Methodological Framework for Modelling the Impact of the Agriculture to Forestry Land Use Change at the Farm Level

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Data Annex: A Distributional Analysis of the Social and Private Return to Farm Afforestation, Accounting for the Cost of Carbon

1. Introduction

There is a growing literature on the use of microsimulation models for agriculture, forestry and land use change (Richardson et al., 2014). Much of this literature addresses issues related to productivity and incomes (O'Donoghue, 2014), however consistent with an increasing global focus on sustainability, there is also increasing interest in combining analyses of both economic and environmental impacts (Ramilan et al., 2011). A sub-field of agricultural microsimulation addresses issues associated with land-use change from agriculture to forestry and vice versa (Ryan and O'Donoghue, 2019; Phimmavong & Keenan, 2020) and vice versa. The former transition is particularly important as it helps to mitigate significant carbon emissions from agriculture. This paper describes the development of a model that incorporates both economic and environmental dimensions of the land-use change from agriculture to forestry.

While afforestation in general is a long-term land-use change that is difficult to reverse, afforestation of farmland is a farm management decision that has particular characteristics. Many farm management decisions have immediate impact, as in the case of how many livestock to keep or which fertiliser to spread. However as it takes a forest perhaps 40 years to reach maturity and generate a harvest, there is a considerable time lag between the period when costs are incurred and when incomes are generated. In plantation forests, income may be generated through planting subsidies, the sale of timber from forest thinnings and finally from final harvest. While direct costs are incurred in forest establishment, management and harvesting, indirect costs relate to the opportunity cost associated with no longer being able to use the land for agriculture, namely the loss of annual agricultural income and the loss of flexibility of land use as a result of the permanence of the land use change.

Therefore, the farm afforestation decision is much more akin to an investment decision. Land use change for afforestation has an impact on the economic return, but also importantly on environmental outcomes, particularly carbon sequestration and emissions. As in the case of the economic dimension, there is an environmental return to both afforestation and agriculture. As a tree grows, carbon dioxide (CO_2) is sequestered in the livewood both above and below ground. On decomposition, needle/leaf and forest-floor litter also contribute to carbon stored in soils. On thinning or final harvest, carbon is removed from the forest as roundwood for processing. However there are also losses to the atmosphere as carbon in wood products is stored until the products eventually decompose are burnt, wood for energy is combusted or un-harvested biomass (e.g. logs, branches, roots) decompose, releasing carbon to the atmosphere. In addition, different harvesting and thinning regimes can impact on total forest carbon, which includes both forest carbon and HWP.

An important element of land use change is the reduced agricultural emissions associated with the substitution of forests for agriculture, whether in the form of reduced methane (CH_4) emissions from livestock or reduced nitrous oxide (N_2O) emissions from fertiliser use. Both methane and nitrous oxide have significantly higher global warming potential than carbon dioxide, so the reduction in more powerful emissions from the land use change is a very significant component of the overall reduction in agricultural greenhouse gas (GHG) emissions.

Environmental emissions are externalities that do not traditionally have a market value and as a result, it has been challenging to link private economic returns and environmental outcomes.

Mechanisms to incorporate the social cost of environmental pollution using, for example, Pigouvian taxes or emission trading schemes, place a value on GHG emissions. This enables integrated analysis and the differentiation of the distribution of social and private returns to carbon sequestration/emissions.

Much of the microsimulation literature that looks at environmental issues, in particular GHG emissions, looks at the household sector (Hynes et al., 2014), particularly in relation to carbon taxes (Symons et al., 1994; Cornwell and Creedy, 1996; Bach et al., 2002; Jacobsen et al., 2003; Serret & Johnstone, 2006; Ysé & Nick, 2006). A number of papers have considered the impact of environmental pollution in agriculture, such as nitrate pollution in the case of Doole et al. (2013), while Hynes et al. (2009, 2013) and O'Donoghue et al., (2019) have developed a microsimulation model to look at agricultural GHG emissions. Non-market values have previously been incorporated in forestry microsimulation models in relation to recreation values (Cullinan et al., 2011; Cullinan, 2011).

While some papers examine forest carbon sequestration, such as Bateman & Lovett, (2000), or the impact of forest management regimes on carbon sequestration (Sedjo, 2001; van Kooten et al., 2009; Im et al., 2007; Yemshanov et al., 2015), such analyses focus solely on the forestry dimension. In assessing land use change, it is also necessary to consider the reduction in agricultural activity (with potential consequential reduction in GHG emissions) in addition to the forest carbon sequestration. This paper extends this literature to incorporate agricultural externalities in the social return to farm forestry, documenting the modelling process and data used in order to facilitate replicability.

2. Methodology

A criticism of the literature is that there is quite a wide variety of assumptions used in calculations (Asada et al. 2020), which can have a considerable impact on the results. Additionally, these assumptions are often poorly documented. The objective of this paper is to describe in detail the complex elements associated with modelling the private (financial) and social (carbon) impact of planting one hectare of new forest (afforestation), substituting for one hectare of an existing agricultural enterprise (in a given year), across a nationally representative distribution of farms. At each stage of the modelling process, we document the sources of data, coefficients and assumptions used in the modelling process in the text and in the Appendix to this paper.

Modelling private and social income requires the estimation of the following:

- Forest income is comprised of both afforestation subsidies and market income arising from timber sales. The estimation of timber sales requires forest growth data as a component of a forest bio-economic model of costs and market income.
- Farm micro data are necessary to estimate the agricultural market and subsidy income foregone on planting
- Sequestration of forest carbon and the agricultural emissions displaced from the superseded enterprise
- Future valuation of carbon.

Private (financial) impact of farm afforestation

Afforestation involves an upfront investment with a long time-frame before a return is realised, given the length of time between planting and harvesting and the legal requirement to replant harvested forests. As a result, comparing or combining the inter-temporal returns to forestry with annual returns from agriculture, necessitates the use of a net present value (NPV) framework, as used by Adams et al. (1993). Additionally, as the planting of former farmland incurs an opportunity cost as modelled by Herbohn et al. (2009) and (Upton et al. (2013), we model the NPV of Net Private Forest Income as private forest income less agricultural income foregone on a per hectare basis where:

NPV Net Private Forest Income (per ha) = (Forest Net Market Income + Forest Subsidies) and Agricultural Income Foregone (per ha) = (Agricultural Net Market Income + Agricultural Subsidies)

and

$$Forest\ Net\ Market\ Income = Harvesting + Thinning\ Income - Establishment\ Costs - Variable\ Management\ Costs - Overhead\ Costs$$

and where agricultural net market income or net margin, is defined similarly by Hennessy et al. (2013) as:

$$Agricultural\ Net\ Market\ Income = Output - Variable\ Costs - Overhead\ Costs$$

As afforestation generally takes place on a small proportion of farms (DAFM, 2015b), farmers continue to incur agricultural overhead costs after planting and forest and farm overhead costs cancel each other out. Thus, we opt instead for a variable costs and income approach to calculating farm income (market gross margin):

$$Agricultural\ Net\ Market\ Income = Output - Variable\ Costs$$

In modelling the replacement of a hectare of an agricultural enterprise with one hectare of forest, a mechanism is necessary to relate the productive potential and physical constraints of individual farms to their forestry potential. This is achieved as described in Ryan et al. (2018) using a categorisation developed by Farrelly (2011), that assigns Sitka spruce productivity or Yield Class (YC)[2] to the Teagasc National Farm Survey (NFS)[3] Soil Code (SC) used to represent the dominant soil class in the NFS. Using this classification, the highly productive YC 24 is associated with farm soils (SC1) that are suitable for a wide range of uses, while the most limiting farm soils (SC6) equate to SS YC (14).

The Social (carbon) Impact of Farm Afforestation

There are three components of the social impact: forest carbon storage, displacement of agricultural GHG emissions from the planted land and placing a value on the sequestered/emitted carbon

Forests gain and lose carbon throughout their life-cycle. Six carbon pools contribute to these gains (carbon sequestered) and losses (of carbon to the atmosphere). Each pool is estimated separately as carbon is stored/lost in different ways:

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[2] Yield Class is a measure of forest site productivity and is reported as the average annual timber production over the life-cycle of a forest measured in cubic metres per hectare per year ($m^3 ha^{-1} yr^{-1}$).

