Carbon Valuation in Forestry

and Prospects for European Harmonisation

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Acknowledgements

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Summary

- Valuing the carbon benefits of forests is essential in guiding public policy and private decision-making in tackling climate change. Carbon valuation is important in comparing the relative merits of forestry projects with other climate mitigation activities, as well as in natural capital accounting.

- Separating out carbon savings relative to the baseline that are due to human causes (‘GHG additionality’) from any due to other causes is a pre-requisite for valuation in most cases. How the baseline is determined is a critical element.

- Approaches to valuing carbon and often associated values, vary depending to whether a societal or market perspective is taken. At present, there is little relationship between the value to society of reducing greenhouse gas emissions or sequestering carbon and the market price, due partly to low emission reduction targets having been set by governments in establishing cap-and-trade schemes.

- The social value of carbon can be based on the marginal damage cost of emissions – also termed the ‘social cost of carbon’, the marginal abatement cost of reducing emissions or sequestering carbon, the shadow price of carbon equating marginal damage cost and abatement cost, or the carbon price required to meet a given climate stabilisation goal.

- Estimates of the social cost of carbon are subject to wide variation between countries, spanning at least three orders of magnitude from zero to over £3600/tCO_2, reflecting different methods, assumptions and models, as well as uncertainty concerning climate change impacts.

- There is also a wide range of market prices for forest carbon. Prices in voluntary carbon markets worldwide in 2011 ranged from under \$1/tCO_2e to over \$100/tCO_2e, highlighting the importance of differences of quality and type.

- Any attempt to harmonise approaches to carbon valuation in cost-benefit analysis could benefit from being underpinned by a common approach to carbon accounting – although this may be particularly challenging for the private sector given the current range of approaches used.

- A common approach to carbon valuation in private sector cost-benefit analysis would require better data on market prices for European forestry carbon, as well as agreement on prices to use for sensitivity analysis.

- There is currently no internationally agreed methodology for estimating the social value of carbon, with existing values partly reflecting national convention. A common approach to estimating social values of carbon would require agreement on the underlying method and model to use. The benchmark value of carbon, and the discount rate or rates applied are critical influences on whether climate change mitigation activities in the forestry sector are considered cost-effective. Low carbon values and high discount rates can make

mitigation unviable, and both a relatively high discount rate and a relatively modest social value of carbon risk undervaluing potential contributions of forestry to climate mitigation.

- Developing a common approach to cost-effectiveness analysis in different EU countries may appear more straight-forward than harmonising approaches to valuing carbon. However, the inclusion of values for ancillary benefits could pose similar challenges in developing a common approach to cost-effectiveness analysis.

- Establishing a framework that adequately values forestry carbon and that provides incentives for businesses and households to incorporate climate change impacts of their activities into their decisions will be important if significant opportunities for climate change mitigation by the forest sector are not to be missed.

1. Why is carbon valuation needed?

Valuing ecosystem services – such as the carbon benefits of forests – is widely viewed as essential both in guiding public policy and private decision-making. Valuing forest carbon is important in appraising the costs and benefits of projects, in comparing forestry with other types of investments, as well as in natural capital accounting.

Climate change has been characterised as "the greatest and widest-ranging market failure ever seen" (The Stern Review (2006, p.i)). The increasing concentration of 'greenhouse gases' (GHGs) in the atmosphere – of which carbon dioxide (CO_2) is the most important, is the primary cause of anthropogenic climate change.

Establishing a framework that values forestry carbon is integral to providing financial incentives for businesses and households to take account of the climate change impacts of their activities. This will be important if significant opportunities for climate change mitigation by the forest sector are not to be missed.

This paper summarises key issues relating to valuing forest carbon and to estimating climate change cost-effectiveness of woodland creation and management options. Although often considered an alternative to cost-benefit analysis, cost-effectiveness analysis can instead be considered a useful complementary approach that is much used in practice in comparing climate change mitigation policies (e.g. through construction of Marginal Abatement Cost Curves) at portfolio level (Price *et al*, 2007).

The paper, including the background section in Annex I on climate change below, the section on carbon accounting issues in Annex II below, and the discussion on discounting in Annex III below, draws heavily upon material presented in Valatin and Price (2013), Valatin (2013) and Valatin (2011a,b). It aims to provide a level of technical detail to allow it to be used as a summary reference document.

After discussing perspectives on carbon valuation, and cost-effectiveness analysis, the final section considers prospects for harmonising approaches to facilitate comparisons across Europe. Annex IV below provides an example illustrating how carbon is valued in practice in cost-benefit analysis.

2. Perspectives on carbon valuation

Private sector and societal perspectives on valuing carbon often share similar concerns about the importance of overcoming market failures associated with insufficient value being placed on carbon (and other ecosystem services), as well as the significance of quality standards underpinning carbon markets. Separating out carbon savings relative to the baseline that are due to human causes (‘GHG additionality’) from any due to other causes is a pre-requisite for valuation in most cases and how the baseline is determined is a key element.

Private sector

Development of forest carbon projects led to the establishment of markets in carbon sequestration and other carbon benefits of forestry. The total volume of forest carbon transacted is estimated to have risen more than twenty-five fold over a 7-year period, from 4 $MtCO_2$ prior to 2005 to 106 $MtCO_2$ in 2011 (Peters-Stanley, Hamilton and Yin 2012, Fig 11, p.7), with forest carbon projects covering an estimated 18 million ha worldwide in 2012.

There is no single market value for forest carbon, with a wide range of prices in practice. Prices in voluntary carbon markets worldwide in 2011 ranged from under \$0.1/tCO_2e to over \$100/tCO_2e (Peters-Stanley and Yin 2013), with those for forest carbon similarly ranging from under \$1/tCO_2e to over \$100/tCO_2e (Peters-Stanley, Hamilton and Yin 2012).

This wide range of prices in part reflects differences in project types, co-benefits (or disbenefits), and wider underlying institutional factors - including regulatory frameworks within which markets operate affecting supply and demand. For example, the market price of carbon benefits lacking institutional additionality would be expected to be lower (due to lack of suitability as offsets reducing demand) than for those with this form of additionality. A discussion of disbenefits associated with some carbon projects approved under the Kyoto Protocol Clean Development Mechanism (CDM) can be found in Wilson (2011).

The price range also reflects other important differences – including whether carbon benefits are sold before they arise (ex-ante) or afterwards (ex-post), and the extent of any third party certification. Other quality differences that can affect prices are different approaches taken to baseline setting, additionality, leakage and non-permanence (see Annex II below). For example, approaches to non-permanence risks involving expiry of carbon credits and a requirement for substitute abatement benefits then to be provided (e.g. along similar lines to temporary credits issued under the CDM), will tend to reduce carbon prices – especially if prices are increasing, and may even reduce carbon prices to zero (see also Chomitz (2000)).

Peters-Stanley and Yin (2013, p.xiv) note “…voluntary buyers are a source of demand for differentiated products that are purchased on the basis of dozens of decision points. These criteria include offset supplier reputation, perceived offset quality, and, more broadly, the health of the buyer’s business, the economy, and their previous experience with offset programs.”

