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ANALYSIS

The social value of carbon sequestered in Great Britain's woodlands

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ABSTRACT

The economic value of carbon storage associated with British woodland is calculated. Models were developed to estimate C flux associated with live trees, forest floor litter, soils, wood products, harvest, fossil fuel used in manufacturing and C displacement from biofuels and products for representative British plantation species: Sitka spruce (*Picea sitchensis*) and beech (*Fagus sylvatica*). Map databases of publicly and privately owned woodlands were compiled for Great Britain. Carbon flux was determined for individual woodland sites, and monetised using candidate parameters for the social discount rate (1, 3, 3.5 or 5%) and social value of carbon (US\$109.5, \$1, \$10 or \$17.10/t). A conventional discount function was applied. Final results are expressed as Net Present Values, for the base year 2001, with discounting commencing in 2002. The minimum suggested NPV (discount rate = 3% and social value of carbon = \$1) of GB woodlands already existing in 2001 is \$82 million, with a further \$72 million that might be added by future afforestation. These figures rise dramatically if a discount rate of 1% and social value of sequestered carbon = \$109.5/t are assumed. The calculated total value of C stored in British woodland depends significantly on parameter assumptions, especially about appropriate discount rate and social value of sequestered carbon.

1. Introduction

Adverse climate change in the next two centuries is widely anticipated due to rises in global temperatures. The anthropogenic emission of greenhouse gases (GHGs) is considered a significant cause of this global warming. The main British GHG is carbon dioxide, which is believed to comprise about 84% of the total global warming potential of GHG releases in the United Kingdom (DEFRA ,2002). One tactic for mitigating these emissions could be to create more woodlands, as both trees and woodland soils take up and store more atmospheric CO_2 than many other land uses. However, a deliberate strategy to offset British GHG releases by promoting massive afforestation of new areas is unlikely to happen, due to many issues including practical problems and competing pressures on land use (Cannell, 1999). Nevertheless, accounting for the economic value of C-storage function in British woodland is still valid from a social benefits perspective. Many government subsidies exist to create and maintain British forests — correctly estimating the value of carbon stored in existing and likely future woodlands can help estimate the true costs and benefits of these taxpayer-funded grants.

Much work has already been undertaken to estimate the total C sequestration into GB woodlands (Cannell and Dewar,

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1995; Cannell and Milne, 1995; Cannell, 1999). Our work builds on the previous research to calculate both the total C that British woodlands might sequester, as well as the total *value* of that C-sequestration function. This is important because the UK government is committed to cost–benefit analysis as a policy making tool (HM Treasury, 2003). Although storing carbon in British woodlands would only be a small stopgap strategy to threatened climate change (Cannell, 1999), the UK government agency responsible for forestry throughout Great Britain (Forestry Commission = FC), receives a substantial public grant (projected to be £94 million annually, in the period 2003–2006; HM Treasury, 2002) for running costs. Meanwhile, taxpayer subsidies to privately owned British woodlands have been estimated at £40 million/year (Henke and Evans, 2005). It is therefore appropriate to assess the social benefits of woodlands (including the carbon sequestration function) when calculating the true cost of these taxpayer-funded subsidies.

This study also takes previous research on C sequestration in British woodlands further by incorporating a detailed lifetime analysis of woodland products, C displacement from products, C storage in thinning products, harvesting and manufacture-related releases of CO_2 and post-afforestation changes of C levels in different types of soils. The analysis also allows for C sequestration under different types of woodland (ancient/semi-natural woods, commercial plantations, Christmas trees and coppice). This work calculates and values marginal C flux for all woodlands in Great Britain (GB), including England, Wales, Scotland, and immediate offshore islands, but not the Channel Islands, Isle of Man, Northern Ireland or Scilly Isles. By 'marginal' flux, we mean all C flux associated with woodland land use for the year 2001 and onwards[3]. Separate statistics are presented for woodland managed under the auspices of either the Forestry Commission (FC) or other bodies. Three species groups in the FC estate were selected for individual analysis: Sitka spruce (*Picea sitchensis*), oak (*Quercus* species) and beech (*Fagus sylvatica*). Sitka spruce is the softwood species with greatest areal coverage (about 49% of the total Forestry Commission area in GB that is planted in conifers; FC, 2001). Oaks are the most common broadleaf species in the FC estate, comprising of approximately 10% of hardwood plantings, with beech making up another 3.5% of broadleaf areas. When considering carbon sequestration in British woodland, or its potential to mitigate global warming, Sitka spruce and beech are the species most commonly used as surrogates for other conifers or broadleaves (e.g. Cannell et al., 1996; Milne et al., 1998). Compared to many countries, carbon inventories and calculation of the value of C sequestered in woodlands is a particularly credible exercise for Britain, due to the completeness of public records on the planting dates and locations of woodland sites.

[3] It may be argued that marginal C flux is inadequate, C flux before base year should be included in calculations of the social benefits woodlands provide now and into the future, because existing trees provide C-storage benefits until they are cut down. Such sunk C in British woodlands was valued in Brainard et al. (2003).

2. Methodology

Our description of methodology, data sources, model construction and verification is brief and readers are referred to Brainard et al. (2003, 2006) for more details. Carbon storage in live wood is dependent on stand-specific variables. Many of these data were available as part of detailed forest records, but other factors had to be assumed or predicted from covariates. Because the work necessarily involved a large number of assumptions, we conducted thorough sensitivity analyses that are described in Brainard et al. (2003, 2006) which address the variability introduced by individual model parameters and assumptions. The implications are discussed in later sections of this paper.

