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Sensitivity analysis in calculating the social value of carbon sequestered in British grown Sitka spruce

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Abstract

We describe a model that estimates the social benefits of carbon (C) sequestered in plantation Sitka spruce in Great Britain. Final net present values (NPV, base year = 2003) resulting from plausible variations in model parameters are calculated. The discount rate, social value of C, timber yield, rate of gain into live wood, length of rotation, lifetime of products, amount of C displaced by products and the changes in C flux on afforested peat soils are the most influential model components. The study predicts that C fluxes in actively managed forests in second or subsequent rotations or planted on peat soils will tend to have low or (on average) negative NPV.

JEL classification: Q230

Keywords: Carbon sequestration; Woodland; Sitka spruce; Uncertainty

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Introduction

The high likelihood of adverse climate change in the next 20–200 years is widely accepted, and much of this threat is attributed to the anthropogenic emission of greenhouse gases (GHGs). The single most important GHG in Great Britain (GB) with respect to both total volume produced and total global warming potential, is carbon dioxide (CO_2). CO_2 has been estimated to comprise 84% of the total global warming potential of UK GHG releases (DEFRA (Department for Environment and Food and Rural Affairs), 2002). Growing more trees has been suggested as a possible means of reducing atmospheric CO_2, and thus mitigating global warming. The tactic has limitations, and is most useful as a short-term stopgap rather than a long-term solution. Cannell (1999) observed that even if the current rate of new woodland creation almost doubled (from about 17,000 to 30,000 ha per annum), the new woodlands would still absorb less than 2% of the UK's annual carbon (C) emissions. Nevertheless, it is valid to include woodlands when calculating the C inventory for the entirety of GB, and to allow for the sequestration function when considering the overall balance of social costs and benefits generated by woodlands.

The authors produced a report (Brainard et al., 2003) for the British Forestry Commission that valued the C storage in GB woodland (GB, for our purposes = all of the UK except Northern Ireland, the Scilly Islands and the Channel Islands). In so doing we became very aware of the multitude of sources of uncertainty in making such valuations. This paper explores the sensitivities of model results to inputs, and attempts to assess the relative influence and importance of individual parameters. Assumptions pervade such modelling. C sequestration is usually modelled on a small unit basis – typically per hectare. These results are then scaled up to produce regional estimates. Even the scaling-up process itself tends to involve many assumptions and uncertainties.[1] As well as estimating net C gain (or loss, in some cases), valuation depends on the social (monetary) value per unit of C sequestered, along with the associated discount rate. The choices of which social value or discount rate to use are themselves highly contentious, and yet have huge impacts on the estimated total value of C sequestration (Price and Willis, 1993).

Thus, to thoroughly assess the uncertainties around an estimate such as the value of woodland C flux, at every step of each component model, is beyond what we can achieve in any short paper. But, we can realistically attempt to identify here which model inputs have most impact on a per hectare basis. This analysis should illuminate which areas of uncertainty would be most valuable to clarify in future efforts to calculate the social value of C sequestered in woodlands anywhere. There is also potential comparisons to be made with respect to other (non-forestry) possible land uses.

We assess impacts of model inputs on valuing sequestered C by focusing on the most commonly planted species in GB, Sitka spruce (*Picea sitchensis*). Sitka spruce

[1]Without stand-specific records, it can be very hard to estimate accurately, over a large afforested area, variables such as the average planting year, which rotation crops are likely to be in (on average), typical productivity, general soil type, the average density of stocking, frequency of thinning, etc.

dominates plantations in GB, comprising 49% of sub-compartments managed by the Forestry Commision (FC).[2] When considering C fluxes in British woodland, or their potential to mitigate global warming, models of Sitka spruce growth are commonly used as a surrogate for all other conifer species (e.g. Cannell et al., 1996; Milne et al., 1998).

We must emphatically stress that any attempt to calculate the value of C sequestration into woodlands for a specific regional or nation is extremely reliant on the quality of forest records. These data quality issues significantly outweigh the uncertainties posed by the variations in model inputs (except, possibly, social value per tonne of C and discount rate) that we describe in this paper. One needs to know, within each region, the location of each woodland, how big it is, what is the stocking density, what kinds of trees are in it, what kind(s) of soils it overlies. The age of the trees is crucial, as is their average growth rate. The value of C flux is very sensitive to plausible variations in these factors (specific examples are also given in Brainard et al., 2003). In practice, any attempts to make regional/national estimates have to be judged individually, with regard to the quality and availability of relevant forest records.

Methodology

All of our simulations consider the case of 1 ha of Sitka spruce planted on land that does not change use or management regime for at least 1000 years.[3] We describe parameter estimates and assumptions that need to be made when constructing a model to calculate the social value of C sequestered on this one hectare. We state what our preferred estimates and assumptions are, treating the adoption of these as a baseline, or most typical management scenario. We then use this paper as an opportunity to explore what happens if we use other credible estimates or assumptions. First, we vary model parameters individually from the preferred assumptions to assess single variable effects. Subsequently, we vary all model parameters simultaneously, which suggests how wide the underlying confidence intervals may be.

The work builds on models of C flux in GB woodlands that appear in Brainard et al. (2003) and Bateman and Lovett (2000). Readers may want to refer to Bateman and Lovett (2000) or Brainard et al. (2003) for more detail than we supply here. Bateman and Lovett (2000) analysed published estimates of C sequestration rates into live Sitka spruce crops with typical productivity ("yield class") characteristics. The authors undertook regression analysis of these published data to derive models that predict annual C flux (into live wood, per hectare, excluding thinnings after the first thinning date[4]) as a function of productivity and plantation age. Date of first

[2]As calculated by the authors, in 2001, from the FC's own inventory database for that same year.

[3]In reality, we expect that land use would indeed change over this long time period.

[4]Bateman and Lovett (2000) were following the example set by Cannell and others, who assumed that after the first thinning date, the net C uptake into timber thinnings could effectively be ignored due to the

thinning, felling dates and proportion of live wood that is removed after first thinning were calculated using regression models dependent on yield class and discount rate. There is good agreement between our model estimates of the C in both thinned and unthinned live wood with calculations made by other researchers (CARBINE and C-FLOW models discussed in Robertson et al., 2003). Lifetime analysis to determine release curves for woodland products was also undertaken by Bateman and Lovett (2000). They further reviewed evidence for likely soil C gains (or losses) following afforestation. The final Bateman and Lovett (2000) models therefore included C in live wood, products and soils. Brainard et al. (2003) extended these models to include leaf litter and C releases due to harvesting. The expected rates of C losses on afforested peat soils were also revised in Brainard et al. (2003). This paper develops the models further by including C stored in thinnings, and displaced C release when wood products substitute for other materials.[5]

C flux is monetised by multiplying annual gains (or losses) in the base year and thereafter, by the social value of C (per metric tonne). Discounting starts in the base year. Discount factors are calculated for each year and are applicable for the entire previous year. Summing these calculations yields a 'net present value' (NPV) for C sequestered in woodland on a per hectare basis, since the start of the year 2003, assuming that (currently) conventional commercial management regimes operate. This study is confined to per hectare values, but our previous works have calculated total NPVs for relevant species types in a region (Wales or GB). Bateman and Lovett (2000) concerned themselves only with C flows starting in their base year (1991), while Brainard et al. (2003) also included 'sunk' benefits in the form of C stores already in British woodland before their base year. Here, we revert to only counting C flows in the base year and thereafter.

In the first part of the results, 21 model parameters (for which there are many credible possible values) are varied one at a time. We discuss the relative individual impact of each model parameter on NPVs. We also briefly consider a range of alternative economic parameters when calculating NPVs. As a result of the observations when varying model parameters in isolation, in subsequent modelling we hold some parameters constant, because they are often known, individually they had very little impact on NPVs, there is a relevant government policy preference (e.g., discount rate), or we have little faith in the proposed alternative values. This still leaves 11 model parameters, which we vary simultaneously to produce hundreds (non-peat soils) or thousands (peat soils) of NPV estimates for each possible planting year, rotation and soil type combination. This gives us an impression of how wide the true underlying confidence intervals in NPVs may be.