The range of carbon prices also partly relates to the formative stage of carbon markets. Hamilton *et al* (2008, p.5), for example, notes that many commentators have likened the early development of voluntary carbon markets to the "wild west", characterised by lack of established rules and limited understanding of potential pitfalls – although this has been changing as participants become more savvy and standards established.

There can be very significant fluctuations in carbon prices between and within years. For example, prices for Certified Emissions Reductions from carbon offset projects under the Kyoto Protocol Clean Development Mechanism reportedly fell from a peak averaging around \$20/tCO_2e in 2008 (Hamilton *et al*, 2008, 2009) to an all-time low of \$0.16/tCO_2e during 2012/13 (Peters-Stanley and Yin, 2013). Average prices for forestry carbon are reported to have fallen by 25% to \$7.8/tCO_2e in 2012 (from \$10.5/tCO_2e in 2011), due to buyers seeking larger volumes for future delivery from more recently established projects (Peters-Stanley and Yin, 2013).

Changing expectations about regulatory factors – such as changes in the 'cap' and allocation mechanisms have also led to carbon price fluctuations in markets like the EU Emissions Trading Scheme (EU ETS). These may also influence carbon prices more widely. In contrast to 'compliance' markets operating in other parts of the world - including in the U.S. and New Zealand, and those relating to developing country projects under the Kyoto Protocol, the EU ETS currently excludes forest carbon - see Ciccarese *et al* (2011) for a discussion.

Costs of participation in carbon markets can be a significant barrier, especially for small-scale projects. Costs can also be a barrier to adopting specific forms of carbon measurement and monitoring. For soil carbon, for example, Smith (2004) suggests that in some cases the costs of demonstrating carbon sequestration can exceed the value of the carbon sequestered.

Although carbon prices in carbon markets may be expected to increase once more stringent climate change policy targets are introduced, the extent to which this occurs will also depend upon a range of other factors.

Societal

A variety of approaches to valuing carbon from a societal perspective exist. Four principle ones are:

1) The marginal damage cost of carbon emissions – also termed the '**Social Cost of Carbon**' (SCC), which aims to quantify the cost to society of emitting an extra unit of carbon dioxide. (For example, the Stern Review (2006, p.287) defines this as "the impact of emitting an extra unit of carbon at any particular time on the present value (at that time) of expected wellbeing or utility.")

2) The **Marginal Abatement Cost** (MAC) of reducing emissions or sequestering carbon that is the cost associated with reducing emissions by one unit or increasing carbon sequestration by one unit.

3) The **Carbon Price** (CP) or pollution tax required to meet a given climate stabilisation goal – or equivalently, the level of MAC consistent with achieving this level of abatement.

4) The **Shadow Price of Carbon** (SPC) – sometimes derived using a cost-benefit analysis approach and defined as the pollution tax that equates the SCC and the MAC (Clarkson and Deyes, 2002), and sometimes considered equivalent to the SCC in the absence of other distortions (Stern Review, 2006).

Under an effective regulatory framework limiting emissions to the socially desired level, market prices might also provide a useful guide to the social value of carbon.

Social values of carbon are often estimated to increase over time. Under the SCC approach, this reflects increasing damage associated with emitting an additional tonne of carbon due to the assumed increase in the atmospheric concentration of GHGs over time – with the level of damage associated with emitting an additional tonne being greater the higher the existing stock of carbon in the atmosphere (Clarkson and Deyes, 2002). Under the MAC approach, it reflects the assumption that more expensive abatement options are progressively needed as the cheapest are taken-up first – although in some models costs eventually fall as cumulative innovation makes low carbon technologies less expensive (Bowen, 2011).

Historically, some countries have used more than one approach. In the UK, for example, government guidance issued in 2002 (Clarkson and Deyes, 2002) recommended adoption of SCC estimates based upon Eyre *et al* (1999). Guidance issued in 2007 (Price *et al* 2007) then recommended adoption of social values ('SPC') based upon SCC estimates from the PAGE2002 model used in the Stern Review (2006) consistent with a scenario of stabilisation of atmospheric GHGs at 550ppm CO_2e (a higher SCC than consistent with meeting a lower stabilisation target such as 450ppm CO_2e).

However, considerable uncertainty exists around climate change damage estimates, with SCC estimates spanning at least three orders of magnitude from zero to over £3600/tCO_2 (£1000/tC) (Downing *et al*., 2005). This partly reflects uncertainties about the extent of climate change and its impacts, different time horizons focused upon and different underlying assumptions about discounting (Tol, 2009) and equity weighting – which attaches a greater weight to damages borne by low income countries as the damages will assume a larger proportion of their income (Watkiss *et al*., 2006). It also reflects different assumptions about the functional form of the climate damage function (Stanton *et al*, 2008), and the trajectory and ultimate concentration of greenhouse gases. In addition, SCC estimates are incomplete to the extent that underlying models exclude impacts on capital stocks – including property and ecosystems (Stanton *et al*, 2008), potentially significant but hard to model impacts – such as impacts associated with tipping points, and second-round impacts – such as costs of mass migrations from inundated countries (DECC, 2009b).

Due to the high level of uncertainty around SCC estimates, and to the possibility of tipping points and potentially catastrophic outcomes for global temperature rises above 2°C, in 2009 the UK switched to adopting a CP based upon estimates of the abatement costs that would need to be incurred to meet its emissions reduction targets. The UK's targets include emissions reductions of 34% compared to 1990 levels by 2020 and of 80% by 2050 (DECC, 2009a).[1] These were set to be consistent with the UK's contribution to ensuring global temperature increase is limited to around 2°C and global atmospheric GHG concentrations constrained to be within the 460-480ppm CO_2e range in 2200, with the existence of separate targets for EU ETS and non-EU ETS sectors, and different marginal costs of meeting these, leading to different social values of carbon being adopted for the two sectors (DECC, 2009a,b). The values adopted draw on modelling by the Committee on Climate Change (CCC, 2008).

Current UK guidance includes central estimates for 2014 of £60/tCO_2e non-EU ETS sectors and £4/tCO_2e for sectors covered by the EU ETS, both rising over time to a peak of £337/tCO_2e in 2077 at 2013 prices, thereafter declining. The two sets of values reflect initially separate targets, being assumed to converge from 2030 as a more comprehensive global carbon market develops. For sensitivity analysis, the estimates are ranged (those for non EU ETS sectors by ±50%).

As illustrative ones cited by Bowen (2011) highlight, CP estimates consistent with meeting the challenging international target of limiting temperature increase due to anthropogenic causes to a maximum of 2°C are also fairly wide ranging. (CP estimates for 2020 cited by Bowen (2011, p.9) range by a factor of almost twenty, from under £10/tCO_2e to over £175/tCO_2e).

