissions during hardwood harvesting are not known, but may be higher due to the typical lower stem density, and will also be affected by the higher timber density. As a first approximation, using the same values from Table 4.11, with the average UK hardwood harvest of 0.542 million green tonnes, and a conversion of 1.11 m^3 per green tonne, gives an estimate of 0.004 $MtCO_2e$ y^{-1}. The total of 0.071 $MtCO_2e$ y^{-1} can be compared to UK estimates for annual sequestration (9–15 $MtCO_2e$ y^{-1}, see Table 2.3) to indicate that harvesting emissions are only approximately 0.4–0.7% of uptake.[28] The combined estimate is approximately a third of that previously published (0.133 $MtCO_2e$ y^{-1} for harvesting and 0.059 $MtCO_2e$ y^{-1} for thinning, Mason, Nicoll and Perks, 2009, p. 111[29]), although that used a different approach. Mason, Nicoll and Perks (2009) categorised the forest area into five different 'Forest Management Alternatives' types, each

Table 4.11 UK operational fuel use GHG emissions for forest harvesting procedures. Estimates for thinning and harvesting are based on m^3 overbark.

Operation	Machinery	Average fuel emissions			Units
		Direct	Indirect	Total	
Thinning	Harvester	4.154	0.788	4.941	kg CO_2e m^{-3}
Harvesting	Harvester	3.206	0.608	3.814	kg CO_2e m^{-3}
	Forwarder	2.405	0.456	2.861	kg CO_2e m^{-3}
Stump extraction (70% removal)	Modified excavator	0.0307	0.0058	0.0366	tCO_2e odt^{-1}
	Forwarder	0.0101	0.0019	0.0121	tCO_2e odt^{-1}
	Shredder	0.0155	0.0029	0.0184	tCO_2e odt^{-}

Sources as in Table 4.9.

28. Note: No fuel-derived emissions from thinning alone have been included explicitly, under the assumption that most thinning results in harvested product. Similarly, the fuel-derived emissions from any fell-to-waste operations are not included, because this only occurs on a small area.
29. Note: The harvesting value is revised from that in their Figure 5.3 of approximately 0.153 $MtCO_2$ due to an error in calculation.

with particular estimated thinning regimes, felling volumes and rotation length, to derive their figures, while the above estimates use actual data on harvested timber from the UK forest estate.

The statistics for forested area and timber production in each country (Forestry Commission, 2010a) can be used to provide an estimate of GHG emissions from harvesting (Table 4.12), although the area estimates include unplanted land on the Forestry Commission estate, and so may slightly overestimate harvesting activity. Emissions from harvesting in Scotland comprise 58% of the total, although the forested area is only 47% of the UK total, while for England the estimated harvesting emissions are only 24% of the total, although that is 40% of the UK forested area, reflecting the differences in management intensity between countries.

Table 4.12 Estimates of GHG emissions from fossil fuel use during harvesting activities in the UK and countries calculated from average timber removals over the period 2000–9 and net forested land area.

Country	Fuel use GHG emissions ($MtCO_2e\ y^{-1}$)		
	Conifer	Broadleaf	Total
England	0.0139	0.0035	0.0174
Wales	0.0087	0.0002	0.0088
Scotland	0.0411	0.0003	0.0414
N. Ireland	0.0033	0.0000	0.0033
UK	0.0670	0.0040	0.0710

Sources: Forestry Commission (2010a) harvest statistics and Table 4.11.

4.6.2 Consequences of clearfelling and harvesting on GHG emissions

Information on the consequences of thinning and clearfelling on tree carbon stocks are explored in Chapter 3, but a wider consideration of the impact of these activities on GHG fluxes is also necessary. Clearfelling alters many factors that may influence GHG fluxes, such as soil mineral N content (Vitousek and Matson, 1985; Smolander *et al.*, 1998), decomposition of organic matter (Binkley, 1986; Hendrickson, Chatarpaul and Burgess, 1989); soil water content and water table depth (Smethurst and Nambiar, 1990; Keenan and Kimmins, 1993) and soil temperature (Chen, Franklin and Spies, 1993).

The effect of forest clearfelling on the fluxes of soil CO_2, CH_4 and N_2O was examined by Zerva and Mencuccini (2005b) in a Sitka spruce plantation on an organic-rich peaty gley soil in northern England. For the first 10 months after clearfelling their results showed a significant decrease in soil CO_2 efflux with an average efflux rate of 14.6 $tCO_2e\ ha^{-1}\ y^{-1}$ in the mature stand (40-year) and 9.9 $tCO_2e\ ha^{-1}\ y^{-1}$ in the clearfelled site. Clearfelling turned the soil from a sink for CH_4 (-0.03 $tCO_2e\ ha^{-1}\ y^{-1}$) to a net source (0.18 $tCO_2e\ ha^{-1}\ y^{-1}$). For the same period, soil N_2O increased significantly after clearfelling from an average of 0.25 $tCO_2e\ ha^{-1}\ y^{-1}$ to 0.62 $tCO_2e\ ha^{-1}\ y^{-1}$. They indicated that, although clearfelling increased the N_2O and CH_4 emissions by 290% and 1650%, respectively, relative to those of a 40-year-old stand, CO_2 remained the dominant component of GHG emission at least in the early stages after clearfelling. Ball, Smith and Moncrieff (2007) also measured CO_2, CH_4 and N_2O annual fluxes across a Sitka spruce chronosequence (20-year and 30-year) on a peaty gley soil in Harwood, England and from a site clearfelled some 18 months before measurement. However, they measured higher CO_2 and CH_4 fluxes after clearfelling but relatively lower N_2O fluxes (Table 4.13). Using empirical modelling Kowalski *et al.* (2004) estimated the net CO_2 uptake for the 30-year and clearfelled sites and showed that harvesting converted a mature forest stand from a net GHG sink (-17.2 $tCO_2e\ ha^{-1}\ y^{-1}$) into a GHG source (4.0 $tCO_2e\ ha^{-1}\ y^{-1}$). Losses from woody residues (brash, stumps) retained on the clearfell site account for the majority of this observed difference (see Ball, Smith and Moncrieff, 2007). Several of these authors highlight the importance of water table depth as an influence on CO_2 losses from the soil.

Table 4.13 Annual soil fluxes (mean 2001 and 2002) of GHGs ($tCO_2e\ ha^{-1}\ y^{-1}$) from Sitka spruce stands in Harwood Forest, Northumberland (Ball, Smith and Moncrieff, 2007).

Gas	20-year	30-year	Clearfelled
CO_2	8.95	17.95	24.85
CH_4	0.05	0.02	0.31
N_2O	0.06	0.98	0.24

Huttunen *et al.* (2003) observed no significant differences in fluxes of CH_4 and N_2O measured during three seasons following clearfelling of drained peatland forests in Finland. Even in situations where forest soils are not emitting CH_4, there are changes in non-CO_2 GHG balances. For example, Castro *et al.* (2000) showed a shift in net forest CH_4 emissions from sink to source in a Florida slash pine stand following clearfelling and Bradford *et al.* (2000) observed reduced oxidation of CH_4 in the soil of clearfelled temperate forests (beech, Japanese larch and oak) in the UK (see Section 4.2, 'Thinning' sub-section).

Jarvis and Linder (2007), citing unpublished work by Clement and Moncrieff, suggested that net CO_2 uptake in a Sitka spruce forest after thinning drops approximately 20% because of the reduced gross photosynthetic production (GPP, see Box 2.1) caused by the reduction of canopy leaf area. This was equivalent to a reduction of 4.8 $tCO_2e\ ha^{-1}$ in the year following thinning. In subsequent years the net uptake was observed to recover. Clearly the extent of such an effect will depend on the thinning intensity, the loss of leaf area, and the rapidity of re-establishing complete light interception.

Kowalski *et al.* (2004) studied the effect of harvesting on European forest net CO_2 uptake by comparing four pairs of mature and harvested sites (Britain, Finland, France and Italy) via a combination of eddy covariance measurements and empirical modelling. They indicated that while every comparison revealed that harvesting temporarily converted a mature forest stand from a carbon sink into a carbon source of similar magnitude, the mechanisms by which this occurred were very different according to species or management practice. They concluded that understanding the effect of harvest on the C exchange of European forest systems is a necessary step towards characterising C exchange for forests on large scales.

4.6.3 Characterising the GHG impacts of forest residue harvesting

As discussed in Sections 3.6 and 3.7, and by Suttie *et al.* (2009), forest product utilisation can increase forestry's contribution to climate change mitigation through the provision of materials and renewable forms of fuel and energy. Woodfuel is sometimes referred to as 'carbon-lean',[30] because emissions of CO_2 during wood combustion are potentially balanced by uptake of CO_2 during subsequent tree regrowth, but there are several environmental aspects to be considered (Lattimore *et al.*, 2009), including energy use in forest management and harvesting, reduction of in-forest C stocks, and the timescale of CO_2 release.

The potential for additional woody residue harvesting to provide biomass for renewable energy generation has led to the development of harvesting techniques which mechanise the removal and conversion of woody residue components (brash, stumps). However, increasing intensity of forest residue harvesting and stump removal leads to reduced second rotation forest growth and an increase in GHG emissions. This will occur during the harvest–restock management phase because intensified forest management activities lead to higher fossil fuel use and increased soil disturbance causes additional soil C losses and more C removal from site. While forest C sequestration through the growth of woody biomass is relatively well quantified, the effect of enhanced woody residue removal and increased soil disturbance upon site C dynamics is much less clear.

Whole-tree harvesting

Pressure is growing to extend whole-tree harvesting (WTH, the removal of all parts of the tree) as opposed to conventional harvesting (CH, where only the tree stem is removed) to upland conifer plantations as a way of maximising woody biomass yields in the UK. In 1997, it was estimated that some 10% of current clearfelling programmes have been achieved through WTH (Nisbet, Dutch and Moffat, 1997). Since then the pressure to employ WTH has increased under the new UK commitments to provide increased renewable resources from woody biomass. McKay (2006) estimated that over 50% of the available woodfuel resource in Britain would be provided by harvesting poor quality stemwood, tree tips and branches.

However, the greater utilisation of forest biomass raises concerns about potential impacts on long-term site productivity and sustainability, due to the removal of increased amounts of nutrients from the forest ecosystem compared with conventional harvesting (Raulund-Rasmussen *et al.*, 2008). Conventional harvesting systems only involve the extraction of the saw-logs and small round-wood from the forest while the small branches and leaves, which contain most of the nutrients, are left on site to decay. The nutrients are reabsorbed into the soil and are therefore available to support the growth of the remaining trees or of a successor stand. By contrast, in WTH the complete tree (i.e. including small branches and leaves) is removed from the site and the various components are processed off site. This can include the chipping of small branches for woodfuel or the bundling of branches into 'brash bales'. A review by Dutch (1994) suggested that WTH of conifers would increase biomass removal by about 34% over CH, but would increase removal of N, P, K, Mg and Ca by 180, 190, 160, 100 and 110%. Proe *et al.* (1996) cited data suggesting that WTH of a 50-year-old stand of Sitka spruce would increase biomass removal by 27%, but removal of N, P, K and Ca would increase by 234, 176, 141 and 69%.

Such increased nutrient removals are of considerable concern because the few studies that have investigated

30. The phrase is not a good one, given that woody material is 50% carbon, but is shorthand for the point that burning wood does not directly produce GHG emissions from fossil fuel, unlike coal, oil or gas-fired heating, only indirectly through the fossil fuels used during planting, harvesting, processing and transport.

the potential impacts of WTH on future growth in British forests have suggested that there may be some loss of productivity in second rotation stands. Proe and Dutch (1994) reported results from an experiment on a peaty gley soil in Kielder forest where there was a decline in height growth of 20% in 10-year-old Sitka spruce grown after a WTH treatment compared to CH. A later study in the same experiment suggested that mean tree volume had fallen by about 30% following WTH (Proe *et al.*, 1996). Assessments over the next 17 years of growth showed a sustained and significant negative effect of WTH on height, mean tree diameter and basal area (Mason *et al.*, in prep.). However, these differences appeared to be greatest at about 15 years after planting and to progressively decline thereafter.

The differences observed in this experiment are larger than those reported from a nearby Kielder site in a more recent series of experiments examining the impact of WTH on subsequent growth of Sitka spruce in upland Britain (Mason *et al.*, 2011). There, after 10 years, the reduction in growth following WTH was between 11 and 15% for mean tree diameter and height. Both sites were peaty gley soils and would be classed as a 'medium risk' for residue harvesting (Forest Research, 2009a). On a peaty ironpan soil in North Wales, Walmsley *et al.* (2009) reported a 10% reduction in mean tree diameter of 23-year-old Sitka spruce planted on plots that had been subject to WTH, which is quite similar to the trend observed in the Kielder experiment where the difference was 16% at year 20 and 9% at year 27. A number of other studies in northern Europe have also shown short- to medium-term declines in growth following WTH (Jacobson *et al.*, 2000; Egnell and Valinger, 2003).

The results from such studies can be ambiguous because the degree of change following WTH may depend on whether the soil is 'sensitive' or 'robust' (Raulund-Rasmussen *et al.*, 2008) to increased nutritional removal. For this reason a number of countries, including the UK, have recently produced guidance to identify those site types which may be more at risk from whole-tree harvesting (Stupak *et al.*, 2007; Forest Research, 2009a). In established long-term Sitka spruce experiments in Scotland, the response of tree growth to WTH was a reduction of 5–9% for height growth and 5–7% for diameter increase after 10 years on a site with medium soil fertility but the reductions were about 9 and 19%, respectively on the poorest site. The results show that the impacts of brash removal during WTH depend on site type and soil fertility, and also that it may take nearly a decade before the impacts of such practices are evident (Mason *et al.*, 2011).

Soil assessment at the long-term experiment in Kielder in 28-year-old second rotation Sitka spruce sites after WTH showed no evidence that WTH decreased soil C and N, but on the contrary there were significantly higher concentrations and stocks in the WTH soils compared with CH, where brash had been left. For example, peat layer and mineral A soil layers contained 396 and 327 tCO_2 ha^{-1}, respectively in WTH plots compared with 264 and 180 tCO_2 ha^{-1} in soils in the CH plots (Vanguelova *et al.*, 2010). The depletion of SOC and N after CH was attributed to much higher mineralisation rates in the brash plots than in the WTH plots, where significantly less soil available NO_3–N was found. These results are in accordance with the evidence of Likens *et al.* (1970), Lungren (1982), Emmett, Anderson and Hornung (1991) Emmett, Stevens and Reynolds (1995) and Moroni *et al.* (2007) that retention of forest residues on site may increase the rate of mineralisation of existing soil C stocks. In other studies, removal of logging residue either did not affect (Brais *et al.*, 2002; Mariani, Chang and Kabzems, 2006) or decreased the net mineralisation of N in the long term (Piatek and Allen, 1999; O'Connell *et al.*, 2004). WTH and bole clearcut treatments with reduction of woody material correlated with decreased microbial activity and less available N in both systems (Hassett and Zak, 2005).

The meta-analysis of Johnson and Curtis (2001) suggests variable effects of harvesting intensity on mineral soil C in coniferous forests. Some studies demonstrate that leaving debris on site has a net positive effect on mineral soil C stock; others show little or no difference between WTH and CH. In a Norway spruce stand in Finland, growing on a relatively fertile soil, the rate of C mineralisation in the humus layer, 10 years after treatment, was lower in WTH than in CH treatments (Smolander, Levula and Kitunen, 2008). The effects of logging residue removal are clearly time, site and soil specifically related (Raulund-Rasmussen *et al.*, 2008). The direction and magnitude of the response of soil C to brash removal or brash retention depends on the SOM quality and quantity. The greatest changes in soil C and N can be expected in soils with deep organic layers and high soil C and N stocks, which is typical of the soil type in the Kielder study by Vanguelova *et al.* (2010). At the Beddgelert site in Wales, 23 years after WTH, the soils under Sitka spruce had a tendency towards higher soil C content in the WTH plots compared to CH plots (Walmsley *et al.*, 2009). The stagnopodzolic soils at Beddgelert have much thinner organic layers than the peaty gley soils at Kielder. Leaving brash on highly organic soils appears to increase C and N mineralisation and in the long term leaves the soil with significantly lower C stocks. Soil type, site quality and environmental conditions are extremely important in selecting the appropriate stands for WTH practices and these are reflected in guidelines for sustainable WTH forest management practices (Forest Research, 2009a).

These results suggest that while WTH leads to a reduction in subsequent above-ground tree biomass compared to CH, these practices on selected soil types and certain sites may be beneficial for soil C and N sequestration. This indicates that cost–benefit analyses are necessary before decisions are made on the appropriate type of forest operations (harvesting and replanting), considering both geology and soils in order to serve environmental benefits, long-term sustainability and the available biomass production for timber and biofuel (Vanguelova *et al.*, 2010).

Stump harvesting

It is likely that the most damaging residue-harvesting activity is stump harvesting due to the high levels of consequent soil disturbance. Removal of stumps for phytosanitary control of root rot fungi, *Heterobasidion annosum* (Fomes) has been practised in Thetford forest but stump harvesting for the bioenergy market is a recent development (Moffat, Nisbet and Nicoll, 2011). Currently it is being practiced in some areas in Scotland, and assumes that tree stumps (and the large woody roots removed with the stump) represent an additional forest biomass component for energy production (primarily electricity), and their use will increase GHG emissions abatement. Additional potential site-based benefits of stump harvesting have been identified as the possibility of reduced pine weevil (*Hylobius*) damage, and improved planting economics. The benefit for pest reduction remains unproven (Walmsley and Godbold, 2010). Mechanised planting with 'single-pass' combined stump removal and mounding systems has been demonstrated in Scandinavia (Saarinen, 2006) although the author noted that a comparison of mound quality should be made, suggesting that the 'single-pass' method may not be as effective as more traditional methods of restock cultivation.

Although Sitka spruce forests accumulate substantial C below-ground (Zerva and Mencuccini, 2005a, b; Black *et al.*, 2009, see Figures 2.7 and 2.8) there are few available peer-reviewed investigations which address medium to high C content soils such as are prevalent in upland Britain, and disturbance effects are poorly represented in soil process models. Knowledge of the impacts of disturbance is necessary to understand and predict forest ecosystem response and impacts on forest C sequestration potential. Jarvis and Linder (2007) have discussed the potential impact of residue removal for biomass energy production, utilising UK site data from Harwood (Zerva *et al.* 2005, Zerva and Mencuccini, 2005a, b; Ball, Smith and Moncrieff, 2007) and Griffin (Clement, Moncrieff and Jarvis, 2003) forests across first and second rotation spruce plantations. Jarvis *et al.* (2009) also interpreted this UK dataset to assess the potential impact of stump harvesting, and suggested that 'stump removal soil disturbance is akin to ploughing a substantial area of the site to a depth of 1 m'. They reported CO_2 flux measurements from a stump harvesting site in Sweden that estimated annual CO_2 emissions of 25 tCO_2 ha^{-1} y^{-1} (A. Grelle pers. comm., cited in Jarvis *et al.*, 2009). This is considerably higher than the estimate for a mineral soil of 5.0 tCO_2 ha^{-1} y^{-1} given by Hope (2007) although soil respiration rates of 25 tCO_2 ha^{-1} y^{-1} are within the range reported for forests on mineral and organo-mineral soils (see Table 4.5). Jarvis *et al.* (2009) postulate that the original ploughing for the first rotation stimulated soil CO_2 emissions, and suggest that stump removal is therefore likely to cause similar substantial soil CO_2 emissions that will continue throughout the next rotation. Their assessment further suggested that 'should enhanced emissions (resulting from stump removal from YC 14–16 Sitka spruce forest) continue for more than 10 years, the gain to the atmosphere from substituting stump-derived chips for fossil fuel would be negated'.

Other published studies have shown varied impacts of stump removal on soil C. Hope (2007) and Zabowski *et al.* (2008) both report a mean reduction in forest floor C of 6% and Zabowski *et al.* (2008) observed an average loss in soil C of 24%, on a mineral soil. Eriksson *et al.* (2007) described a system-wide LCA net carbon benefit analysis, which included a scenario of intensive forest residue removal (including stumps), for Norway spruce on a low C soil. However, Eriksson *et al.* (2007) assumed that existing soil C was lost at a constant rate, irrespective of harvesting regime, which means that the losses measured by Grelle (referred to by Jarvis *et al.*, 2009) were not considered. Eriksson and Gustavsson (2008) made a similar series of comparisons between removal of stumps, small round-wood bundles or chipping residues at thinning. They suggested that stump biomass utilisation at thinning can provide up to a four-fold net CO_2 benefit, but because they also omitted the effect of soil disturbance on CO_2 emissions their findings are likely to overestimate the benefit of stump removal. Repo, Tuomi and Liski (2011) assessed the potential impact of harvest residue removal by the additional consideration of 'indirect emissions',[31]

31. This is an unfortunate term, as this has been widely used to refer to emissions resulting from production of fuels, prior to actual use, see Section 4.4.2.

which they define as the difference between the emissions during the combustion of the harvest residues, which releases CO_2 to the atmosphere soon after harvesting, and the decomposition of the residues if they had been left *in situ*. They make the point that the longer lived the residues are, the less net benefit there is in removal for combustion. These 'indirect emissions' are significant and Repo, Tuomi and Liski (2011) report that the difference between immediate bioenergy CO_2 release and slower release during decomposition can result in effective emissions from stump-derived biomass energy similar to those of fossil fuels for the first two decades of stump extraction delivered at a forest landscape scale, and account for at least 85% of the total emissions of harvesting stump material, excluding stump combustion CO_2 loss.

It is possible to make an outline calculation of the potential net CO_2 emissions of stump harvesting using currently available published values. While not detailed, and necessarily combining data from different sites and countries, it indicates the possible effects, in the absence of UK-specific evidence (Table 4.14). The calculations assume that 65% of stump + root biomass is extracted (Pitman, 2008), the depth of soil disturbance is 0.8 m, stump moisture content is 59% and some soil is removed with the stumps (Saunders, 2008). Additional 'indirect emissions' (*sensu* Repo, Tuomi and Liski, 2011) have not been calculated due to lack of information on stump decomposition rates, which are likely to be higher than those quoted for Scandinavia because of higher temperatures in the UK, so 'indirect emissions' will be lower than the Repo, Tuomi and Liski estimates.

Based on the values calculated in Table 4.14, potential stump removal benefits would be negated by soil C losses of >15% for a mineral soil and for a peaty gley a loss of >5% over the next rotation. The reports of soil CO_2 loss following large-scale disturbance are approximately equivalent to 1–2% loss per year (Hope, 2007; Jarvis *et al.*, 2009). As soil CO_2 emissions are substantial even without disturbance (see Table 5.6), a conservative estimate might be an **additional** C loss of 1% y^{-1} for both these soil types, for the area directly disturbed (assumed 65% here as a minimum, Strandström, 2006). Thus net gains would be void at around 5 years for peaty gleys and about 15 years for mineral soils. Clearly, measurements of soil C changes during and after stump harvesting in UK conditions are urgently required to improve these estimates.

Lindholm, Berg and Hansson (2010) have described a comprehensive LCA assessment of fuel use in all parts of the biomass utilisation chain for softwood extraction in Sweden. They identify the importance that soil C losses may play in the net benefit of intensive stump residue extraction, but do not currently include this component in the assessment, reflecting the paucity of data available. Walmsley and Godbold (2010) also reviewed the potential impacts of stump removal for woody biomass provision. They conclude that biomass available from the stump–root system offers the potential to gain an additional 20% beyond that obtainable from the stem (Richardson *et al.*, 2002), but that a reduction in future forest productivity may be expected, due primarily to nutrient removals in harvested forest residues.

Looking more generally at woodfuel production systems, Lattimore *et al.* (2009) identified six potential environmental issues requiring consideration: impacts on soils, hydrology and water quality, site productivity, forest biodiversity and greenhouse gas balances. Very little work has been conducted on issues of site productivity and biodiversity impacts. Some evidence may be gathered from models of forest growth and soil nutrient cycling, but considerable

Table 4.14 Calculation of the potential contribution to GHG emissions reduction through fossil fuel substitution by wood chips from stump harvesting of Sitka spruce.

	Yield class 20	Yield class 14
	Brown earth	Peaty gley
[A]Soil C stock, $tCO_2\ ha^{-1}$ (peat depth, cm)	495 (0 cm)	1174 (40 cm)
[B]Stump material harvested, odt ha^{-1}	69.1	55.4
Stump material harvested, $tCO_2\ ha^{-1}$	127	101
[C]Emissions from harvesting and chipping, $tCO_2e\ ha^{-1}$	4.3	3.5
[D]Emissions from transport, $tCO_2e\ ha^{-1}$	1.9	1.5
[E] Soil C removed from site, $tCO_2\ ha^{-1}$	1.2	3.0
[F]Potential benefit of stump biomass, $tCO_2e\ ha^{-1}$	56.0	44.9
Net potential benefit (less fossil fuel emissions), $tCO_2e\ ha^{-1}$	48.5	36.8

Notes and assumptions:
[A] See Section 3.5.1, Table 3.8.
[B] 65% of stump and root biomass harvested, Pitman (2008).
[C] 67 kg CO_2e odt^{-1}, see Section 4.6.1, Table 4.11.
[D] 11.4 kg CO_2e fwt^{-1}, direct and indirect, round trip of 164 km, see Section 4.7.1.
[E] 12% of fwt is soil (<17% for freshly harvested stumps, Saunders, 2008).
[F] Net benefit of chipped material is 0.81 tCO_2e odt^{-1} (Suttie *et al.*, 2009).

effort is required in order to further understand the impact of forest residue harvesting including stump removal on such factors. For example, it is likely that re-vegetation will be more rapid on sites cleared for stump removal, which will promote C sequestration by the herb layer and help to reduce the effects of soil nutrient removals.

Current guidance from Forest Research (Forest Research, 2009b) for the Forestry Commission addresses many of the major environmental concerns highlighted in both Lattimore *et al.* (2009) and Walmsley and Godbold (2010), by considering compaction, acidification, site nutrient capital and soil C status to present a 'precautionary' principle in the deployment of stump harvesting practices on UK forest sites. In the Forest Research guidance a matrix of potential losses by soil type was developed (see Table 4.15). In addition a Research Information Note (Moffat *et al.*, in prep.) providing interpretation of the existing scientific evidence is in preparation. The current paucity of empirical data means that it is difficult to accurately predict the impact of stump removal on the exchange of CO_2 and other greenhouse gases for different soil types. Therefore, a simple soil classification based on the expectation that the potential scale of C loss is likely to be directly related to the soil C content was adopted in the development of the guidance. Soils were classified into three risk categories based on the depth of peat layer (Table 4.15).

In response to the existing concerns over the sustainability of stump harvesting Forest Research is implementing a field assessment of the various component fluxes and stocks impacted by stump harvesting operations, for both carbon and soil nutrients, to inform this debate and including an assessment of the effect on *Hylobius* population dynamics.

Table 4.15 Distribution of soil types by risk of soil carbon loss. Reproduced from Forest Research (2009b).

Risk category	Forestry Commission soil types
Low	Brown earths, podzols (except peaty type (3p)), calcareous soils, intergrade and podzolic ironpan soils (4b and 4z), ground water gleys (except peaty phase (5p)), surface water gleys, littoral soils, rankers (except peaty type (13p)) and skeletal soils
Medium	Peaty podzol (3p), ironpan soils (except intergrade (4b) and podzolic (4z) types), peaty ground water gleys (5p), peaty gley soils and peaty rankers (13p)
High	*Juncus* bogs, unflushed peatland/bog soils and *Molinia* bogs

4.7 Timber transport and road building

4.7.1 Timber haulage

Recent information on emissions during timber transport comes from the report by Whittaker, Mortimer and Matthews (2010). From a detailed consideration of the factors affecting fuel consumption in timber haulage, including terrain, poorer aerodynamics compared to typical haulage, and differences in consumption on public and forest roads, they have estimated appropriate figures for direct GHG emissions per tonne-kilometre (i.e. tonne of material moved 1 km, Table 4.16). Direct GHG emissions are estimated at 0.052 kg CO_2e per t-km round trip on forest roads and 0.039 kg CO_2e per t-km on public roads. The indirect emissions from vehicle manufacture and maintenance were also estimated. These add substantially, and Whittaker, Mortimer and Matthews (2010) point out that these are higher than in other haulage operations because of the shorter assumed life of vehicles and the more frequent tyre replacement necessary.

If the average distance per round trip is 164 km, and with 20% of the journey on forest roads (quoted by Whittaker, Mortimer and Matthews, 2010, from FR Timber Miles Survey, 2007), this can be combined with the values in Table 4.16 to estimate direct GHG emissions per tonne transported over a typical trip of 6.82 kg CO_2e t^{-1}, and indirect emissions of 4.59 kg CO_2e t^{-1}, totalling 11.41 kg CO_2e t^{-1}. These figures are for timber haulage from the forest to processing, and do not include emissions due to subsequent transport of timber, which are also considered in the Whittaker, Mortimer and Matthews (2010) report.

Table 4.16 GHG emission values for timber haulage from forest to processing, taken from Whittaker, Mortimer and Matthews (2010, Tables 3 and 6).

	kg CO_2e per t-km
Direct emissions	
On forest roads	0.052
On public roads	0.039
Indirect emissions	
Vehicle manufacture	0.015
Vehicle maintenance	0.013

Assumptions: 40 t gross vehicle, load 25.5 t, round trip, fully laden outward journey, empty on return.

Combining the average emissions value with average figures for the total UK harvest (production) of 8.23 Mt softwood and 0.54 Mt hardwood (= 8.77 Mt green tonnes, see Section 4.6.1, Forestry Commission, 2010a) gives an estimate of total timber haulage emissions in the UK of approximately 0.100 $MtCO_2e$ (direct and indirect). These emissions are relatively small, and only a little more than the emissions from harvesting operations (see Section 4.6). While timber production figures have been relatively stable in the past 5 years, they are forecast to increase by perhaps 33% for 2012–16 (Forestry Commission, 2010a). This will presumably increase the emissions, but they will still remain a very small component in the overall UK forestry GHG balance (Table 2.3).

4.7.2 Road building

Berg and Lindholm (2005) identified that energy use during forest management differed mainly as a result of the carbon cost of 'secondary' haulage (from forest gate to industry) and with the impacts of road building to facilitate this delivery. Karjalainen and Asikainen (1996) reported that the highest emissions from forest operations were associated with the building of forest roads. They calculated that forest road building used fuel equivalent to emissions of 7 $tCO_2e\ km^{-1}$. This is about one-third of the value estimated by Whittaker, Mortimer and Matthews (23.8 $tCO_2e\ km^{-1}$, 2010, Table 12) after performing an LCA of UK forest roads, including the acquisition and laying of aggregate. However, in order to assess the GHG balance of forest road building the key quantities are how long the roads last, and how much maintenance work is carried out (and consequently how much road material used). Mason *et al.* (2009) used Forestry Commission Type A and Type B road density information and fossil fuel emission rates calculated in an LCA by Whittaker *et al.* (2008) to calculate for a mix of five different 'Forest Management Alternatives' a total UK GHG emission for forest road construction and maintenance of 0.0425 $MtCO_2e\ y^{-1}$ for a road network estimated at 43 300 km. Using the same assumptions about road longevity and maintenance requirements with the Forestry Commission road network information, emissions from road construction and maintenance can be estimated at country scale (Table 4.17), giving a similar UK total of 0.0410 $MtCO_2e\ y^{-1}$. While very sensitive to assumptions over longevity and maintenance frequency, this shows that emissions due to roads are less than those from harvesting and thinning (0.071 $MtCO_2e\ y^{-1}$, Section 4.6.1), ground preparation (0.009 $MtCO_2e\ y^{-1}$, Section 4.4.1) or from timber haulage (0.100 $MtCO_2e\ y^{-1}$, previous section).

Table 4.17 Estimates of total GHG emissions from forest road construction and maintenance in the UK and countries calculated from road type, road density and forest area estimates (recalculated from Mason *et al.*, 2009, Table 5.7, and Whittaker *et al.*, 2008).

Country	Road length (km)		GHG emissions (kt $CO_2e\ y^{-1}$)
	Type A	Type B	
England	5014	12496	15.1
Wales	1539	5254	5.4
Scotland	8147	10349	19.2
N. Ireland	483	879	1.3
UK	15182	28978	41.0

Notes and assumptions:
1. Road density in private forest areas is the same as in Forestry Commission areas.
2. Northern Ireland values estimated using GB average densities.
3. Emissions are 1.67 and 0.541 $tCO_2e\ km^{-1}$ year for Type A and Type B roads.
4. Type A re-surfaced every 10 years, and re-graded twice a year; Type B are re-surfaced once every 50 years and re-graded every 10 years.

While Whittaker, Mortimer and Matthews (2010) separately estimated GHG losses from soil disturbance during road building, their estimates were limited to a non-peaty gley (41 $tCO_2e\ km^{-1}$ road, Whittaker, Mortimer and Matthews, 2010, Table 12), and they assumed that if a road had to be constructed on peat soil it would be overlain with no consequent disturbance. Such assumptions should be revisited and estimates improved, and they need to consider the effect of the soil disturbed indirectly through drainage changes caused by the road. For example, in the UK work modelling the impacts of siting turbines on deep peat soils, Nayak *et al.* (2008) have calculated the potential impact of floating road construction. For floating roads (width 3.4 m, depth 0.5 m) developed on an acid raised bog deep peat site the impact is estimated at approximately 345 $tCO_2e\ km^{-1}$ for construction at afforestation, clearly much larger than the emissions estimated for construction and maintenance described above.