(*footnote continued*)
relatively short life-time of their main product (paper). This article presents further evidence why this assumption is reasonable.

[5]In this paper, when we refer to "displacement" we mean to say that the use of wood products has prevented the volatisation of C into the atmosphere, because other materials or fuels were not used; this might also be thought of in terms of C-loss prevented or awarded because of the use of wood products.

Review of model construction and parameters

This paper examines the effect of varying underlying assumptions for the many physical, and some economic parameters included in our previous models. Ideally, we would review many studies containing estimates of the relevant model inputs, all containing research done to an equivalent standard. This review would yield a probability distribution for the candidate values, enabling us to make the statement: "There is a 95% probability that parameter Z is truly between X and Y." Such data would enable us to undertake a Monte Carlo analysis (Gentile, 1998) of model results. Unfortunately, the data don't exist for us to reliably assign probabilities to each model parameter. Instead, we decide on a preferred parameter value, i.e., what seems the most likely value, based on evidence to date.

There are various ways to choose credible variations on our preferred parameter values. Alternatives could be one or two standard deviations from the mean, a small constant fluctuation ($\pm$25%) away from the preferred value, or commonly used alternatives from previous papers. Another way to experiment with the model inputs is on the basis of most extreme (but still plausible) assumptions.

C sequestration associated with woodlands can be broken down into these main areas:

- *Live wood,* including all biomass in plantation trees.
- *Wood products,* including harvest, storage, displacement factors (when wood substitutes for other materials) and fossil fuels used in manufacturing.
- *Leaf litter and debris.*
- *Soils.*

In order to assign value, one also includes economic factors, namely

- *Discount rate.*
- *Social value of C.*

Next we describe in detail the parameters necessary to estimate C flux associated with woodland. Our best guess estimate for each non-economic parameter is denoted by bold text in Table 1, along with proposed alternative values. For reasons explained subsequently, we tend to hold economic parameters constant at 3.5%/$10 per metric tonne C. Fig. 1 shows the cumulative C fluxes predicted for the main model components using our preferred assumptions.

Live wood

Overview

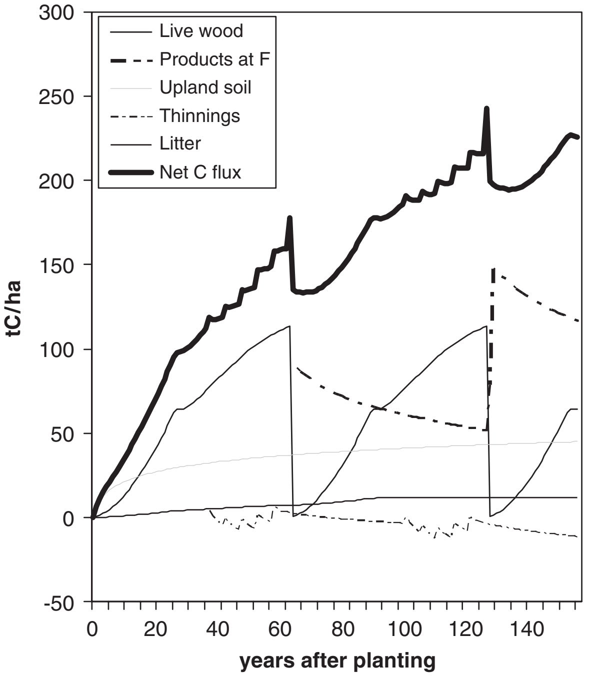
C mass in a tree is closely related to its rate of growth. Timber yield in GB is conventionally rounded to the nearest even integer, and denotes (in cubic metres) the maximum mean annual increment, averaged over the entire rotation to date, in timber production, per hectare. This categorisation is referred to as general yield

Table 1. Estimates of parameter values for input variables relevant to C sequestration in Sitka Spruce

	Lower estimate	Middle estimate	Upper estimate
Harvest releases (% of timber)	0.94	**1.25**	1.56
Leaf litter equilibrium levels (tC/ha)	9	**12**	15
Carbon gains in live wood, ±1 standard deviation (tC/ha per annum)	Middle −30%	**As suggested by Bateman and Lovett (2000) models**	Middle + 30%
Release period for products (years)	50	2 × rotation length	**200**
% C displaced by long-lived products	12.5	**50**	200
% C release from fossil fuels required to make paper	30	**45**	60
Variation in amount of C (tC/ha per annum) in thinnings	Middle −25%	**Authors' own function**	Middle + 25%
Rotation length (years)	Middle −25%	**Regression to predict FIAP dates**	Middle + 25%
Yield class, regional mean	10	**12**	14
Rate of releases from afforested peat	CEH −25%	**CEH figures (generalised)**	CEH + 25%
Whether peat reverts to function as C sink each time tree canopy reopens	**NO**	NA	YES
Pattern of C flux on thin peat, slow and perpetual or fast and temporary	**Slow… (CEH) Hargreaves et al. (2003)**	NA	Fast… Zerva et al. (2005)
Cumulative soil carbon flux (gains, tC/ha)			
Non peat, lowland	75	**100**	125
Non peat, upland	37.5	**50**	62.5
Thin peat	**−112**	NA	−230
Usual annual sink in undisturbed peat (tC/ha per annum)			
Thin peat	0.0375	**0.125**	0.25
Thick peat	0.175	**0.25**	0.325
Year that sink ceases	2013	**2023**	2043

Notes: NA = no middle assumption (only 2 values) tested. Preferred model inputs in bold text. FIAP = Forest Inventory Appraisal Package; a computer program that calculates optimal (in commercial terms) felling dates depending on species, yield class and discount rate. CEH = Centre for Ecology and Hydrology, see text for description of relevant study.

class (GYC or general YC). Thus, YC14 indicates a stand that, at its productivity peak, has produced an average between 13 and 15 m^3/ha per annum during the rotation to date. Most conifer stands in GB have YC = 10–18.



Notes: F=felling date. Species=Sitka spruce, YC=12, first rotation, upland planting. Live wood and products are the biggest C stores. Lines for C storage in products/thinnings include offsets for harvest, transport and C displacement. Sudden, short-term rises in C stocks after F are caused by C displacement in products followed by high wastage and fast decay of short-lived products.

Fig. 1. Cumulative C flux associated with woodland, from our model estimates.

YC refers only to merchantable timber; the total biomass of a tree can be directly calculated from YC using age- and species-dependent multipliers (Matthews, 1991). The proportion of a tree's dry mass that is C is fairly constant across species, at about 49–50% (Matthews, 1993).

Uncertainty about yield class

When assessing C stores in the woodland of an entire region, knowledge of exact YC values for each forest would therefore be ideal, but in reality, such data are often unavailable. Moreover, the restricted number of categories in the British General YC scheme makes it unlikely that a single YC value will perfectly represent productivity in an entire region. A more accurate representative yield description might be "11.5", or "12.2", but these fractional expressions of yield are unconventional and rarely available. Milne et al. (1998) argued that YC 12 was "an acceptable approximation to the C fluxes for GB as a whole" (p. 30). Here, we examine the effects on final NPV of varying YC by just one yield class either side of YC = 12 for Sitka spruce. This should capture some of the uncertainty related to the use of a coarse-resolution, single YC value.