Forest inventory records were linked with other mapped environmental data within a geographic information system. Mean numerical climate and topography variables, as well as majority categorical soil characteristics were determined for each woodland record. Monetising carbon stocks requires the specification of various economic parameters, all of which have inspired lengthy academic debate. We include a range of suggestions for the social value per tonne of C (svtC), and the discount rate. Computer programs written in the Perl language (Christiansen and Torkington, 1998) calculated the corresponding amounts of sequestered carbon and to derive Net Present Values (NPV) under different economic assumptions. NPVs are the present gains from carbon sequestration, plus all future benefits minus both initial and future costs (carbon losses in some cases, as on disturbed peat soils). Rather than NPVs, we would have preferred to list annuities (per annum benefits). Unfortunately, annuities were not straightforward to calculate. Carbon sequestration, unlike other forest benefits, changes over time. C is taken up faster in early rotations, resulting in a benefit stream that is inconsistent over multiple rotations. Application of standard annuity formulae (Price, 1989) would result in over-estimates of per annum benefits for any sites afforested relatively recently. The errors would probably be small, but we don't know of an easy and practical method for circumventing this problem to produce reliable annuity equivalents of NPV for thousands of woodland sites in varying rotations, on varying soil types, of variable species composition, etc.

The next section describes both data sources and model assumptions and how these were incorporated into the models of carbon storage/release. This study builds upon works published previously (Bateman and Lovett, 2000; Brainard et al., 2003, 2006). The analysis calculates the total value of carbon stored in soils, trees and wood products, by species group (conifer or broadleaves), per hectare.

3. Data and model

3.1. *Data on woodland cover*

Digital woodland cover in GB came from three sources: the FC sub-compartment database (FC SCDB), the Woodland Inventory (WI), and ancient woodland cover in England. The FC SCDB is a digital catalogue of all land overseen by the FC's management arm, Forest Enterprise. The data are nominally accurate to the year 2001. The FC SCDB includes information on SC size, yield class, proportion of area planted in trees, planting year for trees currently on site and rotation (whether this is the first, second or later crop of trees in this SC). The WI is an extract of a larger database, the National Inventory of Woodland and Trees (NIWT; FC, 2003), derived from aerial photography dated 1988 to 2001. Only areas of 2 ha or larger were mapped in the WI, which is believed to exclude no more than 5% of GB woodland. The data categorise woodlands using feature codes and descriptors that usually give some indication of likely species and age. We treat individual, contiguous WI areas with the same descriptor code as WI "sub-compartments" (i.e., WI SCs), although it is unlikely that most such areas were actually managed as single units.

We assumed that any ancient woodland site being managed under the auspices of the FC would already be recorded in the FC SCDB, with complete and accurate records for current planting date and number of previous rotations. Therefore, we were mostly concerned about identifying ancient woodlands on WI (non-FC) land. Point grid references for 22,572 ancient woodland sites (AW) in England were downloaded from the National Digital Archive of Datasets held at the University of London computer centre. The NDAD data were originally compiled by English Nature in the period 1986–2002, and designate areas in England that were in continuous woodland cover since the year 1600 or earlier until at least the 1920s. Attributes provided with the AW data included total afforested area and current condition ("Good", "Unknown" or "Poor"). Boundary information was unavailable. We approximated the boundaries by assuming that each AW had a perfectly circular shape with the given point grid reference at its centre. The given total area of a specific AW was used to derive the circle radius. Centroids for WI areas were overlain with AW circles to identify WI sites that our model treats as ancient woodlands. If no WI centroid fell within an AW area, we assumed that the land use at that site was no longer woodland, and the record was ignored for carbon accounting purposes.

Four woodland types were recognized: plantation, coppice, Christmas trees and ancient. These distinctions have most impact on rotation length and release period for products. In most cases we treat stands as actively managed plantation: expected to grow to maturity, clear-felled and restocked according to market demand. The percentage of unthinned forests is not accurately known, but approximate figures of 50–70% have been suggested by colleagues (Colin Price, pers. comm., 2005). We therefore assume that 60% of stands are unthinned, while 40% of sites are actively managed and regularly (intermediate intensity) thinned. How felling and thinning dates were calculated for plantation woodlands is described at length in Bateman and Lovett (2000). Following advice from FC staff and the British Christmas Tree Growers Association (BCTGA), the models assume coppicing every 12 years, and that Christmas trees are felled every 10 years. We assume that carbon in coppice products has a release period of 22 years, of which 50% occurs within the first 3 years after felling. It is assumed that nine previous rotations have occurred among coppice, and four among Christmas trees plantations. All carbon in Christmas trees is released in the year of harvest. After consultation with the BCTGA, we assumed that 10,000 ha of WI areas are in Christmas tree production, but acknowledge that the true figure may be 30–40% more or less. The justifications for these assumptions are explained in greater detail in Brainard et al. (2003).

Where the WI codes indicated mature trees, ancient woodlands were assigned a planting date of 1874 (this accords with the age profiles of non-FC woodland in FC, 2001). If the site was also labeled as being in "Good" condition, a very long rotation period (350 years) was assumed. If stocked with mature trees but labeled as "Poor" or "Unknown" condition the model cut the trees down in 2002. An average felling date of 2002 is plausible, as earlier rather than later felling dates seems more likely on sites not well-managed for conservation value (where the condition was known for ancient woodlands, it was more likely to be "Poor" than "Good"), and bearing in mind that some of the sites were identified from aerial photographs dating from the late 1980s. An alternative would have been to assign random felling dates to such sites, but this would have been awkward to implement, and arguably just as arbitrary as felling all such sites in 2002. When the WI codes indicated a site stocked with young trees, these were assigned recent planting dates (1980 for broadleaves, 1990 for conifers), and presumed to be managed like any commercial plantation, although rotation value was still set at 3.