4.8 Afforestation

4.8.1 Introduction

Although the C accumulation in trees and their component stocks can be reasonably well predicted and assessed, as discussed in earlier sections, the impacts of afforestation on the complete site C balance including soils are currently poorly understood. In particular, the impact on soil C stocks as a result of disturbance to the soil profile such as is recommended for establishment of new woodland is not well quantified (see Sections 4.4

and 4.5). While there are a number of studies that have investigated the impacts of woodland creation on ex-agricultural (normally unimproved) grazing land, their results are variable and difficult to generalise from. This analysis of published research is focused on studies of direct relevance to the UK.

The major determinants of the extent of soil C change under afforestation are soil type and prior land use, from which we can estimate present C stocks and predict long-term equilibrium values. The soil C content is the result of the net balance between large inputs and outputs, and in many soils there are very large C stocks (see Section 3.5). Small changes in the large C stock in soils are difficult to detect, but even small changes in the stock can result in relatively large changes in fluxes of CO_2 to the atmosphere and DOC to waters. It can be viewed simplistically that the inputs are derived from net photosynthesis and the outputs determined in large part by respiration. However, it is very complex to model carbon changes as the balance between photosynthesis and respiration is crucially dependent on the nutrient status and dynamics of the forest ecosystems, as well as on other variables, particularly temperature and precipitation (e.g. Hyvönen *et al.*, 2007). In addition, in peat soils with very high C stocks much of the C is in deep, relatively inactive layers, so fluxes may be small; for such site types information on peat depth is therefore required.

4.8.2 Soil C changes with afforestation on mineral soils

Increases in soil C stocks after afforestation are usually reported as being larger for lowland compared to upland areas due to a variety of factors including the prevalence of more intensive previous agricultural uses (Bateman and Lovett, 2000). Although no studies in the UK specifically have been published on the soil C consequences of the afforestation of mineral soils, there are some data on more gradual land-use change from arable to woodland. A much-quoted example is the long-running experiment at Rothamsted where the Geescroft arable land has reverted to acid soil woodland after being abandoned in 1886. Soil organic C (to 69 cm) increased from 29 tC ha^{-1} in the 1880s to 62 tC ha^{-1} in the 1980s (Poulton, 1996), with an average rate of accumulation in soil and litter of 0.38 tC ha^{-1} y^{-1}, or 1.4 tCO_2 ha^{-1} y^{-1} (Poulton, 2006). This shows the extent of change possible, but as this was natural regeneration of a 1.3 ha area, the rates are presumably much slower than would occur after deliberate tree planting. Studies of old-field succession over century or more timescales in the USA suggest soil C accumulation rates considerably less than this in the upper layers (0.34–0.55 tCO_2 ha^{-1} y^{-1}, Robertson and Vitousek, 1981; Robertson and Tiedje, 1984; Zak *et al.*, 1990), but given the differences in climate and the lack of information on changes below 15 cm, this is of limited relevance to the UK.

The recent Countryside Survey (2007) has determined average topsoil C stocks across different habitats, and these could be used to estimate the potential changes in soil C that could occur under afforestation (Table 4.18, Emmett *et al.*, 2010). However, these data are only for the topsoil (0–15 cm depth), which, although arguably most vulnerable to C changes, does not necessarily reflect total soil C stocks (see e.g. Figure 3.10). In addition, comparison of these large-scale aggregated values between habitats is not straightforward, as the stock values depend to a great extent on the soil types in each habitat (e.g. Figure 3.8) and the age structure of the various land uses and habitats in this GB-wide survey. Nevertheless, the results in Table 4.18 indicate that arable and horticultural land typically have topsoil C stocks that are around 60% of those in woodland. Simplistically, if broadleaved woodland was created on arable land and reached a new topsoil C content in 50–100 years, then this could represent an annual increase of 1–2 tCO_2 ha^{-1} y^{-1}, similar to the rate of increase observed at Geescroft (although those data are for 69 cm soil depth).

Table 4.18 Topsoil (0–15 cm) C stocks across broad habitat types in the Countryside Survey 2007. Data from Emmett *et al.* (2010, Table 2.5).

Habitat type	Mean topsoil C stocks	
	tC ha^{-1}	tCO_2 ha^{-1}
Broadleaved, mixed and yew woodland	72.9	268
Coniferous woodland	81.4	298
Arable and horticulture	47.3	173
Improved grassland	67.2	246
Neutral grassland	68.6	252
Acid grassland	90.6	332
Bracken	84.7	311
Dwarf shrub heath	89.9	330
Fen, marsh and swamp	82.8	304
Bog	85.6	314
All habitat types	69.3	254

There are few real data with which to compare such generalised approximations. Vesterdal, Ritter and Gundersen (2002) examined a 29 year chronosequence on ex-agricultural mineral soil (sandy loam) afforested with oak or Norway spruce in Denmark, and compared it with an adjacent 200-year-old deciduous stand and a permanent pasture. The old stand had a substantially higher soil C content (297 compared to 238 CO_2 ha^{-1}) than the pasture, implying a longer-term accumulation of approximately 0.3 tCO_2 ha^{-1} y^{-1}. In the newly planted sites, Vesterdal, Ritter and Gundersen (2002) found that after the first 8–10 years the forest floor C content increased, particularly under the spruce (1.1 tCO_2 ha^{-1} y^{-1}). However, in the soil the top 5 cm showed increased C content, while the lower layers (5–15, 15–25 cm) lost C, leading to an overall **loss** of soil C of 1.5 tCO_2 ha^{-1} y^{-1} under both species. The findings of Vesterdal, Ritter and Gundersen (2002) support an early study by Ovington (1956) of the effects of tree planting on an East Anglia heathland, where despite increases in litter C, the soil C was reduced in upper layers, compared to an unplanted site after 21 years (decline of approximately 0.6 tCO_2 ha^{-1} y^{-1}, average of Corsican pine, alder and birch stands).

An important study of soil and vegetation C stocks on mineral soils after afforestation in Ireland has recently been published by Black *et al.* (2009). This chronosequence approach examined a set of sites on surface water gley soils, previously rough grassland or pasture, now with Sitka spruce trees aged 9–47 years, high yield class (22–24). Ground preparation varied across sites from ripping and surface drains in the more recent sites to mouldboard ploughed in the older stands. While vegetation (mostly tree) C stocks clearly increased as expected with stand age (Table 4.19), there is no clear pattern in soil C stocks. Black *et al.* (2009) calculated that the difference in soil C stocks between 0 and 16-year-old stands represents an average soil C accumulation rate of 8.1–9.2 tCO_2 ha^{-1} y^{-1} (2.2–2.5 tC ha^{-1} y^{-1}); however, the 30-year-old stand C stock value suggests only a tenth of that rate (0.8 tCO_2 ha^{-1} y^{-1}), and the 45-year-old stand only 0.4 tCO_2 ha^{-1} y^{-1}. In part, the differences may be attributed to soil texture differences, degree of slope on each site and probably the detail of the original ground preparation methods. This illustrates the difficulties in chronosequence studies of finding completely comparable sites. Nevertheless, Black *et al.* (2009) have estimated that at their sites, with these high yield class stands, 31–43% of the GPP (total photosynthetic uptake, see Box 2.1) was allocated to below ground and soil through litter and photosynthate transfer, resulting in high soil C accumulation rates. Similar below-ground allocation **proportions** have been estimated in mature, lower yield class (YC 12) Sitka spruce on peaty gley soils in Harwood Forest in northern England (Zerva *et al.*, 2005).

Table 4.19 Vegetation and soil C stocks in a chronosequence of Sitka spruce afforestation sites in Ireland (from Black *et al.*, 2009, Table 5).

Stand age (years)	C stock (tCO_2 ha^{-1})	
	Vegetation	Soil[B]
0[A]	22	356 ± 50
9	72	430 ± 43
14	132[C]	n.d.
16	268	503 ± 102
22	436[C]	n.d.
30	703	382 ± 69
45	904	374 ± 57
47	629	752 ± 245

[A] Grassland; [B] ± s.e.m. shown; n.d. – not determined.
'Vegetation' is above- and below-ground trees and/or grass, plus litter and harvesting residue, except for [C] where litter was not determined.

In contrast, **total** below-ground C allocation for mature Sitka spruce on the peaty gley soil in Harwood Forest was estimated at 3.65 tCO_2 ha^{-1} y^{-1} (Zerva *et al.*, 2005), much smaller than the 18.0 tCO_2 ha^{-1} y^{-1} for the more productive Sitka stand in Ireland (Black *et al.*, 2009), largely because of reduced litter input. At both sites total soil respiration was approximately 15–18 tCO_2 ha^{-1} y^{-1}, resulting in net accumulation of soil C in the surface water gley soil but net losses from the peaty gley soil, over the first rotation. These contrasting examples emphasise the need to examine the whole C balance of forestry, and the importance of soil type and tree productivity in producing net C sequestration or C emission.

4.8.3 Soil C changes with afforestation on organic soils

The few investigations quantifying the C changes following afforestation tend to be restricted to integration across chronosequence data, rather than measurements taken through time. Afforestation on peatland could have significant impacts on the C balance through stimulating soil C loss due to drainage and ploughing (see section on site preparation, Section 4.4). Cumulative soil C loss from afforested peat soils has been estimated at about 20–25% of the total C in the peat (Harrison *et al.*, 1997; Jones *et al.*, 2000), and some modelling analysis has assumed even higher losses over long time periods (Bateman and Lovett, 2000). However, there is great variability and uncertainty in such estimates (see e.g. Reynolds, 2007), in part due to varying time periods studied and assumptions about equilibrium conditions. Furthermore, these estimates were derived from sites with very intensive ground preparation and drainage techniques which are likely to have caused substantial soil disturbance, and are not necessarily appropriate to later practices (see Morison *et al.*, 2010, p. 24).

Hargreaves, Milne and Cannell (2003) measured fluxes on a deep peat site in Scotland following ploughing for afforestation and found the soil became a net **source** of C peaking with an emission of approximately 14.6 tCO_2 ha^{-1} y^{-1} 2 years after planting. A site 8 and 9 years after planting was a net **sink** of carbon with a maximum uptake value of approximately 7.3 tCO_2 ha^{-1} y^{-1}. This result does not agree well with other studies of soils with lower C contents (M. Mencuccini, pers. comm.). Also a review of several peat land afforestation studies by Reynolds (2007) coupled to modelling of biomass C accumulation showed that net ecosystem productivity (NEP) was around 165 tCO_2 ha^{-1} over a 26-year period (average 6.3 tCO_2 ha^{-1} y^{-1}) including accounting for a soil C loss rate of 1.83 tCO_2 ha^{-1} y^{-1}.

Reynolds (2007) identified, in addition, the importance of non-CO_2 flux losses through stream water, as dissolved (DOC) or particulate organic matter (POC) (see also Section 4.3). A review by Hope, Billett and Cresser (1994) indicated that temperate forest catchments export slightly less DOC in rivers (121 kg CO_2 ha^{-1} y^{-1}) than moorland and grassland catchments (157 kg CO_2 ha^{-1} y^{-1}). Buckingham, Tipping and Hamilton-Taylor (2008) also found higher DOC flux in the topsoil of moorland compared to topsoil of forest ecosystems.

Zerva *et al.* (2005) examined the effects of afforestation of a peaty gley site in northern England upon soil C balances, and found that the number of the rotation was of importance. While the investigation lacked detailed information on early fluxes following afforestation, the underlying trend was that the first 40-year rotation resulted in a decrease in soil C of approximately 12.5 tCO_2e ha^{-1} y^{-1} with a 73% lower soil C content in the H layer[32] of afforested areas compared to that of unplanted moorland. They attributed this decline to the accelerated decomposition caused by the site preparation for drainage and planting. Subsequently, in the second rotation there was a substantial recovery of soil C.

4.8.4 GHG fluxes and peatland afforestation

Afforestation and drainage of peat leads to lowering of the water level and may increase decomposition and nutrient mineralisation and hence may lead to increase N_2O emissions. Similarly, application of fertiliser at early stages of afforestation (or high N deposition throughout) helps accelerate the decomposition of soil organic matter and, in turn, this may increase CO_2 emissions from soil microbial respiration, although this is not always the case (Glatzel *et al.*, 2008). The CO_2 emissions follow an optimum function, with highest rates at intermediate water table levels (Glatzel, Lemke and Gerold, 2006). Nykänen *et al.* (1998) indicated that lowering the water table by 10 cm would induce a 70% reduction in CH_4 emissions from fens and a 45% reduction from bogs. Studies in boreal peatland have shown that CH_4 emissions decreased (or the soil became a net sink) following drainage and afforestation (e.g. Maljanen *et al.*, 2003; von Arnold *et al.*, 2005c). Similarly, Mojeremane, Mencuccini and Rees (2010) studied the effects of site preparation for afforestation (effect of drainage, mounding and fertiliser application) on CH_4 fluxes from peaty gley soil at Harwood Forest, NE England (see Table 4.20), and indicated that the overall soil CH_4 emissions were significantly decreased by drainage (-64%) but increased by mounding and fertilisation (+41 and +44%, respectively).

Table 4.20 Mean CH_4 fluxes (also expressed as tCO_2e ha^{-1} y^{-1}) from the different site preparation treatments at Harwood Forest, measured over 2 years, from Mojeremane, Mencuccini and Rees (2010).

Treatment	CH_4 kg CH_4 ha^{-1} y^{-1}	CO_2e tCO_2e ha^{-1} y^{-1}
Drained	6.3	0.16
Undrained	17.6	0.44
Mounded	14.0	0.35
Unmounded	9.9	0.25
Fertilised	14.1	0.35
Unfertilised	9.8	0.25

Saari, Smolander and Martikainen (2004) investigated the practice of 'mounding' after felling (which is commonly used to promote seedling establishment) of a boreal forest (podzol soil with an organic horizon 2–5 cm thick). In the year immediately following mounding, the mounded plots oxidised 33% more CH_4 than the unmounded plots. This is because of the high capacity for CH_4 oxidation on the surface of the mound without an organic horizon to act as a diffusion barrier. However, in years two and three the converse occurred, with mounded plots oxidising 22% less than their unmounded counterparts, which cannot be adequately explained. The environment for the methanotrophs may have become unfavourable in subsequent years, possibly as a result of drying out of the mound.

Freeman, Lock and Reynolds (1993) simulated, experimentally, the effect of climate change-induced water

32. See Box 3.1 for explanation of soil layers.

table draw-down using undisturbed intact monoliths from a Welsh peatland. Lowering the water table decreased CH_4 efflux (maximum -80%) and increased both CO_2 and N_2O efflux (maximum 146 and 936%, respectively).

Byrne and Farrell (2005) studied the effect of afforestation on soil CO_2 emissions from undrained blanket peatland sites, sites that had been drained and established at 3, 19, 23, 27, 33 and 39 years previously, and from two clearfelled sites in Ireland. Generally, the average annual soil CO_2 emissions from the different sites showed no clear differences between the different sites (ages) or with soil type, and Byrne and Farrell (2005) suggested that afforestation does not always lead to an increase in soil CO_2 emissions. They also showed that soil CO_2 emissions at the most recently afforested site were approximately 6.2 $tCO_2e\ ha^{-1}\ y^{-1}$ where drainage had not lowered the water table, a substantially lower rate than that observed after planting by Hargreaves, Milne and Cannell (2003).

In a recent study from Finland, Mäkiranta *et al.* (2008) concluded that afforestation of previously cultivated peatlands lowered peat decomposition rates. Measurements of CO_2 release from soil showed that temperature was the main factor determining soil respiration rates, whereas water levels at these sites had little effect on the decomposition rates of organic material. Nevertheless, it was estimated that the effect of previous agricultural practices would lead to higher soil respiration in these areas than undisturbed forested peatland for decades after forests were reintroduced. Afforestation of those peatlands abandoned from cultivation or peat harvesting in Finland could reduce total CO_2e (based on measurements of CO_2, CH_4 and N_2O fluxes) at least during the first tree generation (Alm *et al.*, 2007).

4.8.5 Changes in C stocks in other vegetation

Afforestation may result in the removal of existing vegetation, either rapidly during ground preparation and planting, or slowly as the trees grow and suppress existing vegetation (or a mixture of both of these). Above-ground biomass stocks in existing heathland, moorland and rough pasture vegetation are quite well characterised and have been modelled (e.g. Armstrong *et al.*, 1997; Terry *et al.*, 2004). Clearly, stocks will differ depending on the vegetation type, the heather stand age (dependent on management, e.g. burning cycles, grazing), and site conditions. There are major differences between lowland heath and upland heather moorland. Gimmingham (1972) produced a summary table of above-ground biomass in contrasting heathlands (Table 4.21). As expected, they vary greatly with age, and can be as low as 1 or as much as 22 t dry weight ha^{-1}. There is less information on below-ground biomass, but at the Moor House site, perhaps typical of much upland moorland, biomass below ground was similar to that above ground (Gimmingham 1972, p. 145).

Table 4.21 Summary of above-ground biomass and C stocks of heathland and moorland in Britain, adapted from Gimmingham (1972, Table 18).

Location	odt ha^{-1}	$tCO_2\ ha^{-1}$
Dartmoor, age 2–18 years	3–22	5.5–40.3
Moor House, Pennines, modal age 8 years	13	23.8
Kincardineshire, 2–37 years	1–21	1.8–38.5

Litter amounts can vary from 20 to 80% of above-ground biomass, depending also on age (Gimmingham, 1972). Estimating conservatively that there is 50% biomass below ground and 30% additional litter biomass, and assuming that the biomass of heather is 50% C, implies that the C stock in vegetation and litter of heathlands and moorlands range from 3 to 72 $tCO_2\ ha^{-1}$, with a typical value of approximately 40 $tCO_2\ ha^{-1}$ (11 $tC\ ha^{-1}$). Model simulations of both upland *Calluna–Molinia* heathland and lowland *Calluna–Deschampsia* heathland under low-intensity 15-year management cycles also suggested maximum C stocks of approximately 35–40 $tCO_2\ ha^{-1}$ (biomass and litter, low N deposition scenario, Terry *et al.*, 2004).

Recent sampling for the ECOSSE report (Smith *et al.*, 2007, Section 2.2) of heather biomass from three upland Scottish sites (the John Miles Birch Plots) found that above-ground C stocks in 2005 were 80–92 $tCO_2\ ha^{-1}$. There were no data on below-ground biomass. Litter stocks at two sites were as large as or larger than above-ground vegetation (92 and 102 $tCO_2\ ha^{-1}$), but at a third site the litter C stock was only 16 $tCO_2\ ha^{-1}$ (Smith *et al.*, 2007). These results suggest heather biomass C stocks in excess of 180 $tCO_2\ ha^{-1}$, values which are four-fold larger than those quoted above. However, the plots were fenced heather 'control' plots, in a birch afforestation experiment established in 1980, so large herbivore grazing had been removed for 25 years, and there had been no burning. It is likely that these figures indicate the maximum C stocks in heather-dominated moorland.

It is evident from the above that even young tree stands contain substantially more C stock in the trees (see e.g. Figure 3.3) than the typical heather communities. Thus for any other than very low yield classes, by 8–15 years after planting the C stock in trees will have exceeded that in even dense, older stands of heathland or moorland vegetation. Clearly, heathlands with scrub vegetation will contain more C than heather stands alone, but there are few data available, and scrub density will be a key determinant, making it very difficult to characterise and generalise.

Vegetation C stocks in rough grassland and pasture will be lower than those for heather moorland, as there is no woody biomass, and therefore much less accumulation of C over time, except in the litter component when sites are ungrazed. Black *et al*. (2009) estimated the rough grassland biomass prior to afforestation of a surface water gley soil in Ireland as 6.2 tCO_2 ha^{-1} above ground, 2.2 below ground and 13.2 in litter (i.e. a total of 21.6 tCO_2 ha^{-1}). Other values of above-ground biomass (including litter) have been compiled by Holland (2001) and for upland grazed *Festuca/Agrostis* grasslands ranged from 2 to 18 tCO_2 ha^{-1}. For grazed *Nardus* and *Juncus* communities values reported ranged from 2 to 32 tCO_2 ha^{-1} (Holland, 2001[33]). These are values recorded during the growing season, and there is a large annual cycle, with much lower values in the winter, so annual average C stock may be only a half the values quoted. However, below-ground biomass is an important component in grasslands, and, although rarely reported, can equal or exceed the above-ground biomass. Thus, while there are few comprehensive data, it is unlikely that the long-term C stock is more than 30 tCO_2 ha^{-1} for these types of vegetation communities. While small compared to the C stocks in closed-canopy woodlands, they are sufficiently large to be important in the C balance in the early stages of woodland establishment, and their changes should be considered (Hargreaves, Milne and Cannell, 2003; Black *et al*., 2009).

4.9 Continuous cover and other management systems

Alternative management systems to patch clearfelling are being adopted in the UK as a means of developing uneven-aged stands, increasing the species diversity and improving the structural diversity of plantation forests. These management systems are often termed continuous cover forestry (CCF). CCF is considered a more sustainable forest management option as the silvicultural regimes employed favour natural regeneration of an understorey tree seedling layer, thereby avoiding the GHG emissions incurred in nursery stock production and reducing soil disturbance impacts, and likely soil C losses (Stokes and Kerr, 2009). The selection of appropriate sites is achieved through the consideration of soils, site classification and wind stability of the stand. Most transformation of even-aged stands to CCF is carried out on freely draining soil types, and in particular on brown earths, for which there is a more varied species range (including conifers such as larch, Scots pine, Douglas fir etc, and productive broadleaved species). These are also the types of soil with lowest rates of soil C loss after disturbance, which may limit the suggested C benefits of CCF practice. Research suggests that forest stands transformed to continuous cover via shelterwood or group selection silviculture may develop improved stability (Gardiner *et al*., 2005), which will obviously benefit long-term C stocks.

Few studies have been reported on the C balance of forest management regimes other than patch clearfell. Thornley and Cannell (2000) investigated the relative soil–plant C balances of a number of forest management regimes, using a model of the growth of Scots pine in a Scottish climate. They found that regular (yearly) interventions to remove 10 or 20% of above-ground biomass provided increased timber production (25 m^3 ha^{-1} y^{-1}) and C storage (200 to 240 tC ha^{-1}) over conventional (no-thin) clearfelling assessed over a 60-year rotation. The study did not account for the operational C costs associated with the various management scenarios, or the implications for crop stability and the consequent longer-term C storage. Seidl *et al*. (2007) assessed the effect of different silvicultural systems on C stocks in Norway spruce stands in Austria and found the largest C stocks in old-growth stands, followed by CCF stands, and the least in even-aged stands. The recent review by Stokes and Kerr (2009) on benefits of CCF under climate change concluded that although there were few studies on C sequestration under different CCF management conditions 'there is some evidence that CCF has the potential to accumulate more carbon than an even-aged woodland, but less than an "old-growth" stand' (Stokes and Kerr, 2009, p. 24), which agrees with the modelled results shown in Figure 3.3. Stokes and Kerr (2009, p. 25) also discuss the implications for C accumulation rates in stands when they are managed for individual tree growth, rather than a focus on mean tree maximum increment (see also Poore and Kerr, 2009). Development of C accounting models that can incorporate stand age-distributions and different individual growth rates, as well as species mixtures are essential to explore and maximise the benefits of such silvicultural approaches.

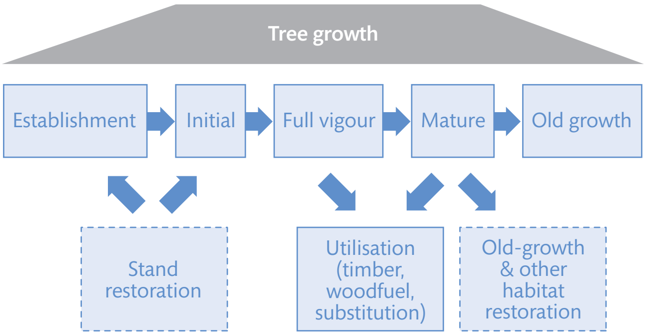
33. Another reference with similar biomass figures for upland vegetation is Milne et al. (2002).

5. Estimating C and GHG fluxes during the management cycle

5.1 Introduction

The information in the previous chapters of this report can be integrated to estimate GHG balances over a complete 'forest management cycle' (FMC), as shown in a simplified block diagram in Figure 5.1. In this part of the report the FMC phases that need to be considered are defined and the approach to the calculation of carbon and GHG balances is described. These calculations are illustrated for two management option examples based on Sitka spruce stands, then a set of results for a range of other options are presented. Finally, the application and interpretation of such results for the evaluation of management options is discussed.

Figure 5.1 Block diagram of the phases in the forest management cycle. The grey 'tree growth' shape indicates the temporal pattern of tree growth rate over the cycle; the dotted boxes indicate other possible management changes (not presently considered).



5.2 Phases in the forest management cycle

It is convenient to define the FMC as a set of phases described in Table 5.1; this is a refinement of the general cycle described in Section 3.2.

Table 5.1 The phases in the forest management cycle (FMC).

Phase name	Description
Establishment	Young trees are planted to create a new stand of trees, or replanted or regenerated to replenish an existing stand of trees after clearfell. In this study, the phase is defined as lasting for the first five growing seasons for the young trees after planting.
Initial	Once established, young trees grow from seedlings through the 'thicket phase' to reach the 'pole phase' and, generally, their annual yield increment (AI – see Symbols and abbreviations) increases during this period. The phase is defined here as lasting from age 5 up until (and including) the age of first thinning as specified in yield tables (the age at which the first thin would normally occur if dealing with a thinned stand). Unless early thinning is carried out, this phase would have zero or one thinning/extraction.
Full vigour	For an even-aged stand of trees, this is the period during which trees grow with the greatest rate of AI. For thinned stands, a number of thinning operations may occur during this phase. The phase is defined as lasting from the nominal age of first thinning until (and including) the time at which maximum mean annual increment (MMAI – see Symbols and abbreviations) occurs. For stands managed with a clearfell regime, clearfelling usually takes place during, or at the conclusion of, this phase. If clearfelling is delayed, then activity beyond MMAI falls into the next phase.
Mature	For an even-aged stand of trees this is the phase following attainment of MMAI during which the biomass accumulation rate and AI declines progressively from its maximum value. The phase is defined as lasting from the age at which MMAI is attained in an even-aged stand of trees up until the point at which mean annual increment (MAI) has dropped to 50% of the maximum value.
Old growth	If the stand is kept beyond the point that AI declines to half of the maximum, we define a phase of 'old growth', with a maximum of 200 years. As a general rule, stands in this phase will not be subject to active management apart from minimum intervention for the purposes of protection. Many such stands will be subject to external influences, e.g. windthrow, these are not modelled here.
Transformation	Stands in this phase are undergoing, or have undergone, transformation from an even-aged composition to a diverse structure of age and size classes. The phase is defined as lasting indefinitely from the point at which transformation of an even-aged stand starts.

For different particular management options the FMC can be defined by a set of phases, as exemplified in Table 5.2 for even-aged stands, continuous cover regime (see Glossary) and in-forest carbon sequestration.

Table 5.2 Example combination of forest management cycle phases for different management options.

Phases included	Even-aged	Continuous cover	In-forest C sequestration
Establishment	•	•	•
Initial	•	•	•
Full vigour	• (may end with clearfell)	•	•
Mature	• (end with clearfell if not done earlier)	• (if transformation does not start earlier)	•
Transformation		•	
Old growth			•

The management cycles for other management options, for example the conversion of an even-aged stand managed for production into a 'carbon reserve', can be defined using different combinations of the phases described in Table 5.1 (although the timing and details of what happens during a particular phase will change).

5.3 Estimation of forest C and GHG balances using CSORT

5.3.1 Overview

Following the precedents of Marland and Marland (1992), Matthews (1994), Marland and Schlamadinger (1995), Nabuurs (1996), Schlamadinger and Marland (1996a, b, c) and Broadmeadow and Matthews (2003), estimates of C and GHG balances for different management options, species and site conditions have been produced. The estimation procedure consists of calculating C stocks and flows of GHGs (principally CO_2, but including N_2O and CH_4) during each phase of the cycle, for each major component associated with the forestry system (see Figure 2.1) i.e.:

- trees
- debris and litter
- soil
- harvested wood products
- forest operations
- 'substitution', i.e. emissions reductions due to utilisation of material (fuel and building materials).

The contribution to the GHG balance of CH_4 and N_2O emissions from fossil fuel combustion during forest operations is included, but the natural fluxes of these to or from forests, other vegetation and soil are not included at this stage, due to their uncertainty (see Section 4.2). Although site conditions and species are key determinants of C stocks and sequestration potential of woodland, the species suitability and particular site conditions are not considered directly, only indirectly through the choice of yield class (YC) and soil type.

5.3.2 CSORT model description

The simulations have been undertaken using the Forest Research CSORT model. CSORT is the successor to the CARBINE model (Thompson and Matthews, 1989a, b) and has common features of structure and functionality with other forest carbon models such as C-FLOW (Dewar, 1990; Dewar and Cannell, 1992), and CO2FIX (Mohren *et al.*, 1999). The model is based on conventional yield models (e.g. Edwards and Christie, 1981), coupled to models of biomass allocation, carbon content, decomposition, soil carbon exchange, product utilisation and empirical data on the GHG balance of forestry operations. It also provides an estimate of harvested wood products (HWP), as a guide to cumulative carbon sinks. The scale on which CSORT works is the stand[34]; this is a key difference from CARBINE, which is a national-scale scenario analysis tool (Matthews, 1996; Matthews and Broadmeadow, 2009).

The forest biomass and management components in CSORT are shown in Figure 5.2 (below). The modelling involves a sequential approach in which an appropriate yield model is selected (dependent on species, yield class and spacing) and used to estimate biomass of the main tree components with the BSORT model. Wood density values (Lavers and Moore, 1983) convert wood volumes to dry weight, 50% of which is assumed to be carbon (Matthews, 1993). The biomass components are roots, stump, round-wood, saw-log, tips, branches and foliage, either as standing (living) biomass, in-forest debris and litter, or extracted material to be processed.

34 Note that C stock estimates from CSORT and BSORT do not therefore take into account any rides, open space or unplanted areas that are included in statistics on forest areas, for example.

BSORT model outline

From around age 20 onwards (the first entry in the yield tables), biomass in trees is calculated on an annual time step by using height, DBH and stem volume values obtained from the appropriate yield model (Edwards and Christie, 1981) as input data to the BSORT biomass model (Matthews and Duckworth, 2005). The published yield models report tabular values with a 5-year time step, but a dynamic algorithm-based yield model (known as M1) gives estimates on an annual time step, and the BSORT model is linked directly to M1 to calculate a sequence of annual biomass estimates from the time of the first entry in the yield table onwards. BSORT estimates biomass separately for the major tree components: roots, stumps, stems, branches, tips and foliage, based on the average tree, using relationships derived from McKay *et al.* (2003) and Matthews and Duckworth (2005). Examples of such output based on BSORT have already been presented (Figures 3.2–3.3). It should be noted that for the establishment and initial phases there are no yield table data, and estimates have to be produced by extrapolation as described later. In addition, there may be limited data on which to base yield tables at old stand ages, and estimates are therefore likely to be extrapolations for mature and old-growth forests. Examples of the relationships used to calculate biomass components in BSORT are given below; other relationships and parameter values have been recently detailed in the Forestry Commission *Woodland Carbon Code: Carbon Assessment Protocol* (Jenkins *et al.*, 2010).

Crown biomass (Bc) is a function of diameter at breast height (DBH), tree height (H) and number of trees (N_t):

$$Bc = (\alpha + \gamma\, \overline{DBH}^{\rho}\, H^{\phi})\, N_t$$

$\overline{DBH}$ is the quadratic mean DBH of all trees in the stand; H is the mean height of the trees in the stand, estimated from DBH and volume via tariff numbers and species-specific parameters. Different functions apply to DBH above and below 7 cm. In the <=7 cm case, the constant α is not required, nor is H, hence parameter ϕ is set to zero.

Figure 5.2 Schematic representation of the components within the CSORT model, with the outputs at the forest roadside.