Rate of C uptake into live wood

Dewar and Cannell (1992) published estimates of annual C gains for Sitka spruce stands of specific yield class and age, using biological and physical models dependent on a large number of inputs. Bateman and Lovett (2000) undertook regression analyses of the published C gains in Dewar and Cannell (1992) to predict C uptake in live wood as a function of only species, YC and plantation age. Our models of C flux are simplistic compared to the underlying biophysical processes, but we believe this is adequate for our purposes, due to the good agreement of the C uptake indicated by our model outputs with relevant literature (Brainard et al., 2003; Robertson et al., 2003). Dewar and Cannell (1992)'s basic models were used again by Milne et al. (1998). Milne et al. (1998) undertook a Monte Carlo analysis with 1000 simulations to assess the effects of varying the many model parameters, including rotation length, initial spacing, year of first thinning, stemwood density, wood decay rate, etc. Milne et al. (1998)'s simulations indicated that the standard deviation around the mean for total sequestration into live wood was $\pm 30\%$. Therefore, we describe here what happens to NPVs if we vary the models for C gains in live wood by $\pm 30\%$.[6]

Rotation length

The longer a tree stands, the more C it accrues into live wood, and the later the C in that tree's products will be released to the atmosphere. However, the rate of C uptake depends on tree age, with trees that are past their growth peak tending to take up C very slowly. It is therefore important to estimate when trees are likely to be felled (rotation length) as reliably as possible. The Forestry Commission uses its own software, the Forestry Investment Appraisal Package (FIAP) to determine optimal (in terms of maximising net profit) felling ages and dates of first thinning. FIAP operates by maximising the net present market value of a stand subject to user-determined parameters, including tree species, yield class and discount rate. In both Bateman and Lovett (2000) and Brainard et al. (2003), the FIAP outputs ($n = 64$) were input to regression analysis to derive a generalised model predicting the optimal felling date from just yield class and discount rate for each species. A regression equation to predict date of first thinning was similarly derived. The FIAP-indicated felling dates inherently assume that a stand is being managed for optimal timber extraction value. This will obviously not always be true; individual site managers may well apply other criteria to choose later (or earlier) felling dates. For instance, some stands are being managed almost purely for amenity or existence purposes (e.g., ancient woodlands), rather than commercial considerations.

We test the sensitivity of model results to varying rotation lengths by a modest variation ($\pm 25\%$) of the dates indicated by the FIAP-derived-regression equations used in Brainard et al. (2003).

[6]Although Milne et al. (1998) calculated that net C flux might vary in live wood by 30%, this does not translate necessarily into a simple $\pm 30\%$ change in all C sequestration NPV associated with live wood and products at felling. Moreover, our model considers interaction effects between different variables, including many aspects of the C sequestration function in woodlands that Milne et al. (1998) did not study.

Products

C release from end products, general decay function

Our model assumes that 70% of the biomass of fully mature trees ends up as merchantable products. The C in the other 30% (roots, needles, bark, etc.) is assumed to decay into the atmosphere over the first 2 years after harvest. Bateman et al. (2003), also described at length in Bateman and Lovett, 2000) undertook a detailed empirical lifetime analysis of hardwood and softwood products from secondary data. This included an assessment of the proportion of wood that went into products with different life-times, and the C release rates from same. The analysis indicated that C would remain in some softwood products for up to 200 years, with the rate of release being greatest in the first decade after felling. Bateman et al. (2003) summed C release curves for products of varying lifetimes, using proportions for the entire UK annual domestic production. Regression analysis was undertaken on these data to generate an inverse power function predicting the proportion of C liberated from the total C in harvested timber in a given year, as a function of years after felling. The function (its derivation is described at greater length in Bateman et al. (2003) and Bateman and Lovett, 2000) takes the form:

$$\begin{aligned}&\text{\% of C liberated from harvested products in year tF}\\&\quad = 0.001746 + 0.110363 \times 1/(1 + \text{tF}),\end{aligned}$$

where tF = years after felling, or tF = 0 at felling and tF = 200 maximum.

This function related to the total UK domestic production, assuming that 49.3% of such production is sawn logs (long-life products), and 21.8% board or other medium-life products. The function predicts that 50% of the C is released from all Sitka spruce products (at thinning and felling, across all YC) after 31 years.

Proportion of products with different lifespans

We modified the basic C-in-products-release-function from Bateman and Lovett (2000) to apply to harvest or thinning, and for specific YCs. Hamilton and Christie (1971) categorise harvested products by log diameter: $\geqslant$24 cm (sawn logs, expected to be used in long-life products), 18–24 cm (medium-life products) and $\leqslant$7 cm (short-life products). This information is presented for Sitka spruce, yield classes 6–24, at both felling (their Table 28) and thinning (their Table 61). From these data we observe, for instance, that upon felling a Sitka spruce stand of YC = 12 would be expected to yield 49% of its products as sawn logs (long-life products) and 33% its products as hardboard, pallets and other medium life products (top diameter = 18 cm). This makes for a total of 82% of products resulting in medium or long-life products. Similarly, thinning a YC = 12 stand 10 years before felling (50–51 years), is forecast (in Hamilton and Christie) to yield 10% 24 cm logs, 40% 18 cm logs and 50% 7 cm diameter logs.

We use the total percentage of medium and long life products to scale the C release from products function reported in Bateman and Lovett (2000), such that the release period for the predicted percentage of short-life products remains (approximately)

6 years, and the 100% total release period remains at 200 years. The point when 50% of C is released from the scaled product functions thus varies, depending on the proportions of short, medium and long-life products that went into each thinning or harvest. For instance, for products from felling, 50% of C is released for YC = 10, 12, and 14 at, respectively, 42, 43, and 58 years.[7]

Bateman and Lovett (2000) observed that 2.4% of the UK domestic annual softwood harvest is used for fuel; lacking further information we assume that the percentage of fuel-wood products holds constant regardless of thinning date or YC. Otherwise, we use the published proportions (Hamilton and Christie, 1971) of how much of the harvest at thinning or felling goes into various types (expected life spans) of products.

Amount of C in thinnings

The production schedule in Hamilton and Christie (1971) is used to calculate the proportion (typically around 18%) of the total C (in live wood) that is removed at each thinning date from conifer stands and that becomes useable as wood products.[8] Under an intermediate, 70% thinning intensity regime, the optimal thinning interval for YC = 8–18 is given in Hamilton and Christie as every 5 years. We only consider products from thinning starting at 25 years before felling. Products extracted earlier are overwhelmingly short-lived (e.g., paper) and were therefore expected to have negligible impacts (or small negative impacts, see later discussion on energy use when making paper) on long-term C stores.[9] Otherwise, the total amount of C removed in thinnings is varied by $\pm 25\%$.

Release period for C in products

We are confident in the ability of Bateman's analysis and resultant function to model releases, as well as our efforts to scale it for different YCs and harvests at thinning or felling. However, the total release period (200 years) is much longer than that adopted by most previous British[10] research. For instance, Cannell and Dewar (1995) assumed that 100% release periods were roughly equal to twice the rotation length (typically 100–140 years). The authors admit this is only an educated guess, in the absence of relevant information. Similarly, Matthews and Heaton (2001) assumed that all C would be released from wood products within 50 years. We believe that 50 or even 100 years is too conservative. Therefore, our analysis only tests to a limited extent the alternative product life-times of 50 years or twice rotation length. This is achieved by scaling our products release function, as described thus far, so that 100% of release occurs within these periods.

[7] As a percentage of the total harvested, there is a large increase in 24 and 18 cm diameter logs, as one moves from YC 12 to 14.

[8] Non-merchantable biomass from thinnings, such as roots, needles and bark are only included in our estimates of C stores before the first thinning date.

[9] Moreover, the financial returns from the earliest thinnings are usually quite low, such that early thinnings may be disposed of immediately near to the site (by burning or shredding, for instance).

[10] C release periods found in life-time analyses of Finnish softwood products in Pingoud et al. (2001) and Karjalainen et al. (1994), and Skog and Nicholson (2000, a North American study) are similar to ours.