Comparing carbon values over time

Discounting is the conventional approach to comparing benefits (and costs) in current prices over time. Rationales include opportunity costs of public finance, a preference for early benefits rather than late ones, future growth of income per capita reducing future opportunity costs of resources to address environmental problems, and the possibility of devastating future events resulting in human extinction (see discussion in Annex III). In addition, concerns about the potential for exceeding critical tipping points combined with uncertainty about precise thresholds can be viewed as providing a reason for prioritising early abatement, either by valuing it more highly than later abatement (because of the longer period of ensuing benefit), or by discounting the later abatement (Valatin, 2011b). From this perspective, early abatement is viewed as having an option value associated with expanding the range of potential policy options to deal with climate change mitigation and adaptation (Rhys, 2011, p.8). For the same reason (i.e. in 'buying time' to develop wider policy options), future emissions may be viewed as preferable to current ones other factors being equal (i.e. for a specific level of atmospheric GHG concentration).

[1] In its latest assessment, the UK committee on Climate Change reports good progress in implementing some measures, but with significant challenges remaining in meeting the targets – as illustrated by the 3.5% increase in UK greenhouse gas emissions in 2012 (attributed to a relatively cold winter) compared to the future annual reductions needed of 3% (CCC, 2013).

Not discounting carbon values would imply that a tonne of carbon sequestered at the end of a rotation is given equal weight to one sequestered immediately. However, they will tend not to be of equal value if the options value argument above (or any of the rationales for discounting discussed in Annex III below) is accepted.

Cost-effectiveness analysis

Cost-effectiveness analysis is an approach to decision-making that does not rely directly upon valuing carbon. The cost-effectiveness of a measure for climate change mitigation is in essence very simple. The extra cost of the measure compared with the cost of *not* deploying it, is divided by the extra reduction in the atmospheric carbon level achieved by deploying the measure compared with the level of carbon reduction ensuing if the measure is *not* deployed. In general terms, this is expressed as (Valatin and Price, 2013):

$$\frac{[\text{Net cost of the measure}] - [\text{Net cost of 'do - nothing'}]}{[\text{Carbon reduction of the measure}] - [\text{Carbon reduction of 'do - nothing'}]}$$

Although cost-effectiveness analysis per se does not involve carbon being valued, and estimates may simply be compared on a unit cost basis (e.g. using a Marginal Abatement Cost Curve (MACC), where a benchmark is used to determine whether estimates are cost-effective, this may be based upon the value of carbon. In UK policy appraisal, for instance, benchmarks used to compare cost-effectiveness estimates are based upon the social value of carbon, being computed as a weighted mean of the social value of carbon in each period when abatement occurs, discounted using the Treasury Green Book (HM Treasury, 2003) protocol:

$$\sum_{t=0}^{t=T}[\{\text{abatement}_t \div \text{lifetime abatement}\} \times \{\text{social value of carbon}_t \times \text{discount factor}_t\}]$$

Key factors in climate change cost-effectiveness analysis include assumptions about opportunity costs – which often represent the main cost of forestry projects, and whether ancillary benefits are included or not (Valatin and Price, 2013). The potential significance of ancillary benefits is highlighted by Richards and Stokes (2004), for example, who note that secondary benefits of converting agricultural land may be as great as the costs, in effect reducing unit costs to zero, and making forestry options extremely attractive as climate change mitigation measures. Case studies (e.g. Nisbet *et al* (2011), Valatin and Saraev (2012)) suggest that this can be the case in the UK for some forestry projects. (There are a range of potentially significant ancillary benefits of woodland projects, including flood risk reduction, sediment and agricultural pollutant absorption, amenity, biodiversity, mental and physical health and wellbeing benefits).

The underlying approach to carbon accounting adopted crucially affects the interpretation of the cost-effectiveness estimates. Of the two principal approaches to carbon accounting (Annex II below), estimates derived using the 'flux' approach can be conceptualised as constituting a price for sequestering carbon. Alternatively, this can be computed by calculating the price per tonne of carbon sequestration which, allowing for any subsequent releases to the atmosphere, would just suffice for the option to break even. Abatement may either be summed - as is the case in the UK approach to policy appraisal (HM Treasury and DECC, 2012), or discounted then summed. By contrast, under the 'stock' approach estimates are essentially for maintaining carbon sequestered in the forest, which similarly can be computed by calculating the break-even cost of maintaining this level of carbon stock (Valatin and Price, 2013).

3. Prospects for European harmonisation?

In conclusion, lack of a common approach to carbon valuation at present implies that forestry options which appear beneficial when judged by approaches in some EU countries, may not be when judged by those adopted in others. This creates a source of potential inconsistencies and could result in misallocation of resources where European funding is being disbursed.

Some governments have seen advantages in establishing common levels of carbon values at European or global level. In relation to carbon prices, Ishwaran and Cimato (2009, p.31), for example, argue that "Interventions at the trans-national level are more likely to achieve a given level of shared commitment to cost effective abatement than separate domestic interventions. Establishing a carbon price at the international level also has the advantage of minimising negative competitiveness effects and the effect of these interventions on economic growth." Furthermore, it is argued that a uniform carbon price could ensure the burden of adjustment costs is spread efficiently across economies (Bowen, 2011).

Private sector

Competitive pressures could be expected to reduce carbon price differentials between European countries for similar types of forest projects. However, the overall range of prices for European forestry carbon is likely to remain wide due to quality and other differences discussed above – including different approaches to carbon accounting, baseline setting, additionality, leakage and non-permanence.

Published carbon price data for European forestry projects appears sparse. However, if reliable price data were available, a common approach to valuing carbon in cost-benefit analysis might simply be based upon existing market prices for European forestry carbon – both average current (spot) prices and average prices for forward delivery at different future dates, ranged for sensitivity analysis. (Note that the market prices paid currently for carbon benefits in future years would be expected to reflect discounted future prices). Price differentials could also be taken into account where information on average prices for the main categories is available. For sensitivity analysis, prices could also be ranged based upon the spread of existing prices and on historic variation in price levels – although agreement would then be needed on the precise approach to use.

Societal approach

Any attempt to harmonise approaches to carbon valuation from a societal perspective or the social values of carbon used in European studies and to facilitate comparisons between national studies would benefit from harmonisation of underlying carbon accounting protocols – including coverage of different carbon pools and baseline setting. Agreement on common GHG accounting protocols for carbon sequestration was achieved in ensuring consistency in meeting national commitments under international agreements such as the Kyoto Protocol. However, to date there has been no similar requirement to develop a common approach at project level – including accounting for wider impacts such as the carbon substitution benefits associated with forestry projects (despite current EU interest and ongoing research).