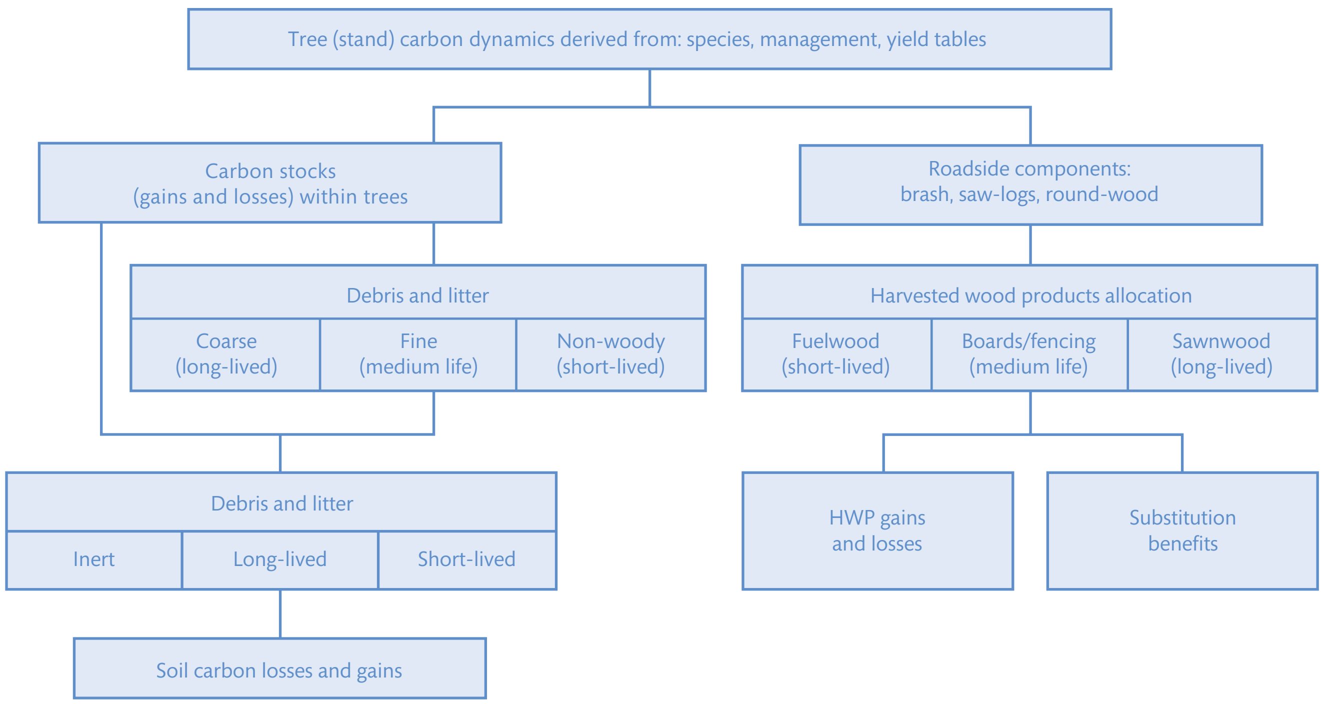


Table 5.3 Empirical coefficient values for calculation of crown and root biomass from DBH.

Species	Crown biomass								Root biomass	
	DBH <= 7 cm				DBH >7 cm					
	α	γ ($\times 10^{-4}$)	ρ	ϕ	α	γ ($\times 10^{-6}$)	ρ	ϕ	γ ($\times 10^{-5}$)	ρ
Sitka spruce	0	5.22	1.459	0	0.00607	9.58	2.5578	0	1.115	2.6836
Oak	0	2.16	2	0	0	54.2	2.3501	-1.022	14.9	2.12

The parameter values used here are in Table 5.3; values for some other common species are given in Jenkins *et al.* (2010, p. 47) reproduced from information in McKay *et al.* (2003):

Since crown biomass contains both branch and foliage, foliage biomass, *Bf*, is separated out as:

For broadleaves and larch: $Bf = 0.0639 - 0.06391 \times 0.1711^{Bc}$
For others species: $Bf = 0.2226 - 0.22265 \times 0.2393^{Bc}$

Root biomass (*Br*) is related to DBH by: $Br = (\gamma\ \overline{DBH}^{\rho})\ N_t$

where γ and ρ are parameters given in Table 5.3.

Stem biomass (*Bs*) is given by: $Bs = \rho V$

Where ρ is the nominal specific gravity of the species, and V is the volume. For Sitka spruce ρ = 0.33, and for oak ρ = 0.56; values for many other common species are given in Jenkins *et al.* (2010, p. 46).

Tips are the unmerchantable amount of the stem (i.e. of diameter less than 7 cm). The volume of these are estimated and then converted to biomass in the same way as *Bs*.

Stumps are also derived from stem biomass, based on the cutting height of the tree.

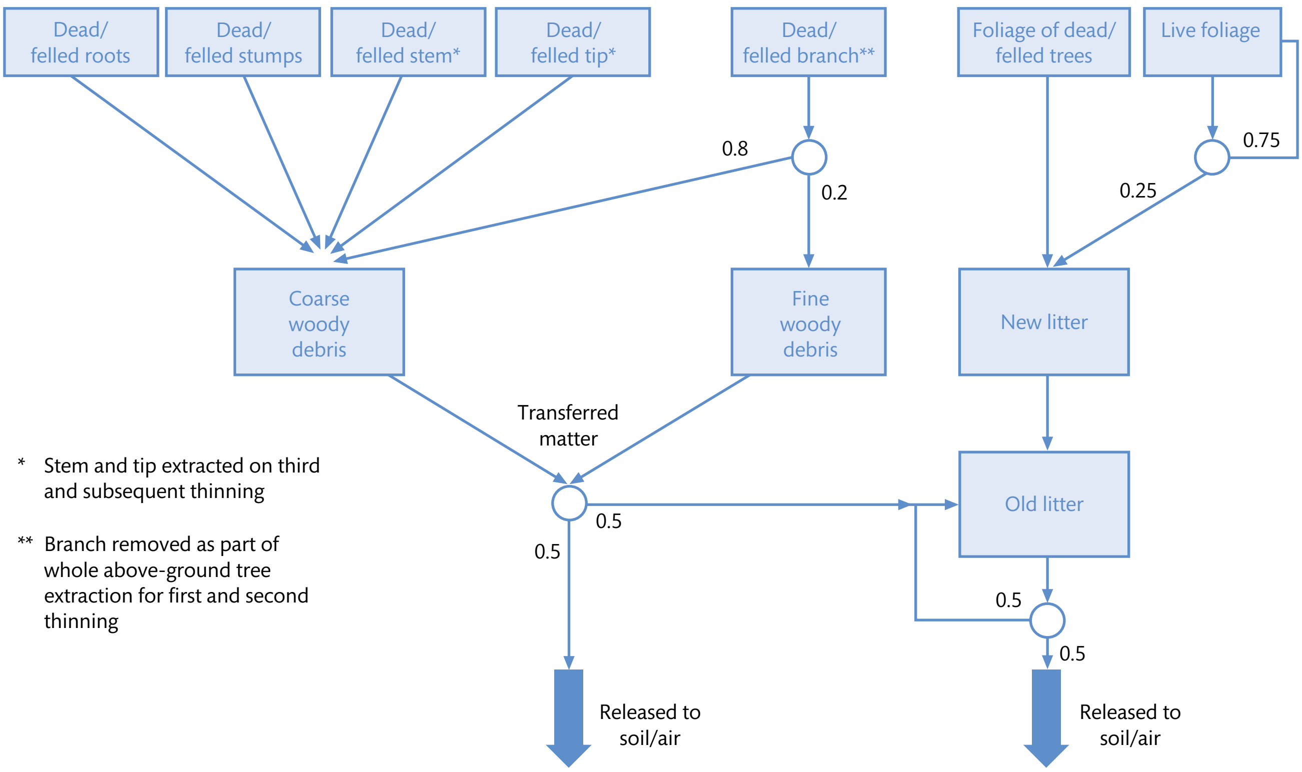
Debris and litter

Carbon in debris accumulates in the forest as dead trees, material left behind after management activities (e.g. after thinning), or as senesced foliage (with the exception of foliage extracted as part of management such as within a brash-bale).

Three separate debris and litter stocks are identified in CSORT: coarse woody debris, fine woody debris and non-woody debris, in order to allow for different residence times and decomposition rates in each stock (see Section 3.4). All foliage is deemed to be non-woody debris as new or old litter; 20% of branch matter is classed as fine woody debris and all remaining material is coarse woody debris. Carbon was assumed to enter these three stocks either as material from dead trees or left behind as 'conversion losses' from harvested trees, or during foliage turnover, as summarised in Figure 5.3. These assumptions vary for other examples, depending on details such as the tree species involved. In the examples presented, broadleaved species are assumed deciduous with 100% leaf loss during the year, for conifers each year 25% of the needle biomass enters the new litter pool. This can be modified if necessary, for example the conifer larch can be set to lose 100% of foliage at the end of the year.

When a given C amount enters one of these debris stocks, it is assumed to have a characteristic residence time during

Figure 5.3 Flow diagram showing C transfer from dead and felled material into the woody debris and non-woody litter pools. Figures shown for the foliage proportions are for conifers assuming 25% turnover per year.



which the C quantity remains constant, after which the C is progressively transferred out of the stock, over a fixed decay period, either to another stock or by CO_2 emission, with the rate determined by transfer coefficients. The decay function for the woody debris may be specified as either linear or exponential (Figure 5.4), though only a linear function is used in the examples here. The decay of non-woody debris is a fixed proportion of material remaining.

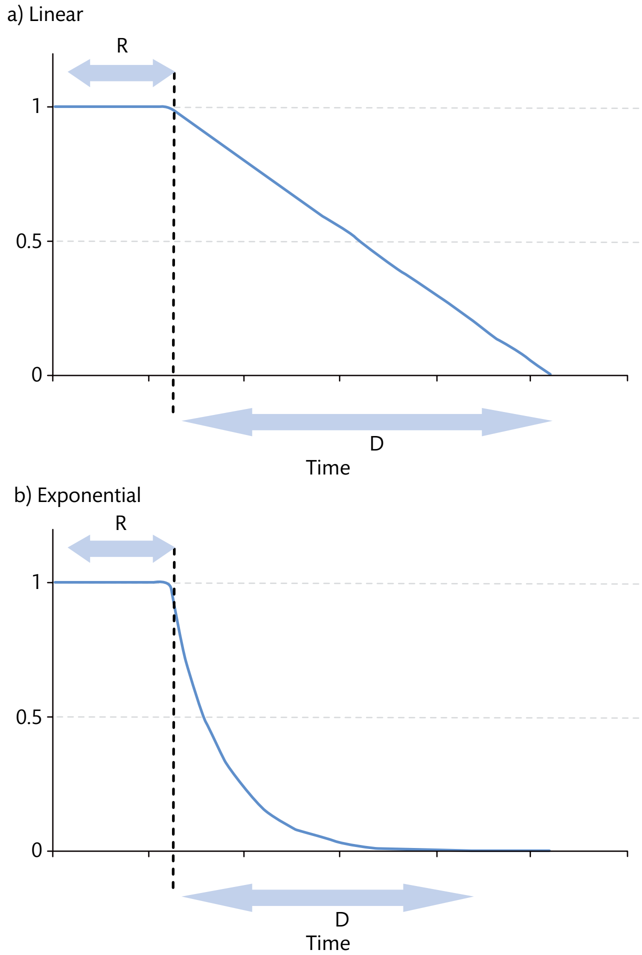
The decomposition of each of the three pools follows the patterns:

- Coarse woody debris remains intact for 2 years, before linearly decomposing over the next 20 years; on decomposition, the matter is transferred to the non-woody debris pool.
- Fine woody debris is treated in a similar way, but remaining intact for 1 year.
- Non-woody debris is assumed to remain intact for 1 year, and then decomposes, releasing CO_2, at the rate of 50% per year.

Decomposition rates can be affected significantly by variations in temperature, moisture content and litter quality (Section 3.4, Coûteaux, Bottner and Berg, 1995), but no allowance is currently made for local variation in environmental conditions in this model.

When thinnings and clearfell are carried out, different proportions and components of biomass are removed depending on whether it is the first thinning, any further thinning or clearfell. This is allowed for by specifying a matrix describing the percentage of biomass from each component that is extracted during each of these management activities (Table 5.4). Material not extracted is added to the debris pools.

Figure 5.4 Proportion remaining in linear and exponential decay functions. R denotes the residence time, and D denotes the decay period.



Soil C changes

The modelling of soil C dynamics was difficult, because of the limited available data and results, as described earlier. Much also depends on the level of disturbance of the soil during tree establishment, and subsequent

Table 5.4 Matrix showing the percentage of different biomass components removed from the forest during different management activities. Early thin activities are assumed to harvest above-ground whole trees.

Component	Conifers			Broadleaves		
	Early thin	Subsequent thin	Clearfell	Early thin	Subsequent thin	Clearfell
Roots	0	0	0	0	0	0
Stump	0	0	0	0	0	0
Round-wood[A]	100	100	100	100	100	100
Saw-log[B]	100	100	100	100	100	100
Branches	100	0	60	100	0	60
Tips[C]	100	0	60	100	0	60
Foliage	0	0	50	35	0	35

[A] Round-wood: minimum top diameter of 7 cm overbark and length of 3 m.
[B] Saw-logs: minimum top diameter of 18 cm overbark for conifers and 25 cm for broadleaf species, with a length of 3 m.
[C] Tips are upper stem parts <7 cm in diameter.

forestry operations. Therefore, the model is presently a simple, empirical one assuming three C pools: inert C, slow changing C and fast changing C. Although relatively simple, this modelling approach is established (FAO, 2004) and requires substantially less parameterisation than complex models. Changes in the soil C pools occur due to soil respiration (CO_2 emissions) and when C is transferred from the standing biomass and debris (through senescence and decomposition) to the fast and slow changing soil C pools. The model is described in detail in Appendix 6.

More complex soil carbon models exist (e.g. ROTH-C: Coleman and Jenkinson, 1999; Yasso: Liski *et al.*, 2005; ForCent: Parton *et al.*, 2010) but these are major models in their own right, and would need significant additional parameterisation and climate input data. Furthermore, it is unclear if such models can simulate likely effects of management change in forests without being too site specific.

Forest operations

Operational fossil fuel use in establishing, maintaining and harvesting trees (including building and maintaining roads, herbicide use etc) are all considered and accounted for. Emissions associated with such operations include those from manufacture, servicing and maintenance, and fuel costs (Whittaker, 2009, pers. comm.; MTRU, 2007).

Emissions from use of diesel: For forestry operations involving use of machinery, diesel consumption values are converted to energy consumption using appropriate density and calorific values (0.853 kg l^{-1} and 45.46 MJ kg^{-1}, Matthews *et al.*, 1994; resulting value of 38.78 MJ l^{-1}) and emissions factors from Elsayed, Matthews and Mortimer (2003) are used to calculate the CO_2, CH_4 and N_2O emissions. Two types of emission are identified: those arising **directly** from combustion of the fuel and those arising **indirectly** from the mining, refining and transport of the fuel to the point of consumption. The conversion factors for both the direct and indirect emissions are given in Table 5.5.

Table 5.5 Factors for estimation of emissions from consumption of diesel fuel.

Greenhouse gas	Emissions factor (note different functional units)	
	Direct (per kg fuel)	Indirect (per MJ direct energy)
Carbon dioxide (kg CO_2)	3.119	8.1×10^{-3}
Methane (kg CH_4)	27.3×10^{-6}	21.0×10^{-6}
Nitrous oxide (kg N_2O)	25.6×10^{-6}	26.0×10^{-9}

The estimates of direct and indirect emissions are summed using CO_2 equivalents (see Box 1.1 for details), to estimate the total emissions from diesel fuel consumption during each specific operation (see Appendix 7).

Harvesting: The discussion of harvesting operations in Section 4.6 suggested that these typically involve diesel fuel consumption of about 1.55 litres per m^3 of felled timber, with a further 0.9 litres per m^3 for forwarding to roadside.

Forest roads: Two types of road are distinguished, heavy and light duty, although emissions from construction of these roads are the same. The data reported here for roads are based on Whittaker *et al.* (2008). The density of each type of road is taken as a UK forest average 0.006 km ha^{-1} and 0.01 km ha^{-1} for the heavy and light duty roads, respectively. Emissions associated with road construction will vary with specification, terrain etc. CSORT assumes a formation construction with crushed rock only and emissions of 42.1 tCO_2e km^{-1} for acquiring and laying aggregate. Forest roads are assumed to be regularly maintained throughout the life of the stand. Heavy duty roads are re-graded yearly and re-surfaced every 10 years, regardless of any other operation carried out in the forest. Maintenance of the light duty roads is less intensive; after construction no maintenance is carried out until the time of first thin, when they are re-surfaced. The roads are then maintained by re-grading, except every third thin when a re-surfacing is carried out.

Harvested wood products

Harvested wood products are modelled using a similar approach to that adopted in the Forest Research C accounting model CARBINE. Estimates of harvested biomass (by tree component) in thinnings or at clearfell were obtained from M1 and BSORT. Table 5.4 above shows the percentage of each tree component that is removed at thinnings or clearfell. This is segregated into four types: roadside chippings, brash-bale, round-wood and saw-log (defined in Table 5.4). The extracted material is assumed to be transported to a processing plant using one-way trips of fully laden 44-tonne-lorry distances of:

saw-logs:	100 km
round-wood:	150 km
brash bales:	200 km
chipped material:	150 km

The extracted materials produce three groupings of wood product:

- **Saw-log timber material** – comprises high grade timber which has a fairly long lifespan; saw-logs are the only extracted materials that contribute to structural timber.
- **Processed wood products** – largely manufactured board material such as chipboard, MDF etc. This category also includes small timber usage such as fence posts. The lifespan is shorter than the saw-log timber group and the products are made from round-wood and some saw-log material.
- **Fuel material** – this category has a short lifespan, and consists of forest chippings, brash-bale material, and a substantial amount of round-wood and by-product/off-cuts from saw-logs and round-wood.

The C in these wood products is then allocated to one of three broadly defined end-use materials, as shown in Figure 5.5. First and second thinnings were assumed to produce woodfuel. Subsequent thinnings involve conversion of trees into round-wood and saw-log material. As each harvested wood product is processed, the waste products from one process are reallocated to another stage; for example, timber off-cuts from the saw-mill may enter the processed wood chain, or be used as fuel material.

Table 5.6 gives 'allocation coefficients' (A–F), which are the percentages of each product arising from each process (E. Mackie, pers. comm., Forestry Industry Council of Great Britain, 1998). These are also labelled on Figure 5.5 to show the flow of raw forest materials to end-use products.

As with litter components, the C in these wood product stocks was assumed to have a characteristic residence time (Table 5.7), during which the quantity remained constant, after which it is progressively transferred out of the stock according to an exponential or linear decay process (see Figure 5.4), and emitted to the atmosphere as CO_2 during the 'decay period'. No account is taken of possible secondary uses of harvested wood or recycling which will tend to underestimate a product's lifespan. The residence times are largely based on the usage of the end-product. BRE (2006) suggested that buildings have a lifespan of 50 years, and internal products, such as fittings, 20 years. Since we consider only domestic buildings, we assume a slightly longer duration. All emissions of C from harvested wood products are assumed to be as CO_2, which is an oversimplification, as material in landfill will release CH_4 as well.

Table 5.6 Percentages of each end-use product emerging from each process.

Factor	Process	Destination	Broadleaf	Conifer
A	Saw-mill	Timber	55	55
B	Saw-mill	Processed wood	25	18
C	Saw-mill	Fuel	20	27
D	Processed wood	Board/posts	17	70
E	Processed wood	Fuel	83	30
F	Fuelwood	Fuel	100	100

Note: These proportions are based on knowledge of the products and efficiency of each step, current market conditions and Forestry Industry Council of Great Britain (1998).

Figure 5.5 Flow-chart of harvested forest products to end-use material. Allocation coefficients A–F are given in Table 5.6.

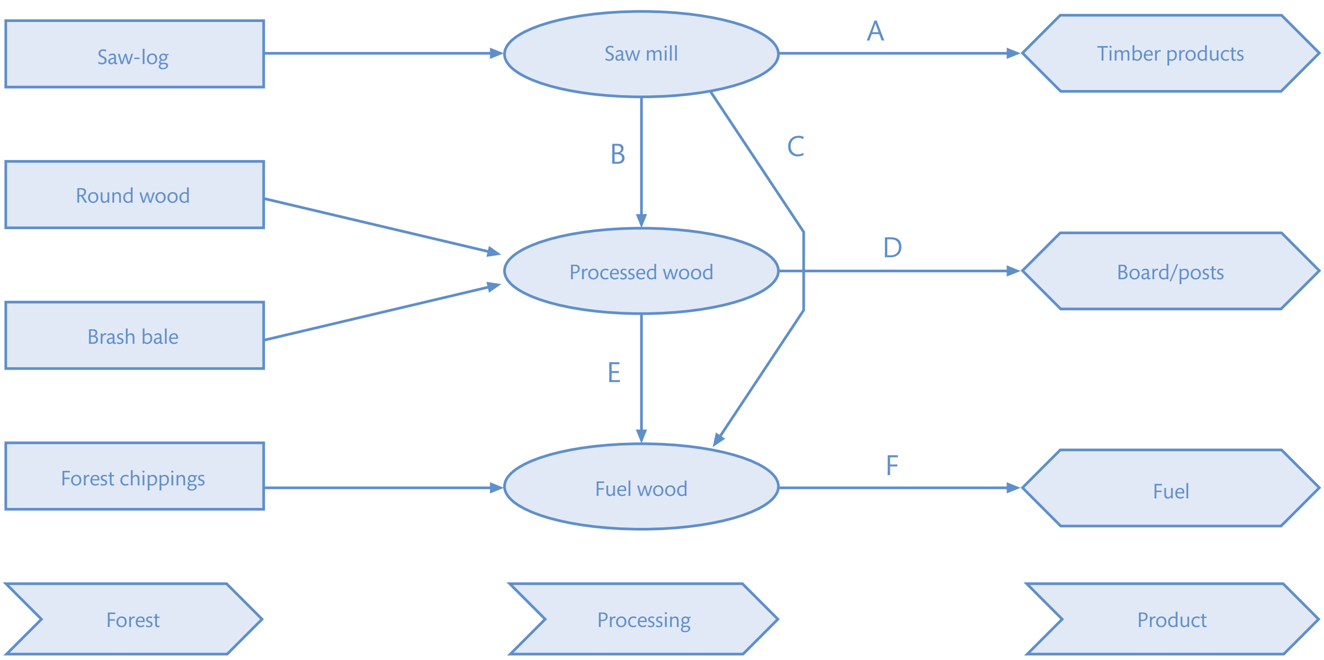


Table 5.7 Decay functions, residence times and decay periods of harvested wood products.

Product	Decay function	Residence time (years)	Decay periods* (years)
Timber products	Exponential	60	30
Board material	Exponential	30	20
Fuel	Linear	0	1

*In exponential decay, decay period is the length of time taken for 95% of the product to decay. In a linear model, this is the total duration.

Processing emissions and substitution

The manufacture of primary wood products involves processing emissions; there is also substitution, as discussed in Section 3.7. Both 'direct' or 'fuel substitution' and 'indirect' or 'building material substitution' are accounted for in the CSORT model.

Woodfuel processing emissions and substitution: The processing of raw harvested wood into chips suitable for utilisation as fuel is assumed to require the consumption of 5 litres of diesel fuel per oven-dried tonne (odt) of material when chipping at roadside, and 2.75 litres of diesel fuel per odt when processing brash bales at a mill. Additional processing of wood chips to achieve a consistent product is assumed to involve emissions of 0.0025 tC per odt (0.0092 tCO_2 per odt) when co-firing and 0.01383 tC per odt (0.0507 tCO_2 per odt) to produce wood chips for utilisation in small-scale heating systems. Production of wood pellets is assumed to involve slightly greater emissions of 0.03136 tC per odt (0.115 tCO_2 per odt). These emissions factors are based on interpretation of results and systems analysis presented in a number of sources, notably Elsayed, Matthews and Mortimer (2003), SDC (2005), Galbraith *et al.*, (2006) and Szendrődi (2006). For the calculations used in this report, it has been assumed that all chipped forest material is utilised for co-firing with coal for electricity generation and in small-scale heat systems. A fixed proportion of 20% of harvested wood fuel by mass is assumed to be used for co-firing, with the remainder used for small-scale heat. The estimated avoided emissions are based on results reported by Szendrődi (2006) for delivered energy and emissions reductions due to utilisation of wood chips for co-firing, or for use in small-scale heating systems, substituting for a mix of possible fossil fuel heating options (see Table 5.8). The assumed emissions avoided when co-firing wood with coal have also been estimated from the results of Szendrődi (2006) at 0.154 tC per odt (0.565 tCO_2 per odt). These results for emissions avoided through utilisation of wood for co-firing make interesting comparison with the results in Table 5.8 for small-scale heat. However, such results must be interpreted with great care. The useful energy delivered by the co-firing system (the output of electricity) is much smaller than for small-scale heating systems when expressed on the basis of 'per odt' of input wood. On the other hand, electricity has a different range of applications when compared with heat energy. Thus simplistic comparisons of emissions (avoidance) factors can be difficult to interpret and potentially misleading. Generally it is important to consider such results expressed with respect to several key functional units, for example per odt harvested wood, per GJ of delivered energy and per ha of forest required to deliver 1 GJ of energy.

Table 5.8 Proportions of fossil fuel energy mixture supplying heating in the UK and carbon emissions substituted by each odt of wood fuel material.

Substituted material	Proportion of energy supplied as heat	C emissions avoided per odt wood (tC)
Coal	0.06	0.557
Oil	0.30	0.448
Gas	0.40	0.268
Electricity	0.24	0.665

Thus, the total substituted emissions per odt wood fuel are estimated as:

$$(0.06 \times 0.557) + (0.30 \times 0.448) + (0.40 \times 0.268) + (0.24 \times 0.665)\ tC = 0.435\ tC\ odt^{-1}\ (1.59\ tCO_2e\ odt^{-1})$$

Building material processing emissions and substitution: The estimation of emissions associated with processing raw timber into primary wood products in use is subject to considerable uncertainty for a number of reasons. These include the many possible routes by which harvested wood may pass through processing chains, variation in processing technologies and in sources of primary energy and feedstocks consumed during manufacture, and other factors such as variation in transport distances. Also, life-cycle analysis (LCA) results are often difficult to interpret because of variation in details of methodology, lack of transparency in calculations and the risks associated in transferring results produced for a specific product manufactured in a particular country to more general application. The requirement for more thorough and systematic review and development of LCA results applicable to UK conditions has already been highlighted.

Currently, the calculations made by CSORT are based on a limited review of readily available relevant LCA literature and results (BRE, 2006; ECCM, 2006; Frühwald, 2007; Wilson, 2010). Many of these results are presented for functional units relevant to end use, for example emissions associated with house construction as considered in the report of ECCM (2006) and summarised in Table 5.9.

Table 5.9 GHG emissions due to house construction for 'typical' and 'increased use of timber'.

Type of building	Typical practice (Scotland) tCO_2	Increased timber content tCO_2e	CO_2e % saving
2 bed semi	12.2	3.1	75
3 bed detached	16.8	2.4	86
4 storey (8) flats	128.3	21.8	83
Average			81

For use in CSORT, it has been necessary to analyse the LCA studies referred to above to derive factors for processing emissions and also emissions avoided through substitution expressed per 'unit' of wood in primary products, as summarised in Table 5.10.

Table 5.10 Factors for processing emissions and emissions avoided due to consumption of selected primary wood products.

Product type	Functional unit	Processing emissions (tC)	Emissions avoided (substitution, tC)
Sawn timber[A]	m^3	0.0368	0.18 (0.16–0.7)
	odt	0.09	0.46 (0.4–1.7)
Particle board	odt	0.167	0.464
Medium density fibreboard[B]	m^3	0.169	-
	odt	0.23	-

[A] Excluding 'upstream' emissions, e.g. due to forest operations.
[B] Including 'upstream' emissions.

It must be emphasised that the factors given in Table 5.10 are provisional and further review and analysis is required. The factors for emissions avoided do not account for the C stocks actually contained in wood-based products, which are frequently included in LCA studies (see discussion in Section 3.7). Their exclusion here is deliberate since CSORT accounts for C stocks in harvested wood products separately. The range given in Table 5.10 for emissions avoided due to consumption of sawn timber indicates the sensitivity of such results to assumptions about the type of alternative material being substituted (e.g. concrete, steel, bricks).

5.4 CSORT model outputs

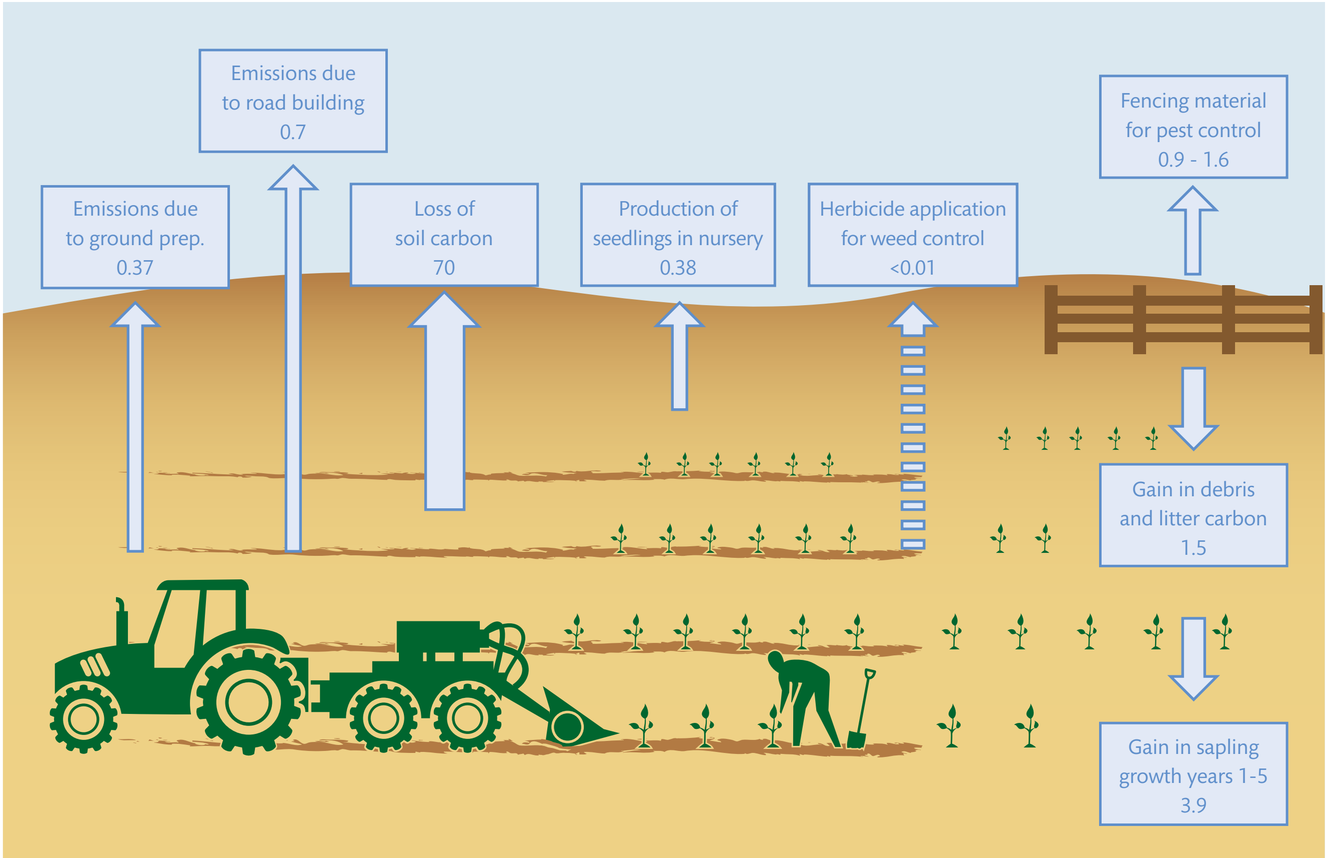
The CSORT model has been used to estimate the C and GHG balances for many different forest management options and situations. One example is described in detail below for the different phases of a new stand of yield class 12 (YC 12) Sitka spruce planted at 2 m spacing on a peaty gley soil previously under grassland. Two management options are considered: the first involving planting the Sitka spruce and then managing the stand as a 'carbon reserve', with minimum intervention ('MinI option'), the second involving management of the stand for timber production through thinning and clearfelling on a 50-year rotation ('T&F option'). For the MinI option, there is no thinning or harvesting, and thus no forest operations (including no road maintenance) after establishment and no substitution benefit. In the T&F option management is carried out according to normal prescription, including the recommended regime of 'management table' recommendations (Edwards and Christie, 1981), and each phase has various forest operations. It should be noted that peaty gley has a high initial soil C stock. The time course for less organic soils may be substantially different (see Section 4.8).

A full calculation of the GHG balance of a forest management cycle should account for the change in C stocks in all vegetation present, not just the trees. However, as discussed in Section 4.8, C stocks in heather moorland vegetation are typically around 40 $tCO_2\ ha^{-1}$ and in grassland typically less than 30 $tCO_2\ ha^{-1}$, considerably smaller than that in established stands of trees. Therefore its contribution is ignored here. During establishment of trees in the creation of new woodland, when other vegetation is being replaced by young trees, the changes in this component of the C stock could be significant. However, for the particular FMCs considered here, the contribution will be small, so it is not included in the current model. In principle, estimates of changes in other vegetation could be included in a later version. Similarly, the GHG balance of any non-forest understorey is presently not considered.

5.4.1 The establishment phase

The establishment phase is taken as the period from zero to 5 years. Figure 5.6 summarises pictorially the main contributions to the GHG balance during this phase, which is the same for both the MinI and T&F options.