Displaced carbon in wood products

Wood products often substitute for other materials that require much more energy to manufacture, such as plastic, steel or aluminium. Marland et al. (1997) reviewed evidence for the C displaced by different types of wood products, considering energy uptake for manufacture and variation in product lifetimes depending on composition of material. They concluded that displacement factors – how much C that 1 kg of C in wood products would displace – was about 0.6 kg for biofuels, 0.25 kg for medium-life products (e.g., packing wood and hardboard), and 0.5 kg for long-lived products in the USA and Europe (e.g., construction beams). This applies only to CO_2 releases related to manufacture and expected life time of the products; inclusion of disposal (or reuse/recycling) options can significantly alter these figures. C displacement estimates made by other researchers include: 90–230% for different types of building construction (in New Zealand) in Buchanan and Honey (1994); 60–80% for multi-storey building construction in Borjesson and Gustavsson (2000); and 12–50% for residential house construction (in the Netherlands) in Goverse et al. (2001). We favour Marland et al. (1997)'s estimate of 50% for long-lived products because it is relatively conservative and approximately agrees with the findings of multiple studies. However, this paper gives us the opportunity to test alternative displacement factors of 0.125 and 2.0 in long-lived products.

Harvest

Harvesting a timber site creates its own C emissions. Karjalainen and Asikainen (1996) observed annual harvest-related releases around 1.4% of the amount of C in Finnish timber. We believe that harvest-related releases in GB are lower (assumed 1.25%), due to milder climate and shorter transport distances. Here, we test the sensitivity of the model to the assumption that C releases are 1.25%, 0.94% or 1.56% of the C stored in harvested timber, and occur during both thinning and felling. In practice releases would be expected to also occur during planting, but we have no means of estimating these.

Fossil fuel use required for making paper

For wood fuels and medium- or long-life products, secondary (manufacturing-related) C emissions are included in the displacement factors described earlier and discussed in Marland et al. (1997). Only the fossil fuel use required to manufacture paper is unaccounted for. Getting a confident calculation of the fossil fuel inputs required to make paper is difficult. Although, it has been claimed that virgin paper production is energy self-sufficient (e.g., waste byproducts such as bark and black liquor can be burnt for electricity and heat; Collins, 1996), most sources (e.g., Ogilvie 1992; Personen 1995; Sundin et al., 2001; Grieg-Gran, 1999; Klungness et al., 1999) agree that some fossil fuel burning is usually required to make paper products. Drying consumes most of that energy requirement (Farla et al., 1997), but generalising how much fossil-fuel energy is needed is difficult, because the efficiency and types of energy sources vary hugely from one mill to the next (Grieg-Gran, 1999). The energy efficiency of paper-making in GB is expected to improve in the future (Sundin et al., 2001; Farla et al., 1997), which adds uncertainty when applying to the industry today any estimate based on historical data of energy offsets and usage.

For our purposes we needed estimates of the energy requirements that relate directly to the amount of C in the original or end products. Pingoud and Lehtilä (2002) estimated that, in Finland, paper production resulted in C releases (from fossil fuel burning) equivalent to 30–60% of the C stored in the end product. The percentage varied depending on the type of paper product and the assumptions about energy sources at individual paper mills. The analysis included consumption of fossil fuels for harvest and transport as well as heating of facilities, drying and other manufacturing activities.

It is hard to know how applicable the 30–60% figures are for Britain. Moreover, it is difficult in Pingoud and Lehtilä (2002) to separate harvest/transport releases (which are known to be a quite small part of the energy total, and which we account for elsewhere) from manufacture. Here, we assess how NPVs vary if paper production is presumed to result in the extra release of 30%, 45% or 60% of the C stored in the end product. We do not consider the energy requirements to manufacture recycled paper products. Note that using a figure of 45% (or greater), long-term C storage in thinning products is negative (Fig. 1). Moreover, if the fossil fuel consumption for paper manufacture is around 25–38% of the C in the final products, then thinnings tend to have little net impact on final valuations; what C storage they provide in products is entirely (or almost entirely) offset by fossil fuel consumption required for harvest and to make paper. The body of work by Cannell and colleagues (1996) (see references for examples), ignores conifer thinnings on the basis that their main products (paper) are over-whelmingly short-lived. Price (pers. comm, 2005) argued that thinnings should not be ignored; there is a high percentage of longer-lived products in late thinnings. We conclude that the decision by Cannell et al. to ignore thinnings in conifers was defensible, but not for the reason they believed it was. It is the fossil fuel energy required to make paper rather than the generally short life of paper products that causes thinnings to provide negligible or (more likely) a small negative C storage. This seems to be true even if both early (overwhelmingly paper) and late thinnings (which include many non-paper products) are considered together. This generalisation probably applies only to softwoods, however. Compared to conifers, thinnings from hardwood trees contain much higher percentages of long-life products.

Litter

Milne and Brown (2002) suggested that first rotation conifer woodlands in GB accrue C in forest floor litter at an average rate of 0.25–0.32 t/ha per annum. This deposition is not static, however. Milne et al. (1998) suggested that up to 50% of C added annually to litter is transferred into soils each year. At the same time, forest floor litter releases CO_2 back to the atmosphere at rates that may be accelerated by global warming (Richey et al., 2002). In contrast to this complex reality, we treat C deposition in the forest floor simplistically. We assume that C gains in forest floor litter are proportional to increases in live wood C, thus annual gains in forest floor litter C are low when trees are very young, and greatest when trees are at the height of their growing cycle. On average, the net gains into leaf litter are 0.16 tC ha^{-1} year^{-1} for a first rotation stand with YC = 14. Dewar and Cannell (1992) suggested long-term equilibrium C stocks in woodland leaf litter might be 9–15 tC/ha under conifers. We describe the results if gains are stopped (reaching an equilibrium state) in leaf litter C at 9, 12 or 15 tC/ha. For YC = 12, our model takes 77–130 years to achieve equilibrium in leaf litter C.

Soils

Overview

After live wood and products, the largest C sink (or loss) associated with woodlands is in soils. Adger et al. (1992) present soil C total levels (or losses) post-afforestation. From these data we tend to conclude (see Brainard et al., 2003) that, on non-peat soils, about 50 tC/ha might be gained post-afforestation in upland areas, and about 100 tC/ha for lowland areas.[11] Bateman and Lovett (2000) assumed a smooth and very slowly diminishing C flux path for C gains in soil, and that all soil C gains occur within 265 years of afforestation. Varying the flux period by $\pm 25\%$ has very little impact on our final results (less than 0.1% change), so we do not consider this assumption further. However, changing either the total gain or the rate of soil C increase by $\pm 25\%$ did affect results significantly and results from these variations are presented here.

There are also soils that characteristically lose C when afforested: temperate wetlands, which we refer to collectively as "peat" soils. The decomposition of C from decaying organic matter is severely delayed under anaerobic conditions, such as when soils are poorly drained or frequently waterlogged, forming peat (Askew et al., 1985). As a result, many British peat bogs have an ongoing net annual increase of soil C. Drainage of peat (a necessary precursor to afforestation) both disrupts the ongoing C sink and removes anaerobic conditions, leading to much lower soil C equilibrium levels. The result is two separate detrimental effects with regard to afforestation on peat. There is the potential loss of an existing sink, in addition to a significant and extended period of C releases on afforested peat wetlands into the atmosphere as CO_2.

C release from afforested peat

Peat soils are typically categorised by depth: deep (or "thick") peat and shallow (or "thin") peat soils. "Thin" peat refers to soils with organic layers less than 45 cm deep; thick peat soils can be up to several metres deep. Both thin and thick peat soils were widely afforested in the 20th century (Zerva et al., 2005).

From reviews of published figures of soil C amounts for peat and forest lands, Bateman and Lovett (2000) derived an expected total C loss on deep peat of 750 tC/ha. Brainard et al. (2003) demonstrated that because the C release from afforested

[11] A further complicating issue is that definitions of 'upland' versus 'lowland' are inconsistent between sources, but this problem has no impact on the calculations that we make here.

peat appears to be relatively slow (see discussion below) the assumed cumulative C release on afforested deep peat had little effect on final NPVs unless the discount rates were virtually negligible, or the expected total C losses were relatively quite small. Therefore, this paper makes only brief comparisons between possible variations on the expected cumulative C releases from afforested thick peat.