Secondly, developing a common approach to carbon valuation for European analyses would require agreement on the underlying basis – whether valuing damage costs through a SCC, focusing upon abatement costs through adopting a MAC, a SPC equating the two, or a pragmatic CP consistent with emissions targets. Member States may still retain their existing approach as the main focus for national projects. However, in providing consistent estimates for European studies agreement on which integrated assessment model (IAM) estimates to use (or developing a more transparent model based upon an agreed approach and assumptions to provide new estimates), would be needed. Discussing and comparing 4 IAMs, for example, Stanton *et al* (2008, p.18) argue "Model results are driven by conjectures and assumptions that do not rest on empirical data and often cannot be tested against data until after the fact. To the extent that climate policy relies on the recommendations of IAMs, it is built on what looks like a "black box" to all but a handful of researchers. Better-informed climate policy decisions might be possible if the effects of controversial economic assumptions and judgements were visible, and were subjected to sensitivity analyses."

Feasibility of using common social values of carbon in different European countries would depend on the underlying approach to valuation selected. Of the four approaches to establishing a social value of carbon, a SCC approach may lend itself most readily to harmonisation as differences in abatement costs and national emissions targets imply other approaches would tend to lead to values differing between countries. However, as the switch away from this approach in the UK highlights, SCC estimates are subject to especial uncertainty, so that even if agreed in principle, precise levels of SCC to adopt could prove controversial.

At first sight, harmonising approaches to cost-effectiveness analysis in European studies may appear more straight-forward than harmonising approaches to carbon valuation, as the former do not rely directly on valuing carbon. However, inclusion of values for ancillary benefits in computing cost-effectiveness estimates could pose similar issues concerning developing common approaches to their valuation.

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Annex I: Background on climate change

Evidence from ice core data indicates that the current concentration of atmospheric carbon dioxide (CO_2) is unprecedented in the past 800,000 years (Lüthi *et al*., 2008), with data from boron-isotope ratios in ancient planktonic shells suggesting that it is likely to be at its highest level for about 23 million years (Pearson and Palmer, 2000; IPCC, 2001, Fig 3.2e, p.201). Anthropogenic carbon emissions rose by 70% between 1970 and 2004, from 29 to 49 thousand million tonnes of carbon dioxide equivalent ($GtCO_2e$)[2] per year (IPCC, 2007), with global emissions rising by 3% a year since 2000 (Peters *et al*., 2013). The current atmospheric concentration of CO_2 of over 390 parts per million (ppm) (Arvizu *et al*., 2011), which is around two-fifths higher than the pre-industrial level of about 280 ppm, is currently rising at an annual rate around 2 ppm (IPCC, 2007; GCP, 2012; CO_2now, 2013).

As atmospheric CO_2 concentrations have increased over the past 150 years, the mean global temperature has risen. In the absence of new policy action, annual world greenhouse gas (GHG) emissions could rise by a further 70% by 2050, and lead to a rise of 4°C, or possibly 6°C, above the pre-industrial global mean temperatures by the end of the century (OECD, 2009), with greater temperature rises likely in some regions, including the Arctic (IPCC, 2007, Fig 3.2, p.46). Likely adverse impacts associated with exceeding a 1.5-2.5°C temperature increase include increased risk of extinction of around 20-30% of plant and animal species, with many millions more people expected to be at risk of floods due to sea level rise by the 2080s (IPCC, 2007). Warming could lead to positive feedbacks that magnify temperature changes. These could include potential dieback of Amazon rainforest if warming exceeds 3°C (see Lenton *et al*. (2008) and discussion in Dresner *et al*. (2007)). Thawing of the permafrost and subsequent soil decomposition could lead to the further release of up to 380 $GtCO_2e$ under a high warming (7.5°C increase) scenario by the end of the century (Schuur *et al*., 2011). Recent evidence shows that warming of the Arctic is occurring faster than had been predicted, with sea level rising more rapidly than expected (Le Page, 2012).

In order to prevent 'dangerous climate change', international agreements reached at Cancun (UNFCCC, 2011, paragraph 4) and under the Copenhagen Accord (UNFCCC, 2010, paragraphs 2 and 12) call for limiting the average global temperature rise to no more than 2°C above pre-industrial levels, with consideration of adopting a limit of 1.5°C. To be confident of limiting the mean global temperature rise to between 2°C to 2.4°C is thought to require stabilisation of atmospheric GHG concentrations in the 445 ppm to 490 ppm range, with reductions in annual global carbon emissions occurring no later than 2015, and emissions 50-85% below 2000 levels by 2050 (Arvizu *et al*., 2011). However, some scientists have argued that even the existing GHG atmospheric concentration, which, including the effect of other GHGs, is equivalent to around 430 ppm CO_2e (Trumper *et al*., 2009), is too high for the temperature rise to stay below the 2°C threshold. Ramanathan and Feng (2008), for example, argue that the increase in atmospheric GHGs since pre-industrial times to date probably commits the world to a warming of 2.4°C (1.4°C to 4.3°C) above the pre-industrial level during the current century – although some underpinning assumptions have been argued to be

[2] A gigatonne (Gt) is the same as a petagram (Pg) – both are 10^{15} grammes.

over-pessimistic (e.g. Schellnhuber, 2008). Hansen *et al*. (2008) also recommend a rapid reduction from the current concentration by around 10% to no higher than 350 ppm of CO_2.

Globally, forests currently cover about 4000 million ha and, excluding woodlands under 0.5 ha, or primarily within agricultural or urban land uses, are estimated to store around 650 GtC, including around 290 GtC both in forest biomass and in soils, and 70 GtC in deadwood and litter. Covering 26 million ha, Forests in Europe (excluding those in the Russian Federation) account for around 5% of the global total area as well as 5% (35 GtC) of the total forest carbon stored (FAO, 2010, Tables 2.1 and 2.21). While comparisons are sensitive to definitional issues such as the depth of soil carbon covered (Jandl *et al*, 2013), by contrast, state that forests store 363 GtC in living biomass and 426 GtC in soils), globally the amount currently stored by forests is of a similar order of magnitude to the total amount of carbon now in the earth's atmosphere: this is currently around 800 GtC (Lorenz and Lal, 2010; Riebeek, 2011).

Temperate forests are currently a carbon sink because both their total area and their productivity is increasing (Jandl *et al*, 2013). However, while estimates of the carbon balance of European forests vary widely (Nabuurs *et al*, 2010), there are reported now to be early signs that unless current forest management practices are altered a maximum will soon be reached due to factors including maturation and greater vulnerability to fires, storms and insects (Nabuurs *et al*, 2013).

Annex II: Carbon accounting

Two main approaches to accounting for carbon are briefly outlined below, followed by discussion of wider issues influencing how carbon benefits are quantified.

Stock approach

Under this approach to carbon accounting (which Richards and Stokes (2004) terms this the 'average storage method'), the mean level of added sequestration over a commercial rotation is defined. A drawback of this method is that the mean may be undefined in cases of indefinite rotations, or if no felling is envisaged.

Flux approach

An alternative approach to carbon accounting is for carbon fluxes to be summed over time. This may be done either by:

a) treating sequestration as equivalent whenever it occurs (Richards and Stokes, 2004) terms this the 'flow summation method') – which, if only the carbon sequestered in the above ground biomass is considered, may imply no net gain in carbon over a rotation; or
b) by applying a discount rate to future emissions and sequestration. (Richards and Stokes, 2004) terms this the 'discounting method' – where the resulting summary statistic may be termed 'present tons equivalent' or 'PTE'). (Griffiths and Jarvis, 2005) note that as it may take 25-40 years for a newly planted forest to become a carbon sink and, depending upon the discount rate applied, some afforestation schemes may take far longer to yield a positive PTE).