Figure 5.6 Illustration of contributions to GHG balance during the establishment phase (0–5 years). For both the minimum intervention (MinI) and thinning and clearfell (T&F) management options. All values in $tCO_2e\ ha^{-1}$ over the phase. Sitka spruce establishment on pasture, peaty gley soil, 2.0 m spacing, YC 12. Soil losses will change substantially, depending on degree of ground disturbance.



Trees

For the examples here, Sitka spruce seedlings are assumed to be planted at 2 m spacing, to achieve an initial planting density of 2500 trees per hectare. Seedling biomass is small: a nursery-grown Sitka spruce seedling, ready for planting, typically has a biomass of 15 oven dry grams (R. Jinks, FR, pers. comm.). Thus, the C in seedlings planted at about 2 m spacing is:

$2500 \times 15 \times 10^{-6}$ t x 50% = 0.01875 tC ha^{-1} or 0.069 $tCO_2\ ha^{-1}$.

The quantity is very small, and therefore is ignored here.

During the 5-year establishment phase the seedlings grow, accumulating biomass and increasing the tree C stock. However, early growth is not described in standard yield tables, which are for established stands, so the CSORT model calculates the annual change in tree C stocks during the establishment phase by extrapolating a growth curve for total biomass. The method does not distinguish between the biomass components in trees (stems, foliage and roots), in order not to accumulate errors in curve fitting to each component. In practice, this will result in some underestimate of debris pools but, given the small biomass at this stage, the effect is not significant. It should be noted that there is little suitable data to test this approach as the data that exist are very site and species specific.

The establishment phase growth curve was generated by fitting a two-part spline: based on a linear regression projected backwards from initial biomass estimates, meeting an exponential curve commencing from time zero. Essentially, tree yield values derived for trees at the initial start point of the yield tables and the BSORT biomass model were extrapolated backwards to earlier ages. In the example of Sitka spruce YC 12, planted at 2 m spacing, the first thin is recommended at age 25. Using the M1 model (see Section 5.3.2) we obtain annual yield table growth values from age 20 onwards. We use the first two estimated total biomass values from ages T_1 and T_2 (in this case, T_1 = age 20 years: biomass = 67.4 odt ha^{-1} and T_2 = age 21 years: biomass = 75.3 odt ha^{-1}) to derive a linear function which is used to estimate total biomass down to an 'intersection point', T_x (age 15 years, in this example). From zero to the intersection point we use an exponential curve. Thus biomass is approximated by:

biomass = A + B.age $\quad T_x < age < T_1$
biomass = $\alpha\ (\beta^{age} - 1) \quad age < T_x$

where age is the stand age in years, α and β are parameters from the exponential curve (this implicitly assumes zero biomass at age zero), A and B are the coefficients of the linear regression fitted through the biomass estimates at times T_1 and T_2, and T_x is evaluated when fitting the spline.

Clearly such an extrapolation from later phases of stand development can only provide a first estimate of tree C stocks and stock changes during the establishment phase. There is a need for better information about the changes in tree C stocks during this establishment phase.

In the example here, by the end of the fifth growing season, the C stock in trees was estimated at 3.9 $tCO_2\ ha^{-1}$, giving an average CO_2 sequestration rate of 0.77 $tCO_2\ ha^{-1}\ y^{-1}$.

Debris and litter

For the establishment phase, a spline fitting approach is also used to estimate debris and litter C stocks. Initial C in the debris and litter pools was assumed to be zero. The possible contribution to the debris and litter pools due to removal of other vegetation (see above) is ignored. In other case studies, for example clearing of scrub woodland to create high forest, it would be necessary to account for the impact of the initial clearance on the debris and litter pools.

In the example here, by age 5, the C stock was estimated to be 1.5 $tCO_2\ ha^{-1}$, giving an average net accumulation rate for the phase of 0.3 $tCO_2\ ha^{-1}\ y^{-1}$.

Soil

A simple approach is taken to estimation of initial soil C stocks and changes due to stand establishment. Based on the BioSoil data (Table 3.8) peaty gley soil under forest is assigned a C stock of 1174 $tCO_2\ ha^{-1}$. It is assumed that under grassland, carbon content is approximately 80% of that in woodland, and the initial soil carbon stock = 250 $tC\ ha^{-1}$ (917 $tCO_2\ ha^{-1}$) before stand establishment. The loss of carbon from forest soils due to removal of existing vegetation and ground disturbance can be highly variable, reaching up to 25% (Harrison *et al.*, 1997; see Section 4.4), but such high values probably occur after a long period and only where deep ploughing and extensive drainage have been carried out. Here it is assumed that the initial impact is a 5% loss, giving in this example an immediate loss of 30 $tCO_2\ ha^{-1}$. Further losses occur throughout the establishment phase and beyond, increasing the total soil loss. At the end of the establishment phase (age 5) 70 $tCO_2\ ha^{-1}$ has been lost at an annual average rate of 14 $tCO_2\ ha^{-1}\ y^{-1}$.

There is considerable uncertainty in these assumptions and with the simplistic approach taken, so further work is required. Specific assumptions about initial soil C stock and changes in response to stand establishment will depend on soil type, previous vegetation, land use, the number of trees being established, and the method of ground preparation and establishment, as discussed in Section 4.4.

Harvested wood products and substitution

The initial C stock in harvested wood products attributable to the stand being created was assumed to be zero. For the two example management options being considered, there is no harvesting of wood during the establishment phase, so the stock of carbon in wood products remained at zero. Similarly, there is no substitution.

Forest operations

During the establishment phase the operations accounted for are:

- ground preparation
- tree planting
- fertilising
- weed control
- pest control
- road creation

More details of the operations involved and the GHG emissions incurred can be found in Appendix 7. A number of other relatively minor operations (e.g. initial site survey) were not considered.

Site preparation

Initial ground preparation: The information in Section 4.4 shows that ground preparation can involve a wide range of methods with very different machinery use and associated GHG emissions. In the following examples it is assumed that the initial site is afforestation of pasture, grassland or arable, thus ploughing is used involving diesel consumption of approximately 21 $l\ ha^{-1}$. Ground preparation at sites requiring excavators and mounding will use higher amounts of fuel (see Table 4.9). Using appropriate density and calorific values (see above) the direct energy consumed during ground preparation is estimated as:

21 l ha^{-1} x 0.853 kg l^{-1} x 45.46 MJ kg^{-1} = 814.5 MJ ha^{-1}.

Using the GHG emissions factors (Table 5.5) gives an estimate for the total emissions arising from diesel consumption during ground preparation of 64.4 kg CO_2e ha^{-1} (0.064 tCO_2e ha^{-1}), although CH_4 and N_2O emissions contribute less than 1% to the total.

Pest control: The main means of pest control usually involves use of a fence or tree shelters and fencing is assumed in CSORT. The emissions associated with construction of fencing materials will vary depending on the fence design and the area and shape of the stand being protected. In CSORT, a nominal rectangular shape and an area of 5 ha is used for scaling purposes. The volume of wood required for fence posts is set to a minimum amount per meter of fence. The emissions associated with the fence posts are indirect, that is, C in the fence itself is not included as this is accounted for in the utilisation of harvested wood products, but the emissions associated with transport to the forest and erecting them are counted. It is assumed that steel wire fencing is used. Elsayed, Matthews and Mortimer (2003) suggest that emissions due to use of materials in fences as part of establishment are typically 0.9–1.6 tCO_2e ha^{-1}, depending on the shape and area enclosed. After erection, fencing is assumed to be un-maintained as its primary purpose is to protect the crops during the establishment phase. If more than one rotation is simulated, occasional replacement/reinstatement of fences is allowed for.

Construction of roads: It is assumed that new forests will need construction of forest roads. Although emissions from road construction are the same, two types of road are distinguished: heavy and light duty. The data reported here for roads are based on Whittaker *et al.* (2008). The density of each type of road is taken as a UK forest average, 0.006 km ha^{-1} and 0.01 km ha^{-1} for the heavy and light duty roads, respectively. Emissions associated with road construction will vary with specification, terrain etc. We assume a formation construction with crushed rock only and emissions of 42.1 tCO_2e km^{-1} for acquiring and laying aggregate. Thus emissions from road building per hectare of forest were estimated as:

(0.006 + 0.01 km ha^{-1}) × 42.1 tCO_2e km^{-1} = 0.7 tCO_2e ha^{-1}.

Assuming a lifespan of 50 years, produces a similar mean annual emissions value to that derived in Section 4.7.2 (0.014 tCO_2e ha^{-1} y^{-1}).

Establishment and beating up

Tree planting: It is assumed that tree planting is carried out manually. By convention, energy and GHG balance calculations ignore processes based on human labour. However, there is an energy and GHG impact due to the production of seedlings in a nursery. A value of 0.126 kg CO_2e per seedling was obtained (see Section 4.5), giving 315 kg CO_2e ha^{-1} (0.315 tCO_2e ha^{-1}) when planting at 2500 trees per hectare. The estimate did not consider GHG emissions due to application of nitrogen fertiliser in nurseries but these would be unlikely to add significantly to the GHG emissions per seedling.

Fertilising: It was assumed that no fertiliser was used during the establishment phase as fertiliser use is now unusual in forest establishment.

Weed control: Stand establishment often requires some weed control through herbicide application. Generally this operation is carried out manually as 'spot application'. The main emissions associated with weed control are therefore those from the manufacture of the herbicide and any subsequent decomposition of the herbicide. Little information is available on the carbon life-cycle analysis of herbicides; however, the results of Elsayed, Matthews and Mortimer (2003) for three example British forestry systems suggest that the total GHG emissions due to herbicide use are less than 0.01 tCO_2e ha^{-1}.

Beating up: Seedlings planted in a forest environment may not all establish successfully; a significant number can fail. These are replaced during 'beating up', early in the crop's lifespan (typically after about 3 years), together with the possible (manual) application of herbicide. We assume that 20% (5% in short rotation coppice, SRC) of seedlings fail to establish and so must be restocked, with an equivalent cost per seedling to the initial planting; this was estimated to be 0.06t CO_2e, giving a total in this phase of 0.315 + 0.06 = 0.38 tCO_2e associated with growing and planting the seedlings.

5.4.2 The initial phase

For the example management options being considered, this phase spanned the stand ages 5 to 25 years (the first thin usually being carried out at age 25). Figures 5.7 and 5.8 summarise pictorially the main contributions to the GHG balance during this phase, for the option of minimum intervention (MinI) and the option of thinning and clearfelling (T&F), respectively.

Figure 5.7 Illustration of contributions to GHG balance during initial phase (minimum intervention option, Sitka spruce, YC 12, 2 m spacing). All values in $tCO_2e\ ha^{-1}$ over the phase, 5–25 years.

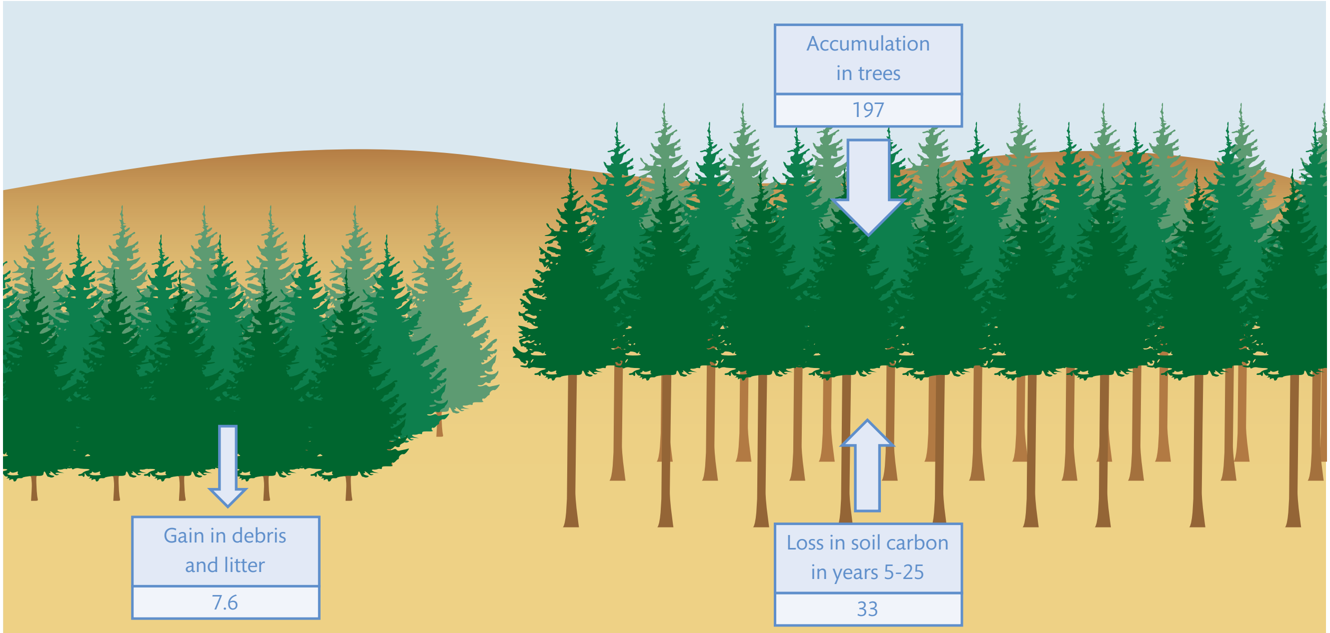
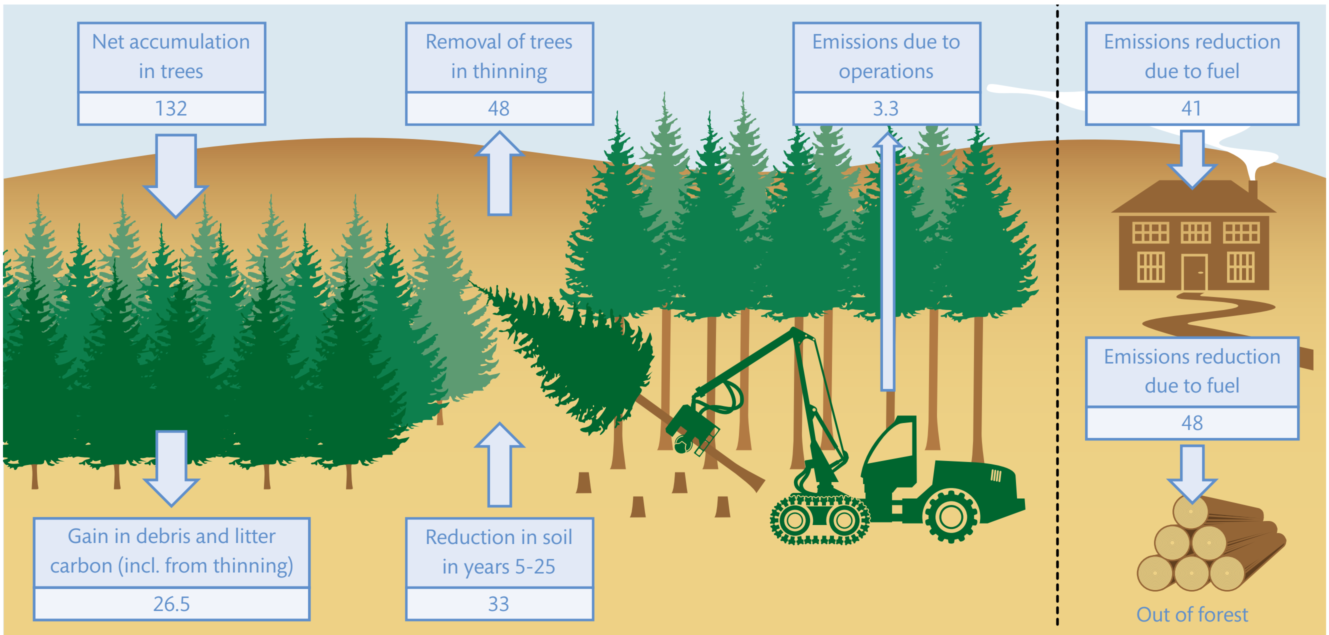


Figure 5.8 Illustration of contributions to GHG balance during initial phase, years 5–25 (thinning and felling option, Sitka spruce, YC 12, 2 m spacing). All values in $tCO_2e\ ha^{-1}$ over the phase. Operations include road maintenance, and thinning, extraction, transport and processing of the first thinning.



Trees

In the period from 5 years up to 20 years (the first entry in the yield table for YC 12 Sitka spruce planted at 2 m spacing), biomass accumulation in trees is estimated by backwards extrapolation as already described for the establishment phase. From age 20 onwards, biomass in trees is calculated on an annual time step by using values obtained from the appropriate yield model as input data to the BSORT biomass model described above.

Over the 20-year period (to age 25), the total C accumulation in trees is estimated to be 197 $tCO_2\ ha^{-1}$, an average sequestration rate of 9.85 $tCO_2\ ha^{-1}\ y^{-1}$. For the T&F option, the first thinning is scheduled to take place at stand age 25 years. The combined Forest Yield/BSORT tree biomass modelling system estimated the biomass in trees before thinning, after thinning, and removed as thinnings. The thinning operation removed 67 $tCO_2\ ha^{-1}$ of living trees, leaving 132 $tCO_2\ ha^{-1}$ in the standing trees. Of the thinned material, 44 tCO_2 was extracted, leaving approximately 23 tCO_2 as residue.

Debris and litter

For the initial phase, estimates for the debris and litter stocks for the full vigour phase are extrapolated back to year zero using a power function, as was done in the establishment phase. By age 25 before thinning, the increase in C stock is estimated to be 7.6 $tCO_2\ ha^{-1}$, giving a total C stock of 9.1 $tCO_2\ ha^{-1}$ and an average net accumulation rate for the period of 0.38 $tCO_2\ ha^{-1}\ y^{-1}$.

For the T&F option, the thinning operation at age 25 resulted in the transfer of a substantial C amount from the tree stock to debris and litter stocks (residue) of 23 $tCO_2\ ha^{-1}$, which in this example increased total litter in the 20-year period to 26.5 $tCO_2\ ha^{-1}$.

Soil

Parameters for the soil sub-model described in Appendix 6 are selected empirically to ensure that soil C stocks saturated at approximately 250 $tC\ ha^{-1}$ (≈920 $tCO_2\ ha^{-1}$), on peaty gley under grassland before tree establishment. In the simulation soil C continues to decrease over the initial phase (5–25 years), reaching a minimum just before the end of the phase. The rate of C loss over the forest management cycle is about 1.6 $tC\ ha^{-1}\ y^{-1}$ during the initial phase, and the soil C stock at the end is estimated at 806 $tCO_2\ ha^{-1}$. The thinning operation is assumed to have no impact on soil C dynamics.

Harvested wood products

For the T&F option only the first thinning was assumed to take place at the end of the initial phase (i.e. at age 25 years). As already discussed, thinning is estimated to result in the extraction of 44 $tCO_2\ ha^{-1}$ from tree stocks, with the remainder of thinned material transferred to the debris and litter stocks. The extracted material is assumed to be whole-tree above ground, and to be utilised entirely as wood fuel. The fractions of tree biomass in needles, branches, stem and roots are estimated for the thinning directly from the outputs of the BSORT model in combination with the appropriate yield model.

Although woodfuel is assumed to have a very short lifespan before being burned, the stock of 44 $tCO_2\ ha^{-1}$ is assumed to be stored for the remainder of the year, with no reduction.

Forest operations

For the T&F option the forest operations considered in this phase are thinning and extraction at age 25, chipping of material at the roadside, transport of material to the mill/processing plant, and road maintenance, as described earlier. The stem volume of harvested timber was given by the relevant yield model as 42 $m^3\ ha^{-1}$, so diesel fuel consumption in harvesting and extraction (see earlier) was estimated as:

$$42\ m^3\ ha^{-1} \times (1.55\ l\ m^{-3} + 0.9\ l\ m^{-3}) = 103\ l\ ha^{-1}.$$

The emission factors given in Table 6.5 were used to work out the GHG emissions from this fuel consumption, equalling 0.305 $tCO_2e\ ha^{-1}$ in this example. Chipping material at roadside incurs an additional fuel cost, leading to total forest operations emissions during the initial phase, including road maintenance, estimated at 3.3 $tCO_2e\ ha^{-1}$.

Substitution

For the T&F option in the initial phase only fuel substitution takes place, using the biomass harvested at the thinning at age 25 in small-scale heat applications, as described above (Table 5.8). Any use of the extracted material will of course reduce the harvested wood pool. The substituted emissions displaced by chipped material used in small-scale heating systems were:

$$26.4\ odt\ ha^{-1} \times 1.55\ tCO_2e\ odt^{-1} = 41\ tCO_2e\ ha^{-1}\ \text{displaced.}$$

5.4.3 The full vigour phase

For the management options being considered, this phase spanned ages 25 to 60 years. Figures 5.9 and 5.10 summarise pictorially the main contributions to the GHG balance during this phase for the MinI and T&F options. The stand was felled at age 60 in the T&F option (time of maximum MAI).

Trees

The C stocks and dynamics in the growing trees are modelled using the M1 and BSORT models as already described. Calculations for the MinI option use the yield model for an unthinned stand, which accounts for losses due to mortality from competition between trees. Carbon in dead trees is not included in tree C stock estimates, being transferred to the debris and litter stocks. By the end of this 35-year phase (25–60 years) a further 501 $tCO_2\ ha^{-1}$ is estimated to have accumulated in tree stocks, suggesting an average sequestration rate of 14.3 $tCO_2\ ha^{-1}\ y^{-1}$, and totalling 703 $tCO_2\ ha^{-1}$ at the end of this phase.

Calculations for the T&F option use the model for 'management table thinning', with clearfell taking place

Figure 5.9 Illustration of contributions to GHG balance during full vigour phase (minimum intervention example, MinI, Sitka spruce, YC 12, 2 m spacing). All values in $tCO_2e\ ha^{-1}$ over the phase, 25–60 years. Note: no operational emissions are incurred.

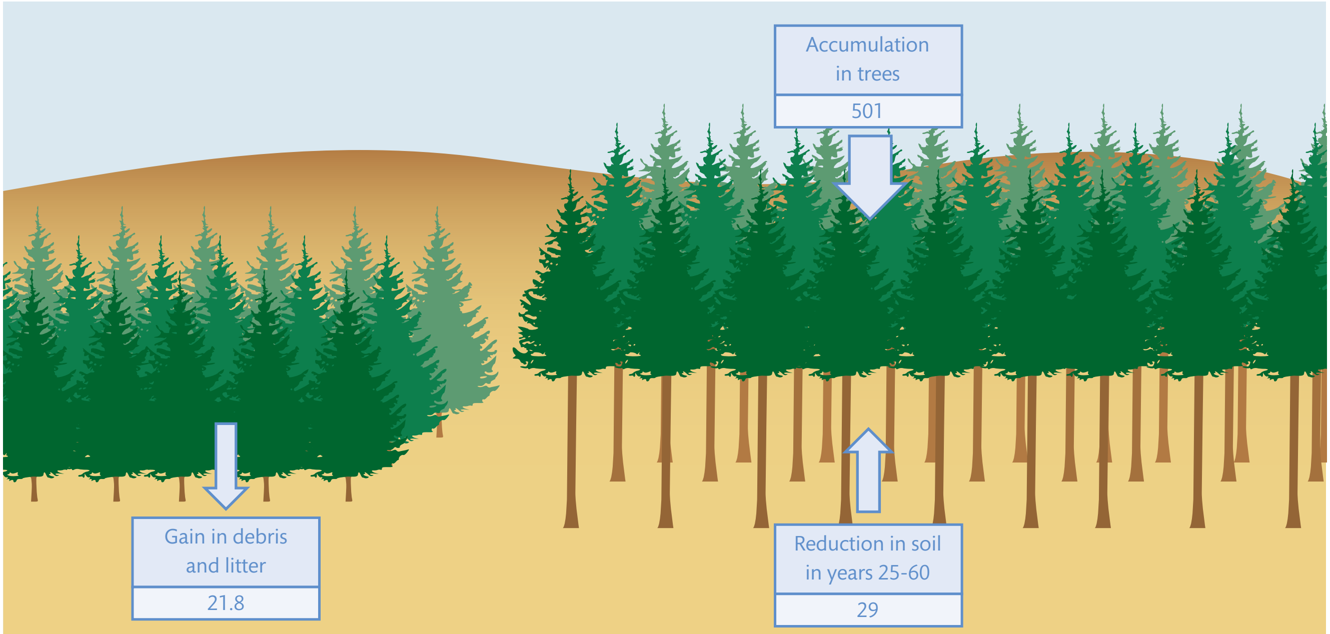
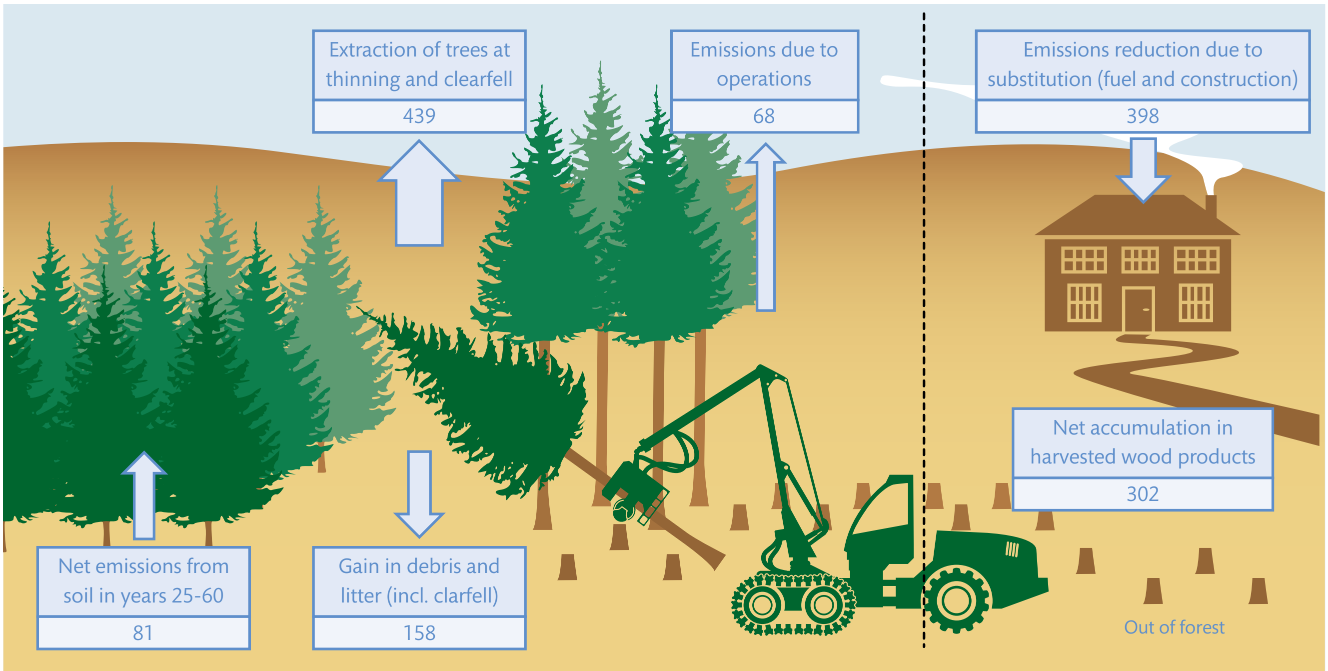


Figure 5.10 Illustration of contributions to GHG balance during full vigour phase (production example, T&F option, Sitka spruce, YC 12, 2 m spacing). All values in $tCO_2e\ ha^{-1}$ over the phase 25–60 years. Clearfell occurs at the end of the phase. Operational costs include thinnings and felling extraction, transportation and processing costs. Road maintenance emissions are also incurred.



before the end of the full vigour phase, at 60 years. During this phase there were a further six thinnings on a 5-year cycle, not including the first thinning at age 25, accounted for in the initial phase. The total removals due to the six thinnings is estimated as 153 $tCO_2\ ha^{-1}$ (approximately 26 $tCO_2e\ ha^{-1}$ per thinning). At the time of clearfell at age 60 the tree C stock is estimated at 450 $tCO_2\ ha^{-1}$, of which 285 $tCO_2\ ha^{-1}$ was extracted.

Debris and litter

Biomass estimates for dead trees (by component) are obtained from M1 and BSORT. For the MinI option, the main inputs to the debris and litter stocks during this phase come from tree mortality due to stand competition processes. From the start to the end of the 35-year period, a total of 21.8 $tCO_2\ ha^{-1}$ is estimated to have accumulated in debris and litter, giving an average rate of sequestration over the period of 0.62 tCO_2 ha^{-1} y^{-1}. The total C stock in debris and litter by the end of the period is 31 tCO_2 ha^{-1}.

For the T&F option for the 35-year period up to and including the time of clearfell, the main inputs to the debris and litter stocks come from conversion loss from trees harvested in the thinnings and final felling. From the start to the end of the 35-year period up to and including clearfell, a total of 158 tCO_2 ha^{-1} is estimated to have been transferred to debris and litter. The total C stock in debris and litter by the end of the period before clearfell is 32 tCO_2 ha^{-1}. The clearfell caused the C stock in debris and litter to rise, giving a total of 184 tCO_2 ha^{-1}.

Soil

For the MinI option soil C increases by an estimated 29 tCO_2 ha^{-1} over the 35-year period, giving a final soil C stock of 835 tCO_2 ha^{-1} and an average sequestration rate of 0.83 tCO_2e ha^{-1} y^{-1}. In the T&F option for the 35-year period up to clearfell, the soil accumulates a total of 81 tCO_2 ha^{-1}, with a final soil C stock before felling of 888 tCO_2 ha^{-1} and an average rate of sequestration over the period of 2.3 tCO_2 ha^{-1} y^{-1}.

Harvested wood products

For the T&F option for the 35-year period from the start of the full vigour phase up to the time of and including clearfell, a further 303 tCO_2 ha^{-1} is estimated to accumulate in HWP, at a rate of 8.6 tCO_2 ha^{-1} y^{-1}, giving total net accumulation of 352 tCO_2 ha^{-1}. The clearfell event causes the HWP stock to rise substantially. Although the T&F option effectively ends at age 60 years, C retained in harvested wood products will be kept out of the atmosphere for several years beyond this period (potentially decades), and should be accounted for in the calculations for successive rotations (where appropriate).

Forest operations

For the T&F option the forest operations included in this phase are thinning, clearfell, extraction, road maintenance, transport and processing of material, which were accounted for as previously described. For the 35-year period prior to clearfell GHG emissions due to forest operations were estimated at 24 tCO_2e ha^{-1}. A further 44 tCO_2e ha^{-1} was emitted during clearfell giving a total of 68 tCO_2e ha^{-1} emitted due to forest operations in this period, including transport and processing.

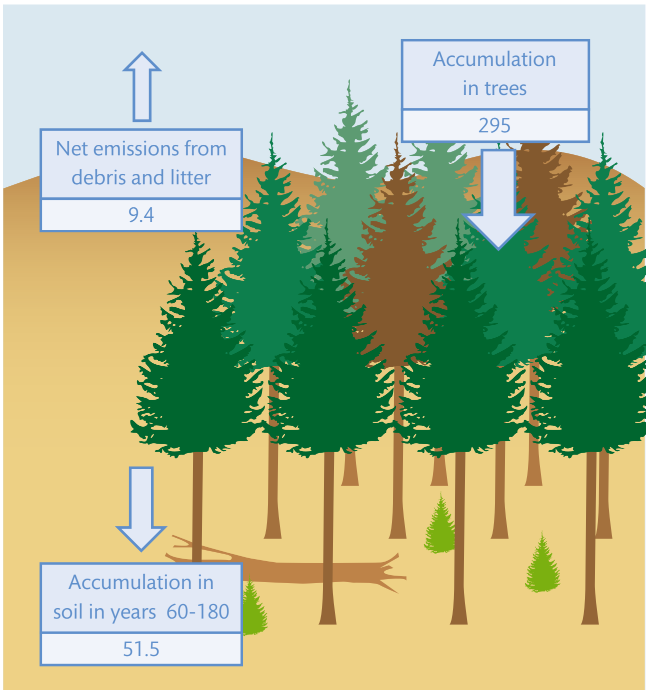
Substitution

For the T&F option the emissions reduction due to utilisation of the material harvested from thinnings and clearfell is estimated from both woodfuel and other HWP. For woodfuel, the approach used is the same as described in the section on the initial phase giving an estimate of a further 168 tCO_2e ha^{-1} emissions reduction, an average rate of 4.8 tCO_2e ha^{-1} y^{-1}. For other HWP, the emissions reduction from substitution in building materials is estimated as described above. For the 35-year period from the onset of the full vigour phase to and including clearfell, total GHG emissions reductions due to substitution of building materials is 230 tCO_2e ha^{-1}, an annual reduction of 6.6 tCO_2e ha^{-1} y^{-1}. Total substitution effect during the period was therefore 398 tCO_2 ha^{-1}.

5.4.4 The mature phase

As the T&F management option example is felled at the end of the full vigour phase, the mature phase is only relevant for the MinI option, and spans stand ages 60 to 180 years. The GHG balance is summarised in Figure 5.11.