[3] The Teagasc NFS is the farm-level micro dataset that is the Irish input for the EU Commission Farm Accountancy Data Network (FADN).

- There is a slow increase in soil organic carbon (SOC) in mineral soils over time, whereas organic (peat) soils lose carbon on drainage and planting
- Carbon is stored incrementally over time in
    a. Above ground biomass (>7cm) and
    b. below ground biomass (roots >5cm), until the wood is either harvested (removed), decomposes or is combusted.
- Decomposing material such as
    a. dead organic matter (DOM) litter (decaying needles/leaves, branches <7cm diameter) and
    b. wood from dead trees (above/below ground >7cm) can decompose and add to SOC over time
- Wood removed as harvested wood products (HWP) (products in current use from domestic harvests)[4] continue to store carbon until they either decompose (rot) or are combusted.

A visualisation of total carbon storage over time is presented for these pools for thinned and unthinned forests and for a range of forest yield classes in Figure 1.

Figure 1. Carbon storage curves for thinned and unthinned Sitka spruce

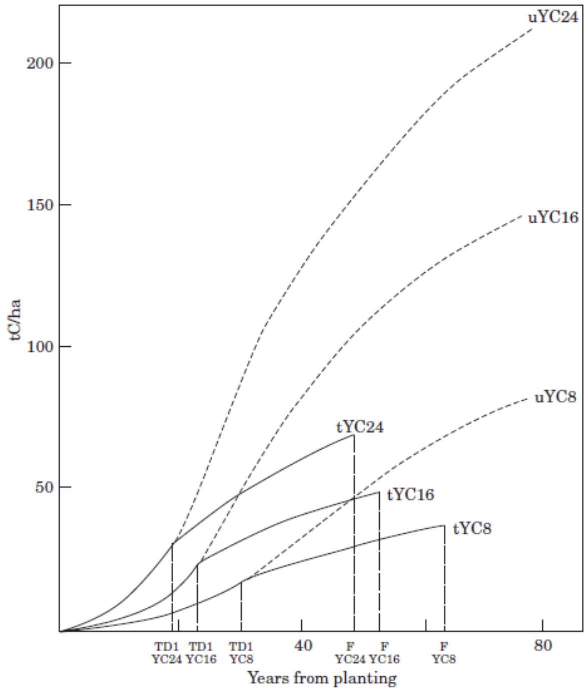


Figure 1. Total carbon storage curves for unthinned and thinned Sitka spruce (discount rate=5%). TD1=date of first thinning (∴TD1 YC24=date of first thinning for yield class 24 Sitka spruce). Dashed line (and solid line prior to TD1) =total carbon stored in unthinned live wood (uTWCS). Solid line (after TD1) =total carbon stored in thinned live wood (tTWCS). ∴uYC24=uTWCS for YC24 Sitka spruce. ∴tYC24=tTWCS for YC24 Sitka spruce.

Source: Bateman & Lovett (2000)

[4] Note: while CO_2 and CH_4 emissions are generated (respectively) in forest harvesting and forest fires, their inclusion is beyond the scope of this analysis.

The agricultural GHGs displaced on planting are (a) methane (CH_4) $e$missions from enteric fermentation in ruminant animals and manure management[5]), (b) direct and indirect nitrous oxide (N_2O) emissions for each livestock sector i.e. manure management, fertiliser application and dung/urine deposition, along with fertiliser emissions from tillage, (c) carbon dioxide (CO_2) from fuel and electricity.

The final component of estimating the social return to farm afforestation is to apply a carbon value to the carbon sequestered/emitted. The concept of the social cost of carbon (SCC) relates to the economic cost caused by an additional tonne of carbon (or equivalent) emissions (Nordhaus, 2017). The cost of additional CO_2 in the atmosphere varies with the level of GHG emissions and as a result, Smith and Braathen (2015) highlight specific assumptions that need to be defined with regard to SCC and the current and future levels of GHG concentrations, against which its effects are to be measured. The SCC estimates the monetary value of the incremental impact of an additional tonne of emissions. Smith and Braathen (2015) further elucidate the elements that should be considered in SCC estimates:

- sea level rise: damage to infrastructure and habitats
- agricultural impact: context specific impacts on crop yields and farmer adaptations
- public health: potential outcomes related to human health
- extreme events: the potential damage from increases in extreme events (fires, floods) and changing weather patterns
- biodiversity: negative impacts on flora and fauna.

Abstracting from the complexity of these elements of the SCC and from tax, regulation and inconsistent assumptions across sectors, the SCC would equal the carbon price, i.e. the marginal cost of emissions reduction (the present value of the damage caused per unit of emissions) (Nordhaus, 2017).

There is significant variation within the literature with regard to the value of carbon (NESC, 2018). Stern (2007) valued carbon at \$85 /tCO_2. A report from the Interagency Working Group (IWG) on Social Cost of greenhouse gases estimated a carbon price of between \$36 and \$42 in 2007 dollar prices per tCO_2, using a discount rate of three percent (IWG, 2013). On the other hand, Moore and Diaz (2015) attribute a lower discount rate as a result of damages caused by climate change and report a SCC value of \$220 per tonne of additional CO_2 (Moore and Diaz 2015). Given the uncertainty around carbon valuation, we utilise the carbon values recommended by the Irish government as the shadow prices of carbon for non-ETS sectors for different years i.e. 2019 (€20), 2020 (€32), 2030 (€100), 2040 (€163) (DPER, 2019)[6].

In estimating the social return to land use change, the value of forest carbon sequestered and GHGs emitted by agriculture are included as follows:

$$\begin{aligned} &NPV\ Social\ Income_{Afforestation\ of\ agricultural\ land} \\ &\qquad = Net\ Private\ Forest\ Income - Agricultural\ Net\ Market\ Income \\ &\qquad + Value\ of\ Net\ Carbon\ Sequestered\ by\ Forests \\ &\qquad + Value\ of\ GHGs\ EEmitted\ by\ Displaced\ Agriculture \end{aligned}$$

[5] Methane accounts for 2/3 of Irish agricultural GHG emissions in 2016 (NIR, 2018)

In order to compare (a) forest returns over different rotation lengths and (b) annual agricultural returns with long-term forest NPVs, an annual equivalent of NPV[7] is required (Herbohn et al., 2009).

C-ForBES Microsimulation Framework

Microsimulation techniques are increasingly used to deal with the complexity of cross-system and cross-country agricultural analyses (Thorne & Fingleton, 2006) and for analyses requiring the use of counterfactual data (O'Donoghue, 2017). The generation of forest and agricultural income and carbon streams in this analysis builds on the farm afforestation microsimulation framework developed by Ryan and O'Donoghue (2019) and the ForBES (Forest Bio-Eeconomic system model) (Ryan et al., 2018) by including a carbon sub-model to estimate forest and agricultural carbon, namely the (carbon) C-ForBES model.

C-ForBES first estimates the private forest income streams associated with planting one hectare of SS forest in 2015, for the land types represented by the Teagasc NFS soil codes. Relative soil productivity for agriculture and SS is incorporated using the relationship between SS yield class and farm soil code (Farrelly, 2011). Annual agricultural private returns are derived using the Teagasc national Farm Survey (NFS) 2015 dataset, which is nationally representative by system and size. C-ForBES incorporates this income foregone as a cost, which is held constant in each year of the forest rotation. A discount rate [5] of 5% is used to generate the NPV of private income streams, which are presented in terms of annual equivalised (AE) NPV of income. In the next stage, agricultural GHG emissions, (methane (CH_4) nitrous oxide (N_2O) and carbon dioxide (CO_2)) are estimated at individual farm level.