As with the stock approach, the time horizon chosen is crucial. Comparisons over incomplete rotations may be misleading, while comparisons over several (or an infinite series of) rotations may be needed to provide a consistent basis for comparing forestry options associated with different rotation lengths.

Hybrid flux/stock approaches are also used. For example, under the Woodland Carbon Code (Forestry Commission, 2011a) developed to underpin the emerging market for UK forest carbon, carbon is accounted for using a flux approach up to the long-run average stock level for the particular forestry option considered, thereafter being capped at this level.

Carbon storage in harvested wood products

Some accounting protocols just focus on carbon within the forest itself, with any harvested wood representing an equivalent reduction in the carbon stock. However, other protocols also account for impacts on carbon stocks in harvested wood products held in the wider economy which, in some cases, may remain significant carbon stores for centuries. For example, millions of tonnes of carbon may be locked up in the roofs and fitments of Europe's medieval churches (c.f. see Valatin and Price, 2013).

Carbon substitution

Forestry projects can lead to generation of carbon benefits in the wider economy by providing a renewable source of energy and materials to substitute for use of fossil fuels and more fossil-carbon-intensive materials. The extent of such savings depends on the emissions that would otherwise have been generated by the specific type of fossil fuel or fossil carbon intensive material substituted for and the specific technology used. Bruges (2009), for example, notes that generating heat from burning wood using pyrolysis (heating the wood without oxygen and burning the expelled gases) and then retaining and crushing the carbon to create biochar to increase soil fertility can significantly increase the carbon benefits associated with substituting woodfuel for burning fossil fuels.

Carbon units

Carbon sequestration can be expressed either in tonnes of carbon, or of carbon dioxide, with straight-forward conversion between the two on the basis that as carbon constitutes 12/44 of the molecular mass of CO_2, one tonne of CO_2 equates to 12/44 tonnes of carbon. Similarly, the cost of a measure quantified per tonne of CO_2 is converted to a cost per tonne of carbon by multiplying by 44/12.

Other fluxes

Forestry operations such as planting, thinning and harvesting are emissions sources – although often relatively small. In the UK, for instance, forestry operations are estimated to result in total annual emissions around 1-2% of the net carbon uptake by forests (Morison et al, 2012).

Other GHG fluxes, where significant, can be converted to a 'per tonne of carbon dioxide equivalent' (tCO_2e) based upon their global warming potential compared to the 'radiative forcing' (resultant increase in the equilibrium temperature) of emitting a tonne of carbon dioxide. Global warming potential is defined as an index, usually computed as the cumulative radiative forcing over an arbitrary 100 years, compared to emitting a unit of carbon dioxide. Over this time-frame, other GHGs have higher (up to 23,900 times higher – in the case of sulphur hexafluoride) global warming potentials than carbon dioxide, molecule for molecule (Brown et al., 2012, p.39).

Other climate impacts

Other climate impacts, including those on solar radiation reflectivity (the 'albedo effect'), and associated with release of water vapour from forests ('evapotranspiration') affecting cloud cover and associated reflection of solar radiation, as well as surface temperatures, can, in principle, similarly be measured in terms of their radiative forcing.

Baseline

In order to quantify the carbon benefits of a project or activity, it is first necessary to determine the baseline 'business-as-usual' scenario. This may not be carbon neutral. For example, abandoned land may lead via a natural succession of vegetation to a mature forest eventually being established that may store more carbon than a planted forest on a commercial rotation – although over a longer time-scale and without providing benefits of carbon storage in harvested wood products, or carbon displacement. Furthermore, background indirect anthropogenic effects (e.g. 'the free-ride' associated with levels of nitrogen deposition and CO_2 fertilisation, or at national scale, associated purely with the changing age structure of existing forests) may imply changing carbon sequestration over time. (Estimates in Watson and Noble, 2005), for example, suggest that globally these factors may be responsible for a total uptake of 6 GtC over the period 2013-2017 compared to the 1990 baseline).

Coverage of different carbon pools and fluxes varies between accounting protocols. This is partly due to differences in the size of impacts between project types or for different environmental conditions, in costs of monitoring and measuring different impacts, and the level of these costs relative to anticipated level of carbon benefits, as well as differences of approach towards trade-offs between cost and precision (Valatin, 2011a).

Table I: Carbon pools and other GHGs covered under voluntary carbon standards" (based upon Valatin, 2013, Table 5), illustrates differences in coverage between existing voluntary market standards covering forestry carbon.

Table I: Carbon pools and other GHGs covered under voluntary carbon standards

		American Carbon Registry Forest Carbon Project Standard	California Air Resources Board				Carbon Fix	Verified Carbon Standard						Woodland Carbon Code
			Reforestation	Improved Forest Management	Avoided Conversion	Urban Forestry		Afforestation, Reforestation and Revegetation	Conversion to reduced impact logging (RIL) with minimal impact on timber	Conversion to RIL with over 25% reduction in timber extracted, or from loged to protected forests	Extended rotation length / Conversion of low productive forests to productive forests	Conversion of forest to non-forest annual crop or pasture	Conversion of forest to perennial tree crop (e.g. oil palm, bananas, fruit trees, spice trees, tea).	
Above Ground Biomass	Tree		√	√	√			√	√	√	√	√	√	√
	Non-tree		√					Δ					√	√
	Woody						√							
	Non-woody						√							
Deadwood	Standing		√	√	√				√	√				
	All													√
Below Ground Biomass	Tree													√
	Non-tree													√
	Woody						√							
	Non-woody						√							
	All							Δ	~	~	~	~	~	
Tree biomass						√								
Litter								Δ	~	~	~	~	~	√
Soil			~	~	~			Δ	~	~	~	Δ	Δ	√
Site preparation	Biological emissions		√	√										
	Fuel emissions		√											
Tree planting/care	Fuel emissions					√								
Woodland management														√
Clearing forest land outside project area	Biological emissions		√		√									
Changes in wood harvesting outside project area	Biological emissions			~										
Harvested Wood products	In-use		√	√	√									
	In landfills		~	~	~									
	Decomposition		√	√	√									
	All							~	~	√	~	~	~	
Unspecified / other		(1)					(2)							

Notes: √ denotes covered in all cases; Δ denotes has to be included where project activities may significantly reduce pool; ~ denotes covered in some cases; (1) All significant changes in carbon pools/GHG sources with exception of litter, and emissions from removal of herbaceous vegetation, fertiliser application, and of nitrous oxide (N_2O) from litter and fine root decomposition; (2) 0.5% of future CO_2 fixation deducted to cover fossil fuel use and 0.005 tCO_2 per kg of nitrogen where fertiliser used. Any biomass burned in land preparation assumed to add 10% to baseline to cover N_2O and CH_4 emissions.