Figure 5.11 Illustration of contributions to GHG balance during mature phase; 60 to 180 years (MinI option, Sitka spruce, YC 12, 2.0 m spacing). All values in tCO_2e ha^{-1} over the phase.



Trees

Yield estimates for stand ages greater than 80 years are dependent on species and may be based on extrapolations of the relevant yield model and then used in BSORT. Over the 120-year period tree C stocks are estimated to increase by 295 tCO_2 ha^{-1}, with an average sequestration rate of 2.46 tCO_2 ha^{-1} y^{-1}. The final C stock was 997 tCO_2 ha^{-1}. The considerable extrapolation at later ages must be acknowledged and is discussed further in the 'old-growth' phase below.

Debris and litter

From just after age 60, C stocks in debris and litter decline progressively, as the growth rate declines from a maximum value of about 31.5 tCO_2 ha^{-1}. Over the 120-year period from age 60 to 180 years, debris and litter C stocks are estimated to decrease by 9.4 tCO_2 ha^{-1}, a net rate of loss of about 0.07 tCO_2 ha^{-1} y^{-1}. The final C stock at the end of the period is 21.4 tCO_2 ha^{-1}.

It is probable that the accumulation of C in debris and litter is underestimated at later stand ages, due to the extrapolation from the relevant yield model. Some aspects of these extrapolations are highly speculative, not least projections of tree mortality. Modelled tree mortality drops off at later ages, because the yield models focus on representing tree death due to competition with neighbours, rather than due to more stochastic processes such as windthrow, storm and snow damage or disease. This results in an underestimate of C transfer to debris and litter.

Soil

Having lost carbon through the establishment and initial phases, the soil C increases during the mature phase. Over the 120-year period tree C stocks are estimated to increase by 51.5 tCO_2 ha^{-1}, with an average sequestration rate of 0.43 tCO_2 ha^{-1} y^{-1}. The final soil C stock is 886 tCO_2 ha^{-1}, a little lower than the initial level of 920 tCO_2 ha^{-1}.

Harvested wood products and substitution

There is no contribution to the GHG balance due to HWP or substitution, as no trees are harvested under a management prescription of minimum intervention.

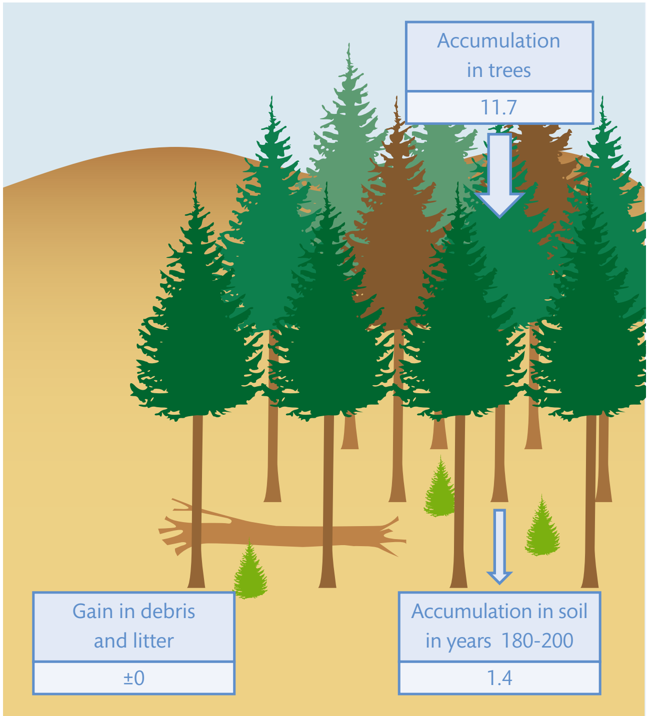
Forest operations

In principle there should be no forest operations for a management prescription of minimum intervention. In practice, a programme of forest protection activities may be needed, but the potential contribution of such activities to the GHG balance was not considered.

5.4.5 The old-growth phase

Consideration of this phase is only relevant to the example of the Sitka spruce stand being managed for 'minimum intervention'. In this case the phase spanned the stand ages 180 to 200 years (the end of the calculations in these examples). Figure 5.12 summarises pictorially the main contributions to the GHG balance during this phase. It is assumed that the stand remains viable, and the model does not consider natural regeneration.

Figure 5.12 Illustration of contributions to GHG balance during old-growth phase, ages 180–200 years (MinI option). All values in tCO_2e ha^{-1} over the phase.



Trees

Over the 20-year phase tree C stocks were estimated to increase by 11.7 tCO_2 ha^{-1}, with a sequestration rate of about 0.58 tCO_2 ha^{-1} y^{-1}. The final C stock at the end of the period was 1009 tCO_2 ha^{-1}.

The extreme extent of this extrapolation of yield must be acknowledged. In reality, by the time a forest has attained the old-growth phase, it is likely that significant disturbance events (windthrow, snow damage, fire) will influence stand development. In addition, a significant understorey and/or diverse stand structure (in terms of tree age and size classes) may develop. Such dynamics are not represented in these calculations, and there is little information on the structure and biomass of old-growth stands relevant to UK conditions, although it might be surmised that any contribution made by understorey biomass would be small compared with the overstorey.

Debris and litter

Over the 20 year period debris and litter C stocks are estimated to marginally decrease by 0.2 tCO_2 ha^{-1} (i.e. negligible annual sequestration rate). The C stock in debris and litter at age 200 is estimated at 21.25 tCO_2 ha^{-1}, which is likely to be an underestimate, as explained in the discussion for the mature phase.

Soil

In this phase, soil C stocks continued to increase towards the saturation level. An increase of 1.4 tCO_2 ha^{-1} occurred during the phase, a very low average sequestration rate of 0.07 tCO_2 ha^{-1} y^{-1}.

Harvested wood products, substitution and forest operations

As in the mature phase, there is no contribution to the GHG balance due to HWP or substitution, as no trees are harvested under a management prescription of minimum intervention. Similarly, it is assumed there are no forest operations.

5.5 Results for example forest management cycles

Results from CSORT of these C and GHG balance calculations for examples of different management options, species and site conditions have been produced, as shown in Figures 5.13 and 5.14. These show, on an annual time step over a 200-year period, the cumulative impact of three example management options on the C and GHG balance of either (a) a new YC 12 Sitka spruce forest on a peaty gley site or (b) a new YC 4 oak forest on a brown earth soil. In both species and site examples the management of the forest is set as either minimum intervention or thinning and felling on medium or long rotations (35 and 80 years for Sitka spruce, and 100 and 150 years for oak, Table 5.11). Further examples can be found in Appendix 8. All of the examples assume that an existing land use of 'pasture' (i.e. heather or grassland) is changed and new forest is created at time zero. However, for those with a shorter rotation length than 200 years, results are illustrated for new woodland creation in the first rotation and restocking and management of existing woodland in subsequent rotations.

Figure 5.13 Time course of GHG balance of forest, set as YC 12 Sitka spruce, on a peaty gley soil, over a 200-year period, managed with (a) minimum intervention, (b), thinning and felling on a 35-year rotation, and (c) thinning and felling on an 80-year rotation. Values are shown as cumulative across components, starting with the forest operations emissions values, which are not visible as they are close to the zero line.

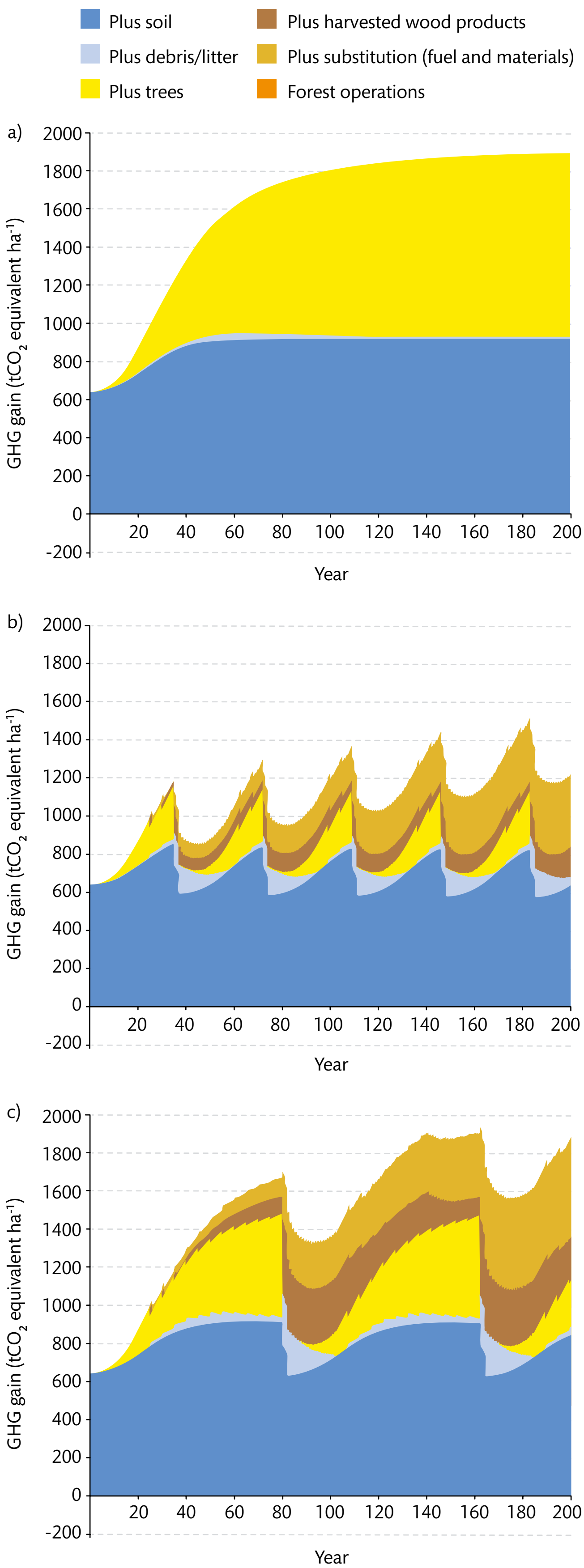


Figure 5.14 Time course of GHG balance of forest, set as YC 4 oak, on a brown earth soil, over a 200-year period, managed with (a) minimum intervention, (b), thinning and felling on a 100-year rotation, and (c) thinning and felling on a 150-year rotation. Values are shown as cumulative across components, starting with the forest operations emissions values, which are not visible as they are close to the zero line.

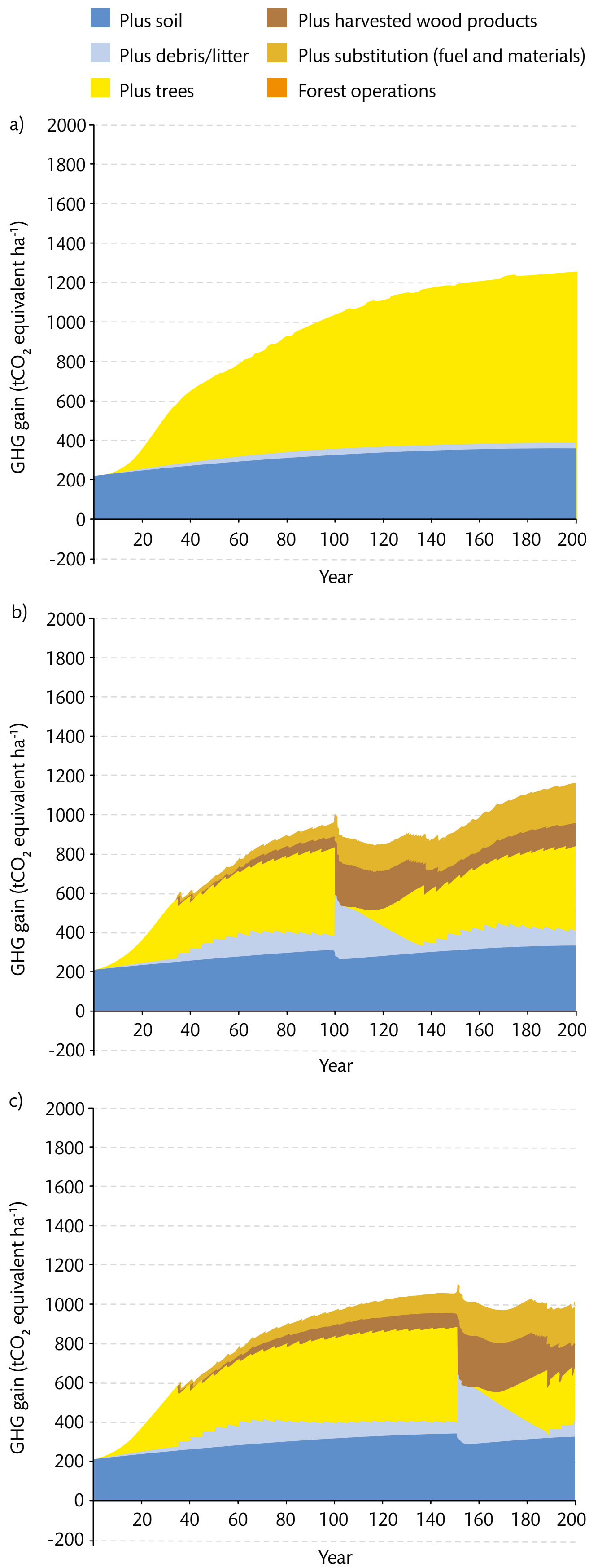


Table 5.11 Forest management cycles illustrated in Figures 5.13 and 5.14 and Tables 5.13 and 5.14.

Species	YC	Soil type	Management regime	Rotation (years)	Figure
Sitka spruce	12	Peaty gley	Min. intervention	-	5.13a
			Thin and fell	35	5.13b
			Thin and fell	80	5.13c
Oak	4	Brown earth	Min. intervention	-	5.14a
			Thin and fell	100	5.14b
			Thin and fell	150	5.14c

Notes: 1. Previous land/cover use assumed to be heather for Sitka spruce; grassland for oak.
2. Assumed established trees per hectare: 2500 for Sitka spruce, and 6750 for oak.

A number of broad conclusions can be drawn from Figures 5.13, 5.14 and A8.1–A8.3 in Appendix 8:

- Compared to other contributions, forest operations have a very small impact on the GHG balance (values are barely visible on graphs).
- Contributions to the GHG balance due to in-forest C stocks (soil, debris and litter, trees) generally involve a change in long-term time-averaged C stock. The size of this change (and the time over which it occurs) depends on a number of factors but most obviously the forest management regime. Factors such as species and yield class are most important through their influence on possible options for forest management.
- Although soil C can represent a very large C stock, in all the examples considered the modelled overall change from initial to final C stock in soils is relatively small.
- Harvested wood products make a larger contribution to the GHG balance through substitution for fossil fuels than through C sequestration in products themselves.
- Ignoring the very small effect of forest operations, substitution is the only contribution to the GHG balance of forestry systems not to exhibit saturation (i.e. positive impacts accrue indefinitely as more and more timber is harvested). However, in these examples, the size of the contribution may not match the in-forest sequestration by the minimum intervention option for many rotations or decades.
- As with forest C stocks, the single biggest influence on the contribution made by substitution is the forest management regime. Factors such as species and yield class are most important through their influence on possible options for forest management.

Generally speaking, these results confirm earlier findings such as those reported by Marland and Marland (1992), Matthews (1994, 1996), Marland and Schlamadinger (1995), Nabuurs (1996) and Schlamadinger and Marland (1996a, b, c).

Previous discussions in Chapters 3 and 4, concerning the uncertainty in the various component estimates that underlie CSORT, should be referred to when considering Figures 5.13 and 5.14 and Figures A8.1–A8.3 in Appendix 8. Definitive estimates of conversion factors for emissions reduction due to substitution of wood for other materials remain elusive and their derivation must be viewed as a high priority. For the time being, the calculations behind Figures 5.13, 5.14 and Figures A8.1–A8.3 in Appendix 8 have been placed on a more defensible but possibly conservative basis.

5.6 Application and interpretation of results for forest management cycles

Results such as presented in Figures 5.13 and 5.14 and Figures A8.1–A8.3 in Appendix 8 can be compared in a number of ways in order to evaluate different land use, forestry and forest management options, but this needs to be done with care. Often decision makers are not particularly interested in the constituent contributions to an overall impact on GHG balance (e.g. trees, soil etc). What is required is a simple value for the overall impact. Such estimates must be constructed carefully, for example including all relevant contributions to the GHG balance of different options, while avoiding double counting and other possible distortions in calculations.

It can also be confusing to follow the comparison of complex impacts over time for different modelled options, such as illustrated in Figures 5.13 and 5.14. It is not unreasonable that the possibility of a long 'payback period' exists since substitution benefits of managing a new stand of Sitka spruce for production may take years to take effect, when compared to management based on minimum intervention.

One way to simplify results is to summarise estimates for key phases in forest management cycles in tables. For example, Tables 5.12 to 5.14 summarise the same scenarios outlined in Table 5.11 and used in Figures 5.13a, b and c. They show (a) the C stocks (or, in the case of forest operations and substitution, cumulative emissions and reductions) for each phase within the management cycles, and (b) the average changes in stocks over the specified phase. For in-forest carbon stocks, the estimates are based on long-term averages of stock estimates, in order to emphasise the long-term impacts of different options. For those examples involving more than one rotation of a specified management cycle, separate tables give outputs for the first rotation and the second rotation. Arguably the key result in each table is shown in the bottom right cell. This is (a) the overall stock change for the period from start to finish for each example forest management cycle or (b) the overall rate of carbon sequestration/emissions reduction for each example forest management cycle. Tables for the oak YC 4 examples and others are given in Appendix 8.

Table 5.12 Total GHG balance of a forest management cycle for first rotation YC 12 Sitka spruce managed for minimum intervention. Time-averaged (a) C stocks and (b) rates of C stock changes in key components. Positive values indicate sequestration, negative values indicate emission.

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Old growth	Full management cycle	Stock change over full management cycle
Years	0-5	5-25	25-60	60-190	190-200	0-200	-
Period	5	20	35	130	10	200	-
Trees	2.0	66.1	299.5	701.5	714.6	714.6	714.6
Other vegetation	-	-	-	-	-	-	-
Debris /litter	0.4	2.5	12.6	15.6	15.4	15.4	15.4
Soil	799.2	723.5	812.3	883.5	885.2	-31.5	
Harvested wood products	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Forest operations	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	0
Substitution	0.0	0.0	0.0	0.0	0.0	0.0	0
Total	**801.5**	**792.0**	**1124.3**	**1600.6**	**1615.1**	**1615.1**	**698.5**

(b) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature		Full management cycle
Years	0-5	5-25	25-60	60-190	190-200	0-200
Period	5	20	35	130	10	200
Trees	0.4	3.2	6.7	3.1	1.3	3.6
Other vegetation	-	-	-	-	-	-
Debris /litter	0.08	0.11	0.29	0.02	-0.02	0.1
Soil	-23.5	-3.8	2.5	0.5	0.2	-0.2
Harvested wood products	0.0	0.0	0.0	0.0	0.0	0.0
Forest operations	0.0	0.0	0.0	0.0	0.0	0.0
Substitution	0.0	0.0	0.0	0.0	0.0	0.0
Total	**-23.0**	**-0.5**	**9.5**	**3.7**	**1.5**	**3.5**

Table 5.13 Total GHG balance of a forest management cycle involving first rotation YC 12 Sitka spruce managed for thinning and felling on a 35-year rotation. Time-averaged (a) C stocks and (b) rates of C stock changes in key components; (c) and d) as (a) and (b) but for second rotation. Positive values indicate sequestration, negative values indicate emission.

First rotation

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-25	25-35	-	0-35	-
Period	0-5	5-25	25-35	-	200	-
Trees	1.9	64.9	103.5	-	103.5	103.5
Other vegetation	-	-	-	-	-	-
Debris /litter	0.4	2.8	10.9	-	15.4	10.9
Soil	799.2	723.5	812.3	-	10.9	
Harvested wood products	0.0	0.9	4.9	-	4.9	4.9
Forest operations	-0.1	-0.4	-2.3	-	-2.3	-2.2
Substitution	0.0	14.1	75.5	-	75.5	75.5
Total	**801.5**	**805.8**	**942.9**	**-**	**942.9**	**26.3**

(b) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-25	25-35	-	0-35
Period	5	20	10	-	35
Trees	0.4	3.1	3.9	-	3.0
Other vegetation	-	-	-	-	-
Debris /litter	0.1	0.1	0.8	-	0.3
Soil	-23.5	-3.8	2.7	-	-4.7
Harvested wood products	0.0	0.0	0.4	-	0.1
Forest operations	0.0	0.0	-0.2	-	-0.1
Substitution	0.0	0.7	6.1	-	2.2
Total	**-23.0**	**-0.2**	**13.7**	**-**	**0.8**

Second rotation

(c) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	35-42	42-62	62-72	-	35-72	-
Period	7	20	10	-	37	-
Trees	86.5	84.6	100.6	-	100.6	-2.9
Other vegetation	-	-	-	-	-	-
Debris /litter	31.9	35.8	35.2	-	35.2	24.3
Soil	729.6	710.7	723.6	-	723.6	-26.9
Harvested wood products	17.6	32.9	38.2	-	38.2	33.3
Forest operations	-2.4	-2.7	-4.6	-	-4.6	-2.3
Substitution	75.5	89.6	150.9	-	150.9	75.5
Total	**938.7**	**950.9**	**1043.9**	**-**	**1043.9**	**101.0**

(d) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	35-42	42-62	62-72	-	35-72
Period	7	20	10	-	37
Trees	-0.4	-0.1	1.6	-	-0.1
Other vegetation	-	-	-	-	-
Debris /litter	0.5	0.2	-0.1	-	0.7
Soil	-0.5	-0.9	1.3	-	-0.7
Harvested wood products	0.3	0.8	0.5	-	0.9
Forest operations	0.0	0.0	-0.2	-	-0.1
Substitution	0.0	0.7	6.1	-	2.0
Total	**-0.1**	**-0.6**	**9.3**	**-**	**2.7**

Table 5.14 Total GHG balance of a forest management cycle involving first rotation YC 12 Sitka spruce managed for thinning and felling on an 80-year rotation. (a) Time-averaged C stocks and (b) Time-averaged rates of C stock changes in key forest components and associated harvested wood products; (c) and (d) as (a) and (b) but for second rotation.

First rotation

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-25	25 -60	60-80	0-80	-
Period	5	20	35	20	80	-
Trees	1.9	64.9	213.7	280.8	280.8	280.8
Other vegetation	-	-	-	-	-	-
Debris /litter	0.4	2.8	24.7	28.8	28.8	28.8
Soil	799.2	723.5	812.3	837.4	837.4	-79.3
Harvested wood products	0.0	0.9	19.3	38.4	38.4	38.4
Forest operations	-0.1	-0.4	-2.4	-6.5	-6.5	-6.5
Substitution	0.0	14.1	80.1	241.0	241.0	241.0
Total	**801.5**	**805.8**	**1147.8**	**1419.8**	**1419.8**	**503.2**

(b) Rates of C stock change ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-25	25-60	60-80	0-80
Period	5	20	35	20	80
Trees	0.4	3.1	4.3	3.4	3.5
Other vegetation	-	-	-	-	-
Debris /litter	0.1	0.1	0.6	0.2	0.4
Soil	-23.5	-3.8	2.5	1.3	-1.0
Harvested wood products	0.0	0.0	-0.1	-0.2	-0.1
Forest operations	0.0	0.0	-0.1	-0.2	-0.1
Substitution	0.0	0.7	1.9	8.0	3.0
Total	**-23.0**	**0.2**	**9.8**	**13.6**	**6.3**

Second rotation

(c) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	80-87	87-107	107-142	142-162	80-162	-
Period	7	20	35	20	82	-
Trees	258.3	225.1	248.5	277.4	277.4	-3.5
Other vegetation	-	-	-	-	-	-
Debris /litter	43.7	47.3	45.6	45.0	45.0	16.2
Soil	823.9	801.6	819.8	831.3	831.3	-6.1
Harvested wood products	62.5	104.6	131.4	130.6	130.6	-6.6
Forest operations	-6.6	-6.9	-8.9	-13.1	-13.1	-6.6
Substitution	241.0	255.1	320.9	482.1	482.1	241.0
Total	**1422.8**	**1426.8**	**1557.3**	**1753.2**	**1753.2**	**333.4**

(d) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	80-87	87-107	107-142	142-167	80-162
Period	7	20	35	20	82
Trees	-0.3	-1.7	0.7	1.4	0.0
Other vegetation	-	-	-	-	-
Debris /litter	0.2	0.2	0.0	0.0	-0.2
Soil	-0.2	-1.1	0.5	0.6	-0.1
Harvested wood products	0.3	2.1	0.8	0.0	1.1
Forest operations	0.0	0.0	-0.1	-0.2	-0.1
Substitution	0.0	0.7	1.9	8.1	2.9
Total	**0.0**	**0.2**	**3.7**	**-**	**4.1**

5.7 Plans to further develop the CSORT model

In principle, the methodology developed in this section could be applied to the calculation of the GHG impacts of a wide range of land use, forestry and forest management options covering different initial land covers, species of tree, yield classes, soil types and management regimes, including restoration of land areas with woodlands to previous land cover (e.g. heathlands and peat bogs). Clarification of a number of key uncertainties and gaps as identified throughout this report would significantly improve the quality of estimates for some options. In addition, a significant barrier on generation of these estimates is one of resources for collecting required input information, running models and the analysis and interpretation of results.

CSORT has great scope for development and the preliminary version of the program is being re-written to include new features and provide more flexibility. This will include compatibility with forecast production needs, improved modelling of continuous cover forestry and to allow for changing management strategies. In order to facilitate the differing requirements, the model will consist of essentially two components: an in-forest element, which will cover material to forest roadside; and 'CSORT plus', which will deal with the out-of-forest elements (transport, processing, substitution and life-cycle analysis), part of which have been presented in this report.

One of the main requirements will also be in establishing the likely users, and in what form to produce output. For example, are a series of output tables or figures appropriate, or is a fully interactive model required, and with what degree of flexibility? Methods will also need to be developed to approximate new yield models (or at least their effect), given that demands will include examples such as broadleaf species at wider spacing than are available from current yield tables.

6. Key conclusions, evidence gaps and research needs

The preceding review allows several key conclusions to be made about our understanding of the main components of the forestry C cycle. It has also highlighted several uncertainties and gaps in evidence and/or understanding. Below the key points are summarised, grouped into the main component or topic areas, together with the important research needs.

6.1 Forestry, the carbon cycle and carbon stocks

1. Carbon stocks and fluxes are central to forestry; both in the functioning of the woodland ecosystem, and in the production of timber, woodfuel and other wood products.

2. Carbon moves continually between the atmosphere and the key pools of the tree, litter, soil and harvested wood products. Although trees are obviously pivotal, there is usually more C held in the soil and litter than in trees. Arguably, understanding the determinants of all the component C stocks and their changes is of central importance in forestry.

3. Compared to other aspects of the C balance of the forestry sector, there is reasonable certainty and comprehensiveness of data to enable the reliable estimation of C stocks, changes and balances in trees in the UK, at least of the major commercial species. Indeed, much of the information available is being used in the Carbon Assessment Protocol (Jenkins *et al.*, 2010) underpinning the development of the Forestry Commission 'Woodland Carbon Code'.

4. Half of the biomass of woody plants is carbon, but as wood density varies between species, the C content per volume of wood varies substantially. For example, hardwood species such as oak have a timber C content of 0.30 tC m^{-3}, while for softwood species such as Sitka spruce it is 0.17 tC m^{-3}. This must be taken into account when comparing yield classes which are based on timber volume.

5. Only about 50% of the carbon in a tree is in the merchantable stem. While assessment and prediction of stem yield is routine and well established, assessing total C stocks relies on a considerably less detailed empirical foundation. More information is required on allocation of biomass to other components than stems (foliage, branches and roots), particularly for broadleaved species, including those that are targets of climate change adaptation, and under newer silvicultural practices and different growth conditions. In addition, in some woodlands woody understorey vegetation is a significant component of the C cycle, which is usually poorly quantified, and is not included in C accounting models.

6. Comprehensive quantification of the C cycle of example woodland stands is still very limited, and there are only two sites in the UK where stand-scale CO_2 fluxes have been directly measured over the medium or long term. Such measurements directly test and enlarge our understanding of woodland processes, and the impacts of a changing climate and environment. They are also key information resources for developing and testing the C cycle models necessary to explore the effect of management decisions and their optimisation and the impact of climate change.

7. National forest C stock estimates need to be revised to take into account research on tree biomass partitioning, as well as new information on forest soil and litter C stocks detailed in this report. Deadwood and coarse woody debris components are presently largely unknown. Furthermore, errors and uncertainties in estimating C stocks in different in-forest and out-of-forest components should be assessed, in readiness for the availability of the new National Forest Inventory information, and the updated spatial information it will provide.

6.2 Woodland stand C dynamics

8. The pattern of C accumulation in trees follows the growth of timber as 50% of the dry weight of wood is carbon. Growth rate and thus CO_2 uptake of even-aged stands peaks early in the growth cycle, and maximising CO_2 uptake rate is very different from maximising C stocks.

9. Stands thinned regularly and harvested on shorter rotations will have lower C stocks, but may sequester more C over time than stands managed on long rotations or no thinning.

10. The growth of new woodland results in the accumulation of substantial C stocks in the trees. Depending on yield class, species and conditions, woodland C stocks in Britain can exceed 1000 tCO_2 ha^{-1}, and CO_2 uptake rates can reach 30 tCO_2 ha^{-1} y^{-1}.

11. The estimation of biomass and C dynamics in young (from planting to pre-pole stage) stands is crude in existing models of biomass accumulation (such as BSORT) and carbon dynamics (such as CSORT, CARBINE and C-FLOW) because these are reliant on yield table data, typically starting at age 20. It is essential that additional information is collected to improve the early growth predictions.

12. In many C accounting models, including CSORT, CARBINE and C-FLOW, the description of C transfer from live trees to dead trees, coarse woody debris, fine woody debris or litter and ultimately back to the atmosphere or into the soil is weak, and handled empirically. These C pools need to be defined more clearly and the C transfer processes between the pools needs additional experimental investigations and improved representation in models. The C dynamics of understorey woody vegetation may also need to be modelled for some woodland types.

13. The BSORT, CARBINE, CSORT and C-FLOW models rely on yield models to provide basic data on stand growth and yield to estimate changes and levels of C stocks. The existing published yield models represent a wide range of possible management regimes, but this is not comprehensive, and is inevitably based on past management practices. In particular, there is almost no representation of 'continuous cover' forest management, other forms of LISS or of 'carbon reserve' management. Furthermore, there are no yield tables appropriate for some of the spacing and silvicultural practices presently in use, and for old-growth stands. These deficiencies need to be addressed.

14. It is likely that continuous cover forestry silvicultural systems have the potential for higher C accumulation than even-aged stands, although there is little real evidence. There are several additional aspects that need to be considered, including examination of the consequence for operation and disturbance 'costs' of more frequent harvesting.

15. In order to assess the impact of climate change on UK forestry, and to examine adaptation options, linking of simplified but robust carbon accounting models to climate change scenarios is urgently required. It may be necessary to use a hierarchy of process-based, semi-empirical and empirical models to tackle these key issues, or to use 'hybrid' models.

6.3 Litter and coarse woody debris C stocks

16. Litter C stocks are significant components of forests, and it is important to make reliable assessment of litter inputs, stocks, C cycling and their changes with management.

17. The BioSoil dataset has provided an important new assessment of the C stocks in litter and organic F layer in GB forests, across the 166 sites sampled. Mean litter C stocks for broadleaves and conifers were 63 and 56 tCO_2 ha^{-1}, respectively (L and F layer together). Measured C stocks in broadleaf and conifer stands were not as different as some reports indicate, because litter bulk density was higher in broadleaved stands.

18. The new estimates of the litter and organic F layer C stock in GB forests for conifers and broadleaves are scaled up with forest areas in this report, and amount to 168 $MtCO_2$ (45.9 MtC). This is considerably more than the value reported in earlier FC statistics of 25 MtC. However, there are uncertainties over whether FC statistics include the F layer.

19. Further analysis of this dataset will hope to identify any systematic variation in litter stocks between tree species, stand age and soil type.

20. There are problems and ambiguities over definitions, particularly between field survey methods, and litter layer representation in models. Some measurements are reported only for the litter layer, but often the data include the litter and F layer together, because in some site and soil types it is difficult to separate the two layers.

21. The C stocks in coarse woody debris (CWD) in British forests are not known reliably, based only on NIWT survey information of deadwood piece counts which were converted to estimated CWD volumes. Preliminary work in the BioSoil programme has shown the feasibility of measuring and estimating CWD stocks and analysis of these data will provide measured CWD biomass and C stocks under both conifers and broadleaved forests in GB. This will be combined with CWD data from the new National Forestry Inventory detailed CWD survey to improve estimates.

22. Typical rates of litterfall in UK forest stands range between 10 and 30 tCO_2 ha^{-1} y^{-1}.

23. The rate of litter decomposition is very important in estimating soil and forest C sequestration potential due to its influence on soil C and N fluxes. To establish litter, organic layer (F and H) and overall soil C balances it is critical that decomposition rate is reported alongside litterfall amounts and soil C stocks, but there are very few published studies with all this information at the same sites.