Tree productivity, species, planting year and rotation data were vital inputs to the carbon sequestration models. Modelling to predict yield from other environmental characteristics was undertaken. Where certain variables were missing, we generally assigned values using the observed proportions as recorded in the FC SCDB or FC (2001). Brainard et al. (2003) give extensive details of variable assignments where data were otherwise missing or ambiguous. Our final estimate of the total woodland area and for specific species types, corresponded closely to FC (2001).

3.2. *Carbon uptake in live trees*

The approach relied on underlying assumptions about carbon accrual in beech woodlands at yield class (YC) = 4, and Sitka spruce at YC = 12. Bateman and Lovett (2000) modelled sequestration rates using regression analysis on published data on the carbon gains in these species at these YC, both before and after thinning, from storage data in Cannell and Cape (1991) and yield data from Edwards and Christie (1981) on merchantable volume (MV). MV can be related directly to total carbon storage (Corbyn et al., 1988; Matthews, 1991).

The models use actual planting year where known, else an estimate of likely afforestation year. Forest stands are assumed to be managed only for commercial criteria. This means that trees are felled when their continued rate of growth ceases to be profitable; rotation lengths and consequent C uptake into biomass, are therefore dependent on discount rate. Optimal rotation lengths, for given YCs and discount rates, were calculated by the Forestry Commission's own woodland management software FIAP. When afforestation year is estimated, the modelling assumes a maximum of two previous rotations prior to the current planting, except for coppice, ancient and Christmas trees. We discuss in our concluding remarks some possible impacts on our final results if noncommercial criteria are used to estimate rotation length and thinning dates.

3.3. C storage and release from products

Bateman et al. (2003) undertook a detailed lifetime analysis of UK domestic production of hardwood and softwood products. This included the study of the proportion of wood that went into products with different lifetimes, and the C release rates from these. Bateman et al. (2003) summed C release curves for different types of products of varying lifetimes, using proportions for the entire UK annual domestic production. Regression analysis was undertaken on the summed data to generate an inverse power function predicting the proportion of C released from the total carbon in harvested timber, as a function of years after felling. About 50% of C from all domestic production (from both thinnings and fellings) is estimated to be released by year 31 (conifers) and year 28 (beech) post harvest.

We adapted the basic release function for harvest or thinning for specific YCs. Hamilton and Christie (1971) give schedules for the expected quantity of harvested materials by log diameter (24 cm, 18 cm or 7 cm) and by yield classes (6–24), at both felling and thinning. From these data we can observe, for instance, that a Sitka spruce stand of YC=12 would be expected to yield 49% of its products as long-life products (24 cm diameter) and 33% of its products as hardboard, pallets and other medium-life products (top diameter=18 cm). This makes for a total of 82% of products as medium- or long-life products. Similarly, thinning a Sitka spruce, YC=12 stand 10 years before felling (50–51 years), is forecasted (in Hamilton and Christie, 1971) to yield 10% 24 cm logs, 40% 18 cm logs, and 50% 7 cm diameter logs.

We use the percentages of short-, medium- and long-life products to scale the products function reported in Bateman and Lovett (2000). The release period for the predicted percentage of short-life products remains six years, but the exact percentage of the harvest that is released in this 6 years varies according to the schedule information in Hamilton and Christie (1971) (for that YC and species). The remaining products (medium+long life) are assumed to be released entirely in years 7–200 (for softwoods) or 7–300 (for hardwoods) after harvest. Wood product release functions that are very similar to ours are given in Karjalainen et al. (1994 ; applicable to Finland) and Skog and Nicholson (2000; a North American study). The half-life of C release from the scaled product functions thus varies, depending on the proportions of short- and medium-life products that went into each thinning or harvest. For instance, C half-lives from the products from felling Sitka spruce YC=10, 12, 16 and 20 are respectively 37, 44, 53 and 56 years. The C half-lives for thinning Sitka spruce, same YCs, 10 years before felling date, are much shorter at (respectively) 3, 6, 23 and 32 years.

Under an intermediate, marginal thinning regime, optimal thinning intervals for YC=8–18 (typical of conifers) are given in Hamilton and Christie (1971) as every 5 years. It is suggested that optimal thinning frequencies for YC=2–6 (typical of broadleaves) would be about 10 years. We only consider thinnings starting at 25 years before felling; products extracted earlier are overwhelmingly short-lived (e.g., paper) and therefore expected to have negligible impact on long-term C stores.

We used the harvest schedule given in Hamilton and Christie (1971) to estimate how much (volume) would be removed at each thinning date (as a percentage of what was left behind). This translates directly as an estimate of what percentage (typically about 15%) of the C in live wood that would be expected to be removed at each thinning date.

3.4. Soils

Woodland tends to have higher equilibrium levels of soil carbon than pasture or farmland, which means that a period of C gains into soils often occurs post-afforestation. Some soils emit C post-afforestation, however. The decomposition of carbon 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). In consequence, most undisturbed British peat bogs have an ongoing net annual increase of soil carbon. Drainage of peat (necessary as a precursor to afforestation) therefore has two detrimental effects with regard to carbon sequestration. It removes both an existing sink, and the anaerobic conditions that otherwise cause C to be retained, leading to an extended period of carbon release.

Data from the National Soil Resources Institute and Macaulay Land Use Research Institute were used to indicate the presence of mineral (i.e., non-peat), shallow (or 'thin', <45 cm depth) and deep (or 'thick', >45 cm depth) organic soils in 1×1 km squares. We treat organic and peat soils as equivalent. Each woodland area (FC or WI SC) was assigned the soil status (non-peat, thin peat or thick peat) of the area coinciding with the majority of that SC.