The underlying basis for setting the baseline is fundamental to determining the level of carbon benefits. For Germany, for example, it has been shown that some approaches may even reverse carbon sinks to sources, or sources to sinks, depending on whether carbon sequestration is compared to carbon stocks in a base year (gross-net accounting), or to previous sequestration rates (net-net accounting), and whether a fixed base year or a moving reference period is used (Krug *et al*, 2009; Ciccarese *et al*., 2011). Some approaches would create liabilities for European forest owners rather than simply offering income opportunities (Ciccarese *et al*., 2011).

Additionality

A key issue in carbon accounting – not least if associated carbon benefits are to be sold, is the net impact on the carbon balance within the project boundary compared to the situation if the project or specific activity had not gone ahead (i.e. compared to the baseline ‘counterfactual’). This relates to the concept of ‘additionality’. The underlying rationale of this concept is to distinguish activities which further contribute to climate change mitigation from those which, although they may appear to be associated with carbon savings, offer no benefits above those expected anyway.

Although a simple concept at first sight, in practice approaches to determining additionality vary widely (Valatin, 2011a). As the taxonomy shown in Table II: Forms of additionality” (based upon Valatin, 2013, Table 8) illustrates, additionality is a multi-faceted concept, that can encompass a range of environmental, institutional and financial aspects. More than twenty different forms can be distinguished.

Table II: Forms of additionality

Type	Description
Environmental:	
GHG	Positive overall impact on GHG balances (net carbon benefit of activity or project).
Unit	Emissions per unit output below specified level (or possibly GHG savings per unit area above a threshold level).
Project	a) Afforestation and reforestation: forests unable to establish themselves in the absence of planned activities or project; b) Avoided deforestation or forest degradation: forests would have been deforested or degraded in the absence of the project.
Intent	GHG abatement a decisive factor in decision to proceed.
Tree	Positive impact on the total number of trees.
Ecological	Positive net impacts on habitats, species and biodiversity.
Legal, regulatory, Institutional:	
Compliance	Exceeds statutory requirements.
Incentive	Exceeds benefits associated with incentives provided by regulatory framework.
Threshold	Does not exceed maximum GHG savings counted as additional.
Norm	Meets voluntary industry standards, or good practice benchmarks.
Technological	Application of specific technology.
Barrier	Overcomes implementation barrier.
Practice	Not common practice.
Reporting	National GHG accounting and reporting additionality rules.
Institutional	Independent of statutory emissions reduction targets.
Date	Activities occur after (or in some cases before) particular date.
Term	Abatement arises within a specified time-scale.
Jurisdiction	Activities in particular location, or undertaken by specific communities or social groups.
Financial and investment:	
Financial	Would not be financed without sale of carbon certificates.
Viability	Not financially viable without sale of carbon certificates.
Investment	Not most attractive option without sale of carbon certificates.
Sales	Income from the sale of carbon credits a decisive factor in decision to proceed.
Gaming	GHG emissions not generated for the purpose of subsequent abatement to claim carbon credits.

Table III: Explicit additionality tests applied to forestry projects" below (based upon Valatin, 2013, Table 2), illustrates differences in the aspects currently covered by explicit additionality tests (i.e. those part of specific additionality protocols) under existing voluntary carbon standards that cover forestry carbon. (However, note that similar tests are sometimes included elsewhere under these standards. For example, tests for GHG, ecological, norm, date term and jurisdiction additionality are also used elsewhere under the UK Woodland Carbon Code - see Valatin, 2013, Table 3).

Table III: Explicit additionality tests applied to forestry projects

Category	Additionality test	American Carbon Registry Forest Carbon Project Standard	California Air Resources Board	Clean Development Mechanism:		CarbonFix	Green-e	Plan Vivo	Verified Carbon Standard	Woodland Carbon Code
				Small-scale	Large-scale					
Environmental	GHG	~	√	√		~				
	Unit	~					~		~	
	Project	√	~			√			~	
	Intent	~		~	~		~			
	Tree		~							
	Ecological									
Legal, regulatory and institutional	Compliance	√	√		√	~	√	√	√	√
	Incentive									√
	Norm					√	~			
	Technological						~		~	
	Barrier	~		~	~	~	~	√	~	√
	Practice	~			~				~	
	Institutional									
	Date	√	~		√	~	√			
	Term									
	Jurisdiction									
Financial & investment	Financial	~							~	~
	Viability	~			√	~	~			√
	Investment		~		√	~	~			
	Sales	~			~	~	~			
	Gaming									

Note: √ denotes test applied in all cases; ~ denotes applies in some cases.

Although not currently subject to testing under specific additionality protocols (Table III), 'gaming additionality' and 'institutional additionality' may be considered among the more troubling aspects. Lack of gaming additionality implies that, although purporting to deliver carbon benefits, a project in fact has a negative impact - increasing GHG emissions or reducing abatement. (For a discussion of perverse incentives in relation to HCF-23 destruction projects, and the potential for perverse incentives to arise with forestry projects - e.g. due to incomplete baselines, see Valatin, 2013). Lack of institutional additionality can imply double-counting if carbon benefits are counted both in meeting national carbon targets, and are sold separately as carbon offsets. (Hence the current UK government position, for example, that forestry carbon from UK projects cannot be sold as offsets, but can be described as contributing to meeting national carbon targets).

However, as with other payments for ecosystem services schemes (see Valatin and Coull, 2008), incentives that only value additional carbon benefits may be considered inequitable by landowners already managing their land for the carbon benefits who – because they are not offering additional benefits – do not qualify for the incentives. Nordén (2013) explores in a wider context how such issues can lead to a reduction in the existing level of ecosystem services benefits provided by landowners not covered by the incentives.

Leakage

The net impact of a project or activity on the carbon balance outside the specified project boundary, compared to the case if it had not gone ahead, is also a key issue in carbon accounting. 'Leakage' refers to negative effects that projects can have on the wider carbon balance. (Note, however, in some cases forestry projects may also create positive spillovers).

Projects can lead to increased emissions elsewhere for a variety of reasons (Valatin and Price, 2013). Afforestation of agricultural land may lead to increased emissions from imports, or intensification of agriculture production elsewhere, affecting fossil fuel use and associated carbon emissions. Similarly, forest conservation projects may lead to intensified wood harvesting elsewhere, affecting carbon stocks and emissions.

US studies report that 'leakage' can range from 5% to 93% of project abatement benefits depending upon the activity and region (Murray *et al.*, 2004; van Kooten *et al.*, 2012). A primary concern is that conserving domestic forests will result in increased timber harvesting and environmental degradation in other countries (i.e. indirect land use change), but also that it may result in use of more energy-intensive materials (Gorte, 2009). However, research quantifying such international leakage effects appears sparse, with a recent review (Henders and Oswald, 2012) relating to REDD projects identifying just two studies – both involved modelling exercises based upon complex data inputs.