C-ForBES Agricultural GHG emissions estimation

In order to estimate the emissions per hectare, the average livestock density per hectare (taking soil code into account), is calculated using methodology consistent with the NFS. Displaced agricultural emissions are calculated for tillage and for each of the livestock systems (dairy, cattle and sheep) on a per animal basis (taking soil code into account). Per hectare livestock emissions are derived using Teagasc NFS methodology and animal numbers and age categories from the 2015 Teagasc NFS, providing livestock equivalent units (LU) per ha (one dairy cow equals 1 LUha^{-1}). NIR (2018) system-specific emission factors (EF) for methane (CH_4) and N_2O emissions for tillage, livestock grazing and manure management practices, fertiliser inputs and fuel and electricity usage are then applied. Tables 12 and 13 (Appendix) present the agricultural activity data by farm system and soil code and table 14 presents the 2015 emission factors from the common Reporting Framework used by the EPA National Inventory Reports. Methane and nitrous oxide are converted to carbon dioxide equivalent CO_2e and the results are presented in table 1.

Table 1. Agricutural Carbon Emission Factors and GHG equivalent conversions

	Energy CO_2	Energy CH_4	Energy CH_4	CH_4	Agri CH_4	Total CO_2e	Total CO_2e
	TCO_2/€	TCH_4/€	TCO_2e/€	tCH_4/ head/yr	tCH_4/head/ yr	tCO_2e/ head	tCO_2e/ LU
Dairy				0.124	0.00012	3.129	3.129

[7] Annual equivalised (AE) NPV (per hectare) = $\frac{r.NPV}{1-(1+r)^{-n}}$ and may not be equal to an average over time.

Cattle				0.051	0.00013	1.308	2.975
Sheep				0.006	0.00001	0.152	0.684
Horses				0.020	0.00015	0.544	1.238
Pigs				0.006	0.00003	0.262	1.176
Poultry				0.000	0.000001	0.00590	0.006
Fuel	0.003	0.00000154	0.00000015				
Fertiliser					0.000032		
Crops					0.00000041		
CO_2 equiv conversion factor	1	25	298	25	298		

Source: Common Reporting Framework 2015[8]

Next C-ForBES estimates carbon in forest soils, live biomass, dead organic matter and harvested wood product pools. As carbon gains and losses can vary considerably depending on forest management practices, this analysis documents all relevant assumptions and equations denoted by [ ] in table 15 (Appendix) for transparency purposes. Forest management assumptions [1] are those used in the ForBES model and based on the Teagasc FIVE (Forest Valuation and Investment Evaluator) (see Ryan et al., 2016). Forest subsidies [2] are provided by the ForSubs model (Ryan et al., 2014). Timber volumes are derived using the merchantable timber volume (MTV) from the Edwards & Christie (1981) yield models for relevant yield classes. Establishment and reforestation costs are derived from Teagasc FIVE [3]. Roundwood from thinnings and final harvest is assumed to be sold standing[9] and income projections are derived using MTV and published timber prices [4].

In addition, we undertake several validations and sensitivity analyses. Finally, in estimating the social value of carbon, C-ForBES replaces the afforestation subsidy with a carbon subsidy, based on carbon values in future periods.

C-ForBES Forest Carbon Estimation

For UNFCCC reporting purposes, Ireland adopts a long-term gains and losses approach in National Inventory Reports (NIR) based on forest and agriculture activity data supplied by government agencies, where national carbon stock changes (CSC) over time are reported annually for all sectors nationally. In this study however, the focus is on the net carbon implications of a land use change from agriculture to forestry in a given year (2015) on a per hectare basis. Where possible and/or feasible, the study utilises the relevant forest and agriculture emission/sequestration factors using the information provided in NIRs. Taking the long-term nature of carbon losses and gains into account, this analysis is undertaken for a 200 year period.

In researching and developing the C-ForBES model utilised in this paper, the authors note considerable variability in the assumptions in the literature in relation to parameters used for forest carbon estimation. On this basis, we undertake sensitivity analysis of the variability in some of these parameters. We reference the assumptions and carbon emission/sequestration factors used in this analysis in the Appendix, along with differing assumptions as used in two particularly relevant but diverse analyses. The Bateman & Lovett (2000) paper is an economic and GIS analysis of the valuation of carbon sequestered by Sitka spruce and Beech in Wales,

[8] http://www.epa.ie/pubs/reports/air/airemissions/ghg/nir2015/

[9] The 'standing' price is the residual price paid to the forest owner, net of harvesting and timber haulage costs

whereas the Irish National Inventory Report (NIR, 2018) is the national accounts framework for GHG emissions reporting (1990-2016) to the United Nations Framework on Climate Change (UNFCCC). Forest carbon is thus estimated for each of the six carbon pools.

According to Wellock et al. (2011) there is little change in *soil organic carbon (SOC)* following planting of forests on (formerly grassland) mineral soils, although there is a gradual build-up of carbon over time. Afforestation on organic soils on the other hand can result in significant emissions (Byrne and Farrell, 2005), that are influenced by water table, soil depth and weather. In reviewing the literature however, values for carbon change in soils vary considerably in different analyses. The soil carbon emissions factor (EF) has increased over successive Irish NIRs, but are lower than those assumed by Bateman & Lovett (2000, p316). While considerable historic planting was undertaken on organic soils in Ireland, current forest policy limits afforestation on peats in environmentally sensitive areas. As there are insufficient data in the Teagasc NFS in relation to the soil composition of individual farms, this analysis makes the simplifying assumption of planting on mineral soils only. However a sensitivity analysis of planting on peats is included (table 3) using the NIR (2018) EF for peats [6].

This section describes the carbon storage per hectare in SS *forest livewood biomass* (both above and below ground), which is estimated where possible using country specific parameters from the 2018 NIR for yield classes 14-24. Total carbon stock (C) is estimated over successive forest rotations from equation 1:

$$C = \sum_{i=1}^{n}\{V_i \cdot D_i \cdot BEF_i \cdot CF\} \cdot P \tag{1}$$

First, the MTV per hectare (denoted by V_i) is derived from the Edwards and Christie (1981) forest yield models. However as these models don't include early annual growth rates, a sigmoid function is utilised to interpolate growth up to the age of first thinning [7]. The carbon mass of live biomass is calculated is the product of volume (MTV), wood density [8] ((ratio of oven dry weight of timber to green volume (D_i) and carbon fraction (CF) [9] (which is generally about 50% of wood biomass) (Dewar & Cannell, 1992).

The dynamic BEFs [10] used in the NIR (2015/2018), derived from research conducted by Black et al. (2004) are used in this analysis. However a sensitivity analysis using both static and dynamic BEFs is also undertaken (table 2). P [11] is the productive forest area.

As only carbon from the above ground part of the tree is used in harvested wood products, we divide total carbon between above ground C_{AG} and below ground C_{BG}:

$$C = C_{AG} + C_{BG} \tag{2}$$

The share of the below ground carbon is defined by the root ratio parameter (denoted by R). In general, broadleaf species have a larger root mass than conifers (Morison et al., 2012). This analysis uses the NIR (2015, 2018) country specific ratio for SS [12] (Black et al., 2009).

The estimation of live biomass carbon relates to above ground biomass:

$$V_{EC,i} = V_i \cdot (1 - R) \tag{3}$$

Therefore, we use an adjusted version of the carbon equation to differentiate above ground (equation 4):

$$C_{AG} = \sum_{i=1}^{n}\{V_{EC,i} \cdot D_i \cdot BEF_i \cdot CF\} \cdot P \tag{4}$$

and below ground carbon stocks (equation 5):

$$C_{BG} = \sum_{i=1}^{n}\left\{\frac{V_{EC,i}}{(1-R)} \cdot R \cdot D_i \cdot BEF_i \cdot CF\right\} \cdot P \tag{5}$$

The *DOM (dead organic matter)* pool is comprised of litter and deadwood. Litter [13] is modelled according to algorithms outlined in NIR (2018) and represents the transfer of carbon from the above ground pool to the litter pool, based on the leaf/needle biomass and the foliage turnover rates (6.7 years for conifers and annually for broadleaves (Tobin et al., 2007). Litter is assumed to decompose at a rate of 14% per year (NIR, 2018-p 222).

The carbon inputs to the deadwood [14] component of DOM in NIR (2018) include tree mortality, tree roots and harvest losses. We follow the assumptions of Black (2016) of a mortality rate of 1.6% of volume per year, with deadwood decomposing at 14% per annum.[10] Although beyond the scope of this analysis, the NIR reports the gradual accumulation of DOM, which can be an important component of carbon stock change over long time horizons.