24. An attempt was made to summarise litter quality of different tree species and link them qualitatively to decomposition rate.

25. The next necessary step will be to complete the synthesis of litter decomposition rates and classify forest soils and forest types according to their C stocks in soils, L and F layer, woody debris along with litter quality and decomposition rate. This will form an important platform for modelling of C dynamics and assessing the effects of management options.

6.4 Woodland soil C stocks

26. New estimates of C stock in UK woodland soils are now available from the BioSoil dataset, which represents a major step forward, and will provide much new information as the dataset is further examined. Mean C stock values for seven main soil groups have been calculated. For mineral soil groups under forest average C stocks in the 0–80 cm soil depth ranged between 487 and 569 tCO_2 ha^{-1} (133 to 155 tC ha^{-1}). Organo-mineral soils (peaty gleys, peaty podzols and peaty rankers) have approximately double those (1174 tCO_2 ha^{-1} or 320 tC ha^{-1}), and organic soils (deep peats) have over three times as much (1644 tCO_2 ha^{-1} or 448 tC ha^{-1}).

27. These soil group C stocks have been used with national soil and forest cover maps to produce a GB estimate of forest C soil stocks. The new estimate is 2302 $MtCO_2$ (627 MtC) down to 1 m depth. For GB this is just 12% smaller than estimates reported in FC statistics but the difference is larger for Wales (17%) and Scotland (15%) than for England (6%). New woodland cover data from the National Forest Inventory will enable an improved estimate.

28. The importance of sampling soils to the correct depth, with measured soil C and bulk density of the corresponding depths, is emphasised through the analysis shown in this report. Precise estimation of soil C stocks is impossible without these measures. Furthermore, estimates of soil C stocks from topsoil or shallow soil depths (e.g. 0–15 cm or 0–30 cm) alone do not give an adequate picture as there is substantial soil C below 40 cm, particularly in organic soils.

29. The classification separation of true peats from peaty soils or organo-mineral soils is different in the different assessments and surveys (e.g. the NSI definition was 45 cm depth, Soil Survey for Scotland uses 50 cm and BioSoil used 40 cm depth), which must be taken into account when comparing or combining soil C stocks measured and calculated from different soil surveys.

30. Carbon stocks in peat soils depend greatly on the depth of the peat layer as well as peat bulk density and its variation with depth. While the recent ECOSSE reports have improved estimates of peat C stocks overall in Scotland and Wales, more information is needed on total peat C stocks under existing forests on deep peats with peat layers more than 80 cm depth.

31. If temporal changes are to be detected in soil C specifically in forestry, detection should be based on comparison of soil characteristics measured at soil horizons, not at specific soil depths due to the substantially different properties and accumulation rates of organic material in the topsoil layers in forests. This is a critical difference between forest soils and agricultural ones; one that is essential to understand in comparison of soil C changes across different land uses.

32. Detecting change in soil C stocks is very difficult; more chronosequence work will be useful to establish the effects of afforestation and different management practices on soil C. However, the approach is inevitably limited by the long timespans required, which may mean that site preparation and silvicultural practices in older sites may not be relevant to current practices.

33. Deriving more accurate relationships between soil C content and bulk density for cases where bulk density is not measured is crucial and BioSoil datasets at EU scale will be used to verify such relationships. GB BioSoil datasets will be used to develop pedotransfer functions for calculating specific bulk density for different forest soils, which can then be used in calculating soil C stocks in plots from the National Forest Inventory.

34. Information is required on soil C fractions and pools to differentiate soil types and soil C stability and also sensitivity to C loss with different management activities.

35. There are many uncertainties when comparing national estimated soil C stocks from different soil survey networks, some of which are highlighted in this review. The uncertainties are due to differences in:

a. soil depth from where the measured and calculated soil C are derived;
b. bulk density estimation methods and measurements used to calculate the soil C stocks;
c. soil association grouping, reflected in the mapping of different areas for the main soil groups;
d. forest type mapping; examples of the discrepancy between Land Cover 2000 and Forestry Commission forest cover databases are reported in this review.

Harmonisation between methods of soil sampling and measurements, soil grouping and use of same source and details of spatial information such as soil maps and forest cover maps should resolve some of the differences found in calculating national soil C stocks between different national soil surveys.

36. In British conditions, the annual flux of dissolved organic carbon (DOC) can be important in the overall C balance of forest stands, although it is usually considerably less than 10% of the typical net CO_2 uptake rates by the trees. Such relatively large fluxes are found in organic soils, but on mineral soils typical annual DOC fluxes are usually <1% of likely tree net CO_2 uptake.

6.5 Quantifying the role of harvested wood products

37. The contribution of harvested wood products (HWP) to overall C stocks for the UK forest sector is significant. Harvested wood is used to make primary products, and at the end of their life these may be made into secondary products, prior to eventual disposal. The size of the total HWP stock in the UK depends on the utilisation of materials and their longevity, and is likely to have an upper limit, although its value is hard to determine. The estimation is complicated as the majority of the HWP stock derives from imports.

38. Models such as CSORT or CARBINE need to be run for many more examples of tree species, yield class and management regime to present a full picture of the contribution made to the C balance by HWP in the UK.

39. There are few examples of detailed analyses of biomass flows through forest sector production and processing systems and these tend to be limited to representation of specific wood products. A much more comprehensive analysis of production and processing systems relevant to the UK forest sector is needed.

40. Understanding biomass flows in harvested wood is complicated in particular by changes in moisture content that occur following harvest, during storage, during processing and while products are in use. These changes influence the ratio of oven dry mass of wood to volume, and therefore result in systematic variation in the C content per unit volume of wood as moisture content changes. This variation is not well represented either in existing forest C accounting models or in analyses of wood production and processing systems, and improvements are required.

41. More information on net GHG emissions during HWP processing is required as these are not presently included in CSORT.

42. Most modelling of C stocks and flows associated with wood products depends on assumptions about the service lives of products. While the results of such modelling exercises appear to be relatively insensitive to quite large variations in these assumptions, there are few hard data to support the selection of values for wood product service lives.

43. There is only limited understanding and little quantification of the C stocks and flows associated with wood in secondary use and following disposal.

6.6 Quantifying the role of material and fuel substitution

44. Forests can provide woodfuel to substitute for fossil fuel derived energy sources, and timber and other materials to substitute for those made from fossil fuels or those manufactured using fossil-fuel derived energy. Such substitution can reduce GHG emissions.

45. Importantly, while C stocks in growing forests tend to reach an asymptote, substitution benefits continue to accrue for as long as substitution occurs, because of the fossil fuel GHG emissions they avoid.

46. The contribution that woodfuel and HWP substitution could make to emissions reduction is substantial,

although the quantification relies on a relatively small information and research base and further work is needed to improve its reliability.

47. Estimates of present wood fuel usage by country and plans for expansion should be incorporated into calculations of the UK forestry C balance and the contributions to emissions abatement.

48. Estimates of emissions reduction factors due to utilisation of wood for fuel or materials tend to represent either 'generic' products or very specific examples of wood production and utilisation systems. A more comprehensive analysis of wood production, processing and installation chains is required in order to understand the range of possible impacts on C and GHG balances and the sensitivity to particular production and processing methods.

49. Existing research on substitution benefits needs to be better linked to the assessment of resulting forest vegetation carbon dynamics so that the impacts of the emissions reduction from wood utilisation can be assessed in relation to on-site C stock changes caused by changes in management.

6.7 Assessment of forest C and GHG fluxes

50. Overall, in most forest types CO_2 fluxes dominate the GHG balance of forest soils; N_2O and CH_4 are small components. N_2O fluxes are usually small in UK forests because fertilisers are rarely used and atmospheric N deposition is not as high as, for example, in central Europe. In stands on wet peat soils CH_4 fluxes can be large, although their annual emissions never exceed, in CO_2-equivalent terms, the CO_2 emissions from the soil. Rates of CH_4 emission from wet soils depend critically on water table depth.

51. There is a lack of simultaneous measurements of CO_2, CH_4 and N_2O fluxes from forests. More information is necessary to quantify reliably the emissions associated with particular silvicultural practices through the whole forest cycle in different conditions and to assess the impact and potential of particular forest management decisions for GHG emissions mitigation.

52. Example datasets of GHG fluxes in a range of appropriate climate and site conditions are also essential in order to use the advanced process-based models now available to examine the impacts of site, environment and management effects at a variety of scales. These are necessary to assess climate change effects, and for GHG inventory verification.

53. Although there are numerous measurements of stand-scale and soil CO_2 fluxes and some for CH_4 and N_2O fluxes from soils in forests, the large spatial and temporal variations, and the sensitivity to environmental conditions and C and N inputs, result in substantial uncertainty in the appropriate value for particular forest locations or types.

6.8 Impacts of forest management

54. The management of forests clearly has a major impact on the C and GHG balance of forestry, at all scales from the stand up to the UK. With 12% of the UK presently forested, decisions about forest extent, design, purpose, management and use influence the national GHG balance.

55. Most C flux data available for UK forests have been measured during the productive growth phase in conifer crops, and mostly in even-aged stands. There has been one reported chronosequence study where C and other GHG flux measurements were taken directly after afforestation on peat during the establishment phase. One assessment reports soil GHG fluxes immediately at cultivation, and one UK site has had assessments of soil GHG flux losses after harvesting with a second rotation chronosequence. The lack of more information is an important gap, as these activities can be expected to change fluxes due to the soil disturbance (as a result of drainage, in particular, and site preparation practices), any fertiliser application, and the consequent changes in water levels and decomposition rate.

56. Data on C and other GHG emissions caused by forest management activities (particularly after thinning and clearfelling) on the major contrasting soil types are very limited. There is also a lack of data on GHG emissions after land-use changes such as open habitat restoration or afforestation. Information on these is essential to identify mitigation options to reduce GHG emissions.

57. The C balance benefits of new forest planting depend substantially on soil type, climate and previous land use. The net emissions mitigation benefits of afforestation will be largest on mineral soils with low existing soil C

stocks, such as those previously intensively cultivated. On highly organic soils, the benefits of increasing the above-ground stock by planting trees, have to be balanced against potential soil C losses due to soil disturbance. Estimation of that balance depends on the specifics of the site and soil type, ground preparation practices, the likely productivity of the trees and their management and eventual use.

58. Although there are few definitive data, afforestation of mineral soils may be expected to lead to long-term increases in soil C, with typical rates of between 0.5 and 1.7 tCO_2 ha^{-1} y^{-1}. For soils with high organic content, such as peaty gleys, evidence from past afforestation suggests that there were substantial soil C losses during the first rotation after afforestation, approximately 5–15 tCO_2 ha^{-1} y^{-1}, but then subsequent increases in soil C stock. This recovery is partly due to incorporation of litter and harvesting residues during the ground preparation for the second rotation. More information is required on the stability of different soil C fractions after disturbance during new planting or restocking. The uncertainty in extrapolating from past ground preparation and planting practices to estimate likely soil C changes should also be recognised.

59. Carbon stocks in other vegetation are usually small, compared with forests and woodlands, but in the early period of tree establishment up to canopy closure they may be significant, and their changes may need to be incorporated for a full site C balance.

6.9 Forest operations effects

60. The management of forest operations can have an impact on the C and GHG balance of forestry, especially on sites where high C content soils predominate. An increased focus on operational activities such as civil engineering (e.g. road building), cultivation and forest residue utilisation (brash, stumps) is required to ensure disturbance losses are minimised.

61. Tree establishment usually requires ground preparation and in many cases the resulting soil disturbance may cause soil C loss. How much and for how long depends on the soil type (particularly initial soil C amount), the degree of disturbance and the environmental and site conditions. There are few reliable estimates for soil C loss during ground preparation, and this gap must be filled as this is a key question in all considerations of afforestation C balances and restocking practices.

62. While many forestry operations are mechanised, the amount of fossil fuel combustion is small per unit area of forest operation, or per unit volume of timber produced. The consequent impact of this fuel use in the forest on the net GHG balance of forestry is usually small.

63. The effect of tree harvesting intensity on fuel use figures require further improvement, and information on harvesting of trees in different forest types (e.g. SRF and CCF).

64. UK GHG emissions from timber transport out of the forest are estimated at approximately 0.10 $MtCO_2e$ y^{-1}, which is more than that estimated for harvesting and thinning (0.071 $MtCO_2e$ y^{-1}), but amounts to <1% of the net annual CO_2 uptake by UK forests.

65. Road building in forests is an important component of the net GHG balance. The annual fossil-fuel derived emissions caused from road construction and maintenance are estimated to be only 0.3–0.5% of the positive GHG contribution of tree growth for typical productive forests. However, potential losses due to soil disturbance could be much larger, although these are not presently well quantified.

66. These investigations suggest that there is potential for improvement in the C and GHG efficiency of forestry operations. This can be achieved by development and improved understanding of 'best practice' C and GHG management either through mechanical–engineering advances or improved planning, although not all forestry operations may be improved to a similar degree.

67. Energy use in timber processing is substantial, but poorly quantified at the moment.

6.10 Use and development of C accounting models

68. Much of the information on the components and processes underlying forest C and GHG balances has been used to inform the development of forest C accounting models, such as CARBINE and CSORT. The modelling effort has itself highlighted gaps in information available – either particular quantities, or on the controls on the rates of key processes.

69. The C accounting model CSORT is described in detail. It uses yield table data combined with allocation relationships to estimate C stocks in the trees, together

with a simple three pool model for soil C stock changes. The GHG emissions from forest operations are included, and the GHG balance effects of harvested wood products and their substitution benefits as woodfuel or timber are estimated.

70. The use of CSORT in this review to contrast the C and GHG balances of a 'minimum intervention' management option with a 'conventional thin' exemplified the complexity of understanding and summarising the resulting differences. It also highlighted that such comparisons cannot be made easily on simple single figures or 'bottom lines'. Analysis and comparison requires looking at parameters like average CO_2 uptake over time, and use of comparable periods, which are appropriate for the management options being considered.

71. There are several important components of CSORT that must be improved:

    a. better modelling of soil C changes during and after planting, and during the subsequent growth of the stand on the major soil types;
    b. the consequences of forest operations on soil C and GHG balance;
    c. modelling build-up and decomposition of litter during stand development on different soil types and the effects on soil C;
    d. incorporation of estimates of soil GHG fluxes in addition to soil C changes;
    e. modelling of other silvicultural management options;
    f. development of methods to model new species and silvicultural practices for which yield tables do not presently exist;
    g. improved modelling of changes in pre-existing vegetation C stocks during stand establishment and inclusion of woody understorey vegetation C stocks where appropriate.

6.11 Concluding points

Overall, this report is a significant step in collating and evaluating available knowledge, data and models in order to evaluate C and GHG balances for UK forests. While there is much that is well understood, there are significant gaps that need to be addressed if policies are to be correctly devised and forest management practices that maximise the effective contribution to national emissions abatement targets are to be deployed. In particular, much of our present information is derived from timber production forests, so our measurements of, and ability to understand and model other woodland types is limited. The report also highlights key evidence gaps on which to focus research resources. While the report could be widened further, reflecting the importance and intricacies of forestry C and GHG balances, this collation of information can be built on using the framework, datasets and modelling tools outlined here.

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Appendices

Appendix 1. Table of standing biomass in GB woodland

Table A1.1 Standing biomass in GB woodlands, in million oven-dried tonnes (M odt); from Table 3 from McKay *et al.* (2003).

Area	Species group	Biomass component											
		Stemwood of different diameters (cm)						Tips	Branches	Foliage	Roots	Stumps	Total
		7–14	14–16	16–18	18+	Poor quality	Total stem						
Scotland	Pines	6.38	2.52	2.06	6.92	0.51	18.39	0.77	4.66	2.10	6.48	0.49	32.88
	Spruces	11.38	5.19	4.40	14.25	0.14	35.35	1.15	10.54	4.78	16.85	0.96	69.62
	Other conifers	1.38	0.67	0.68	4.03	0.19	6.96	0.16	1.35	0.61	2.26	0.15	11.50
	Broadleaves	1.19	0.56	0.57	4.11	4.04	10.48	0.26	4.95	0.00	8.05	0.26	23.99
Total Scotland		20.33	8.94	7.72	29.30	4.89	71.18	2.34	21.50	7.48	33.64	1.86	137.99
Wales	Pines	0.39	0.17	0.14	0.52	0.00	1.22	0.04	0.27	0.12	0.37	0.03	2.06
	Spruces	2.01	0.95	0.87	3.47	0.00	7.30	0.21	1.93	0.87	3.17	0.17	13.65
	Other conifers	0.62	0.33	0.37	2.85	0.04	4.21	0.07	0.72	0.32	1.26	0.08	6.65
	Broadleaves	0.76	0.37	0.40	2.95	3.05	7.54	0.15	3.41	0.00	5.51	0.18	16.79
Total Wales		3.78	1.82	1.79	9.79	3.09	20.27	0.48	6.32	1.31	10.32	0.46	39.15
England	Pines	2.35	1.17	1.14	7.13	0.21	11.99	0.27	2.40	1.05	3.21	0.26	19.18
	Spruces	2.17	0.99	0.88	3.56	0.02	7.61	0.23	2.10	0.95	3.16	0.19	14.24
	Other conifers	1.23	0.67	0.76	6.83	0.34	9.83	0.14	1.73	0.75	2.82	0.18	15.46
	Broadleaves	5.89	2.76	2.96	25.20	4.42	41.24	0.94	18.77	0.00	30.76	0.97	92.68
Total England		11.64	5.58	5.74	42.72	4.99	70.67	1.58	25.00	2.75	39.95	1.60	141.56
GB Total		**35.75**	**16.34**	**15.24**	**81.81**	**12.97**	**162.12**	**4.40**	**52.82**	**11.54**	**83.91**	**3.91**	**318.70**

Appendix 2. The Forestry Commission soil classification system

Taken from Kennedy (2002)

Table A2.1 The Forestry Commission classification system for the main mineral and shallow peaty soils.

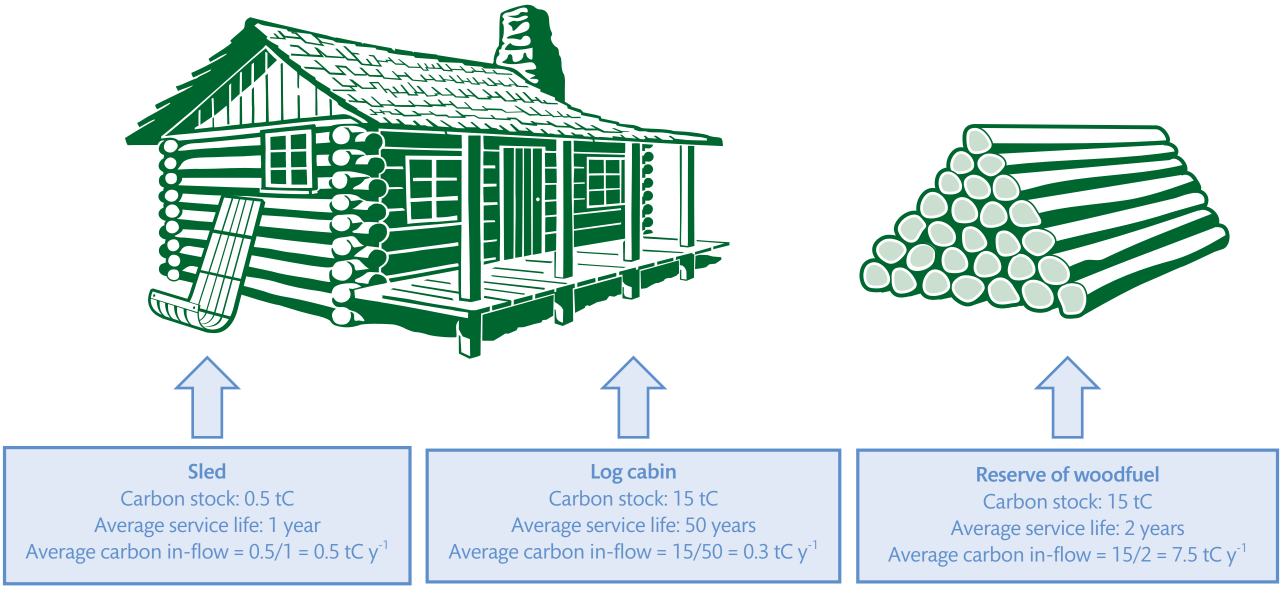
	Soil group	Soil type	Code
Soils with well aerated subsoil	1. Brown earths	Typical brown earth	1
		Basic brown earth	1d
		Upland brown earth	1u
		Podzolic brown earth	1z
	3. Podzols	Typical podzol	3
		Hardpan podzol	3m
	4. Ironpan soils	Typical ironpan soil	4
		Podzolic ironpan soil	4z
		Intergrade ironpan soil	4b
	12. Calcareous soils	Rendzina	12a
		Calcareous brown earth	12b
		Argillic brown earth	12t
Soils with poorly aerated subsoil / gleys	5. Ground water gley soils	Typical ground water gley	5
	6. Peaty (surface water) gley soils	Typical peaty surface water gley	6
		Podzolic peaty surface water gley	6z
	7. Surface water gley soils	Typical surface water gley	7
		Podzolic surface water gley	7z
		Brown surface water gley	7b

Table A2.2 The Forestry Commission classification system for deep peats.

	Soil group	Soil type	Code
Flushed peatlands	8. *Juncus* (or basin) bogs	*Phragmites* (or fen) bog	8a
		Juncus articulatus or *acutiflorus* bog	8b
		Juncus effusus bog	8c
		Carex bog	8d
	9. *Molinia* (or flushed blanket) bogs	*Molinia*, *Myrica*, *Salix* bog	9a
		Tussocky *Molinia* bog; *Molinia*, *Calluna* bog	9b
		Tussocky *Molinia*, *Eriophorum* bog	9c
		Non-tussocky *Molinia*, *Eriophorum*, *Trichophorum* bog	9d
		Trichophorum, *Calluna*, *Eriophorum*, *Molinia* bog (weakly flushed)	9e
Unflushed peatlands	10. *Sphagnum* (or flat or raised) bogs	Lowland *Sphagnum* bog	10a
		Upland *Sphagnum* bog	10b
	11. *Calluna*, *Eriophorum*, *Trichophorum* (or unflushed blanket) bogs	*Calluna* blanket bog	11a
		Calluna, *Eriophorum* blanket bog	11b
		Trichophorum, *Calluna* blanket bog	11c
		Eriophorum blanket bog	11d
	14. Eroded bogs	Shallow hagged eroded bog	14
		Deeply hagged eroded bog	14h
		Pooled eroded bog	14w

Appendix 3. The C dynamics in harvested wood products

Figure A3.1 A log cabin, a sled and a stock of woodfuel illustrate the relationships among carbon stocks, flows and the service lives of wood products.



Suppose that a family moves to a previously uninhabited area of forest. The family harvests some of the trees to make essential items from wood, as illustrated in Figure A3.1 (after Matthews et al., 2007) represented by three examples of harvested wood products requiring different amounts of wood to make, and lasting for different lifespans or service lives:

These properties of lifespan and amount can be used to estimate the size of C stocks in harvested wood products, but there can be complications. Critically, it should be realised that such calculations do not describe the processes that cause C stocks in wood products to increase or decrease. For example, it might be suggested that if the lifespan of wood products can be extended then the quantity of C in wood products should increase (e.g. if the lifespan of timber used to build the house in Figure A3.1 can be extended from 50 years to 100 years, then the C stock in houses should double). However, this does not take into account the actual requirement for that product. For example, the family illustrated in Figure A3.1 use just one house. If the house lifespan is doubled somehow to 100 years, it is unlikely that the family would build and maintain two houses instead of one. It is more likely that the family would replace their single house every 100 years instead of every 50 years. As a consequence, the number of houses and C stock stays the same, even though the lifespan has been doubled. In addition, the C flow into the wood products pool in houses would decrease, because less wood needs to be harvested to maintain the same number of houses over time.

Appendix 4. Volume, carbon content and density of timber

The carbon content of timber varies between approximately 45 and 55% of the oven dry weight and is conventionally taken as being 50% (Matthews, 1993). To estimate the carbon content of timber, assumptions need to be made on the relative humidity (RH), to allow the moisture content (MC) of timber to be determined. The convention is to assume a RH of 65%, which gives an equilibrium MC at 20°C of approximately 12%. The moisture content needs to be known so that the weight of the water in the timber can be calculated and used to correct the volume and density of the timber (which changes with MC due to shrinkage and swelling).

The values in Table A4.1 are based on those given by Lavers and Moore (1983). The density shown is the mean for the timber of each species grown in the UK at 12% MC, expressed in tonnes. The weight excluding moisture is based on the nominal specific gravity (NSG) at test as this is usually the oven dry weight of the timber for the volume at point of testing expressed as a ratio. Generally the testing was carried out at an MC of 12% so there is no need for correction in NSG to account for a difference in the volume of the timber when tested. However, for a few of the species the tests were carried out at slightly higher MC. In these cases the NSG was corrected to that for 12% MC using the procedure outlined in the Forest Products Laboratory *Wood Handbook* (1999). NSG on a 'green' basis allows for the swelling of the wood and so has a lower weight per unit of volume. It is used when converting standing timber volumes to biomass (and carbon), and has been used in several places in this report.

The contents in Table A4.1 are expressed in tonnes weight per cubic metre of timber at 12% MC (t m^{-3}). To convert to carbon the weight is divided by two; for CO_2 per cubic metre the carbon value should be multiplied by 44/12.

Table A4.1 Density (t m^{-3}) of common UK timber species (after Lavers and Moore 1983).

Species	Density of timber @12% MC	Density based on oven dried weight and	
		air dried volume	green volume
Sitka spruce	0.384	0.340	0.330
Norway spruce	0.400	0.350	0.330
Scots pine	0.513	0.460	0.420
Corsican pine	0.481	0.420	0.400
Douglas fir	0.497	0.440	0.410
Japanese larch	0.481	0.440	0.410
Hybrid larch	0.465	0.405	0.380
European larch	0.545	0.480	0.450
Ash	0.689	0.600	0.530
Birch	0.673	0.600	0.530
Oak	0.689	0.610	0.560

Appendix 5. Summary of measured forest soil GHG fluxes

Table A5.1 Greenhouse gas fluxes reported for UK forest soils from standing forests on mineral, organo-mineral and deep peat sites and from clearfelled and unafforested deep peat and other vegetation sites. Negative values indicate uptake by the soil, positive values indicate emissions.

Activity/Site	Soil type	Species or treatment	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	CO_2e (t ha^{-1} y^{-1})	References
Standing forest, mineral soils							
Straits Enclosure, Hampshire	Eutric vertisol gley	Oak	1.86	-3.24	25.06	25.5	Yamulki et al., (in prep.)
Wytham Woods, Oxford	Stagni-vertic cambisol with area of arenihaplic luvisols and calcaric cambisols	Mixed deciduous woodland.			22.66 (sum of soil and litter)		Fenn et al., 2010
Central Ireland	Low humic mineral gley	Sitka spruce 10, 15, 31 and 47-year-old stand			25.56 (mean of all ages)		Saiz et al., 2006b
		- undisturbed ground - ridges - furrow			25.7–48.5 21.3–34.1 35.2–36.9		Saiz et al., 2006a
Gisburn, Lancashire clearfelled and replanted in 1991	Cambic stagnogleys to stagnohumic gleys	Alder Oak Spruce Pine Mean forest		-0.67 -0.25 -0.52 -0.58 -0.51			McNamara et al., 2008
(Semi-natural soils in South and Central Scotland) Glencorse = = Dunslair Heights Devilla North Berwick	Silty loam, brown forest soil	 Sitka spruce Alder Birch Sitka spruce Pine Sycamore	 0.55 2.07 0.94 0.2 0.57 1.04				Skiba et al., 1996
Glencorse	Silty loam, brown forest soil		0.12–0.29				Kesik et al., 2005
7 UK forest and woodland sites				-0.1 to -9.1 Median: -2.4			Smith et al., 2000

Table A5.1 (Continued)

Activity/Site	Soil type	Species or treatment	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	CO_2e (t ha^{-1} y^{-1})	References
Standing forest, organo-mineral soils							
Harwood Forest, Northumberland		Grassland between two second rotation Sitka spruce stands					Mojeremane, Mencuccini and Rees, 2010
	Peaty gley soils	Drained Undrained Mounded Unmounded Fertilised Unfertilised		6.27 17.52 13.92 9.87 14.09 9.70			
Harwood Forest, Northumberland, 20–30 year and year 1–2	Fine loam over clay with peat surface horizon 35–50 cm		0.2–4.7	0.2–2.8	7.8–22.3	7.9–23.8	Ball, Smith and Moncrieff, 2007
Harwood Forest, Northumberland	Organic-rich peaty gley (histic gleysols)	40 year Sitka spruce	0.74	0.48	25.48	25.7	Zerva and Mencuccini, 2005b
Kershope, Cumbria	Peaty gley		2.0–4.1				Dutch and Ineson, 1990
Dunslair Heights, Peebleshire	Peaty podzol	Sitka spruce	0.49				Skiba *et al.*, 1996
Southeast Scotland	Montane soil very peaty with enhanced N deposition of 24.3 kg N ha^{-1} y^{-1}	Sitka spruce	0.8				Skiba, Fowler and Smith, 1994
Standing forest, deep peat							
Flanders Moss, Stirling	Deep peat (nutrient poor)	Lodgepole pine Drained Undrained	 0.76 0.68	 1.49 6.43	 16.57 12.31	 16.8 12.7	Yamulki *et al.* (submitted)
Cloosh Forest, Co. Galway, Ireland	Drained ombrotrophic blanket peatland 0.9 to 5.5 m depth	- recently planted Sitka spruce - lodgepole pine 19, 23, 27 and 33 years old - mature Sitka spruce			6.2 3.7–5.1 9.5		Byrne and Farrell, 2005
Auchencorth Moss, Midlothian	Drained peat	Beech	0.36				Skiba *et al.*, 1996
Clearfelled sites, organo-mineral							
Harwood Forest, Northumberland, year 1–2	Fine loam over clay with peat surface horizon 35–50 cm		0.7-2.0	6.8–18	23.7–26.0	24.1–27.1	Ball, Smith and Moncrieff, 2007

Table A5.1 (Continued)

Activity/Site	Soil type	Species or treatment	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	CO_2e (t ha^{-1} y^{-1})	References
Harwood Forest, Northumberland	Organic-rich peaty gley (histic gleysols)		2.16	7.92	18.15	19.0	Zerva and Mencuccini, 2005b
Clearfelled sites, deep peat							
Cloosh Forest, Co. Galway, Ireland	Ombrotrophic blanket peatland 0.9 to 5.5 m depth				5.1–5.9		Byrne and Farrell, 2005
Unafforested deep peatland/wetland							
Flanders Moss, Stirling	Deep peat (nutrient poor	Undrained–unplanted Restored	0.15 0.86	77.0 226.3	25.5 18.2	27.5 24.1	Yamulki *et al.* (submitted)
Auchencorth Moss, Midlothian	Acid peatland, 85% histosols 'peats'; 9% gleysol; 3% humic gleysol, 3% cambisol; peat depth <0.5 to >5 m with low-intensity sheep grazing and peat extraction	Riparian	Mean ±SE 0.34 ± 0.12	Mean ±SE 51.3 ± 27.2	Mean ±SE 39.4 ± 5.3	Mean 40.8	Dinsmore *et al.*, 2009
UK	Nutrient-poor bogs	Unmanaged wetland		13.3–53.3			Joosten and Clark, 2002
Other vegetation sites, organo-mineral							
Harwood Forest, Northumberland year 1–2	Fine loam over clay with peat surface horizon 35–50-cm	Unplanted grassland	0.3 0.3	1.2–2.6	33.1–55.8	33.2–56.0	Ball, Smith and Moncrieff, 2007
Great Dunfell, Cumbria	Peaty podzol	Grass/heather	0.41				Skiba *et al.*, 1996
Southeast Scotland	Montaine soil very peaty with enhanced N deposition of 24.3 kg N ha^{-1} y^{-1}	Moorland vegetation (*Calluna* sp., grasses, mosses)	1.86				Skiba, Fowler and Smith, 1994
Other vegetation sites, peatland							
Auchencorth Moss, Midlothian	Acid peatland, 85% histosols 'peats'; 9% glaysol; 3% humic gleysol, 3% cambisol; peat depth <0.5 to >5 m with low-intensity sheep grazing and peat extraction	Calluna Hollow Sedge/Hummock Juncus/Hummock	Mean ±SE 0.13 ± 0.29 -0.10 ± 0.13 0.18 ± 0.17 -0.06 ± 0.12	Mean ±SE 0.7 ± 0.5 1.8 ± 2.1 0.2± 0.6 0.4 ± 0.6	Mean ±SE 25.4 ± 3.5 21.0 ± 0.9 21.0 ± 1.8 23.4 ± 9.6	Mean 25.5 21.0 21.1 23.4	Dinsmore *et al.*, 2009
Northern England Auchencorth Moss	Peat Peat	Upland grass Grass/sphagnum	0.83 0.16				Skiba *et al.*, 1996

Table A5.2 Greenhouse gas fluxes reported for other European and worldwide forest soils (mainly temperate regions). Negative values indicate uptake by the soil, positive values indicate emissions.