It is very unclear what the cumulative C releases on afforested thin peat might be. Lacking further information, we assume that these are approximately 15% of the suggested maximum from thick peat soils, or 112 tC/ha. This estimate is supported by observations in Zerva et al. (2005) but it may still be too low, given that peat soils in Wales have been estimated (CEH (Centre for Ecology and Hydrology), 2002) to contain a mean value of 250 tC/ha in just the top 15 cm, compared to only 20 tC/ha to the same soil depth for Welsh agricultural soils. We therefore also consider how NPVs change if a higher cumulative loss is assumed, namely 230 tC/ha.

Studies of the rate of C releases on afforested peat are preliminary. From original field measurements and computer modelling, Hargreaves et al. (2003) of the Centre for Ecology and Hydrology (CEH) constructed a descriptive model of C exchanges on afforested deep peat soils in Scotland. The model is based on observations of trees of different ages on the same soil type, or a chronosequence. CEH staff supplied us with their estimates of annual C flux on afforested peat (including ground vegetation and litter, but not trees). We have generalised the data for our own modelling and list them in Table 2. The data only extend to year 26 after afforestation. After that it was believed that afforested peat might emit C long- term at a rate of about 0.3 t ha^{-1} year^{-1}. This loss would be expected to continue in perpetuity, or until the total expected maximum was reached. The CEH studies were only on two sites, and may not be representative of C releases on afforested peat across the whole of GB.

Similarly, Zerva et al. (2005) studied a thin peat (peaty gley) soil in northern England. Soil C stocks were again measured under trees of varying ages, in different rotations, to construct a chronosequence of C flux post-afforestation. Zerva et al. (2005) noted soil C levels of (mean values for many samples) 274 tC/ha on undisturbed peaty gley. This dropped to 140 tC/ha on 40-year-old Sitka spruce in its first rotation. However, mean soil C levels were measured at 147, 181 and 249 tC/ha in years 12, 20 and 30, respectively, for trees in their the second rotation. Zerva et al. (2005) hypothesised that although the initial reduction in soil C was fast and great

Table 2. Generalised carbon losses from peat, leaf litter and ground vegetation on afforested deep peat soils

Year	0	1	2	3	4	5	6	7	8	9	10	11	12	13
C loss	0.08	2.2	3.8	2.5	1	−0.27	−1.2	−1.6	−1.6	−1.3	−1.1	−0.8	−0.6	−0.5
Year	14	15	16	17	18	19	20	21	22	23	24	25	26	
C loss	−0.3	−0.2	−0.1	−0.04	0.03	0.08	0.12	0.15	0.18	0.2	0.22	0.24	0.25	

Source: Generalised from figures supplied by Milne (pers. comm., 2003). C loss = tC/ha. Year starts at zero, meaning the moment that trees are planted, and otherwise = end of year x after initial afforestation. Negative values denote periods of carbon absorption in the peat (i.e., gains rather than losses). We assume a constant loss of 0.3 tC/ha per annum from year 27 onwards.

(134 tC/ha over the 40 years in the first rotation), by late in the second rotation soil C stocks had recovered to close to pre-afforestation levels.

There are inconsistencies between the chronosequences generated by the CEH (Hargreaves et al., 2003) and Zerva et al. (2005). The C loss for the thin peat soils in Zerva et al. (2005) is much greater and faster than that observed by Hargreaves et al. for thick peat. Also, there is no indication from Hargreaves et al. (2003) that soil C on deep peat might return to pre-afforestation levels. This may happen on thin peat soils because of the penetration of fine roots into the soil layer below the thin peat layer; in effect, C losses from the thin peat at the top of the soil profile may be compensated for by C gains further down. Very clearly there is a need for further research on these types of soils. The CEH study seems more detailed and we are inclined to adopt theirs as our preferred assumptions. However, in this paper we also consider whether the C flux on afforested thin peat soils might follow the sequence suggested by Zerva et al. (2005): initial steady and fast but temporary releases, with C stocks recovering to near pre-afforestation levels by 30 years into the second rotation. We do not consider the possibility of C stores in thick peat soils recovering to pre-afforestation levels.

Moreover, we do not know how accurate our estimates are of the rates of C flux on afforested soils. Therefore, this paper considers what happens if the rates of gain/release from the CEH and Zerva et al. (2005)'s studies are accelerated or decelerated by 25%.

The usual C sink on undisturbed peat

On deep peat soils, annual ongoing C sinks of 0.4–0.7 $\mathrm{tC\,ha^{-1}\,year^{-1}}$ were suggested by Cannell and Milne (1995), and 0.2–0.5 $\mathrm{tC\,ha^{-1}\,year^{-1}}$ (Cannell, 1999). Hargreaves et al. (2003) observed actual C accumulation rates on undisturbed peat of 0.22 and 0.25 $\mathrm{tC\,ha^{-1}\,year^{-1}}$ at two Scottish sites and Crill et al. (2000) report a somewhat lower observed value for Finnish peatland, of just 0.205 $\mathrm{tC\,ha^{-1}\,year^{-1}}$. From these data we are inclined to assume that the ongoing sink in British peat bogs is about 0.25 tC/ha. We test the sensitivity of this assumption by varying it by 30%, i.e., ± 0.075, or 0.175 and 0.325 tC/ha. We can also ask how important is the lifetime of the usual C sink on peat soils. There is evidence that peat bogs will tend to lose their C fixing function as global temperatures rise (Chapman and Thurlow, 1998; Bousquet et al., 2000). When including this potential negative impact of forestry on peat soils, we examine what happens if the C sink function on undisturbed peat stops 10, 20 or 40 years after our base year (2013, 2023, or 2043). The loss of the sink can otherwise be directly subtracted from possible C gains in woodlands planted on peat, up until these dates.

Information about the usual (if any) C sink on shallow peat soils is unavailable. We are inclined to assume that, in current climatic conditions, the rate of C absorption into thin peat soils is half of that for thick soils (or, 0.125 instead of 0.25 $\mathrm{tC\,ha^{-1}\,year^{-1}}$). This paper describes what happens to NPVs if the usual C sink is assumed to be equal to that for thick peat (0.25 $\mathrm{tC\,ha^{-1}\,year^{-1}}$) or much smaller (15% of the deep peat value, or 0.0375 $\mathrm{tC\,ha^{-1}\,year^{-1}}$).

Returning to a C sink

To enable afforestation, peat lands are drained and ploughed, which leads to immediate large releases of C, as shown in Table 2. A very interesting aspect of Table 2, however, is that after an initial large release, deep peat soils were observed to revert to their function of scavenging C from the atmosphere in years 5–17 – i.e., until the tree canopy closed. No one knows if afforested peat soils will repeatedly revert to a C sink function each time the canopy is reopened (following felling). Here we take the opportunity to run the model both ways, with peat returning to C absorption whenever the canopy is still open, or not.

Discount rate and social value per tonne of carbon (svtC)

Most of this analysis relies on a discount rate of 3.5%, which is the current UK Treasury Green Book discount rate for public sector projects (*http://greenbook.treasury.gov.uk/chapter05.htm#discounting*). In Table 5, we consider other discount rates, with their associated social values of C. We apply a standard discount function (Price, 1993).

The ''social value'' of C sequestration may be defined as the benefit in savings from damage avoidance. This benefit can be calculated by observation of compensatory costs to reveal its cost to society, or ''shadow price''. The lack of observable markets introduces significant uncertainties in estimates of the true social cost of sequestered C. We follow the conventions of describing svtC in US dollars, and with reference to potential damage caused by the most common GHG, CO_2. Recent estimates as to the true svtC value range from \$1–\$2 (Mendelsohn, 2003), to \$109 (Clarkson and Deyes, 2002). Rather than delve into the merits of the arguments supporting any particular value, we generally adopt a notional value of \$10 (in the year 2003) for each metric tonne (1000 kg) of sequestered C starting in 2003. In our final table we explicitly vary the starting svtC for 2003 with associated discount rates. The initial svtC value increments by 1% (\$0.10) each year after 2003, until the year 2033, in accordance with a widespread perception that damage caused by each new tonne of CO_2 released will be greater as CO_2 concentrations rise. The svtC is held constant from 2033 in case scientific or technical innovations are adequate to mitigate the acceleration in harm from climate change.