Carbon is also transferred between DOM pools through complex mechanisms including decay, transfer and disturbance, however, these transfers are beyond the scope of this analysis.

At the point of thinning or final harvest H_i, harvest losses (waste) L_i [15] are incurred and are assumed to be oxidised. These losses can be significant, particularly in first thinning and it is important to adequately take account of these losses to the system (Ryan et al., 2016). The above ground biomass C_{AU} remaining on the site and available for removal from the forest therefore is:

$$C_{AU} = C_{AG} \cdot (1 - L_i) = \sum_{i=1}^{n}\{V_{EC,i} \cdot D_i \cdot BEF_i \cdot CF\} \cdot P \cdot (1 - L_i) \tag{6}$$

Energy wood is considered to be immediately oxidised to the atmosphere [16] and accounts for a significant proportion (34%) of the biomass C removed from forests, according to the Knaggs & O'Driscoll, (2017) 'Wood Flow' report.

A significant amount of non-forest carbon is stored in *harvested wood products (HWP)* for differing lengths of time, depending largely on the end-use and its half-life (IPCC, 2006). The inflow to HWP employed here is modelled as:

$$Inflow = C_{AU} \cdot (1 - E_i) \tag{7}$$

where E_i represents wood energy losses. The proportions of sawnwood, wood-based panels and paper and paper-board making up the HWP on an annual per hectare basis are taken from NIR (2018) [17]. Saw-milling losses are taken into account on the basis of the Knaggs and O'Driscoll (2017) report [18]. In estimating the liberation of carbon from HWP we follow the method of Pingoud & Wagner (2006) which is used in NIR (2018) as follows:

[10] We do not assume a difference between thinned and non-thinned forests, although this is indirectly captured in the growth curves.

$$CHWP_j(i+1) = e^{-k} \cdot CHWP_j(i) + \left[\frac{(1-e^{-k})}{k}\right] \cdot Inflow(i) \qquad (8)$$

where $CHWP_j$ is the carbon stock of HWP in category j, k is the first order decay constant (units yr^{-1}) for HWP category, k is the decay rate ln(2) divided by the associated half-life (HL) of HWPj. $Inflow_i$ is the inflow of HWPj in year i.

Combining the carbon pools, total forest carbon stock is expressed as tCO_2e ha^{-1} yr^{-1} using IPCC conversion factors [19].

3. Data and Summary Statistics

The primary data source in this paper is the Teagasc NFS, which collects detailed information from a representative sample of farms and is Ireland's input to the EU Farm Accountancy Data Network (FADN). Using data for 2015, agricultural cost and revenue streams are generated for each of six agricultural systems (dairy, cattle rearing, cattle other, sheep, tillage and mixed livestock[11]) on six NFS soil codes. C-ForBES then links these agricultural data with the forest yield classes (Edwards & Christie, 1981) and forest management costs and revenues in Teagasc FIVE (Teagasc, 2012). Table 2 describes the average agricultural gross margin (including subsidies) per hectare for the livestock farm systems by soil type and corresponding yield class. There is a broad correlation between soil code and the gross margin per hectare as the highest gross margins are achieved on dairy and tillage farming (particularly on the less limiting soil codes), while cattle and sheep farming have lower gross margins per hectare.

Table 2. Average Private Returns (Gross Margin € ha^{-1}) to Agriculture by Soil Code and Farm System (2015)

SC	YC	Specialist Dairy	Cattle Rearing	Cattle Other	Sheep	Tillage
1	24	1566	1029	996	954	1098
2	24	1181	855	882	1066	1186
3	20	1145	766	764	740	1103
4	20	1082	656	878	859	803
5	18	670	497	550	471	
6	14	584	583	356	681	
All	All	1237	784	867	811	1114

Source: Teagasc NFS (2015)
Note: SC: Teagasc NFS Soil Code (1 – Best, 6 – Worst), YC: Forest Yield Class

Table 3 reports the average return per hectare from planting Sitka spruce (SS) for a range of yield classes, reflecting the opportunity cost of agricultural income foregone. The impact of better quality land as represented by high yield class is clearly evident.

Table 3. Private Returns to Planting SS Forest in 2015 (annual equivalised (AE) of Average NPV per ha) at 5% discount rate

Yield Class	Average AE of NPV (€ ha^{-1})
24	563
20	473
18	425
14	355

Source: C-ForBES

[11] Given the relatively small size and heterogeneity, the results of the Mixed Livestock system are not reported here

4. Results

In this section, we first present the results of the carbon sequestration analysis and discuss these in the context of the modelling assumptions used for forest carbon pools and validating against other analyses. Next we present the private and social returns to the land use change with sensitivity analyses around the parameters used and lastly, the distributional analysis is presented.

Carbon Sequestration

First we illustrate the accumulation and loss of carbon over a 200 year period for Sitka spruce yield class 18 in figure 1. For simplicity, we present an unthinned (No Thin) scenario. The largest increase in carbon is evident in the above ground livewood, particularly in the early years, with an acute loss of carbon at the point of harvesting, when timber is removed from the forest. The lower rate of sequestration in the below ground livewood reflects the 80/20 above/below ground ratio.

At clearfell (final harvest), the above ground biomass declines to zero, while the below ground biomass transfers slowly to the DOM pool. There is an immediate decline in carbon in HWP at harvest, relative to above ground livewood, as just over a third of the livewood is used for energy (Knaggs & O'Driscoll, 2017) and is immediately oxidised, while there are also considerable harvest losses, particularly for first thinnings.

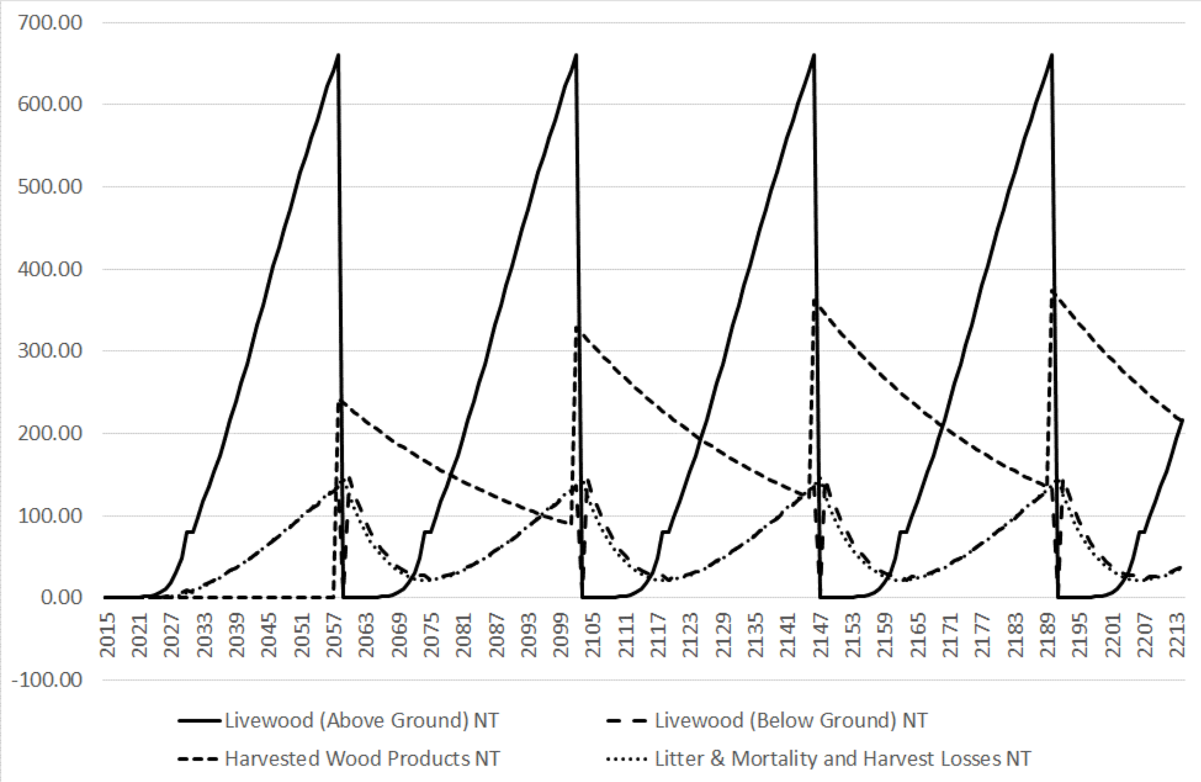
This analysis makes the simplifying assumption that all extractions from forests have the same share of final uses. It is likely however that a high proportion of early thinnings are used for energy, while wood from later thinnings and clearfell is more likely to be used for longer half-life HWP such as sawn-wood and wood-based panels. More comprehensive and further research would be required to disaggregate these allocations, however a sensitivity analysis of different value chain assumptions is undertaken later (see table 9).