Adger et al. (1992) reported equilibrium soil carbon levels for a variety of soils and land uses, suggesting that post-afforestation gains on non-peat soils might typically be about 100 t C ha^{-1} on lowlands (<150 m elevation), and 50 t C ha^{-1} in upland areas (which we define as ≥150 m elevation). This distinction between upland and lowland is inevitably somewhat arbitrary, but a map of these areas has close correspondence with upland and lowland habitats as identified by the Countryside Survey 2000 (Haines-Young et al., 2000).

The rate of gain in non-peat soils was modelled by consulting published sources and soil science experts. Robert Sheil (pers. comm., 1994) suggested that about 95% of the net gains in soil carbon will occur within 200 years of planting. Bateman and Lovett (2000) undertook regression analysis of published soil carbon uptake curves (Sampson, 1992; Dewar and Cannell, 1992; Matthews, 1993), to produce a model predicting percentage of per annum soil carbon gain over time. The cumulative gain was constrained to equal 95% at 200 years post-afforestation. The model predicted 100% of total carbon gain at 265 years post-afforestation.

From various sources Bateman and Lovett (2000) concluded that the maximum C release from newly afforested deep peat bogs was 750 t C ha^{-1}. We assume that cumulative total C releases on thin peat soils are approximately 15% of the suggested maximum from thick peat soils, or 112 t C ha^{-1}, which is consistent with observations (Zerva et al., 2005) of C soil levels pre- and post-afforestation on peaty gley (a common shallow organic soil in northern England).

From original field measurements and computer modelling, Hargreaves et al. (2003) constructed a descriptive model of carbon exchanges on afforested peat soils in Scotland. The models indicate, based on a chronosequence, that afforested deep peat might emit carbon long term at a rate of about 0.3 t ha^{-1} $year^{-1}$. We treat this value (0.3 t C ha^{-1}) as the expected annual C release on afforested peat (either thick or thin peat) after 26 years. This loss continues in perpetuity, or until the total expected maximum is reached, i.e., 750 t C ha^{-1} for deep peat soils, 112 t C ha^{-1} for shallow peat.

Hargreaves et al. (2003) observed actual carbon accumulation rates on undisturbed deep peat of 0.22 and 0.25 t C ha^{-1} $year^{-1}$ at two Scottish sites. Accordingly we assume that the usual sink in British peat bogs is about 0.25 t C ha^{-1} $year^{-1}$. We assume that the usual C sink on thin peat is half of that for thick peat (0.125 t C ha^{-1} $year^{-1}$). However, peat soils may lose their carbon fixing function as global temperatures rise (Chapman and Thurlow, 1998; Bousquet et al., 2000), which could theoretically happen as soon as the year 2024. The sinks of 0.25 and 0.125 t C ha^{-1} $year^{-1}$ are thus directly subtracted from possible C gains in woodlands planted on deep peat, but only until the year 2024. Sensitivity analysis in Brainard et al. (2006) showed that varying these assumptions (by ±25%, or ±10 years) only has very small impacts on final valuations.

3.5. *Other C sources and sinks associated with woodlands*

3.5.1. *Litter*

Conifer woodlands in the UK accrue carbon in forest floor litter at around 0.25–0.32 t C ha^{-1} $year^{-1}$ (Milne and Brown, 2002). Annually, up to 50% of C in this deposition may be transferred into soils (Milne et al., 1998). 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 carbon deposition in the forest floor very simply. We assume that carbon gains in forest floor litter are proportional to increases in live wood carbon, scaled such that the maximum rate of net gain into leaf litter is 0.25 t C ha^{-1} $year^{-1}$, for a stand with YC=12. The long-term equilibrium is assumed to be 12 t C ha^{-1} (middle estimate by Dewar and Cannell,1992). The model thus ceases C accrual into leaf litter when the cumulative total C reaches 12 t C ha^{-1} for YC=14. Equilibrium levels for C in leaf litter at other yield classes are scaled as a proportion of YC=14. The addition of C in litter applies only to non-peat soils; the chronosequence developed by Hargreaves et al. (2003) included C in leaf litter and ground flora.

3.5.2. *Harvest and primary transport*

Harvesting and management of a timber site create its own carbon emissions. Karjalainen and Asikainen (1996) estimated that annual harvest-related releases were 1.4% of the amount of carbon in harvested Finnish timber. We believe that harvest-related releases in the UK are somewhat lower, due to milder climate and shorter transport distances. We assume that C releases associated with harvest are slightly less, at 1.25% of the C in the harvested timber. This only refers to primary silvicultural and harvesting activities.

3.5.3. *Displacement in, medium-life, long-life and fuel products*

Wood products often displace carbon releases from the energy used to manufacture other materials (such as fuel, plastic, steel or aluminium). Inclusion of such displacement can significantly reduce C emissions, particularly for early rotation crops (see Brainard et al., 2006 for detailed examples). Marland et al. (1997) provided estimates of the displaced C associated with different types of wood products. They compared the C required (in energy use) to manufacture wood versus substitute products, concluding that about 25% of the carbon in medium-lived products (such as packaging, pallets and board) would displace the C used for substitutes. However, estimates for the carbon displaced by long-term wood products (such as construction beams, which may replace concrete or steel) varied widely, from 50% to 230% of the C in harvested wood products. We adopt a value of 50% for long-lived products, which is what Marland et al. (1997) used in their own work, and is relatively conservative whilst being in approximate agreement with other sources (Brainard et al., 2006).