Impacts on NPV of a changing any single model parameter

We examine the effect on NPV of each individual model parameter, using three example planting years. The Forestry Commission's own sub-compartment database (SCDB) indicates a mean[12] planting year for Sitka spruce of 1970. Sixty per cent of

[12]The choice of this year is somewhat for convenience, but it is representative. The mean planting year for all (not adjusted by size) Sitka spruce FC subcompartments, (SCs) in the FC SC database was 1970. Median = 1965. Adjusting by area, the mean is 1976 for all SCs. The mean planting date for Sitka spruce in its first rotation is 1965, the median planting year (again, first rotations only) is 1996. Adjusting by size of the afforested area, the mean for first rotation SCs is 1971.

Table 3. Per ha NPVs (US $) for Sitka spruce, using preferred model paramater inputs

Planting year/rotation	Lowland non-peat	Upland non-peat	Thin peat	Thick peat
1950/1	264	223	28	8
1970/1	489	432	215	196
1990/1	864	763	526	507

Notes: YC = 12, rotation = 1, discount rate = 3.5%, svtC = $10.

Sitka spruce sites are first rotation, planted in the period 1950–1990. Thus, we assess planting in 1950, 1970 and 1990. We assume (again following the FC SCDB) that Sitka spruce is usually in its first rotation, and (except where otherwise indicated) planted on an upland non-peat soil (C gains of 50 tC/ha post-afforestation). In fact, our own map overlays suggest that most existing FC Sitka spruce plantations are on peat soils, especially shallow peat (Brainard et al., 2003), but comparisons of single variable impacts are made simpler by assuming a non-peat soil.

In reality, the exact soil type, planting date and rotation for any particular subcompartment are critical pieces of information. The quality of actual forest records (see Brainard et al., 2003 for more discussion on this point) has much more impact on C flux valuations than any of the variations described in this paper for individual non-economic model parameters.

Table 3 shows the calculated NPVs using all of our preferred values and assumptions (in bold text in Tables 1 and 4a,b), for Sitka spruce on various soils, planted in the given years. Forests on lowland rather than upland soils have NPVs which are greater by $41–$57/ha for sites afforested in 1950 or 1970, and by $101 for a site planted in 1990. This is because of the fast gains in soil C for very young sites. Even when planted on peat, Sitka spruce is still expected to produce net social gains for all years, because the rate of C absorption into live wood and litter is greater than the rates of loss from the peat soils.

Otherwise, it is worth noting how much NPVs vary by year. Young plantations have the benefits of faster accrual of C into soil, as well as distant thinning and felling dates (and thus release of C from products). Differences between soil types are also considerable. The variations in Table 3 by row or column (year or soil type) underscore the importance of accurate forestry inventory data in any attempt to calculate the social value of C flux in British woodlands.

Tables 4a and b show further variations on these baseline scenario NPVs (in italics) for each of the model parameters in Table 1. To facilitate discussion rows (and model parameters) are numbered. NPVs for different years are comma-separated on the same line (i.e., 1950 NPV, 1970 NPV, 1990 NPV), below the parameter value. We discuss Table 4a (non-peat soils) first. For instance, if C releases associated with harvest (row 1 of Table 4a) are 0.94% of the C stored in harvested timber, a stand planted in 1950 has a NPV of $225. Corresponding NPVs for stands planted in 1970 and 1990 are $434 and $764. These values only change slightly as one reads across the row, considering other proposed proportions (1.25% or 1.56%) of C

Table 4a. Per ha NPVs of sequestered carbon, when varying each model input, non-peat soils

Row		Lower estimate	Middle	Upper estimate
1	Harvest releases (% of live wood)	0.94 *\$225,\$434,\$764*	**1.25** *\$223,\$432,\$763*	1.56 *\$223,\$430,\$762*
2	Leaf litter equilibrium levels (tC/ha)	9 *\$210,\$425,\$759*	**12** *\$223,\$432,\$763*	15 *\$229,\$435,\$764*
3	Carbon gain in live wood, 1 standard deviation (tC/ha per annum)	Middle −30% *\$174,\$322,\$566*	**Bateman and Lovett (2000)** *\$223,\$432,\$763*	*Middle+30%* *\$263,\$537,\$958*
4	Release period for products (years)	50 *−\$30,\$283,\$687*	Twice rotation length *\$84,\$351,\$722*	**200** *\$223,\$432,\$763*
5	% C displaced by long-lived products	12.5 *\$79,\$344,\$718*	**50** *\$223,\$432,\$763*	200 *\$802,\$784,\$942*
6	% C release from fossil fuels required to make paper (%)	30 *\$269,\$513,\$808*	**45** *\$223,\$432,\$763*	60 *\$178,\$351,\$717*
7	Amount of C in thinnings	Middle −25% *\$230,\$437,\$766*	**Authors' own function** *\$223,\$432,\$763*	Middle + 25% *\$216,\$426,\$760*
8	Rotation length (years)	FIAP −25% *\$61,\$321,\$600*	**FIAP-based** *\$223,\$432,\$763*	FIAP + 25% *\$229,\$371,\$869*
9	Yield Class, mean regional	10 *\$174,\$311,\$642*	**12** *\$223,\$432,\$763*	14 *\$515,\$764,\$999*
Cumulative soil carbon flux (gains, tC/ha)				
10	Non-peat, lowland.	75 *\$244,\$460,\$813*	**100** *\$264,\$489,\$864*	125 *\$285,\$517,\$914*
11	Non-peat, upland	37.5 *\$213,\$417,\$738*	**50** *\$223,\$432,\$763*	62.5 *\$233,\$446,\$788*

Notes: NPVs are in italics below model parameter values. NPVs for multiple planting years are listed comma-separated in date order on the same line, or *1950 NPV, 1970 NPV, 1990 NPV*. Discount rate = 3.5%, svtC = \$10, rotation = 1, soil type is upland non-peat except where otherwise stated. Preferred model inputs are the values in bold text, such that NPV only varies due to the parameter value/assumption shown above each group of NPVs. See text for other model assumptions.

released by harvesting machinery. Thus the 1950 value (\$225) only reduces to \$223, and the 1970 and 1990 values also only fluctuate by \$1–\$2. The model NPV outputs are obviously not very sensitive to the proposed variations in this parameter.

Three parameters in Table 4a have modest effects on NPVs: the assumptions about leaf litter equilibrium levels (row 2); the amount of C that goes into thinnings (row 7) and the possible gains into upland, non-peat soil (row 11). Varying these

Table 4b. Per ha NPVs of sequestered carbon, when varying each model input, thin and thick peat soils

Row		Lowest estimate	Middle	Upper estimate
Thin peat				
12	Rate of releases from afforested peat	CEH figures −25% *$59,$247,$549*	**CEH figures** *$28,$215,$526*	CEH figures + 25% *−$4,$184,$504*
13	Whether peat reverts to C sink	**NO** *$28,$215,$526*	NA	YES *$148,$215,$526*
14	Cumulative releases from thin peat (tC/ha)	**112** *$28,$215,$526*	NA	230 *$28,$215,$526*
15	Usual annual sink (lost post-afforestation), tC/ha per annum	0.0375 *$42,$229,$540*	**0.125** *$28,$215,$526*	0.25 *$8,$196,$507*
16	Year that sink ceases	2013 *$36,$224,$535*	**2023** *$28,$215,$526*	2043 *$16,$204,$515*
17	Pattern of C losses, slow, uneven, perpetual or fast, steady, temporary	**Slow….(CEH)** *$28,$215,$526*	NA	Fast… (Zerva et al.) *$424,$205,$248*
Thick peat				
18	Rate of releases from afforested peat	CEH figures −25% *$44,$232,$534*	**CEH figures** *$8,$196,$507*	CEH figures + 25% *−$29,$159,$479*
19	Whether peat reverts to C sink	**NO** *$8,$196,$507*	NA	Yes *$128,$196,$507*
20	Usual annual sink (lost post-afforestation, tC/ha per annum	0.175 *$20,$207,$519*	**0.25** *$8,$196,$507*	0.325 *-$4,$184,$495*
21	Year that sink ceases	2013 *$25,$213,$524*	**2023** *$8,$196,$507*	2043 *-$16,$172,$483*