On thinning, there is an initial drop in above ground biomass, followed by a subsequent increase in growth due to the greater availability of light, moisture and nutrients for the remaining trees. As each thinning (removal of trees) occurs, livewood carbon declines and the cumulative carbon in HWP increases, albeit declining if the biomass is combusted (oxidised).

Carbon Models, Assumptions and Validation

The calculation of carbon sequestration in forests is sensitive to the data, yield model and forest management assumptions used. For transparency purposes, all forest management assumptions, information on parameters such as productive forest area, species, species mix, rotation length, allocation of timber to different end-uses and other practical aspects of forest management are documented in table 15 (Appendix).

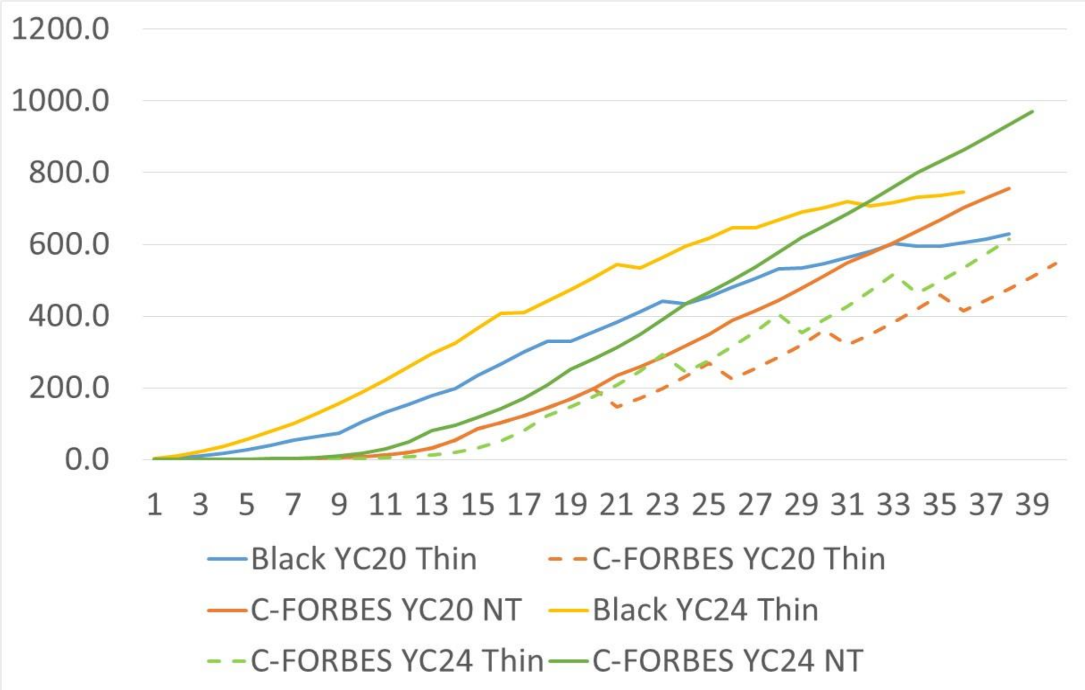
Figure 2. Carbon Sequestration/Loss for Unthinned (NT) Yield Class 18 over 200 years



In this paper, we utilise the Edwards and Christie (1981) static forest yield models, also used by the Forestry Commission (Britain) CARBINE model (Thompson & Matthews, 1989) and in the Forest Greenhouse Gases Model of the UK NEVO (Natural Environment Valuation Online) tool (Binner et al., 2019). The carbon valuation model developed by Bateman & Lovett (2000) combines data from Edwards & Christie (1981) yield models with carbon storage data, to estimate functional relationships for carbon storage in livewood on a per hectare per annum basis. The carbon accounting framework in NIR (2018) uses the CARBWARE single-tree model model (Black et al., 2016) along with the stand-based Edwards & Christie (1981) FORCARB model (Black et al., 2012). In more recent analyses by Black (such as NIR (2019) and the Teagasc Forest Carbon navigator (Teagasc, 2019a), the Carbon Budget Model (Kurz et al., 2009) is parameterised for Irish forest carbon estimations.

Figure 3 compares C-ForBES livewood (above ground) carbon estimation (based on Edwards & Christie (1981) yield models and NIR (2018) accounting rules, against those produced in the Teagasc Forest Carbon Navigator, using the CBM and National Forest Inventory activity data (2004-2006) (DAFM, 2006). The resulting carbon estimations for accumulated carbon dioxide over one rotation is quite similar in both models, albeit differing in relation to the estimation of early growth. Growth prior to age of first thinning is not recorded in Edwards & Christie (1981), thus differences arise between carbon models in relation to imputation of early growth and are sensitive to the data or assumptions employed.

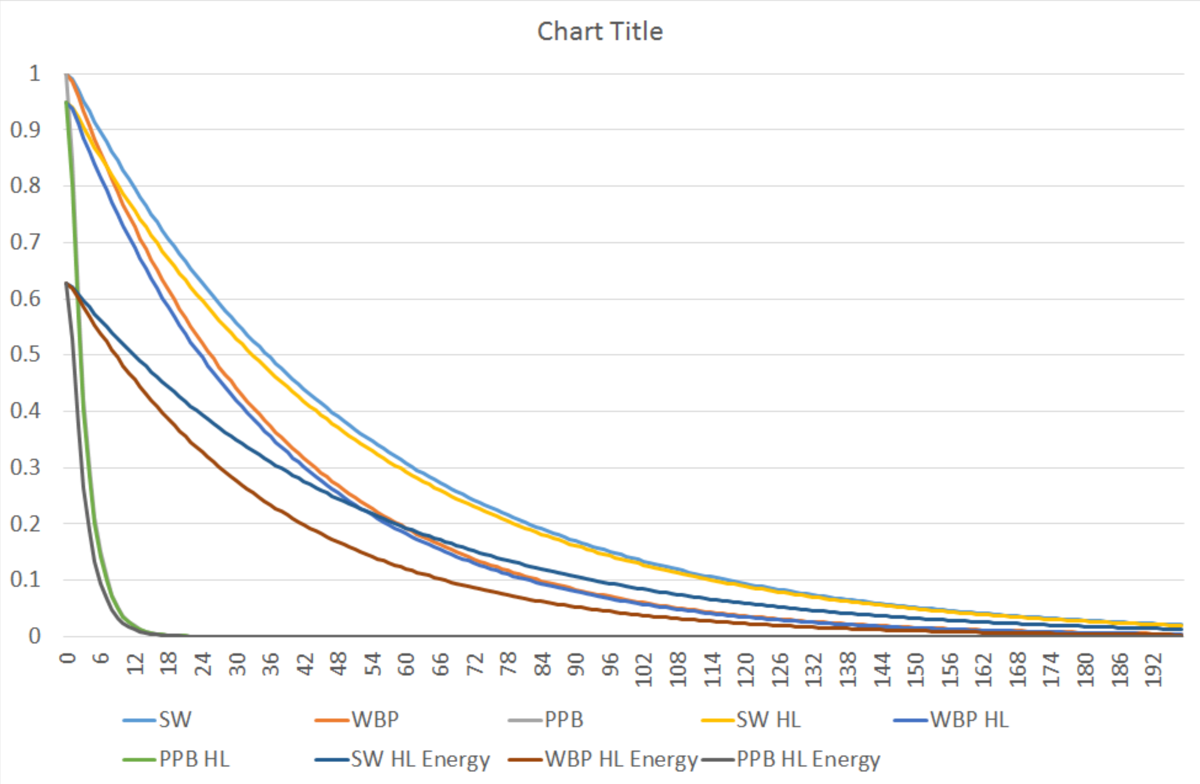
Figure 3. Comparison of Above Ground tCO_2 (C-ForBES and Teagasc Forest Carbon Navigator) over one rotation



Modelling carbon for different purposes can lead to differences in approaches. For example, C-ForBES looks at returns to planting on a hectare basis, (before conversion to volume after thinning and harvest) and has a longitudinal focus. The NIR approach on the other hand takes a cross-sectional perspective on biomass volumes (rather than hectares) and models livewood and wood products separately, using different data sources. Thus, C-ForBES follows the wood-flows from livewood to HWP and wood energy using the Knaggs & O'Driscoll, (2017) approach, making particular assumptions about forest rotation and harvest losses etc. The NIR approach meanwhile, derives livewood (and DOM) sequestration from National Forest Inventory (NFI) data and calculates HWP liberation on the basis of separate roundwood use totals (the volume of the domestic harvest in cubic metres). Due to heterogeneity in forest management practices and differing perspectives on optimal harvest rotation lengths, it can prove difficult to reconcile the resulting outputs.