Biofuels are another benefit, particularly of coppice woodlands. For each 1 kg of carbon in wood fuels, Marland et al. (1997) estimated that 60% (600 g) displaces fossil fuels that might otherwise have been used. In our models, this displacement only applies to the first 50% (released in the first 3 years) of C in coppice products, and to 3% of the total hardwood or conifer harvest on plantation stands.

The model treats the C displacement as occurring during the first 3 years after harvest for coppice biofuels, and during the year after felling for other products.

3.5.4. *Energy to manufacture paper*

The displacement factors estimated by Marland et al. (1997) for medium- and long-life products also account for secondary (manufacturing-related) C emissions. Only the fossil fuel use required to manufacture short-lived (overwhelming paper) products is unaccounted for. It is difficult to confidently calculate the fossil fuel inputs required to make paper. Although it has been claimed that virgin paper production is energy self-sufficient (e.g., waste by-products 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 fossil fuel burning is usually needed to make paper products. Drying consumes most of that energy requirement (Farla et al., 1997). The efficiency and types of energy sources vary greatly from one mill to the next (Grieg-Gran, 1999), making it difficult to generalise about the energy uptake. Moreover, the energy efficiency of paper-making in the UK is expected to improve over time (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 requirement that related 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. We opt for a central estimate of C releases associated with fossil fuel consumption to make paper (45% of the C in end product).

3.6. *The social value of carbon*

The “social value” per tonne (1000 kg) of sequestered carbon (svtC) can be defined as the benefit in savings from damage avoidance. This benefit is calculated by observation of compensatory costs to reveal its cost to society, or “shadow price”. The true social value of sequestered C, as well as the appropriate discount rate and discount function to associate with it, have been hotly debated (Weitzman, 1998; Pearce, 2003). We are not able to reconsider these debates thoroughly, and instead choose to experiment with the most realistic options in actual policy making in the UK; (these choices have very large impacts in C-accounting exercises, and are discussed at much greater length in Brainard et al., 2003, 2006. We employ here a conventional discount function (Price, 1993), and experiment with four candidate discount rate/svtC combinations, all expressed in US dollars (starting with year=2001): 3%/\$1 (Mendelsohn, 2003; pers. comm., 2005), 3.5% (UK government's preferred discount rate for projects with social benefits; HM Treasury, 2003) and \$10 (a notional svtC), 5%/\$17.1 (Tol, 1999) and 1%/\$109.5 (Clarkson and Deyes, 2002). For a brief period the Clarkson and Deyes (2002) figure (\$109.5) was the UK Treasury's preferred svtC (Pearce, 2003). In our modelling, the svtC increments 1% each year for the first 30 years, based on expected increases in the magnitude of GHG-related damages. Values are held constant from 2031 in the assumption that the trend towards increased damages will be mitigated by adaptive change.

3.7. *Woodland created in future*

The woodland data records only referred to planting before 2002. It is important to consider later plantings, as well. FC (2001) stated that an additional 13.4 kha of new broadleaf and 4.7 kha of new conifer woodlands were created in 2001, which is broadly consistent with longer-term trends (Cannell, 1999). Our own map overlays (Brainard et al., 2003) suggest that among new Sitka spruce plantings in the period 1980–2001, 4% were on lowland non-peat, 11% on upland non-peat, 68% on thin peat and 17% on thick peat. Corresponding percentages for broadleaves in all planting years were 56% on lowlands, 33% on upland and 11% on thin peat. These findings are consistent with observations by others (Chapman et al., 2001). We therefore assume that these percentages and total planting areas and rates in each species in 2001 will hold true in future

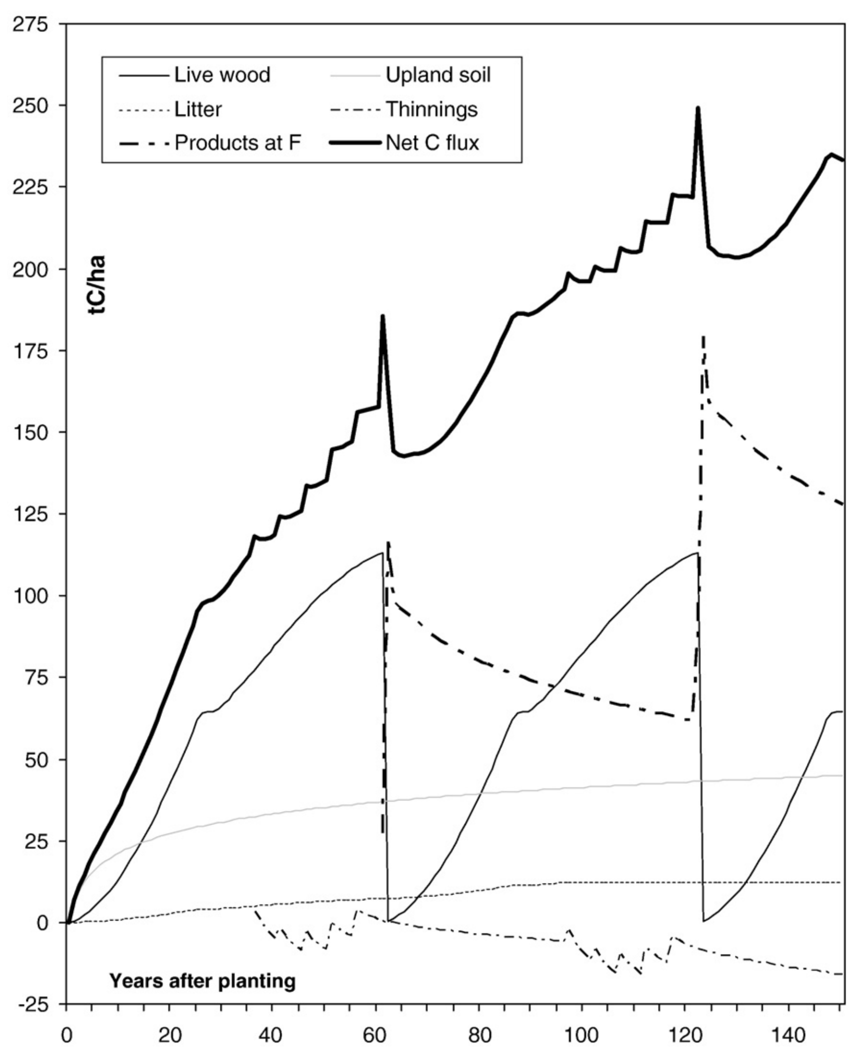

Fig. 1 – C flux associated with woodland.