Europe and worldwide	Soil type	Species	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	References
Organo-mineral						
Austria	Calcareous mountain forest soil; rendzic leptosols and chromic cambisol. High clay content in upper 10 cm	Spruce-fir-beech forest year 1 year 2	 0.47 0.64		 3.74 6.0	Kitzler *et al.*, 2006a
Netherlands		Mixed oak-beech forest	31.43			Tietema, Bouten and Wartenbergh, 1991
Denmark		Detailed study on N cycle in old beech forest.	0.79			Beier *et al.*, 2001
Austria	Moderately well drained dystric cambisol over sandstone	142-year beech forest N deposition = 20.2 kg N ha^{-1} y^{-1} 62-year-old beech N deposition = 12.6 kg N ha^{-1} y^{-1}	1.24 1.01		10.62 8.67	Kitzler *et al.*, 2006b
Denmark		Spruce forest Beech forest	1.21 1.26	1.21 0.49		Ambus and Christensen, 1995
Denmark		Small beech, drained	0.79	-1.25 to -3.78		Ambus *et al.*, 2001
Germany	Soil of temperate forest ecosystems	Spruce Douglas fir Pine Beech Mixed oak–beech etc	0.16 0.16 0.02 0.63 0.03			Papen and Butterbach-Bahl, 1999
European forest soils	Continental temperate Climate Lithosol Coarse soil Medium soil Fine Gleysol	Broadleaved or conifer Broadleaved Conifer Broadleaved or conifer = =		-0.47 4.12 -7.76 -2.27 -1.51 8.60		Lindner *et al.*, 2004
Germany	Upper mineral soil above an organic layer	Different experimental sites of the Höglwald Forest: Spruce, control Spruce, limed Beech		-2.43 -1.74 -6.45		Butterbach-Bahl, Rothe and Papen, 2002b

Table A5.2 (Continued)

Europe and worldwide	Soil type	Species	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	References
The Netherlands (Speulderbos)	Various forest soils	Douglas fir	0.32			Pilegaard *et al.*, 2006
Germany (Höglwald)		Spruce	1.24			
		Beech	1.79			
Austria (Schottenwald)		Beech	0.88			
Austria (Achenkirch)		Spruce	0.39			
Austria (Klausen-L.)		Beech	0.74			
Denmark (Sorø)		Beech	1.16			
Hungary (Matra mtns.)		Oak	2.79			
=		Spruce	2.04			
Italy (Parco Ticino)		Mixed	0.67			
=		Poplar	2.71			
Italy (San Rossore)		Pine	0.29			
Scotland (Glencorse)		Sitka spruce	2.05			
=		Birch	1.64			
Finland (Hyytiälä)		Scots pine	0.04			
Belgian Ardennes		Douglas fir (35 m tall)			14.67	Longdoz, Yernaux and Aubinet, 2000
Northeast Germany		Norway spruce 47 years old			25.67	Buchmann, 2000
Florence, Italy		Silver fir 75 years old			44.00	Certini *et al.*, 2003
Canada (British Columbia)	Temperate coniferous forest type	Douglas fir (55 years old) with smaller amounts of western red cedar and western hemlock			44.00	Lalonde and Prescott, 2007
Canada (British Columbia)		Douglas fir 55 years old			69.67	Drewitt *et al.*, 2002
USA (North Carolina)		Loblolly pine			51.33	Maier and Kress, 2000
USA (Florida)		Slash pine 11 years old			47.67	Ewel, Cropper and Gholz, 1987
		Slash pine 9 years old			29.33	
USA (Pennsylvania)		Black cherry-sugar maple forest	0.36	11.87	18.66	Bowden *et al.*, 2000
USA (North Carolina and Tennessee border)	Inceptisol with spodic characteristics as Dystrochrepts or Haplumbrepts	Uneven-aged forest with red spruce and Fraser fir	5.35			Tewksbury and Van Miegroet, 2007
New Zealand		Nothofagus forest	0.0	-10.57		Price *et al.*, 2004

Table A5.2 (Continued)

Europe and worldwide	Soil type	Species	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	References
Temperate locations		Review of coniferous forests			25.67	Raich and Schlesinger, 1992
Europe and worldwide organic						
Finland	Effect of environmental variables on GHG fluxes from 68 forestry-drained peatlands in Finland		-0.30 to 9.20	-9.70 to 125	11.65 to 44.37	Ojanen *et al.*, 2010
Finland	Peatlands drained for forestry	Nutrient-rich Nutrient-poor Eriophorum vaginatum communities Sphagnum forest moss community Continuously water-covered ditches in drained fen In bog Drained fen Drained bog Undrained fen Undrained bog		-2.1 to -0.7 67.40 19.10 4.80 420 to 1640 70.00 17 to 45 34.00 21 to 131 65.00		Minkkinen and Laine, 2006
Finland	Restoration of peatland by clear cutting the trees and block draining	Minerotrophic fen – rewetted – untreated Ombrotrophic bog – rewetted – untreated			15.15 25.84 8.85 14.76	Komulainen *et al.*, 1999
Netherlands	Abandoned agricultural peat meadow converted into a wetland nature reserve			416.93 to 430.27		Hendriks *et al.*, 2007
European soils reviewed from different authors						
South Sweden	Organic forestlands/ sites with varying effect of water table	Different conifer sites (water table range 12–70 cm)	1.0 to 29.0 mean 9.78 (n=9)	-1.1 to 47.3 mean 14.64 (n=26)	5.13 to 65.16 mean 14.35 (n=33)	von Arnold *et al.*, 2005a
		Deciduous sites (water table range 15–129 cm)	0.0 to 2.2 mean 0.55 (n=24)	-5.2 to 17.2 mean 2.42 (n=15)	14.6 to 27.6 mean 19.55 (n=7)	
	Histosols	Alder drained Birch drained Alder undrained	9.00 2.00 1.00	9.00 9.00 76.00	17.00 19.00 10.00	
	Histosols	Drained spruce with young trees Drained spruce with old trees Drained pines Undrained mire	0.80 0.50 0.40 0.30	0.30 3.00 11.00 114.00	14.00 12.00 15.00 10.00	

Table A5.2 (Continued)

Europe and worldwide	Soil type	Species	N_2O (kg ha^{-1} y^{-1})	CH_4 (kg ha^{-1} y^{-1})	CO_2 (t ha^{-1} y^{-1})	References
Germany	Histosols	Peatland restored Peatland drained	7.94 to 10.1 -0.69	-1.20 to -0.6 1006.00	15.10 to 15.8 18.30	Meyer, 1999
Sweden Boreal sites	Whole country	Ombrotrophic (nutrient-poor bogs) Minerotrophic (nutrient-rich fens)		49.33 120.00		Nilsson *et al.*, 2001
Finland		Ombrotrophic (nutrient-poor bogs) Minerotrophic (nutrient-rich fens)	0.18	133 to 160 100 410.7		Minkkinen *et al.*, 2002 Minkkinen *et al.*, 2002 Joosten and Clark, 2002
Germany		Ombrotrophic (nutrient-poor bogs) Minerotrophic (nutrient-rich fens)	-0.02	232.00 290.67		Drössler, 2005
Sweden Finland Germany	Effect of C:N ratio on N_2O emissions from forested sites (n) on drained peat	Coniferous (n=5) Birch Deciduous Pine (n=3) Alder	0.4 to 9.0 30 6.60 0.05 to 0.5 11.00			Klemedtsson *et al.*, 2005

Appendix 6. CSORT soil C sub-model

A6.1 Introduction

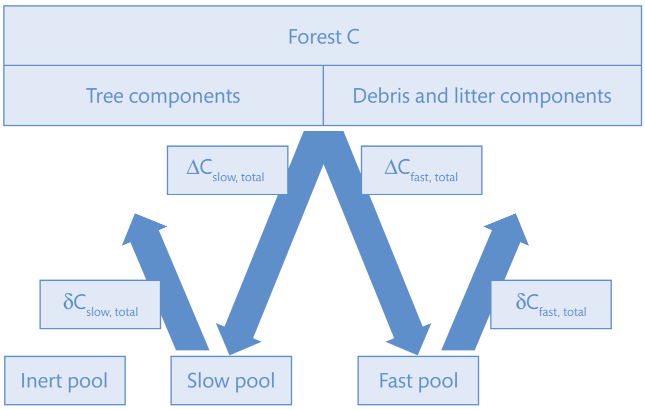
This Appendix describes in full the soil C sub-model of CSORT used in the calculation of C balances for forest management cycles (see Chapter 5). Values are quoted for parameters as they are introduced in equations, and the rationale for the selection of these values is presented in a subsequent section. For consistency with the model, soil organic carbon (SOC) stocks are in tC ha^{-1}, and fluxes in tC ha^{-1} y^{-1}. Where the model is used within forest stand simulations, outputs are presented in tCO_2e.

A6.2 Model structure

The approach to soil C modelling is kept as simple as possible. The model consists of three SOC pools (Figure A6.1) representing different turnover rates: 'fast', 'slow' and 'inert' (i.e. no turnover). Dynamics of soil C are represented using an annual time step.

A single soil property, soil texture, is distinguished as characterising the main differences in the levels and rates of change of SOC. Four broad categories of soil texture are defined: sand, loam, gley, peat/high peat gley.

Figure A6.1 Outline of simplistic model of soil carbon dynamics. Tree component pools are foliage, branch, stem, stump and root; debris and litter components are coarse woody debris, fine woody debris and non-woody litter.



Fast soil carbon pool

The C input to the fast soil C pool at any time step t ($\Delta C_{fast,total}(t)$) is assumed to be related to the total C stock in living trees and in litter (including dead biomass):

$$\Delta C_{fast,total}(t) = \Delta C_{fast,trees}(t) + \Delta C_{fast,litter}(t) \quad [1]$$

where: $\Delta C_{fast,trees}(t)$ is the rate of C input to the fast pool from biomass in standing trees and $\Delta C_{fast,litter}(t)$ is the rate of C input to the fast pool from debris and litter (equation 2).

The value of $\Delta C_{fast,litter}(t)$ is assumed to be directly proportional to the total mass of C in litter at the time step, i.e. the sum of C stocks in coarse woody debris, fine woody debris and non-woody debris, with no distinction being made between the three categories of debris for soil C:

$$\Delta C_{fast,litter}(t) = kf_1\ (C_{CWD}(t) + C_{FWD}(t) + C_{NWD}(t)) \quad [2]$$

where: $C_{CWD}(t)$ is the C stock in coarse woody debris at time step t years; $C_{FWD}(t)$ is the C stock in fine woody debris; $C_{NWD}(t)$ is the C stock in non-woody debris and kf_1 is a parameter. The value of parameter kf_1 is set at 0.15 independent of soil texture.

The value of $\Delta C_{fast,trees}(t)$ is assumed to be related to the total C stock in standing trees at the time step, i.e. the sum of C stocks in roots, stumps, stems, branches and foliage with no distinction made between the five categories of biomass. The relationship between the input to the fast pool and tree C stocks is assumed to follow an exponential approach to a maximum specified by kf_2:

$$\Delta C_{fast,trees}(t) = kf_2\ (1 - \exp\ (-kf_3\ C_{TREE}\ (t))) \quad [3]$$

where: $C_{TREE}(t) = C_{FOLIAGE}(t) + C_{BRANCH}(t) + C_{STEM}(t) + C_{STUMP}(t) + C_{ROOT}(t)$
and kf_2 and kf_3 are parameters. The variables $C_{TREE}(t)$, $C_{FOLIAGE}(t)$, $C_{BRANCH}(t)$, $C_{STEM}(t)$, $C_{STUMP}(t)$ and $C_{ROOT}(t)$ refer to C stocks (tC ha^{-1}) at time step *t* years in tree components with obvious meanings. The values of parameters kf_2 and kf_3 are set at 4.0 and 1.8, respectively, independent of soil texture (see below).

The C loss from the fast pool at any time step t ($\delta C_{fast,total}(t)$, tC ha^{-1} y^{-1}) was assumed to be related to the C stock in the pool at that time step:

$$\delta C_{fast,total}(t) = kf_4\ C_{fast}(t) \quad [4]$$

where: kf_4 is a parameter, with a value of parameter that depends on soil texture as given in Table A6.1.

Slow soil carbon pool

The C input to the slow soil C pool at any time step t ($\Delta C_{slow,total}(t)$) was assumed to be related to the total C stock in living trees and in litter (including dead biomass) as with the fast pool:

$$\Delta C_{slow,total}(t) = \Delta C_{slow,trees}(t) + \Delta C_{slow,litter}(t) \qquad [5]$$

where the variables have similar meanings to those in equation 1, but for the slow C pool. The values of $\Delta C_{slow,litter}(t)$ and $\Delta C_{slow,trees}(t)$ were assumed to be determined by the same sort of relationships as defined for $\Delta C_{fast,litter}(t)$ and $\Delta C_{fast,trees}(t)$ (see equations 2 and 3, respectively) but the values of parameters ks_1 and ks_2 were different and were assumed to depend on soil texture as shown in Table A6.1. The value for parameter ks_3 was set to the same value as used to describe the fast pool (i.e. 1.8).

Table A6.1 Assumed values of parameters kf_4 (fast soil C pool), and ks_1 and ks_2 (slow soil C pool).

Soil texture	Parameter		
	kf_4	ks_1	ks_2
Sand	0.1	0.002	0.057
Loam	0.06	0.0037	0.1
Gley	0.03	0.0075	0.2
Peat/high peat gley	0.03	0.0075	0.2

The C loss from the slow pool at any time step t ($\delta C_{slow,total}(t)$, tC ha^{-1} y^{-1}) was assumed to be determined by the same sort of relationship as for the fast pool (see equation 4) and parameter ks_4 was assumed to take a value of 0.00015, independent of soil texture.

Inert soil carbon pool

The inert soil C pool was taken to contain a characteristic, constant quantity of SOC dependent on soil texture (Table A6.2), with the pool neither gaining nor losing C due to specific site conditions, vegetation dynamics or management. These sizes of soil C stock have been reported for arable soils. A small increase in the C stock from sand to loam to gley/peaty gley was assumed on the basis that higher C stocks are generally associated with finer soil textures.

Table A6.2 Assumed values of SOC stock in the inert soil carbon pool.

Soil texture	Carbon stock (tC ha^{-1})
Sand	10
Loam	15
Gley	20
Peat/high peat gley	20

A6.3 Model initialisation and selection of model parameters
Values needed to be chosen for the parameters:

- kf_1, kf_2, kf_3, ks_3, ks_4 (single values)
- kf_4, ks_1, ks_2 (by soil texture category)

and for pool initial values:

- total soil C stocks under grassland (by soil texture category)
- total soil C stocks under arable land (by soil texture category)
- C stock in the inert pool (by soil texture category)
- soil C inputs for soils under arable land
- soil C inputs for soils under grassland or pasture (by soil texture category).

The rationale for the selection of parameter values was simplistic, with an emphasis on tuning the model to produce projections of soil C dynamics that conformed reasonably well to what has been reported in the literature. This was achieved by manual variation of the parameters, broadly following the procedure described below.

Many of the CSORT model simulations involve the establishment of new woodland on land previously under some other vegetation and use. Two types of previous vegetation/use were considered: grassland or pasture, and arable cropping. The initialisation of soil C stocks prior to forest establishment assumed that C stocks in the fast and slow pools had achieved steady state, i.e. inputs of soil C were balanced by outputs.

For soils subject to arable cropping, the steady-state soil C stocks in the fast and slow pools were assumed to be zero, and soil C stock was equal to the inert soil pool (see Table A6.2) only. Inputs of C to arable soils were assumed to be zero as the model time step is one year, and most arable crops are grown on an annual cycle, so while there may be exchanges of soil C between vegetation, soil and the atmosphere associated with crop growth, these are taken to be zero over a full year.

For soils under grassland or pasture, the steady-state soil C stocks for the four soil texture categories of sand, loam, gley and peat/high peat gley were selected to result in predictions of steady-state C stocks close to observed values of 60, 100, 150 and 250 tC ha^{-1}, respectively. These were based on estimates reported in the literature. Note that for deep peats (not necessarily under grass), a C stock of 250 tC ha^{-1} (917 tCO_2 ha^{-1}) represents a significant underestimate of potential C stocks, but afforestation of deep peats was considered not relevant to contemporary afforestation policy options. For soil textures of sand, loam and gley, the fast pool was set at approximately 70% of the total stock in the fast and slow pools combined. For peat and peaty gley, this was set at approximately 80%. Given these proportions, the values for annual inputs of soil C needed to give total C stocks (including the inert pool, specified in Table A6.2) around the values specified above for each soil texture are given in Table A6.3.

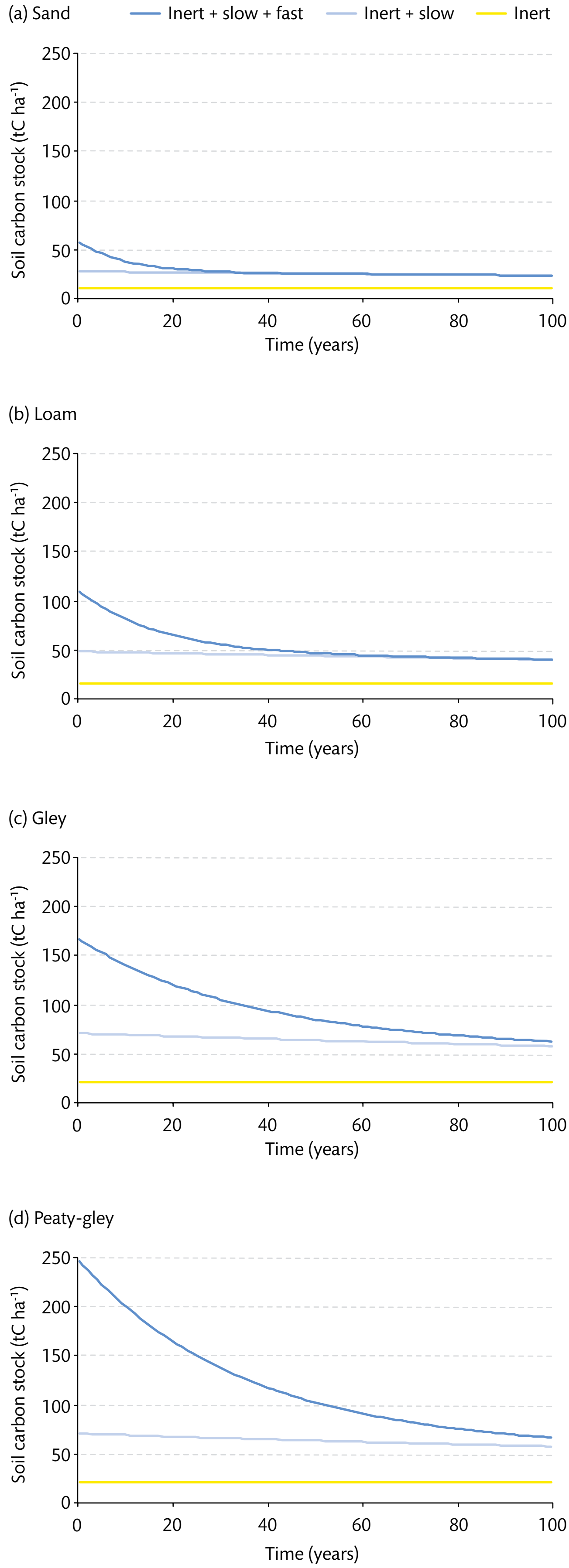
Table A6.3 Assumed values of initial carbon inputs to soils under grassland or pasture prior to forest establishment.

Soil texture	Soil C input (tC ha^{-1} y^{-1})		Soil C stock (tC ha^{-1})	
	Fast pool	Slow pool	Fast pool	Slow pool
Sand	3.5	0.05	15	35
Loam	4.0	0.1	25	60
Gley	3.0	0.15	40	90
Peat/high peat gley	5.0	0.15	45	185

Parameters kf_4 and ks_4

Parameters ks_4 was set at 0.00015 and kf_4 varied with soil texture as given in Table A6.1. The literature suggests that when relatively undisturbed soils (such as grassland/pasture) are converted to arable land, there is a loss of soil C, with C stocks eventually reaching a steady state at a much lower level. As discussed above, this lower level was taken to be the same as the C stock in the inert pool. Thus C stocks in the fast and slow pools were assumed to be completely lost if the land was maintained under arable management for long enough. The literature suggests that the bulk of the soil C loss takes place over the first few decades (say 20 to 40 years), with the rate of loss being faster in soils with more coarse texture (such as sands). Loss of soil C then continues at a much slower rate (perhaps over very many decades) until the new steady state is reached. The values for parameters kf_4 and ks_4 were selected to give this pattern of soil C dynamics if pasture/grassland was converted to arable land (i.e. by 'switching off' inputs of soil C), as illustrated in Figure A6.2.

Figure A6.2 Predicted time course of soil C stocks in soil textures of (a) sand, (b) loam, (c) gley and (d) peat/peaty gley as a result of conversion of pasture/grassland to arable land. These dynamics are determined largely by values of parameters kf_4 and ks_4.



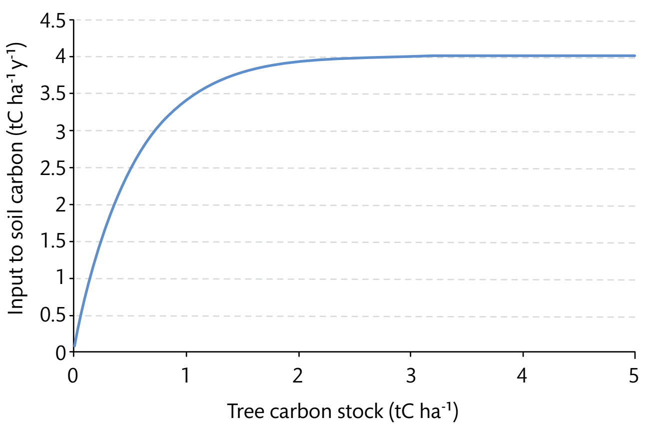
Parameters kf_2, ks_2, kf_3 and ks_3

The values of parameters kf_2, ks_2, kf_3 and ks_3 determine the shape of the relationship between the input of soil C due to tree biomass and the C stock in those trees (equation 3). Parameter kf_2 was set at 4.0, and the value of ks_2 varied with soil texture, as given in Table A6.1. The following principles were used in selecting values.

- The value of ks_2 should be a lot smaller than kf_2, perhaps around 1% of this value.
- The steady-state C stocks ultimately obtained as a result of forest establishment should be in the region of 60, 100, 150 and 250 tC ha^{-1}, respectively for soil textures of sand, loam, gley and peat/peaty gley.
- The change in steady-state C stocks from the previous land use (pasture/grassland/arable) to the new land use (arable) should take between 30 and 100 years (Figure A6.2).

Figure A6.3 illustrates how the values chosen result in maximum inputs to the soil C pools occurring when the tree C stock exceeds 2 tC ha^{-1}, a relatively low value, as tree C stocks can easily exceed 200 tC ha^{-1}. The saturation of soil C inputs at a relatively low value is based on the assumption that the magnitude of the inputs will be related to the establishment of an effective root system by the trees, with the main input being due to fast turnover of fine roots. This is assumed to be achieved relatively early in the growth of a stand of trees.

Figure A6.3 Assumed relationship between $\Delta C_{\mathrm{fast,trees}}(t)$ and total tree carbon stocks.



Parameters kf_1 and ks_1

Parameter kf_1 was set at 0.15, and ks_1 varied with soil texture as given in Table A6.1, using the same principles for selecting values as for parameters kf_2 and ks_2.

Ultimately, the aim was to select values empirically in order to obtain projections of soil C which were consistent with the literature, notably when considering transformations between land use such as pasture and grassland to arable (Figure A6.2) and either arable or pasture and grassland to woodland (Figures A6.4–5).

Figure A6.4 Predicted time course of soil C stocks in soil textures of sand, loam, gley and peat/peaty gley as a result of conversion of pasture/grassland to woodland. Woodland type: Sitka spruce, yield class 12, 2 m spacing, MT thinning, rotation of 60 years.

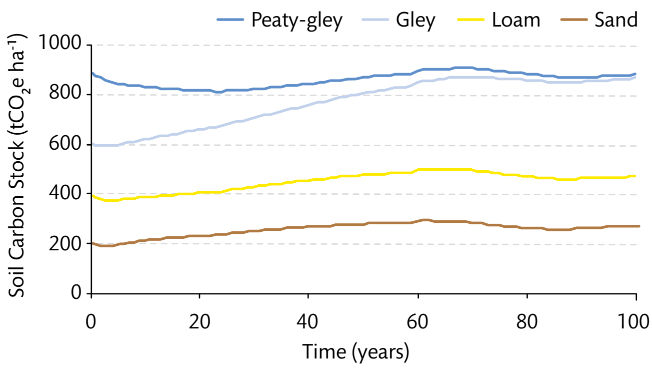
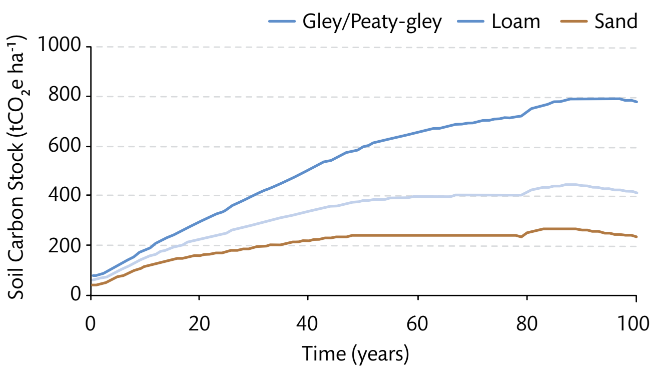


Figure A6.5 Predicted time course of soil C stocks in soil textures of sand, loam, gley and peat/peaty gley as a result of conversion of arable land to woodland. Woodland type: sycamore, ash and birch, yield class 6, 1.5 m spacing, MT thinning, rotation of 80 years. Note: Gley and peaty gley are indistinguishable.



Appendix 7. CSORT calculations of forest operations and wood processing

This appendix gives in more detail the values assumed in calculating GHG emissions from operations, substitution etc. In some cases emissions are in GHGs other than CO_2, in which case conversions are made using the values below; simulations outputs are in tCO_2e (tonnes carbon equivalent).

Some differences in operation emissions occur between high forest stands, short rotation forests (SRF) and short rotation coppice (SRC). Table A7.1 describes the operational emissions for all scenarios, and annotates the exceptions of SRF and SRC in the notes column. Another notable difference is that after felling, urea is only applied to conifer stumps.

General:

Atomic masses: H = 1, C = 12, N = 14, O = 16

1 oven dry tonne (odt) of biomass contains 0.5 tC

1 tCO_2 contains 12/44 tC, i.e. 0.273 tC

Fuel:

FuelMass (Diesel) = 0.8532 kg per litre

FuelMJ (Energy) = 45.46 x FuelMass (= 38.79 MJ per litre)

GHG equivalents:

kg CH_4 x 25 (kg CO_2e)

kg N_2O x 298 (kg CO_2e)

Direct emissions:

CO_2 = FuelMass x 3.1186 (kg CO_2)

CH_4 = FuelMass x 27.3 x10^{-6} (kg CH_4)

N_2O = FuelMass x 25.6 x10^{-6} (kg N_2O)

Indirect emissions:

CO_2 = 0.0081 x FuelMJ (kg CO_2)

CH_4 = 21.0 x 10^{-6} x FuelMJ (kg CH_4)

N_2O = 26.0 x 10^{-9} x FuelMJ (kg N_2O)

It should be noted that in Table A7.1, a number of cited references are as 'pers. comm.' Many of these are variable amounts (e.g. fuel consumption, which will vary from machinery to machinery, operator to operator etc. Although in places the values are broken down, values usually lie within accepted ranges of values. Similarly emission values may be cited as direct emission values (e.g. CO_2e per litre of diesel) or as total litres, from which we subsequently evaluate total emissions.

Table A7.1 Calculation of GHG emissions attributable to forest operations over forest management cycles.

Operation	Notes	Calculations
Establishment		
Type A and Type B road construction	High Forest, SRF only (no roads in SRC)	Construction emission: A and B roads: 42 121 kg CO_2e km^{-1} A-road density: 0.006 km ha^{-1} B-road density: 0.01 km ha^{-1} (C. Whittaker, pers. comm.)
Ground preparation	Forest	Mowing and ploughing of pasture/arable land: 21 litres fuel (Elsayed, Matthews and Mortimer, 2003)
Ground preparation	SRC, SRF	Coppice, SRF: Ploughing: 2 passes at 30 litres diesel ha^{-1} each (Matthews *et al.*, 1994)
Fencing		Based on a rectangular shape enclosing 5 hectares TotalFenceLength = 3.2 x √(5 x 1000/0.6) metres Scale to per hectare basis FenceLength = TotalFenceLength/5 Volume of wood consumed for fence (VolWood) VolWood = 0.00517 x FenceLength (m^3 ha^{-1}) Mass of steel wire consumed per ha of fence (MassWire) as: MassWire = 1.4 x FenceLength (kg ha^{-1}) Indirect emissions: GHG emissions from wood in fencing (CF1): CF1 = 0.001 x VolWood x FenceLength (tC ha^{-1}) Emissions from steel wire in fencing (CF2): CF2 = 1.72 x MassWire / 1000 (tC ha^{-1}) BEAT2 (Bates et al. 2011) Total indirect emissions due to fencing per hectare: Total emissions = CF1 + CF2 (tC ha^{-1})

Table A7.1 (Continued)

Operation	Notes	Calculations
Herbicide application		Direct emission: Diesel consumption for herbicide broadcasting 0.75 litres ha^{-1} for forests, 2 passes for coppice. Indirect emissions: Herbicide application of 1.62 kg Active Ingredient (AI) ha^{-1}. CO_2 emission factor of 4.613 kg CO_2 per kg AI CH_4 emission factor of 0.01667 kg CH_4 per kg AI N_2O emission factor of 0.000153 kg N_2O per kg AI
Fertiliser		Mass of fertiliser per hectare (MassF) = 0.000075 t x No. trees Nitrogen applied (N) = 0.21 x MassF Phosphorous applied (P_2O_5) = 0.14 x MassF Potassium applied (K_2O)= 0.07 x MassF CO_2 emissions per t: N, P_2O_5, K_2O = 1.904, 0.7, 0.453 tCO_2 CH_4 emissions per t: N, P_2O_5, K_2O = $3.6x10^{-3}$, $2.3x10^{-5}$, $2.1x10^{-5}$ tCH_4 N_2O emissions per t: N, P_2O_5, K_2O = 0.0183, $4.2x10^{-5}$, $9.4x10^{-6}$ tN_2O BEAT2 (Bates et al. 2011)
Planting		Forest: 126 kg CO_2 per 1000 seedlings Coppice plants: 0.0011 tC per cutting (Matthews *et al.*, 1994)
Cut-back (coppice)	SRC only	Initial cut-back of coppice, 25 litres diesel ha^{-1} (Matthews *et al.*, 1994)
Beat up		
Restock	Forests: 20% restock	Forests: same material as establishment (126 kg CO_2 per 1000) (Matthews *et al.*, 1994)
Restock	SRC: 5% restock	Coppice: material grown on, 0.0047 tC per sett (Matthews *et al.*, 1994)
Herbicide application		Direct emissions: Zero (manual spot application) Indirect emissions: Herbicide application of 0.48 kg Active Ingredient (AI) ha^{-1} CO_2, CH_4, N_2O emission factors per kg AI as above
Early thins		
Felling and extraction		Felling of trees (by harvesting machine) 0.4 litres diesel fuel m^{-3} of stemwood felled Extraction 0.9 litres diesel fuel m^{-3} of stemwood extracted (M. Perks, pers. comm.)
Chipping at roadside		1.0 litre diesel fuel consumption per odt of material being chipped (Elsayed, Matthews and Mortimer, 2003)
Harvest, carting, chipping	SRC	4.9 litres fuel per odt (harvested and chipped) (I. Tubby, pers. comm.)
Stump treatment	Conifers only	Approximate basal area: use stem volume being thinned (VolThin) and multiply by a factor (Fact) to give approximate stump basal area (StumpBasal) StumpBasal = Fact x VolThin The factor is related to tree age at the time of the fell, Fact = 0.1 + 0.9 x 0.95^{AGE} Urea: 2.5 litres applied to this stump basal area (K.V. Tubby, pers. comm.) Concentration of solution is 0.37 kg per litre (K.V. Tubby, pers. comm.), giving: Quantity of urea applied per hectare Q_U = 0.37 x 2.5 x BasalArea Equivalent quantity of nitrogen Q_N applied per hectare as Q_N = 0.4665 x Q_U kg N GHG emission factor for urea of 0.3627 kg C-equivalent per kg N-equivalent of urea BEAT2 (Bates et al. 2011) Emissions = 0.3627 x Q_N kg C per hectare

Table A7.1 (Continued)

Operation	Notes	Calculations
Main thins		
Felling, conversion and extraction		Assume felling and conversion of trees consumes 1.55 litres diesel fuel m^{-3} of stemwood extracted (0.9 litres) = 2.54 litres (M. Perks, pers. comm.)
Stump treatment		As for early thins
Final felling		
Felling, conversion and extraction		As for main thins
Brash baling		Assume 5 litres diesel consumption per odt of bale (I. Tubby, pers. comm.)
Brash-bale extraction		Assume extraction of brash bales consumes 4 litres diesel fuel per odt of brash extracted in bales (I. Tubby, pers. comm.)
Stump treatment	Conifers only	As previous thins
Grubbing up	SRC only	31.5 litres fuel per hectare (Matthews *et al.*, 1994)
Transport		
Chip transport	Transport distance (one way: 150 km)	Load density 40% by volume, material density 97% of wood odt density Based on 44-tonne lorry at full capacity: Payload volume 69 m^3; Payload weight 28.5 t Fuel consumption: 0.38 litres km^{-1} (C. Whittaker, pers. comm.; MTRU, 2007)
Saw-log transport	Transport distance (one way, 100 km)	Based on 44-tonne lorry at full capacity: emissions as above
Round-wood transport	Transport distance (one way, 150 km)	Load density 65% by volume Based on 44-tonne lorry at full capacity: emissions as above
Brash-bale transport	Transport distance (one way, 200 km)	Load density 70% by volume Based on 44-tonne lorry at full capacity: emissions as above
Off-site processing		
Processing of timber into sawnwood		Emissions: 0.142 tCO_2e m^{-3} of final product (i.e. sawn timber going into primary use) (R. Matthews, pers. comm.)
Processing timber into board		Emissions: 0.612 tCO_2e per odt of final product (i.e. board going into primary use) (R. Matthews, pers. comm.)
Brash-bale chipping (at mill)		0.9 litres diesel per odt of bale (I. Tubby, pers. comm; Elsayed, Matthews and Mortimer, 2003 – assuming bale density 70–80% of solid material)
Chip processing/ conversion	Proportion of biomass going to co-firing rather than to small-scale heat (20%)	Co-firing: For a given quantity of biomass (chips) delivered to power station (odt): Emissions = 0.0025 x Biomass tC Small-scale heat: For a given quantity of biomass (chips) delivered to heating system (odt): Emissions = 0.01383 x Biomass tC For a given quantity of biomass (pellets) delivered to heating system (odt): Emissions = 0.03136 x Biomass tC (Elsayed, Matthews and Mortimer, 2003)
Ongoing Maintenance		
Roads	Re-grade (if not resurface) (zero in SRC)	A-roads, two visits per year, 7269 kg CO_2 equivalent km^{-1} per visit B-roads, each thin/fell 314 kg CO_2 km^{-1} (C. Whittaker, pers. comm.)
	Re-surface (if not re-grade) (zero in SRC)	A-roads on a 10-year cycle, 17 953 kg CO_2 equivalent km^{-1} B-roads every third thin/fell, 7049 kg CO_2 equivalent km^{-1} (C. Whittaker, pers. comm.)