A general equilibrium approach might help quantify complex secondary impacts on carbon balances such as the effect of increasing the supply of wood products to the world market. (For example, to what extent would an increase in EU output of construction-grade timber lead to more timber being is used in buildings and displacement of fossil-carbon-intensive materials, and to what extent does it displace imports from Russia or North America? To what extent would any reduction in timber exports to the EU lead to a greater accumulation of carbon in Russian and US forests, or to conversion of forests to agriculture as forestry became less profitable, affecting forest carbon stocks in these countries?). Richards and Andersson (2001) note that estimating off-site effects of individual carbon projects is an onerous task as it requires analysing shifts in supply functions for forest products, agricultural products and agricultural land, and suitable general equilibrium models (or even the requisite time-series datasets to build such models) for some countries may not exist at present.

The significance of carbon leakage effects could be expected to vary between project types and countries, partly as a consequence of differences in wider institutional arrangements. Within the UK, for example, leakage due to the potential for afforestation to result in deforestation of other areas is not considered a significant issue owing to the existing regulatory requirements for an environmental impact assessment for deforestation over 1 ha (0.5 ha in sensitive areas), for re-stocking of areas felled, and for protection of biodiversity and semi-natural habitats. The approach to leakage adopted under the Woodland Carbon Code

(Forestry Commission, 2012) developed to help underpin the emerging market for UK forest carbon does not account for any reductions in GHG emissions associated with the cessation of the previous land use. This allows for the potential intensification of activities (e.g. agriculture) elsewhere in the UK. Currently just covering afforestation, leakage associated with forest conservation (the main focus of US studies) is not an issue at present for projects under the Code.

Non-permanence

Non-permanence risks, including the risk of fire or other events releasing the carbon stored back into the atmosphere, are often considered primarily to affect forestry and other land use sector projects, although this view is not universally accepted. Herzog *et al.* (2003, p.306) for instance, argue that permanence issues apply to virtually all carbon mitigation options, essentially being “a function of the policy regime,” so that, in the absence of globally binding emissions restrictions, avoiding burning fossil fuel today increases its future use by enhancing its availability and reducing its future price. Thus, they argue, it does not permanently reduce atmospheric greenhouse gas concentrations as often thought. For a discussion of different approaches to taking non-permanence risks into account, see: Valatin (2011b).

Annex III: Discounting perspectives

For social values, the main justifications for applying discounting to carbon values include (Valatin and Price, 2013):

- The opportunity cost of public finance devoted to climate change mitigation. However, the return on investment funds can be dealt with in other, more appropriate, ways through using an opportunity cost (Price, 2003);
- An innate *time preference* for early rather than late consumption, which government should respect. However, this has no relevance in valuing future climate impacts *to* future generations;
- Assumed future growth of income and consumption per capita which entails diminishing marginal utility – reducing the significance of additional units of future consumption, or lowering opportunity cost of resources to deal with environmental problems. However, under some scenarios (e.g. catastrophic disruption of the world economy due to severe climate change) future income and consumption may fall;
- The possibility that devastating events will eliminate, or radically and unpredictably alter, future returns (HM Treasury, 2003), or result in human extinction (Lowe, 2008). However, discounting on this basis risks creating a self-fulfilling prophecy by reducing the weight given to future costs of climate change, and thus, perversely, increasing the attractiveness of the most risky strategy (business-as-usual).

Approaches differ not just with regard to whether to discount carbon and if so, at what rate, but also concerning whether to use a constant discount rate. Declining discount rates are recommended for UK policy appraisal (HM Treasury, 2003), for example, based upon uncertainty about future values of time preference (Lowe, 2008). For a discussion of the use of declining discount rates for policy, see Pearce (2001), OXERA (2002), Hepburn and Koundouri (2007), and Gerlagh and Liski (2012), and for critiques, see Price (2005, 2010, 2011).

Annex IV: Training material

This Annex provides an example of how the UK government's current approach to valuing carbon and estimating the cost-effectiveness of a forestry option is used.

Problem set

What is the present value of carbon sequestration associated with creating 1,000 ha of new woodland this year? How cost-effective is it as a climate change mitigation measure?[3] For illustrative purposes, we assume that the baseline level of woodland creation in the absence of the change being considered (e.g. new woodland grant structure) is 250 ha. The time horizon of interest is assumed to be 100 years, with the first year of planting (year 1) 2014. The assumed time profile of net carbon sequestration (allowing for initial emissions associated with establishment) for the woodland created is assumed to be as shown in Table IV: Net carbon sequestration over 100 years (tCO2e/ha/year)" below.[4]

For sensitivity analysis, the carbon estimates will be ranged by +/-20% to allow for uncertainties. Thus the low estimates will be based upon increasing initial establishment emissions by 20% and reducing the subsequent sequestration emissions by 20%, while the high estimates will be based upon reducing the initial establishment emissions and increasing the subsequent sequestration emissions by 20% (with no change in the central estimates).

For simplicity, the carbon sequestration estimates assume no thinning, harvesting or other management intervention after planting during the time fame considered, and that the longevity of the particular tree species in the locations planted exceeds 100 years.

Three forest management scenarios will be considered. Under the high scenario the woodland is assumed to be left without further management indefinitely. For the other two scenarios it is assumed that the maximum level of net carbon benefits valued is capped at the long-run average for the species and future management regime assumed. In this example, it is assumed the maximum levels are 438 tCO_2/ha (low scenario) and 639 tCO_2/ha (central scenario).[5]

Carbon sequestration is valued either at private sector values (assumed to be £3/tCO_2) or, from a societal perspective, by applying UK government guidance on valuing carbon.

[3] For an example addressing questions with respect of the Welsh government's afforestation target of 100,000 ha over 20 years, see: Valatin and Saraev (2012).

[4] Although the series used here is fictitious, it displays some similarities to patterns in 5-yearly means published in the Forestry Commission Woodland Carbon Code Carbon Lookup Tables – such as that for oak yield class 4, planted at 1.2m spacing on an indefinite rotation (see: http://www.forestry.gov.uk/forestry/infd-8jue9t), but with net emissions in the first few years to also reflecting establishment impacts on soil carbon, etc.

[5] These correspond, for example, to the long run average for 110-year and 200-year rotations, respectively, for oak yield class 4, at 1.2m spacing, with no thinning in the Forestry Commission Woodland Carbon Code Carbon Lookup Tables.