plantings, and that all new plantings will be Sitka spruce, average YC=12, or beech, average YC=6.

3.8. *Other model assumptions*

Carbon gains in live trees are assumed to stop when a rotation reaches the age of 200 years (conifers) or 300 years (broadleaves). We only include C flux in soils, litter, live wood, and products, in the year 2001 (base year) and thereafter. Discounting applies from the year 2002 onwards, until the year 3001, when final tallies are taken. This long period should provide ample time to calculate net carbon sequestration over multiple rotations. The year 3001 is only used as a consistent endpoint, and is not significant in itself.

At felling, only 70% of the C in live wood is retained in products (merchantable wood). 30% of the C in living trees — found in bark, foliage, roots and harvest shavings not otherwise input to products — is wasted and the C in these waste by-products is presumed to be released back to the atmosphere almost immediately. About 2.4% of harvested wood biomass is used immediately as fuel (Bateman and Lovett, 2000). We assume that some other wood products also have a secondary usage (disposed of by burning to generate heat or energy) as fuel. Our model assumes that a total of 3% of products are used primarily or secondarily as fuel. We do not consider the energy requirements to recycle or reuse paper products.

4. Model verification

Fig. 1 shows model forecasts of cumulative C flux over the first 150 years of a Sitka spruce plantation, YC=12, discount rate=3.5%, intermediate thinning intensity, on an upland non-peat soil. Note that C flux associated with thinnings is almost entirely negative, due to the high fossil fuel consumption required to make the primary product of thinnings (paper). Upward spikes occur at felling because of displaced C in products followed by high wastage of biomass materials. Brainard et al. (2003, 2006) report detailed comparisons of our model outputs with other published estimates on the rates of uptake and per hectare totals of carbon sequestered in live trees and litter in British woodland. Generally our rates of C sequestration are very close to or slightly underestimate calculations by others. Table 1 gives some examples of comparative calculations from our work and that of others. For instance, Cannell and Dewar (1995) suggested that the long-term (after three rotations or so) equilibrium carbon storage in conifer trees and litter in GB would be about 75 t C ha^{-1}. Our model calculates 73.2 t C ha^{-1} (for YC=12, or 82 t C ha^{-1} for YC=14). Cannell et al. (1996) implied per annum rates of 2.7 t C ha^{-1} year^{-1} for live wood Sitka spruce 25–40 years old, YC=14/16. Our model suggests gains of 2.1–2.6 t C ha^{-1} year^{-1}. Cannell (1999) reported that a model of Sitka spruce, YC=16, felled at 55 years accumulated carbon in live wood, soil and litter at a rate of 3.6 t C ha^{-1} year^{-1}. Our model predicts an average per annum carbon sink of 3.46 t ha^{-1}. We have a similar degree of correspondence for rates of C accumulation into broadleaf species. Cannell and Milne (1995) implied that a 55 year old oak woodland would hold 63.6 t C ha^{-1} and a 55 year old beech woodland would hold 70.6 t C ha^{-1} in live vegetation; we have 59.7 t C ha^{-1} for 55 year old beech, YC=4, or 85.04 t C ha^{-1} for YC=6. Among ancient broadleaves in southern England, live overstory biomass measurements of up to 187.2 t C ha^{-1} in densely stocked semi-natural woodland have been observed (Patenaude et al., 2003). This is not far above our estimate of 169.4 t C ha^{-1} for 125 year old beech, YC=6. We also note good agreement between our calculations of the C in both unthinned and thinned live wood with comparable estimates generated by the CARBINE and C-FLOW models (Robertson et al., 2003).

Table 1 – Comparisons of C flux from our work and that of others

Example	Source	Source estimate	Our estimate
Long-term equilibrium C storage (t C/ha) in conifers and litter	Cannell and Dewar (1995)	75	73.2 (YC=12) 82.0 (YC=14)
Per annum rates (t C/ha) of C uptake, live Sitka spruce, 25–40 years old, YC=14–16	Cannell et al. (1996)	2.7	2.1 (YC=14) 2.6 (YC=16)
Per annum C uptake (t C/ha) in to live Sitka spruce (YC=16), soil and litter	Cannell (1999)	3.6	3.46
Total C (t C/ha) in live vegetation, 55 year old oak or beech	Cannell and Milne (1995)	63.6 (oak) 70.6 (beech)	59.7 (YC=4) 85.0 (YC=6)
t C/ha in overstory biomass (broadleaf, densely stocked semi-natural woodland)	Patenaude et al. (2003)	187.2	169.4

We further consider previous estimates of total C sequestered in live wood throughout GB. Cannell and Milne (1995) estimated that British trees (live wood alone) in 1990 contained 92.1 Mt C. Our models estimate 121.17 Mt C in the year 2001, if discount rates=3.5%. The difference can probably be accounted for by variations in discount rate[4], planting dates and species type, as well as growth and new plantings in the 1990–2001 period.