Notes: NA: no middle assumption (only 2 variable assumptions) tested. NPVs are in italics below model parameter values. NPVs for multiple planting years are usually listed comma-separated in date order on the same line, or *1950 NPV, 1970 NPV, 1990 NPV*. Discount rate = 3.5%, svtC = $10, first rotation. Preferred model inputs are the values in bold text, such that NPV only varies due to the parameter value/assumption shown above each group of NPVs. See text for other model assumptions.

parameters alters the models by $50/ha or less, within any given planting year. Somewhat more impact (up to $101/ha) on output NPVs results from the assumed gain in lowland non-peat soils (row 10), or the percentage of fossil fuel C required to make paper (row 6). The fluctuations are not great in percentage terms in rows 6 and 10, however.

How much NPV estimates change depends greatly on planting year with respect to the release period for C in products (row 4). For plantations planted in 1990, there is moderate change in row 4. But for 1970 and 1950 plantations, NPVs vary dramatically depending on the total life-time of products. The 1950 plantation, in particular, changes from a net cost (-\$30) to a moderate benefit (\$223). A similar effect is shown when rotation length is varied (row 8) or uncertainty about the total C gain in live wood (row 3) is accounted for. However, the largest variations (in absolute or percentage terms, and for all years) are to be seen with respect to the amount of C displaced by long-lived products (row 5), and the regional YC (row 9).

Table 4b relates to both thin peat (top of table) and thick peat soils (bottom part of table). In most cases, the effects of the given variables are relatively small (usually ⇐\$40/ha, or about 7% change between assumptions). The assumption about cumulative releases from thin peat (row 14) has virtually no effect on NPVs, which is reassuring given the paucity of information about what the true value might be. Whether peat reverts to a C sink (rows 13 and 19), and the rate of C release on afforested peat soils (rows 12 and 18) affect the 1950 NPV significantly, but not the 1970 and 1990 NPV. The most dramatic changes to NPVs for all planting dates occur with respect to the C flux on afforested thin peat (whether to assume the CEH or Zerva et al. (2005) patterns; row 17). The 1950 value changes from \$28 (CEH) to \$424 (Zerva et al., 2005), because of the different patterns of C flux in the second rotation. The 1970 value barely changes, but the 1990 value more than halves, falling from \$526 to \$248, because of high releases in the first rotation under the assumptions suggested by Zerva et al. (2005). There would be much utility in verifying either or even both the CEH and Zerva et al. (2005) chronosequences of C release rates on afforested peat.

As a final consideration of single-variable inputs to the models, we list in Table 5 variations in NPV from the preferred model assumptions (bold text in Table 1) if we allow both discount rate (r) and the social value per tonne of C (svtC) to vary; otherwise, model assumptions are the same as the preferred assumptions used to derive Tables 4a and b. For the 2001–2010 period, Pearce (2003) reports $r = 5\%$ and \$17.10 from Tol (1999) and 1% and \$109.50 from Eyre et al. (1997). We further include values of 3% and \$1.50 as argued for by Mendelsohn (2003). These choices are not meant to be thoroughly representative of the literature or to give definitive

Table 5. Variations in NPVs with discount rate (r) and svtC (\$US)

Origin and parameters/ soils	UK Treasury $r = 3.5\%$, svtC = \$10	Mendelsohn (2003) $r = 3\%$, svtC = \$1.50	Tol (1999, FUND 1.6) $r = 5\%$, svtC = \$17.10	Eyre et al. (1997) $r = 1\%$, svtC = \$109.5
Lowland	489	86	695	11,615
Upland	432	76	619	9976
Thin peat	215	36	335	1298
Thick peat	196	33	304	524

Notes: YC = 12, rotation = 1, r = discount rate, svtC = social value per tonne of carbon, planting year = 1970. See Table 1 and text for other model assumptions.

answers; they are chosen only to give some indication of the variations that result from different combinations of *r* and svtC.

Reading across each row, the variations in NPV estimates are very large. The range of variation in each row is much greater than that seen across any row in Tables 4a and b.[13] In terms of valuing the C sequestration function of woodlands, greatest emphasis should be placed on accurately estimating the true social value of sequestered C, given that the svtC exerts the most impact on estimates of NPV. The importance of other model assumptions, e.g., those that we explore in Tables 4a and b, is secondary. Otherwise, we only make general observations about Table 5. The value of NPVs varies little between plantings on thick or thin peat soils, or types of non-peat soils. Afforestation on all types of soils is positive, regardless of *r*/svtC combination, but NPVs derived using Mendelsohn's values for *r*/svtC are relatively low. These estimates are particularly likely to push below zero in response to small changes in our preferred model parameters and assumptions.

Varying multiple model parameters to predict NPV

We next assess the combined effect of the parameter variations in calculated NPVs. Most variations on model parameter values are included; however, for many reasons, *r*/svtC are held constant at 3.5%/\$10. The question of which *r*/svtC values to use is difficult to resolve, and yet, as Table 5 demonstrates, has dramatic effects on the results. Moreover, the British government is committed to a discount rate of 3.5% for public sector investments. The UK Treasury's recommended value for svtC is still under review, but NPVs resulting from alternative svtCs can easily be derived from our results by applying the relevant arithmetic factor (e.g., to apply an svtC of \$1, divide our NPVs by ten).

In the following results we hold certain variables constant. This is either because they tend to be known (YC, rotation, planting date), or because in Tables 4a and b they had very little impact on NPVs (leaf litter equilibrium levels, C releases associated with harvest, cumulative releases from thin peat). We set the release periods for products to equal 200 years in all cases, because we have little confidence in the alternative values of 50 years or twice rotation length. Otherwise, for a given planting year and rotation, there were 729 model runs for non-peat soils, and 6561 (thick peat) or 19,683 (thin peat) permutations on peat. Computer programs were written in the Perl language (Christiansen and Torkington, 1998) for calculating NPVs on each soil type, using the parameter variations listed in Table 1. First, second and third rotations were included for planting every twenty years in the period 1950–1990. The outputs are similar to what would result from a Monte Carlo analysis, but cannot be correctly described as such because we cannot assume that

[13]It is beyond the scope of this paper to thoroughly discuss whether the ranges of variables in our physical model (of C uptake) are as plausible as the suggested variations in possible discount rate and svtC. If we were able to reliably quantify the probability of any economic or physical model parameter being the "true" value, then this paper would comprise a genuine Monte Carlo approach rather than a mere sensitivity analysis.

Table 6. NPV medians (5th and 95th percentiles) from simulations where most model parameters vary at the same time

Planting	Soil type			
Year/ rotation	Lowland	Upland	Thin peat	Thick peat
1950/1	**224 (−74, 357)**	**181 (−118, 306)**	**136 (−245, 479)**	**−12 (−310, 168)**
1970/1	419 (287, 615)	359 (233, 544)	**177 (−16, 388)**	143 (3, 338)
1990/1	813 (549, 1197)	721 (452, 1096)	**368 (−11, 796)**	460 (208, 828)
1950/2	**37 (−134, 112)**	**19 (−158, 90)**	**−80 (−259, 48)**	**−121 (−313, 17)**
1970/2	193 (95, 321)	167 (72, 288)	**70 (−75, 222)**	**0 (−121, 156)**
1990/2	478 (319, 670)	444 (289, 637)	520 (142, 927)	266 (88, 501)
1950/3	**−87 (−247, −6)**	**−100 (−259, −20)**	**−172 (−352,−51)**	**−220 (−406, −77)**
1970/3	**76 (−46, 188)**	**57 (−60, 169)**	**−26 (−184, 103)**	**−95 (−236, 46)**
1990/3	320 (215, 479)	301 (194, 460)	218 (66, 399)	161 (17, 336)

Notes: Base year = 2003, currency = $US, discount rate = 3.5%, svtC = $10/tC, YC = 12. Numbers are median NPV (5th percentile NPV, 95th percentile NPV). Bold text indicates cases where at least 5% of the NPV predictions were negative.

each of the input parameters has equal likelihood of being the "true" value. Because the outputs tend to have bimodal or skewed distributions, they are best treated non-parametrically. Table 6 lists the median NPVs, with fifth and 95th percentiles, for example years and rotations, on given soil types.