The literature on harvested wood products remains relatively sensitive to assumptions such as the degree of losses of trees/logs to oxidation during harvesting process can have a considerable impact on the proportion of wood that ends up as wood products. In addition, a critical assumption in relation to life-cycle carbon sequestration, is the proportion of the harvested wood that is used for energy. When wood is burnt for energy, the carbon is immediately released into the atmosphere, while other wood products (e.g. sawnwood and wood-based panels) store the carbon for considerable periods of time. Thus assumptions in relation to both harvest losses and allocations to wood energy can impact on the domestic harvested wood inputs to the HWP pool. Figures 4 and 5 present sensitivity analyses of the Harvested Wood Flow Carbon Liberation depending on whether a proportion (34%) of wood is used for energy.

Figure 4. Impact of different approaches to determine the wood inputs to HWP on SW, WBP, PPB Carbon Liberation Curves for NIR (2018) approach, C-ForBES harvest losses (HL) and C-ForBES harvest losses + wood energy losses (HL Energy)



Note: see Table 15 (appendix) for further detail on HWP assumptions.

Figure 4 compares the NIR (2018) approach with the C-ForBES approach, plotting the carbon liberation curves for the HWP categories (SW: sawnwood, WBP: wood-based panels and PPB: paper and paperboard. Three scenarios are presented, namely (a) NIR (2018) approach (b) C-ForBES harvest losses from Teagasc FIVE and (c) C-ForBES harvest losses plus 34% allocation of harvested wood to energy, based on Wood Flow (Knaggs & O'Driscoll, 2017).

Comparing the carbon liberation curves for different uses (SW, WBP and PPB harvested wood product categories) we see that the impact of harvest losses is quite small, while the impact of the wood energy use is considerable. In all cases, the Paper and Paperboard HWP is negligible.As expected, sawnwood stores the greatest amount of carbon over time. There is little difference between the sawnwood curves for NIR (2018) (SW) and C-ForBES (SW HL). Similarly, there is little difference between the liberation curves for WBP (NIR) and WBP HL (C-ForBES harvest losses. However, the reduction of the harvested wood input to HWP as aresult of the 34% allocation to wood energy, pulls the C-ForBES carbon liberation curves (SW HL Energy, WBP HL Energy) well below the NIR curves in the earlier years, with the curves converging only towards the end of the 200 year period.

The Bateman & Lovett (2000) approach is calculated on a per hectare basis but with a different methodological approach to C-ForBES, as Bateman & Lovett (2000) estimate equations for livewood sequestration, emissions incurred in thinning/harvesting and carbon storage/release from HWP. Comparing Bateman and Lovett (2000) with the NIR (2018) in table 4 and plotting the curves in figure 5, we find that the carbon profile for SW and PPB are very similar (they are indistinguishable)), when harvest losses and energy are excluded. However, the profile is lower for wood-based panels in NIR (2018) than in Bateman & Lovett (2000).

Figure 5. Harvested Wood Flow Carbon Liberation Curves: Comparison of NIR (2018) (SW, WBP, PPB) with Bateman & Lovett (2000) (Bateman SW, Bateman WBP, Bateman PPB)

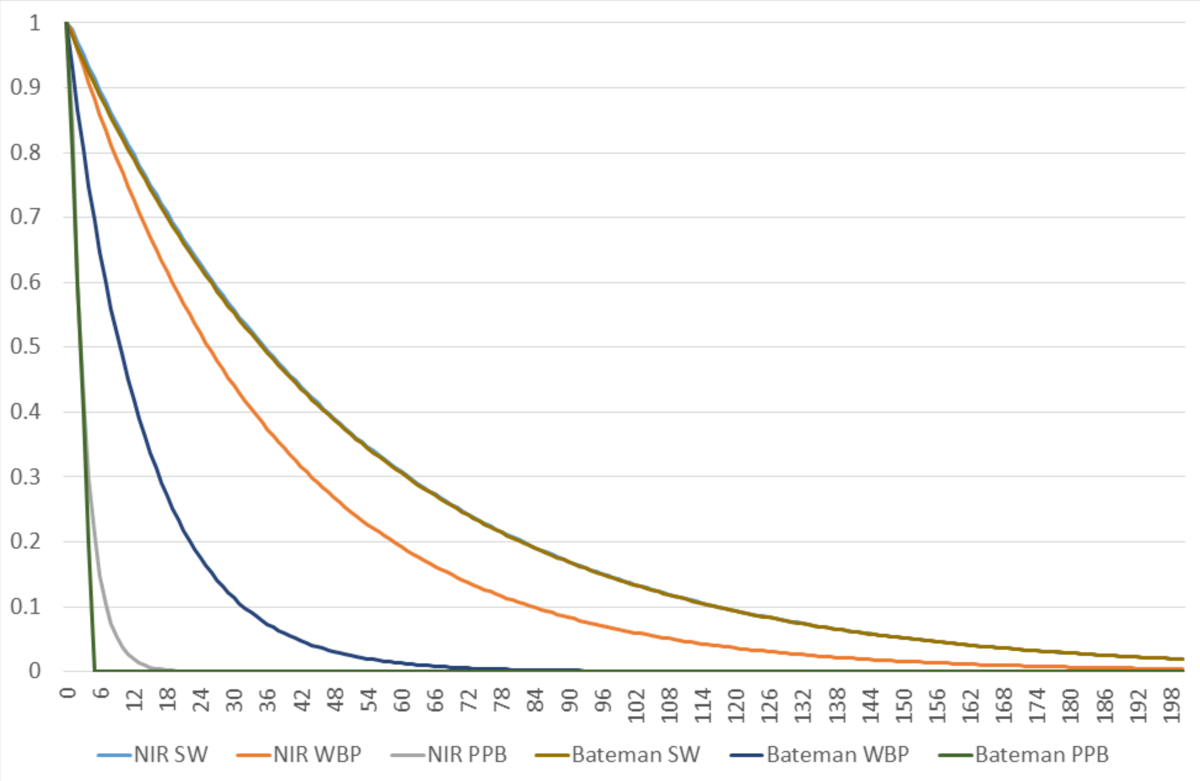


Table 4. Harvested Wood Flow Carbon Liberation Curves: Comparison of NIR (2018) (SW, WBP, PPB) with Bateman & Lovett (2000) (Bateman SW, Bateman WBP, Bateman PPB)

Bateman	0	0	0	1
Harvest Loss	0	1	1	0
Energy	0	0	1	0
Sawnwood (SW)	0.255	0.242	0.160	0.253
Wood-based panels (WBP)	0.186	0.177	0.117	0.072
Paper and paper board (PPB)	0.020	0.019	0.012	0.015

Private and Social Returns to Land Use Change from Agriculture to Forestry

In order to model the social impact of land use change, it is necessary to include the alternative land use, namely agriculture, and to combine the private economic component with the social component (including the value of carbon). Table 5 describes the economic dimension of planting one hectare of forestry. Columns A, B, C and D respectively report market gross margin and direct payments/subsidies for agriculture and for forestry. Market gross margins for both agriculture and forestry are correlated with soil type. Direct payments have a redistributive focus and are uncorrelated with soil type.