There is another source we can consult to confirm the validity of our estimates. The total area of woodland in specific age groups, according to FC (2002) (in kha, per species type) are shown in Table 2. In adjacent columns, we list our model predictions of average C storage in live trees (for each species type) per hectare. Our estimates presume that average YC values for broadleaves and conifers are respectively YC=4 and YC=12 (following suggestions by Milne et al., 1998), and that the discount rate=3%.

The far right column in Table 2 gives the expected total Mt C (from our models) for trees in the given age brackets, net of thinnings (Cannell and Milne, 1995) also omit C in thinnings). Thus, among trees planted before 1861, our model suggests 6.35 Mt C (=(6*169+46*115.8)/10^3). The total expected C

[4] Empirically, we derive the same estimated felling dates cited in the body of work by Cannell et al. (1996) by using a discount rate of 3.8 or 3.9%. The slightly lower rate of 3.5% in our comparisons increases felling age by less than 5 years.

Table 2 – Area of woodland, by planting year classes with predicted t C ha^{-1} (in live wood only) on expected average YCs

Planting year	Conifers '000 ha (FC 2002)	Predicted t C ha^{-1} in live wood	Broadleaves '000 ha (FC 2002)	Predicted t C ha^{-1} in live wood	Total Mt C
<1861	6	169.0	46	115.80	6.35
1861–1900	14	141.5	144	112.80	18.33
1901–1910	3	127.5	27	97.91	3.04
1911–1920	13	123.3	75	89.36	8.31
1921–1930	22	119.4	85	79.87	9.42
1931–1940	37	116.2	91	69.93	10.66
1941–1950	89	109.8	128	60.49	17.52
1951–1960	228	96.4	121	58.34	29.04
1961–1970	314	78.6	90	41.90	28.44
1971–1980	317	66.0	63	24.56	22.46
1981–1990	273	30.3	52	11.15	8.85
1991–	89	6.3	50	2.525	0.68
Totals	1405	1184	972	761	163

Notes: FC (2002) is the source for hectares planted in conifers or broadleaves in given year range. Model assumptions used to predict C per hectare in live wood: presumed YC=4, all beech, for broadleaves; YC=12, all Sitka spruce, for all conifers. Year taken as midpoint of given range or 1850 if before 1861. Discount rate=3%. Expected t C ha^{-1} when discount rate=1% or 5% differ from given figures by (on average) ±1.1%. See text for results if average YC expectations are reduced by one class.

in all live wood present in 2001 in GB is estimated at 163 Mt in 2001, assuming representative conifer and broadleaf YCs equal respectively to 12 and 4, as suggested by Milne et al. (1998). However, if the presumed average YC values are reduced to 10 (conifers) and 2 (broadleaves) the total estimated C in live wood in 2001 falls to 117.3 Mt, using the per hectare data in Table 2 (and our model predictions based on per hectare C storage for the specified YC). 117.3 Mt is extremely close to the figure (121.17 Mt C) our model calculates from individual records of actual woodland cover in Great Britain. This analysis suggests that YC values of 10 (conifers) and 2 (broadleaves) may be more representative of woodland in the UK than the YC values suggested by Milne et al. (1998). We are also left with the impression that our estimates of total C stored presently in British woodland are realistic.

5. Results

Table 3 presents the calculated NPVs for Britain under four discount rate/svtC combinations, with possible GBP equivalents, for both present and anticipated future woodlands, using a conventional (exponential) discount function.

Adjusting any of the total NPVs under each discount rate to reflect a different svtC only requires scaling by the relevant factor. For instance, to apply a svtC of $10 when using a 3% discount rate, one only needs to multiply the given figure ($154 Million) by 10 (=$1.54 billion).

If we include future new afforestation, the minimum total NPV (when the svtC=$1 and discount rate is 3%) is $154 million. Most estimates of the true svtC (Pearce, 2003) are in the range of $10–$17.1, which makes the NPV estimates using these svtC figures seem quite plausible, at $1.25–$1.3 billion for existing and future woodland. Changing the estimates of svtC to $109.50 and the discount rate to 1% yields an estimate of $76.3 billion. Although this last figure may seem unlikely because of the magnitude of difference between it and the other estimates, we cannot dismiss it as entirely implausible; there is still considerable uncertainty about the true value of the social cost of CO_2 releases. It is worth noting (Brainard et al., 2006) that the svtC exerts much more influence on NPV estimates than any of the uncertainties in the underlying biophysical models to calculate rates or totals of carbon sequestration.

6. Discussion and conclusion

This paper presents results from a model that calculates the social value of carbon sequestration in woodlands. To our knowledge, no such analysis has been previously undertaken with such a complete inventory of actual species types, planting dates and management regimes. The study is also

Table 3 – NPV estimate ($millions) for social value of carbon in existing and future woodland in Great Britain

	Discount rate/svtC			
	3%/$1	3.5%/$10	5%/$17.1	1%/$109.5
FC Beech	0.7	6.0	6.4	197.7
FC Oak	0.3	2.6	2.0	126.5
FC Sitka spruce	28.6	250.5	301.3	6643.2
Other FC broadleaf	0.5	4.8	5.7	154.5
Other FC conifer	6.7	57.1	57.1	1697.7
WI broadleaf	3.0	26.0	30.3	840.1
WI conifer	42.5	372.7	450.1	10,012
Sub-total	82	720	853	19,672
Future broadleaf	57	422	361	46,190
Future conifer	15	110	90	10,483
Total	154	1252	1304	76,344

Notes: 2001 social value of sequestered carbon (svtC) as noted, with annual increments of 1% through the year 2031. Unless otherwise known, broadleaf assumed to be beech, YC=6; conifers assumed to be Sitka spruce, YC=12. Proportions of planting on different soil types assumed to be same as that observed for beech and Sitka spruce (post 1980) in FC (2001), as summarised in text.

especially thorough in including many aspects of carbon accounting that have often been overlooked in previous studies, including C in thinning products, C displacement in wood products, harvesting and manufacture releases and the lost C sink on disturbed peat soils. The work helps place in context the annual publicly-funded subsidy that the Forestry Commission receives. It should inform any other attempts to undertaken cost–benefit analysis of woodland in the UK. The minimum social value of C sequestration associated with existing woodland is $82 million, while a valuation of $720–$853 million is plausible. The true value of C storage associated with existing woodland may even be much higher. Future new afforestation would also increase these figures considerably.