Appendix 8. Example CSORT C balances for forest management cycles

Below are the results for example forest management cycles (FMCs), additional to those presented in Sections 5.4 and 5.5. Graphs A8.1–A8.3 show, on an annual time step, the cumulative forest C and GHG balances of example forest management cycles. To permit comparison, a common time horizon of 200 years has been used. All of the examples assume that an existing land use is changed and new woodland is created at time zero. However, for management examples on a rotation of less than 200 years, results are illustrated for new woodland creation for the first rotation and then restocking for subsequent rotations. The same results are then presented as summary tables in Tables A8.2–A8.12.

Table A8.1

Species	Yield class	Soil type	Management regime	Rotation (years)	Figure number
Sitka spruce	12	Peaty gley	Thin and fell	50	A8.1
Oak	4	Brown earth	Minimum intervention	200	5.14 earlier
Oak	4	Brown earth	Thin and fell	80	A8.2
Oak	6	Brown earth	Thin and fell	80	A8.3a
Oak	6	Brown earth	Thin and fell	100	A8.3b
Oak	6	Brown earth	Thin and fell	150	A8.3c

Notes: (1) Previous land/cover use assumed to be heather in all cases involving Sitka spruce; grassland in all cases involving oak, both treated as 'pasture'; (2) assumed established trees per hectare: 2500 for Sitka spruce, and 6750 for Oak.

Key to the components shown in Figures A8.1–A8.3. Values are shown as cumulative across components, starting with the forest operations emissions values, which are barely visible as they are close to the zero line.

- Plus soil
- Plus debris/litter
- Plus trees
- Plus harvested wood products
- Plus substitution (fuel and materials)
- Forest operations

Figure A8.1 Time course of GHG balance of forest, set as YC 12 Sitka spruce, on a peaty gley soil, over a 200 year period, managed with thinning and felling on a 50 year rotation.

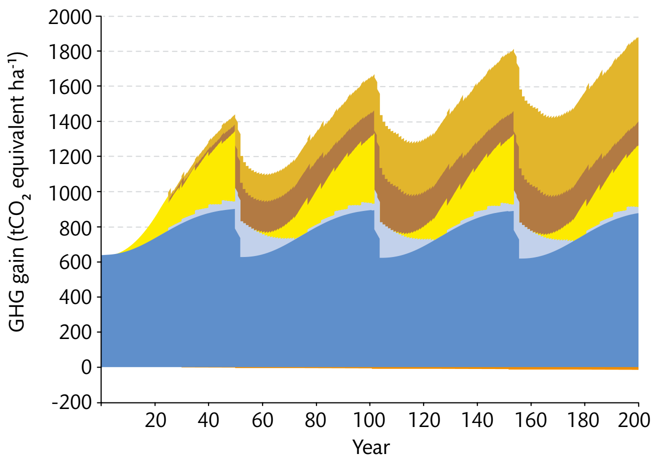


Figure A8.2 Time course of GHG balance of forest, set as YC 4 oak, on a peaty gley soil, over a 200 year period, managed with thinning and felling on an 80 year rotation.

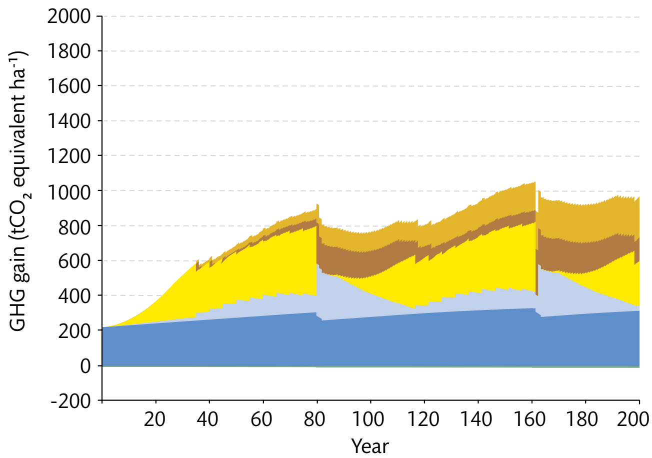
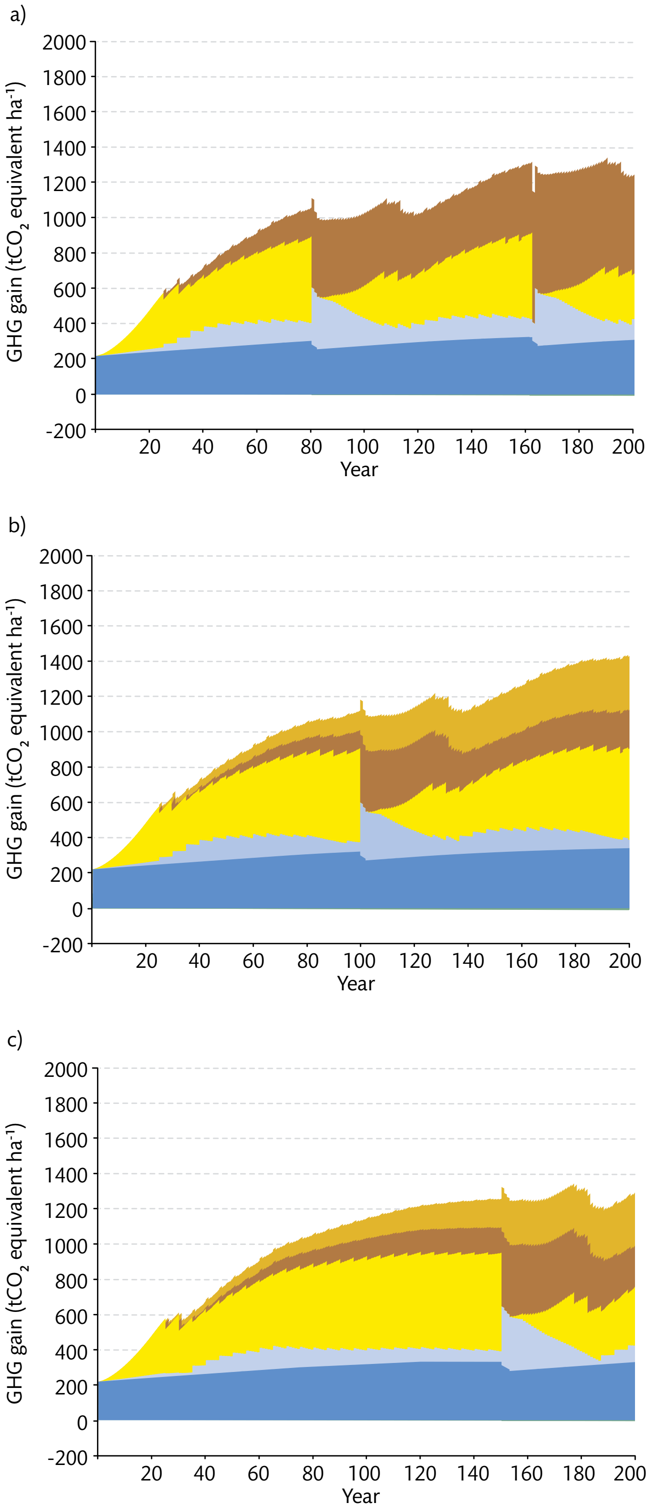


Figure A8.3 Time course of GHG balance of forest, set as YC 6 oak, on a peaty gley soil, over a 200 year period, managed with thinning and felling on a rotation of (a) 80 years, (b), 100 years, and (c) 150 years. Values are shown as cumulative across components, starting with the forest operations emissions values which are barely visible as they are close to the zero line.



The following tables give (a) the carbon stocks (or, in the case of forest operations/substitution, cumulative emissions/reductions) for each phase within the management cycles, and (b) the average source/sink over the specified phase. For in-forest carbon stocks, the estimates in the tables are based on long-term averages of stock estimates, in order to emphasise the long-term impacts of different options over short-term fluctuations in stocks. For those examples involving more than one rotation of a specified management cycle, it was possible to prepare separate tables based on the first rotation and the second rotation.

Table A8.2 Total GHG balance of a forest management cycle involving first rotation YC 12 Sitka spruce managed for thinning and felling on a 50 year rotation. Time-averaged (a) C stocks and (b) rates of C stock changes in key components. (c) and d) as (a) and (b) but for second rotation. Positive values indicate sequestration, negative values indicate emission.

First rotation

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-25	25-50	-	0-50	-
Period	5	20	25	-	50	-
Trees	1.9	64.9	169.1	-	169.1	169.1
Other vegetation	-	-	-	-	-	-
Debris /litter	0.4	2.8	22.3	-	22.3	22.3
Soil	799.2	723.5	791.6	-	791.6	-125.1
Harvested wood products	0.0	0.9	12.9	-	12.9	12.9
Forest operations	-0.1	-0.4	-4.2	-	-4.2	-4.2
Substitution	0.0	14.1	148.5	-	148.5	148.5
Total	**801.5**	**805.8**	**1140.1**	**-**	**1140.1**	**223.5**

(b) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-25	25-50	-	0-50
Period	5	20	25	-	50
Trees	0.4	3.1	4.2	-	3.4
Other vegetation	-	-	-	-	-
Debris /litter	0.1	0.1	0.8	-	0.4
Soil	-23.5	-3.8	2.7	-	-2.5
Harvested wood products	0.0	0.0	0.5	-	0.3
Forest operations	0.0	0.0	-0.2	-	-0.1
Substitution	0.0	0.7	5.4	-	3.0
Total	**-23.0**	**-0.2**	**13.4**	**-**	**4.5**

Second rotation

(c) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	50-57	57-77	77-102	-	50-102	-
Period	7	20	25	-	52	-
Trees	148.5	130.9	165.8	-	165.8	-3.3
Other vegetation	-	-	-	-	-	-
Debris /litter	43.0	47.2	45.6	-	45.6	23.3
Soil	775.7	755.6	780.7	-	780.7	-10.9
Harvested wood products	36.0	72.1	90.5	-	90.5	77.7
Forest operations	-4.3	-4.6	-8.5	-	-8.5	-4.2
Substitution	148.5	162.6	297.0	-	297.0	148.5
Total	**1147.4**	**1163.8**	**1371.2**	**-**	**1371.2**	**231.0**

(d) Rates of C stock changes (tCO_2e ha^{-1} y^{-1})

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	50-57	57-77	77-102	-	50-102
Period	7	20	25	-	52
Trees	-0.4	-0.9	1.4	-	-0.1
Other vegetation	-	-	-	-	-
Debris /litter	0.4	0.2	-0.1	-	0.4
Soil	-0.3	-1.0	1.0	-	-0.2
Harvested wood products	0.4	1.8	0.7	-	1.5
Forest operations	0.0	0.0	-0.2	-	-0.1
Substitution	0.0	0.7	5.4	-	2.9
Total	**0.1**	**0.8**	**8.3**	**-**	**4.4**

Table A8.3 Total GHG balance of a forest management cycle involving YC 4 oak managed for minimum intervention. Time averaged (a) C stocks and (b) rates of C stock changes in key components. Positive values indicate sequestration, negative values indicate emission.

First rotation

(a) C stocks (tCO_2e ha^{-1})

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-35	35-90	90-100	0-200	-
Period	5	30	55	110	200	-
Trees	0.6	112.0	341.2	583.0	583.0	583.0
Other vegetation	-	-	-	-	-	-
Debris /litter	0.0	8.1	21.0	27.2	27.2	27.2
Soil	265.9	244.3	270.4	309.3	309.3	83.4
Harvested wood products	0.0	0.0	0.0	0.0	0.0	0.0
Forest operations	-0.1	-0.1	-0.1	-0.1	-0.1	0
Substitution	0.0	0.0	0.0	0.0	0.0	0.0
Total	**265.5**	**364.4**	**632.6**	**919.4**	**919.4**	**693.6**

(b) Rates of C stock changes (tCO_2e ha^{-1} y^{-1})

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-35	35-90	90-200	0-200
Period	5	30	55	110	200
Trees	0.1	3.7	4.2	2.2	2.9
Other vegetation	-	-	-	-	-
Debris /litter	0.0	0.3	0.2	0.1	0.1
Soil	8.0	-0.7	0.5	0.4	0.4
Harvested wood products	0.0	0.0	0.0	0.0	0.0
Forest operations	0.0	0.0	0.0	0.0	0.0
Substitution	0.0	0.0	0.0	0.0	0.0
Total	**8.1**	**3.3**	**4.9**	**2.6**	**3.5**

Table A8.4 Total GHG balance of a forest management cycle involving first rotation YC 4 oak managed for thinning and felling on an 80 year rotation. Time-averaged (a) C stocks and (b) rates of C stock changes in key components. (c) and d) as (a) and (b) but for second rotation. Positive values indicate sequestration, negative values indicate emission.

First rotation

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-35	35-80	-	0-80	-
Period	5	30	45	-	80	-
Trees	3.5	118.2	226.7	-	226.7	226.7
Other vegetation	-	-	-	-	-	-
Debris /litter	0.6	9.8	58.4	-	58.4	58.4
Soil	265.9	244.3	265.3	-	265.3	29.5
Harvested wood products	0.0	0.6	14.5	-	14.5	14.5
Forest operations	-0.1	-0.2	-2.4	-	-2.4	-2.3
Substitution	0.0	14.1	109.3	-	109.3	109.3
Total	**269.9**	**386.9**	**671.8**	**-**	**671.8**	**446.0**

(b) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-35	35-80	-	0-80
Period	5	30	45	-	80
Trees	0.7	3.8	2.4	-	2.8
Other vegetation	-	-	-	-	-
Debris /litter	0.1	0.3	1.1	-	0.7
Soil	8.0	-0.7	0.5	-	0.5
Harvested wood products	0.0	0.0	0.3	-	0.2
Forest operations	0.0	0.0	0.0	-	0.0
Substitution	0.0	0.5	2.1	-	1.4
Total	**8.8**	**3.9**	**6.3**	**-**	**5.6**

Second rotation

(c) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	80-87	87-117	117-162	-	80-162	-
Period	7	30	45	-	82	-
Trees	208.7	190.4	223.9	-	223.9	-2.8
Other vegetation	-	-	-	-	-	-
Debris /litter	75.1	88.3	89.9	-	89.9	31.5
Soil	265.3	269.6	282.7	-	282.7	17.4
Harvested wood products	27.2	55.6	57.6	-	57.6	43.1
Forest operations	-2.4	-2.5	-4.8	-	-4.8	-2.4
Substitution	109.3	123.4	218.6	-	218.6	109.3
Total	**683.1**	**724.7**	**867.9**	**-**	**867.9**	**196.1**

(d) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	80-87	87-117	117-162	-	80-162
Period	7	30	45	-	82
Trees	-0.2	-0.6	0.7	-	0.0
Other vegetation	-	-	-	-	-
Debris /litter	0.2	0.4	0.0	-	0.4
Soil	0.0	0.1	0.3	-	0.2
Harvested wood products	0.1	0.9	0.0	-	0.5
Forest operations	0.0	0.0	0.0	-	0.0
Substitution	0.0	0.5	2.1	-	1.3
Total	**0.1**	**1.4**	**3.2**	**-**	**2.4**

Table A8.5 Total GHG balance of a forest management cycle involving first rotation YC 4 oak managed for thinning and felling on a 150 year rotation. Time-averaged (a) C stocks and (b) rates of C stock changes in key components. Positive values indicate sequestration, negative values indicate emission.

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-35	35-90	90-150	0-150	-
Period	5	30	55	60	150	-
Trees	3.5	118.2	248.4	330.0	330.0	330.0
Other vegetation	-	-	-	-	-	-
Debris /litter	0.6	9.7	61.2	66.0	66.0	66.0
Soil	265.9	244.3	270.4	295.0	295.0	69.2
Harvested wood products	0.0	0.6	17.7	38.8	38.8	38.8
Forest operations	-0.1	-0.2	-1.3	-3.8	-3.8	3.7
Substitution	0.0	14.1	65.5	170.6	170.6	170.6
Total	**269.9**	**386.8**	**661.8**	**896.7**	**869.7**	**670.9**

(b) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-35	35-90	90-150	0-150
Period	5	30	55	60	150
Trees	0.7	3.8	2.4	1.4	2.2
Other vegetation	-	-	-	-	-
Debris /litter	0.1	0.3	0.9	0.1	0.4
Soil	8.0	-0.7	0.5	0.4	0.5
Harvested wood products	0.0	0.0	0.3	0.4	0.3
Forest operations	0.0	0.0	0.0	0.0	0.0
Substitution	0.0	0.5	0.9	1.8	1.1
Total	**8.8**	**3.9**	**5.0**	**3.9**	**4.5**

Table A8.6 Total GHG balance of a forest management cycle involving first rotation YC 6 oak managed for thinning and felling on an 80 year rotation. Time-averaged (a) C stocks and (b) rates of C stock changes in key components. Positive values indicate sequestration, negative values indicate emission. (c) and d) as (a) and (b) but for second rotation.

First rotation

(a) C stocks (tCO_2e ha^{-1})

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-25	25-75	75-80	0-80	-
Period	5	20	50	5	80	-
Trees	12.3	127.8	276.3	285.2	285.2	285.2
Other vegetation	-	-	-	-	-	-
Debris /litter	1.4	10.8	76.7	80.1	80.1	80.1
Soil	265.9	241.3	262.9	265.3	265.3	39.5
Harvested wood products	0.0	0.7	25.5	30.6	30.6	30.6
Forest operations	-0.1	-0.1	-1.6	-3.5	-3.5	-3.5
Substitution	0.0	11.5	85.4	164.6	164.6	164.6
Total	**279.5**	**392.1**	**725.1**	**822.3**	**822.3**	**596.5**

(b) Rates of C stock changes (tCO_2e ha^{-1} y^{-1})

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-25	25-75	75-80	0-80
Period	5	20	50	5	80
Trees	2.5	5.8	3.0	1.8	3.6
Other vegetation	-	-	-	-	-
Debris /litter	0.3	0.5	1.3	0.7	1.0
Soil	8.0	-1.2	0.4	0.5	0.5
Harvested wood products	0.0	0.0	0.5	1.0	0.4
Forest operations	0.0	0.0	0.0	-0.4	0.0
Substitution	0.0	0.6	1.5	15.8	2.1
Total	**10.7**	**5.6**	**6.7**	**19.4**	**7.5**

Second rotation

(c) C stocks (tCO_2e ha^{-1})

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	80-87	87-107	107-157	157-162	80-162	-
Period	7	20	50	5	82	-
Trees	262.9	243.1	277.3	281.7	281.7	-3.5
Other vegetation	-	-	-	-	-	-
Debris /litter	96.9	112.4	114.1	113.9	113.9	33.8
Soil	265.3	267.4	281.2	282.7	282.7	17.4
Harvested wood products	50.8	88.1	107.2	109.4	109.4	78.8
Forest operations	-3.6	-3.7	-5.2	-7.1	-7.1	-3.5
Substitution	164.6	176.1	249.9	329.2	329.2	164.6
Total	**836.9**	**883.5**	**1024.6**	**1109.8**	**1109.8**	**287.5**

(d) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-25	25-75	75-80	0-80
Period	5	20	50	5	80
Trees	-0.3	-1.0	0.7	0.9	0.0
Other vegetation	-	-	-	-	-
Debris /litter	0.2	0.8	0.0	0.0	0.4
Soil	0.0	0.1	0.3	0.3	0.2
Harvested wood products	0.2	1.9	0.4	0.4	1.0
Forest operations	0.0	0.0	0.0	-0.4	0.0
Substitution	0.0	0.6	1.5	15.8	2.0
Total	**0.2**	**2.3**	**2.8**	**17.0**	**3.5**

Table A8.7 Total GHG balance of a forest management cycle involving first rotation YC 6 oak managed for thinning and felling on a 100 year rotation. Time-averaged (a) C stocks and (b) rates of C stock changes in key components. Positive values indicate sequestration, negative values indicate emission

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-25	25-75	75-100	0-100	-
Period	5	20	50	25	100	-
Trees	12.3	127.8	276..3	327.4	327.4	327.4
Other vegetation	-	-	-	-	-	-
Debris /litter	1.4	10.8	75.2	77.5	77.5	77.5
Soil	265.9	241.3	262.9	275.1	275.1	49.2
Harvested wood products	0.0	0.7	25.5	44.4	44.4	44.4
Forest operations	-0.1	-0.1	-1.6	-4.3	-4.3	-4.2
Substitution	0.0	11.5	85.4	198.3	198.3	198.3
Total	**279.5**	**392.1**	**723.6**	**918.4**	**918.4**	**692.6**

(b) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-25	25-75	75-100	0-100
Period	5	20	50	25	100
Trees	2.5	5.8	3.0	2.0	3.3
Other vegetation	-	-	-	-	-
Debris /litter	0.3	0.5	1.3	0.1	0.8
Soil	8.0	-1.2	0.4	0.5	0.5
Harvested wood products	0.0	0.0	0.5	0.8	0.4
Forest operations	0.0	0.0	0.0	-0.1	0.0
Substitution	0.0	0.6	1.5	4.5	2.0
Total	**10.7**	**5.6**	**6.6**	**7.8**	**6.9**

Table A8.8 Total GHG balance of a forest management cycle involving first rotation YC 6 oak managed for thinning and felling on a 150 year rotation. Time-averaged (a) C stocks and (b) rates of C stock changes in key components. Positive values indicate sequestration, negative values indicate emission.

(a) C stocks ($tCO_2e\ ha^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle	Stock change over full management cycle
Years	0-5	5-25	25-75	75-150	0-150	-
Period	5	20	50	75	150	-
Trees	12.3	127.8	276..3	396.9	396.9	396.9
Other vegetation	-	-	-	-	-	-
Debris /litter	1.4	10.4	58.9	71.1	71.1	71.1
Soil	265.9	241.3	262.9	293.4	293.4	68.8
Harvested wood products	0.0	0.7	25.5	74.9	74.9	74.9
Forest operations	-0.1	-0.1	-1.6	-5.5	-5.5	-5.4
Substitution	0.0	11.5	85.4	251.5	251.5	251.5
Total	**279.4**	**391.6**	**707.3**	**1082.4**	**1082.4**	**857.9**

(b) Rates of C stock changes ($tCO_2e\ ha^{-1}\ y^{-1}$)

	Establishment	Initial	Full vigour	Mature	Full management cycle
Years	0-5	5-25	25-75	75-150	0-150
Period	5	20	50	75	150
Trees	2.5	5.8	3.0	1.6	2.6
Other vegetation	-	-	-	-	-
Debris /litter	0.3	0.5	1.0	0.2	0.5
Soil	8.2	-1.2	0.4	0.4	0.5
Harvested wood products	0.0	0.0	0.5	0.7	0.5
Forest operations	0.0	0.0	0.0	-0.1	0.0
Substitution	0.0	0.6	1.5	2.2	1.7
Total	**11.0**	**5.6**	**6.3**	**5.0**	**5.7**

Notes, symbols and abbreviations and glossary

Notes

1. Carbon emissions are usually quoted in units of mass of CO_2. To convert to units of mass of C the value is divided by the ratio of molecular weights 44/12, thus divided by 3.667.

Symbols and abbreviations

Symbol/ abbreviation	
AI	Annual yield increment. The annual increase in volume of timber in a tree, which will vary over time
BD	Bulk density, mass of soil per unit volume
BEF	Biomass expansion factor; usually the ratio of total above-ground biomass to merchantable stem
CCF	Continuous cover forestry
CEH	Centre for Ecology and Hydrology
CH	Conventional harvesting (compared with WTH, whole-tree harvesting)
CO_2e	CO_2 equivalents: expressing the combined effect of several GHGs by weighting each by its global warming potential
CS2000	Countryside Survey 2000
CS2007	Countryside Survey 2007
CWD	Coarse woody debris
DBH	Diameter at breast height, in cm, measured 1.3 m above ground level
DECC	Department of Energy and Climate Change
DIC	Dissolved inorganic carbon
DOC	Dissolved organic carbon
ECN	Environmental Change Network; a UK network of terrestrial and aquatic sites where long-term, coordinated environmental and biological monitoring is continuing (see www.ecn.org.uk)
ECOSSE	Estimating Carbon in Organic Soils Sequestration and Emission
FC	Forestry Commission
FMC	Forest management cycle
FR	Forest Research
FRA	FAO Forest Resources Assessment
fwt	Fresh weight tonne
GHG	Greenhouse gas (main gases: CO_2, water vapour, CH_4, N_2O, O_3, CFCs and HCFCs)
GPP	Gross primary productivity (photosynthetic CO_2 assimilation)
GWP	Global warming potential, defined as the contribution to cumulative warming over time, usually 100 years, for a particular GHG, relative to CO_2. According to the IPCC (2007) the 'global warming potential', GWP, of the 3 GHGs considered here is equal to 1, 25 and 298 for CO_2, CH_4 and N_2O, respectively
HWP	Harvested wood products
IPCC	Intergovernmental Panel on Climate Change
LCA	Life-cycle analysis

Symbols and abbreviations	
LISS	Low impact silvicultural system; also known as 'continuous cover forestry' or 'alternatives to clearfelling'. Encompasses a range of silvicultural methods including shelterwood or selection systems, the key characteristics being retaining some mature trees on the site during the regeneration phase
LULUCF	Land Use, Land-use Change and Forestry, a UN GHG inventory sector that covers emissions and removals of greenhouse gases resulting from direct human-induced land use, land-use change and forestry activities
MAI	Mean annual increment – the average rate of volume growth from planting/regeneration to any point of time
MinI	An example management option used in this report – minimum intervention
MMAI	Maximum mean annual increment. The MAI increases during the early phases of stand development before reaching a peak and then declining; the peak value is the 'maximum mean annual increment' (MMAI). MAI is an estimate of the maximum average rate of volume production that can be maintained on a site. In conifer species, the age of MMAI usually occurs towards the end of the 'stem exclusion' phase
MT	[standard] management tables (see Edwards and Christie, 1981)
NEE	Net ecosystem exchange. NEE = -NEP
NEP	Net ecosystem productivity: a measure of the net uptake of CO_2 into vegetation and the underlying soil. NEP = NPP - R_H
NFI	National Forest Inventory
NIWT	National Inventory of Woodland and Trees
NPP	Net primary productivity. NPP = GPP - R_A
NSG	Nominal specific gravity
NSI	National Soils Inventory
NSRI	National Soils Research Institute
odt	Oven-dried tonne, equivalent to 't dry weight' used in biology
POC	Particulate organic carbon
R	Ratio of root biomass to above-ground biomass
R_A	Autotrophic respiration (results in emission of CO_2 by plants)
R_H	Heterotrophic respiration (animals and micro-organisms)
SOC	Soil organic carbon
SoEF	State of Europe's Forests report
SOM	Soil organic matter/material (i.e. not just C component)
SRC	Short rotation coppice
SRF	Short rotation forestry
T&F	An example management option – standard thinning and felling
WRiB	Woodfuel Resource in Britain report, see McKay *et al.* (2003)
WTH	Whole-tree harvesting (compared with conventional harvesting, CH)
VOC	Volatile organic compound – see Glossary
YC	Yield class – see Glossary

Glossary

Term	Definition
Autotrophic	Organisms synthesising organic compounds for energy growth such as sugars from simple inorganic molecules such as CO_2 using light or chemical energy from the environment
Beating up	Replacing tree saplings that have died, a few years after original planting
Biome	A major habitat category, based on distinct plant assemblages which depend on particular temperature and rainfall patterns
Clearfell	Or clearcut – when a large area of forest is felled over a short period
Continuous cover	Silvicultural systems whereby the forest canopy is maintained at one or more levels without clearfelling
Eddy covariance	Methodology for measuring the net vertical flux of a gas (or energy) between a surface and the atmosphere by measuring the rapid and small fluctuations in concentration (or air temperature) and wind speed as turbulence mixes the air. It is using the way that air movement and concentration 'covary'
Heterotrophic	Organisms relying on organic compounds synthesised by autotrophs for energy and growth
Inorganic carbon	Carbon held in inorganic compounds, as in minerals like carbonates
Level II	Network across Europe of Intensive Monitoring Sites in woodlands, set up under European Union Regulations and the ICP Forests programme
Methanotrophs	Microbes that metabolise (oxidise) methane
Minerotrophic	Ecosystems deriving all water and nutrients from groundwater or streams, e.g. fens
Oligotrophic	Ecosystem with very low inorganic nutrients such as N, P, K
Ombrotrophic	Ecosystems deriving all water and nutrients from precipitation, e.g. much blanket bog
Organic carbon	Carbon held in organic compounds (i.e. derived from plant, animal or microbial process), not minerals such as carbonates
Productivity	Growth of organisms, expressed as mass of material or C accumulated per unit area and per unit time, i.e. a rate
Screefing	Removal of herbaceous vegetation and soil organic matter to expose a soil surface for planting
Spline	A smooth curve that joins a set of points by solving a number of equations
UK Assessment	The report commissioned by the Forestry Commission entitled *Combating climate change* (Read *et al.*, 2009)
Volatile organic carbon (VOC)	Gaseous organic compounds emitted by plants such as terpenoids and isoprene. They are important controls on atmospheric chemistry reactions such as formation of tropospheric ozone (O_3)
Yield class	The yield class (in the British system) is the MMAI (maximum mean annual increment, see Symbols and abbreviations) of a stand of trees, expressed in steps of 2 m^3 ha^{-1} y^{-1}

Forests and woodlands represent a substantial stock of carbon that is contained in soil, trees and other vegetation. They are a key component of the global carbon cycle and their effective management, at both global and regional scales, is an important mechanism for reducing greenhouse gases in the atmosphere. Understanding what determines the size of forest and woodland carbon stocks, and the processes and controls on the exchanges of carbon dioxide and other greenhouse gases, is critical in helping the forestry sector to contribute to reducing anthropogenic climate change. The objective of this review is to provide that understanding by summarising key information on carbon stocks in British forests, the fluxes of carbon dioxide and other greenhouse gases, how these are affected by changes as trees grow, and how they are affected by forest operations and other forest management decisions. This report will be of interest to forest managers, policymakers and researchers involved in estimating and understanding forest carbon and greenhouse balances, particularly in British conditions, how the balances can be affected by management, and what the limitations are to our knowledge.

Forestry Commission

Silvan House
231 Corstorphine Road
Edinburgh
EH12 7AT

www.forestry.gov.uk