Table 5. Economic Components of Agriculture and Forestry (2015) (Annual Equivalised NPV per ha (€)

Soil Code	Agriculture		Forestry	
	A	B	C	D
	€	€	€	€
SC1/YC24	1200	366	224	306
SC2/YC22	792	388	224	306
SC3/YC20	803	342	154	302
SC4/YC18	731	351	154	302
SC5/YC16	356	314	124	300
SC6/YC14	258	326	52	298
Average	878	359	155	296

A – Market Gross Margin per hectare (Agriculture)
B – Direct Payments per hectare (Agriculture)
C – Market Gross Margin per hectare (Forestry)
D – Direct Payments per hectare (Forestry)

The carbon sequestration/emissions resulting from planting one hectare of SS forest replacing agricultural enterprises, are calculated by applying the carbon dioxide equivalent coefficients per hectare from table 1 to farm-level activity data (both direct and indirect emissions) and forest life-cycle (200 years) data. In facilitating comparison between agriculture and forestry, life-cycle sequestration from forestry and emissions from agriculture are annualised using a 5% discount factor to produce the average discounted tonnes of carbon dioxide equivalent per hectare $(tCO_2e)\ ha^{-1}$ for the farm systems and soil types. These are reported as average annual equivalised values in table 6 (columns E and F). The components that result in these returns are also reported, showing that for higher quality soil types, the quantity of carbon sequestered per hectare is almost twice that of the animal emissions displaced. For SC5, the ratio is closer to a factor of three. The implication here is that replacing agriculture with forestry on the best land would result in a reduction of net carbon dioxide emissions of 24.1 tCO_2-e on average over the life-cycle.

In terms of the corollary (replacing agriculture on poorer soils with forestry), the net reduction or sequestration is lower, because of (a) the lower carbon sequestration from forests and (b) because the substituted enterprise had lower emissions. Table 6 also reports the average private and social returns from the afforestation of one hectare of agricultural land and shows that on average, the private return to forestry is lower than the return from agriculture in the current policy environment, but becomes increasingly positive when increasing carbon values are substituted for afforestation premium.

Table 6. Average Annual Equivalised Social Return to Planting one hectare of Unthinned Forest (2015) Displacing Agriculture

Soil Code	Agriculture	Forestry	Private Return	Social Return			
	E	F	(C+D) – (A+B)	C – (A+B) + (E+F)*P			
tCO_2 Value (P)	tCO_2	tCO_2	0	20	32	100	163
SC1	-9.2	14.9	-1036	-556	-268	1365	2878
SC2	-8.4	14.9	-651	-185	95	1680	3148
SC3	-7.5	11.8	-690	-304	-72	1239	2455
SC4	-7.4	11.8	-627	-242	-11	1298	2511
SC5	-4.5	10.8	-246	59	242	1281	2243
SC6	-4.9	7.8	-234	19	171	1031	1828
Total	-8.0	13.4	-785	-359	-103	1347	2691

1. No-thinning assumed
2. BEF Factors from the 2015/18 National Inventory Reports are used

3. All values are discounted using a 5% discount rate
E – tCO_2– Agriculture
F – tCO_2 – Forestry

P – Cost per tCO_2However, we would like to examine the impact of taking carbon into account in the returns to planting. Therefore, we replace the afforestation subsidy with a carbon subsidy (based on a range of carbon values) for the afforestation subsidy. At low values of €20 and €32, (similar to the lower bound carbon price in the national agricultural GHG Marginal Abatement Cost (MAC) curve (Lanigan et al., 2018) of €25 per tonne of CO_2), the social return for planting a hectare of forest exceeds that of agriculture on the poorest soils. Once the carbon value is increased to €100 and €163 (reflecting the Irish government shadow price of carbon for 2030 and 2040 respectively) all soil codes have a higher social return from forestry than from agriculture, with the highest returns on the most productive soils.

Sensitivity Analyses

The science behind forest carbon accounting and modelling is complex and values change over time as the science evolves. One such example is the choice of the biomass expansion factors (BEF), which is the ratio of total tree biomass to merchantable timber volume which changed quite substantially between the 2012 National Inventory Report and the 2015/2018 reports. The earlier report assumes a constant BEF of 1.64, whilst the latter uses a minimum of 1.68, with variation over time as described in table 15 (Appendix). Table 7 compares results for annual equivalised tCO_2 using different BEF coefficients, showing both the greater tCO_2 for the later coefficients and the change in the ratio between thin and no thin forests.

Table 7. Comparing the Annual Equivalised tCO_2 for Thin and No Thin by Soil Code/Yield Class for 2012 and 2015/18 National Inventory Report Assumptions

Soil Code (SC)/ Yield Class (YC)	2012 BEF	2015/18 BEF	Ratio	2012 BEF	2015/18 BEF	Ratio
	No Thin	No Thin	No Thin	Thin	Thin	Thin
SC1/YC24	12.69	14.86	1.17	8.46	10.77	1.27
SC2YC/24	12.69	14.86	1.17	8.46	10.77	1.27
SC3/YC20	10.44	11.83	1.13	7.58	9.04	1.19
SC4/YC20	10.44	11.83	1.13	7.58	9.04	1.19
SC5/YC18	9.34	10.75	1.15	7.43	9.14	1.23
SC6/YC14	7.60	7.80	1.03	5.14	5.37	1.04
Average	11.56	13.37	1.16	8.04	9.98	1.24

Note BEF – Biomass expansion Factor

The analysis undertaken here assumes the same allocation of wood to the value chain from thinnings and from final harvest; in other words, the share of wood uses allocated to energy, sawn wood and wood-based panels is consistent across thinnings and final harvest. Actual allocations can vary considerably, depending on management regime, market conditions and proximity to wood energy and sawnwood processors. In reality, a large proportion of early thinnings is likely to be used for wood energy, with the proportion allocated to sawnwood increasing as tree size increases. However there are no data sources for these allocations, therefore, we would like to test the impact of allocating all thinnings to wood energy.

In table 8, annual equivalised tCO_2 is compared for two extreme assumptions: (a) all harvested wood (thinnings and clearfell) is allocated proportionally to energy and to harvested wood products and (b) all thinnings are allocated to energy and immediately combusted. These assumptions can be regarded as an upper and a lower bound. We see that when all thinnings are used for energy, the amount of clearfell wood used for energy decreases by about 40%. The annual equivalised total $tCO_2$is about 11% higher for the original assumption, reflecting greater earlier storage than when all thinnings are used for energy. If the values were undiscounted, there would be no difference, as the same amount of wood is burnt for energy under both scenarios. In order to get a more precise estimate, a more detailed evaluation of different wood flows over different parts of the life-cycle would be required.

Table 8. Comparing the Annual Equivalised tCO_2 for Thinned Forests by Soil Type if all thinnings are used for energy

Soil Code	Equal Allocation from Thinnings and Clearfell to Energy	Thinnings only Allocated to Energy
1	10.77	9.60
2	10.77	9.60
3	9.04	8.16
4	9.04	8.16
5	9.14	8.37
6	5.37	4.83
Average	9.98	8.95

Note: 2015/18 National Inventory Report Assumptions

The impact of the land use change from agriculture to forestry is at the core of this analysis. While afforestation on blanket peats is no longer grant-aided, many farms have areas of peaty (organo-mineral) soils. However, the available farm-level data are insufficient to distinguish between peaty and minerals soils. C-ForBEs assumes that the SS forest is planted on mineral soils, however we are interested in the sensitivity of planting on peat and mineral farmland. Table 9 highlights the lower carbon sequestration on peats by about 2tCO_2 per ha, given soil carbon losses at the time of planting due to drainage on peaty soils.