As with any model, there are things we omit and things we might improve. A more complete assessment of the global warming impacts of forestry would need to consider forestry or soil greenhouse-gas (GHG) fluxes other than carbon dioxide. For instance, peat soils may act as a sink or source of other important GHGs: methane (CH_4) and nitrous oxide (N_2O). The global warming potential (GWP) of methane is roughly 21 times as high as carbon dioxide, nitrous oxide has about 300 times the GWP of CO_2. Undisturbed peat is generally a net source of methane emissions, while disturbed peat (including afforested peat) is likely to become a methane sink. Whilst many soils have been observed to emit N_2O, drained peatlands are noted as particularly important sources (Leffelaar et al., 2000; Chapman et al., 2001). The picture is complicated by nitrous oxide and related NOx emissions from routine forestry operations; i.e., from thinning, harvesting and milling machinery. Moreover, climate change (including warmer temperatures and higher atmospheric concentrations of CO_2) may increase timber yields and the rate of carbon uptake in living trees, as well as nitrogen deposition in GB woodlands (Centritto et al., 1999; Murray et al., 2000; Murray et al., 2002).

CH_4 is also relevant with regard to disposal options. We assume that all C in disposed of 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. In reality, most solid wood and paper 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). Methane can be captured and used as fuel; burning CH_4 for fuel 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 UK landfills have capture facilities, and there is great uncertainty about what percentage of CH_4 releases can potentially be collected (UKHL, 1998). Barlaz (2004) summarises previous studies which calculated that 50–84% of the carbon in paper and other wood products may remain long-term in landfills. Miner (2003) concluded that, even adjusting for methane releases, approximately 40% of the C (or equivalents, including un-captured CH_4) in paper/wood products is stored at least 100 years. Unfortunately, as Barlaz (2004) 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. Moreover, there are high social costs of waste disposal in landfills (Powell and Brisson, 1994) that might need to be accounted for.

Expected rotation lengths and thinning dates in our analysis depended entirely on current conventional and commercial considerations. A strong emphasis on C-storage potential would probably lead to rather different management regimes (see Creedy and Wurzbacher, 2001; Stainback and Alavalapati, 2002; Cacho et al., 2003), and much more C storage associated in woodlands. We have considered the impacts on our valuations of altering rotation lengths, expected C in thinnings and C release periods for products elsewhere in our sensitivity analysis (Brainard et al., 2006); the effects can be dramatic, and are not always obvious. For instance, earlier or later thinning and rotation lengths could lead to fewer or more long- and medium-life products, with associated changes in releases of CO_2 associated with manufacture, C displacement by product use and product lifetimes.

However, we believe that an actual and significant shift towards criteria other than commercial concerns to be unrealistic at the moment; there is no political appetite for them to be imposed. Voluntary changes in woodland management could be achieved, to result in later thinning dates and longer rotations, by introduction of a centrally managed carbon trading scheme. Again however, this seems unrealistic given the recognized difficulties in administering such trading schemes, as well as the very limited scope for current or future British woodlands to substantially reduce national GHG emissions (Cannell, 1999).

We ignore C in ground flora, which is significant on some sites (Patenaude et al., 2003). It can also be argued that increases in the carbon stocks in GB woodlands serve only to displace carbon in woodland stocks overseas, and thus growing more trees in the UK does nothing on a global scale to reduce global warming. We do not attempt to address how changes in this trade balance might impact our results. The mapped database generally omitted woodland on sites of less than about 2 ha. This means small groups of trees (including most in urban areas or lining roads) were unaccounted for. What if climate change leads to different predominant timber species being grown in the UK? Or if wood grown as biofuel becomes more popular and profitable as other energy prices generally rise? It is unclear and difficult to assess what the net effect might be of these omitted factors: disposal options, C in ground flora, imports displacement, overlooked trees, climate change feedback, and price energy-related changes.

Sensitivity analysis in Brainard et al. (2003, 2006) suggests that wide uncertainty intervals apply to the valuations here. Much of this uncertainty comes from imprecise information about species type, underlying soil, planting year, rotation, management regime and other variables on non-FC woodlands. However, the ongoing disputes in the environmental economics community over the correct social value of carbon (per tonne), appropriate discount rate and even the right form for the discounting function (Henderson and Bateman, 1995; Weitzman, 1998) have the biggest impacts on attempts to value the C-sequestration function of woodlands. This is amply demonstrated by the wide range of NPV estimates in Table 3.

This paper focuses only on C-storage function of forest land use, and how it may be thoroughly accounted for. British woodlands offer many other social benefits (biodiversity, recreation, water quality protection, etc.) that may be validly compared to public subsidies to assess the value for money that taxpayers receive. We don't attempt such an ambitious calculation here. Interested readers are referred to ERM and Willis (2004) for a complete analysis of all the social benefits from Great Britain's woodland.

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