Table IV: Net carbon sequestration over 100 years (tCO_2e/ha/year)

	Central			Central
2014	-10.0		**2064**	4.8
2015	-4.0		**2065**	4.7
2016	0.2		**2066**	4.6
2017	0.3		**2067**	4.5
2018	0.4		**2068**	4.4
2019	0.5		**2069**	4.3
2020	0.6		**2070**	4.3
2021	0.7		**2071**	4.3
2022	0.8		**2072**	4.2
2023	0.9		**2073**	4.2
2024	1.0		**2074**	4.2
2025	1.1		**2075**	4.2
2026	1.2		**2076**	4.2
2027	1.3		**2077**	4.2
2028	1.4		**2078**	4.2
2029	1.6		**2079**	4.2
2030	1.8		**2080**	4.1
2031	2.0		**2081**	4.1
2032	2.3		**2082**	4.1
2033	2.5		**2083**	4.1
2034	2.8		**2084**	4.1
2035	3.2		**2085**	4.1
2036	3.7		**2086**	4.1
2037	4.2		**2087**	4.1
2038	5.0		**2088**	4.1
2039	6.0		**2089**	4.1
2040	7.5		**2090**	4.1
2041	9.0		**2091**	4.1
2042	11.0		**2092**	4.1
2043	13.5		**2093**	4.1
2044	17.0		**2094**	4.1
2045	21.0		**2095**	4.1
2046	25.0		**2096**	4.1
2047	30.0		**2097**	4.1
2048	36.0		**2098**	4.1
2049	41.0		**2099**	4.1
2050	40.0		**2100**	4.1
2051	38.0		**2101**	4.1
2052	35.0		**2102**	4.1
2053	32.0		**2103**	4.1
2054	29.0		**2104**	4.1
2055	26.0		**2105**	4.1
2056	23.0		**2106**	4.1
2057	20.0		**2107**	4.0
2058	17.0		**2108**	4.0
2059	14.0		**2109**	4.0
2060	11.0		**2110**	4.0
2061	8.5		**2111**	4.0
2062	6.8		**2112**	4.0
2063	5.5		**2113**	4.0

Table V: Social values of Carbon and sensitivities 2014-2100 (£/tCO_2e at 2013 prices)

	Non-traded					Non-traded		
	Low	Central	High			Low	Central	High
2014	30	60	90		**2058**	128	278	428
2015	30	61	91		**2059**	130	285	440
2016	31	62	93		**2060**	131	291	452
2017	31	63	94		**2061**	132	297	461
2018	32	64	96		**2062**	133	302	471
2019	32	65	97		**2063**	133	307	480
2020	33	66	99		**2064**	134	311	488
2021	33	67	100		**2065**	134	315	496
2022	34	68	102		**2066**	134	319	504
2023	34	69	103		**2067**	134	322	510
2024	35	70	105		**2068**	133	325	517
2025	36	71	107		**2069**	133	327	522
2026	36	72	108		**2070**	132	330	527
2027	37	73	110		**2071**	131	332	532
2028	37	74	112		**2072**	130	334	537
2029	38	76	113		**2073**	129	335	541
2030	38	76	114		**2074**	128	336	544
2031	42	84	126		**2075**	126	337	547
2032	45	91	136		**2076**	125	337	549
2033	49	98	147		**2077**	123	337	551
2034	53	105	158		**2078**	121	337	552
2035	56	112	168		**2079**	119	336	553
2036	60	119	179		**2080**	117	335	552
2037	63	126	190		**2081**	115	335	554
2038	67	134	200		**2082**	114	334	554
2039	70	141	211		**2083**	112	333	554
2040	74	148	222		**2084**	109	332	554
2041	77	155	232		**2085**	107	331	554
2042	81	162	243		**2086**	105	329	552
2043	85	169	254		**2087**	103	327	551
2044	88	176	264		**2088**	101	325	549
2045	92	183	275		**2089**	98	322	547
2046	95	191	286		**2090**	96	320	544
2047	99	198	296		**2091**	94	318	542
2048	102	205	307		**2092**	92	316	541
2049	106	212	318		**2093**	89	314	538
2050	110	219	329		**2094**	87	311	535
2051	112	227	341		**2095**	85	308	532
2052	115	234	354		**2096**	82	305	528
2053	117	242	366		**2097**	80	303	525
2054	120	249	379		**2098**	78	300	521
2055	122	257	391		**2099**	76	297	518
2056	124	264	404		**2100**	73	293	514
2057	126	271	416					

The relevant social values are those published for the 'non-traded' sector as carbon sequestration is not covered by the EU emissions trading scheme at present. Current values (as of March 2014) for these are shown in Table V above. Government guidance has yet to be published providing post-2100 carbon values. For this exercise, post-2100 carbon values will be assumed to remain at the 2100 level.

To allow for non-permanence risks (e.g. associated with fires) and potential leakage a buffer of 30% (low scenario), 20% (central scenario), or 15% (high scenario) will be applied to the positive net carbon sequestration by the woodland. These will not be applied to the net carbon losses in the first two years due to emissions from forestry operations and soil disturbance, however.

To estimate the cost-effectiveness of the woodland creation option, indicative establishment costs of £8000/ha (low scenario), £5000/ha (central scenario) and £3000/ha (high scenario) at 2013 prices incurred in the year of planting will be assumed. For simplicity, no ancillary benefits, are assumed. The cost-effectiveness estimates will then be compared with cost-effectiveness benchmarks based upon current UK government guidance (DECC, 2013), computed using the low, central and high estimates of the social value of carbon, respectively.

The Treasury Green Book discounting protocol (HM Treasury 2003, Table 6.1, p.99) will be adopted in computing present values and cost-effectiveness benchmarks. This entails applying a discount rate of 3.5% in the first 30 years after the year of planting (i.e. in years 2-31), declining to 3% in the following 45 years (i.e. years 32-76), and then to 2.5% in the remainder of the period to year 100.

Solutions

As the baseline level of woodland creation is assumed to be 250 ha, the total new woodland of 1,000 ha equates to an additional area of 750 ha. The long-run average net carbon sequestration of 438 tCO_2/ha under the low scenario is reached in this example (allowing for the uncertainty bound) during year 52, and the long-run average net sequestration of 639 tCO_2/ha under the central scenario during year 72.

Net carbon sequestration, present values of the carbon, and cost-effectiveness estimates over a 100 year time horizon for the additional woodland created for the three scenarios are shown in Table VI: Carbon sequestered, present values and cost-effectiveness (2013 prices)".

Table VI: Carbon sequestered, present values and cost-effectiveness (2013 prices)

	Low	Central	High
Carbon sequestered by additional woodland ($MtCO_2$)	0.23	0.38	0.58
Social value of the carbon (£m)	7	24	50
Market value of the carbon (£m)	0.03	0.09	0.20
Cost-effectiveness (£/tCO_2)	27	10	4
DECC cost-effectiveness benchmark (£/tCO_2)	31	62	86

Discussion

The results illustrate the wide gulf between the social values and market values of carbon sequestration. Were a higher discount rate applied in estimating the market value of the carbon sequestration, this disparity in present values could be expected to increase further.

From a societal perspective, the cost-effectiveness estimates are below the benchmark level in each case, and indicate that the woodland creation is highly cost-effective as a climate change mitigation option. Were ancillary benefits (e.g. biodiversity and amenity) included, it would be expected to become even more cost-effective.

A challenge for governments remains to narrow the gap between social and market values of carbon to help overcome market failures resulting in under-provision of carbon sequestration and climate change mitigation.