Although the 5th to 95th percentile intervals tend to be large ($200–$500), most of the median NPVs are positive. But a small majority of stands, particularly older (1950 or 1970-planted) stands in later rotations on peat, appear to have at least a 5% chance of being net C sources rather than sinks (bold text). This is because the C gains to be made into soils under these stands are small, and felling (with its associated high C releases) is not far in the future.

It is useful to see how NPVs based on our preferred assumptions (Table 3) compare with the results in Table 6. Most often the values in Table 3 are greater than the median estimates in Table 6, for the same year/rotation/soil type combination. An example is 1990 planting, first rotation on lowland soils. The estimate using our preferred parameters is $864 (in Table 3), which compares to a 50th percentile value of $813 in Table 6. In general the differences are around $40-$75 for non-peat soils or thick peat. The discrepancies are greatest (usually more than $100) on shallow peat soils, which is to be expected with the larger number of uncertainties for this soil type. However, it is reassuring to note that in all cases the NPVs from our preferred model assumptions (in Table 3) are comfortably within the 5th–95th percentile ranges (in Table 6).

Final comments

"Uncertainty abounds in climate change" wrote Tol (2003, pg. 265). This statement holds very true in any attempt to quantify or value a C sink. This paper represents a start at trying to gauge how the uncertainties may affect our attempts to calculate the value of C sequestration associated with British woodlands. We believe that our work:

- Demonstrates which assumptions in C valuation work have the most impact when calculating the NPV of future C flux in woodlands. The most important information is forestry inventory records and economic parameters (appropriate discount rate and social value per tonne of C). Secondary to those considerations are assumptions about displacement factors, disposal options and the rate of and total uptake of C into live wood.
- Indicates (Table 4a) which information for specific types of plantations would be most valuable in attempting to accurately estimate the social value of C flux. This includes yield class on young plantations, and C flux on all soil types, but especially peats.
- Demonstrates (Table 6) that input of our preferred parameter values to the modelling produces NPVs that are representative of NPV estimates derived from a range of plausible model parameter inputs.
- Strongly supports (Table 6) the FC policy that afforestation on deep peat soils should be discouraged.
- Suggests (Table 6) that most FC-owned Sitka spruce plantations provide C sequestration benefits, although it is certainly possible (at least 5% of model predictions) that there is nil or negative utility associated with sites planted on peat many decades ago, or those currently in their second or later rotation.
- Strongly suggests (Table 6) that by the time Sitka spruce stands are well into their 3rd or later rotation, the net C flux from these sites and their products will usually be negative, regardless of soil type.
- Provides evidence (Fig. 1; Table 4a) that thinnings can generally be ignored in C accounting studies of conifer woodlands because of their negligible net impacts on C storage.
- Provides a range of estimates for the value of the ongoing C sequestration function of British woodlands, with which to compare to other possible land uses.

Our modelling attempts to describe and quantify uncertainties that surround estimates of C sequestration benefits into woodland. The modelling is not perfect, but it is hopefully adequate for the purpose of investigating which of many variables that enter such modelling matter most. There are inevitably factors that we omit. For instance, our work ignores completely the non-merchantable tree biomass in thinnings after the first thinning date. This omission imposes a downward bias on our NPV estimates. What if elevated atmospheric levels of CO_2 alter forest productivity? There is research to suggest that increased atmospheric CO_2 may reduce tree growth rates, at least for some species and in some climates (Cox et al., 2001; Clark et al., 2003). Other recent research suggests that C uptake into afforested soils may decrease with increases in the concentration of atmospheric CO_2 (Heath et al., 2005). Our work also does not attempt to include such interaction effects. What about C in the ground flora? Patenaude et al. (2003) showed that this can add significantly to the C stores in semi-natural woodlands, but significant undergrowth and C flux seem unlikely within conifer plantations.

The modelling is simplistic in many other ways. For instance, we also don't vary any of the elements underpinning calculation of the exact growth rate of trees. The work further assumes that the C stocks in roots volatise into the atmosphere within two years of harvest; in reality the C in tree roots slowly decays into soil and is transferred from soil into the atmosphere (see Matamala et al., 2003 for relevant discussion).

We take no account of greenhouse-gas emissions other than CO_2; a complete GHG budget for forestry would need to also include the more potent GHGs, nitrous oxide and methane (CH_4). This is particularly relevant with regard to disposal options. We assume that all C in disposed wood products quickly and entirely decays to CO_2. In reality, disposal options significantly affect how C is released. For instance, burning wood products results in immediate release of almost all C as CO_2. Currently, most solid wood and newspaper products are buried in landfill, where decay is very slow: some C will effectively never be released at all, while a small proportion is released as CH_4, due to complex physical and chemical processes (Skog and Nicholson, 2000). CH_4 can be captured and used as fuel; burning CH_4 for energy has the benefits of both displacing fossil fuel sources, and releasing C in the less potent GHG form of CO_2. However, only a minority of British landfills have capture facilities, and there is great uncertainty about what percentage of landfill CH_4 could potentially be collected, anyway (United Kingdom House of Lords, 1998). Barlaz (2004) summarises previous studies, which calculated that 50–84% of the C in paper and other wood products may remain long-term in landfills. Reviewing Barlaz, Miner (2003) concluded that, even in the absence of facilities to capture CH_4, approximately 40% of the C (or equivalents, as CH_4 releases) in paper/wood products is stored at least 100 years. Unfortunately, as Barlaz highlights, understanding of C storage functions and decay processes in landfills is still poor. Such research has not progressed to a point that we could confidently include different disposal options in the modelling in this paper.[14]

It can be argued that increases in the C stocks in GB woodlands serve only to displace C in woodland stocks overseas, and thus growing more trees in GB does nothing on a global scale to reduce global warming. We do not attempt to address how changes in this trade balance might affect our results. We also ignore the possibilities of reuse and recycling of wood products.

Most of all, in this study, we do not attempt to estimate the overall value of C flux in existing GB forests. There are many uncertainties inherent in the existing inventories of Britain's woodlands (FC SCDB and The National Inventory of Woodland and Trees). Prerequisite data, such as planting year, species type, underlying soils, or rotation are often not available within these sources and would have to be subject to educated guesses.

[14]Moreover, there are high social costs associated with landfill disposal that might need to be considered (Powell and Brisson, 1994).

Hopefully our study provides a range of credible estimates – with some indication of associated uncertainty – of the potential value of C sequestration into woodlands. These may be useful to compare with the possible value of C sequestration associated with other land uses. The models and the estimates of NPV are predicated upon the assumption that forests are managed using (currently) conventional economic criteria, and that the land use would not change for the next 1000 years. This is an unrealistic assumption; historically land in GB has passed in and out of many types of uses in the last millenium. Even if the land use did not change over this long time period, there are radical alternative management regimes, which would significantly alter the results. For instance, NPVs might decrease dramatically if harvests were only used to produce biofuels (trees were not left to grow to maturity but were cut down at relatively young young ages). Similarly, leaving all crops to develop into old-growth forests would probably also decrease NPVs. Although we do not directly address these possibilities, this work does provide a basis for comparisons to be made with many possible alternative land uses.

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