Table 9. Total Annual Equivalised tCO_2 Forest Carbon Sequestration for Mineral and Peat Soils by Yield Class

Yield	Mineral	Peat
24	12.69	10.56
24	12.69	10.56
20	10.44	8.32
20	10.44	8.32
18	9.34	7.21
14	7.60	5.48
Average	11.56	9.43

Note: For simplicity purposes in peat calculations, these values are calculated using EFs from NIR (2012)

Distribution of Private and Social Returns

In the paper thus far, we have described average carbon figures for different soils and sectors. However, these averages mask a wide distribution. Table 10 presents distributional assumptions in relation to private returns and social returns. On average 32.4% of farms have positive private returns to planting, including forest subsidies. Replacing the afforestation subsidy with low carbon (subsidies) values of €20 and €32 per hectare, and using the most recent NIR (2015/18) assumptions for biomass expansion factors and also incorporating agricultural subsidies, the share of farms with a positive social return is 30.4% and 46.6% respectively. Using a carbon value of €100 per hectare, the share rises to 96.5%, while at a carbon value of €163 per ha, nearly all farms (99.9%) have positive social returns. As a sensitivity analysis, we utilise earlier NIR assumptions, finding an impact of about five percentage points for lower carbon values. We also test the sensitivity to whether afforestation occurs on land with or without agricultural subsidies, finding an unsurprisingly high impact.

Table 10. Share of Farms with a positive Private and Social return to Forestry by BEF Assumption and inclusion of Forest Subsidy

	Private Return	Social Return			
Carbon value (€)	0	20	32	100	163
BEF (NIR, 2012)					
Incl Farm Subsidy	0.324	0.264	0.427	0.943	0.998
Excl Farm Subsidy	0.551	0.577	0.672	0.983	0.999
BEF (NIR, 2015/18)					
Incl Farm Subsidy	0.324	0.304	0.466	0.965	0.999
Excl Farm Subsidy	0.551	0.594	0.697	0.991	0.999

Relating the livestock units per hectare on different soil codes, table11 reports the average areas of livestock emissions that could be displaced by the carbon sequestered by one hectare of SS forest, measured in terms of land area (hectares of forest) and livestock emissions (LU per ha). For simplicity, values are reported for no thin (NT) forests. Results are reported for (a) planting a forest only, without a land use change (No LUC), and (b) a land use change, accounting for carbon sequestration from forestry and the displacement of GHG emissions from agriculture. On average, the emissions from 1.68 hectares of land (with animals) is equivalent to the carbon sequestration on one hectare of forest without land use change. When forestry displaces agriculture, the net sequestration increases to the equivalent of 2.68 hectares. This ratio increases with stocking rate, with a 25% gap between the highest and lowest soil codes. In terms of livestock units, one hectare of forestry displacing agriculture (with LUC), replaces 3.79 livestock units per hectare on average, with the number of livestock units displaced varying from 2.34 for the poorest soil types to 4.18 on the best soil types, and 2.38 LU per ha on average without LUC.

Finally, we note that one hectare of thinned forest displaces fewer animal emissions than an unthinned forest, due to the lower sequestration associated with thinned forests. It should be noted also that this information relates to all land, including tillage. As tillage land is primarily used for animal feed, this is a reasonable assumption.

Table 11. Areas of livestock production that could be offset by one hectare of SS, at different livestock densities ($LUha^{-1}$), with/without accounting for land use change (LUC[1]) by soil code

Soil Code	Stocking Rate	Ha of Forest per Ha of Animals	Ha of Forest per Ha of Animals	LU per Ha of Forest	LU per Ha of Forest	LU per Ha of Forest
	LU per Ha	No LUC	With LUC	No LUC	With LUC (No Thin)	With LUC (Thin)
1	1.57	1.62	2.62	2.56	4.13	3.03
2	1.52	1.76	2.76	2.67	4.18	3.03
3	1.33	1.59	2.59	2.12	3.45	2.69
4	1.36	1.59	2.59	2.17	3.52	2.75
5	0.87	2.38	3.38	2.06	2.93	2.29
6	0.90	1.61	2.61	1.44	2.34	1.85
Average	1.41	1.68	2.68	2.38	3.79	2.84

5. Conclusions

In this paper, we describe in detail the development of a microsimulation model to simulate the life-cycle impact of the conversion of agricultural land to forestry. Reflecting the negative and positive externalities associated with agriculture in terms of carbon emissions and forestry in terms of carbon sequestration respectively, we model both private returns and social returns.

These reflect respectively the return from the market and the returns to society (accounting for carbon emissions and sequestration). The paper is novel in that it brings together the literature on land use change, which focused mainly on the social returns to land at an aggregate spatial level, and farm level simulation analysis that focuses on private returns. Given the importance of afforestation as a driver of carbon sequestration and the economic impact of alternative land uses, it is important to combine both dimensions.

Incorporating information from many fields, there is a wide variety of assumptions made in calculating social returns. In this paper, we document all assumptions, validate components against other analyses and test the sensitivity of the results to different assumptions. In comparing the growth curves from the UK-based Edwards & Christie (1981) stand-level yield models, with more recent growth curves based on Irish forest inventory data and single-tree plots (Teagasc, 2019a), the end point in terms of tree biomass is similar, but the growth occurs earlier in the Irish curves.

Recognising the variety of assumptions that are used in modelling land use change to forestry, we assessed the impact of using the changing assumptions in relation to Biomass Expansion Factors in the National Inventory Report over time, finding a material difference for lower carbon values. We found that our harvested wood products simulation based upon the Irish National Inventory Report compares quite favourably with an alternative method used by Bateman and Lovett (2000). However, we note that assumptions in relation to harvest losses and the allocation of harvested wood to energy can result in sizeable impacts on carbon storage. Our analysis focused mainly on afforestation of mineral soils, but highlighted the lower carbon sequestration on peat soils. Looking at extreme assumptions in relation to the use of thinnings for energy production, we find that allocating equal shares of thinings and final harvest to energy (upper bound), to allocating all thinnings to energy (lower bound), resulted in a moderate difference of about 12%, with the actual allocation lying somewhere between these bounds.

Given the wider variety of assumptions used in this type of inter-disciplinary analysis, there is merit in a more coordinated approach, both nationally and internationally in relation to model assumptions. The IPCC makes assumptions at a particular level, but there is a need for greater consistency in more detailed assumptions. For example, at national level, agreement around what happens at the point of harvest in terms of losses at international level, greater consistency in terms of the harvested wood value chain and its relationship with the characteristics of the trees in the forest, would both be beneficial to modellers working in this complex field. The US government Interagency Working Group (IWG, 2013) on the Social Cost of Carbon (Nordhaus, 2017) provides an influential example of such a coordinated approach.

There are limitations in relation to the modelling approach employed here that relate largely to the availability of forestry activity data, as forest data collection was historically focused on the science of timber rather than carbon production. The analysis does not take into account the accumulation of carbon in soils over very long time horizons. The sole focus on pure Sitka spruce is also a limitation of this analysis, and relates to the robustness of production and economic SS data, compared to other conifer and broadleaf species. Thus, this analysis ignores the carbon contribution of broadleaf species planted as areas of biodiversity enhancement within Irish forests. However, these limitations do not affect the qualitative conclusions of this analysis, namely that unthinned livewood is the largest forest carbon pool while the HWP pool is the largest non-forest pool and is perhaps where the greatest advances could be made in relation to the substitution of long-lived sawnwood products for concrete and steel.

Finally, in considering the distributional impact of the private and social returns to agriculture, this paper finds significant heterogeneity between the private and social return across farms, with the incorporation of a value for carbon resulting in many more farms with positive social return than private returns. This paper also contributes to the growing fields of microsimulation modelling for both environmental policy analysis and agricultural policy analysis. Utilising farm survey data that is collected uniformly in many countries, whether it be the European FADN or OECD data in countries like the USA, Australia, New Zealand, and utilising information in relation to afforestation that is available as part of national carbon accounting frameworks, this methodology is replicable and scalable to other countries. It thus has the capacity to be utilised for ex-ante analyses of afforestation initiatives associated with GHG reduction and increased carbon sequestration.

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Appendix

Table 12. Enteric Fermentation and Nitrous Oxide Emission Factors

	CH_4		N_2O
	Enteric Fermentation	Manure Management	Manure Management (kt)
Emission Factor	(kg /headCH_4/yr)	(kg CH_4/head/yr)	(kg/N_2O /head)
Dairy	113.41	10.30	0.12
Cattle	46.39	4.43	0.13
Sheep	5.61	0.39	0.01
Horses	18.00	1.99	0.15
Pigs	1.33	5.04	0.03
Poultry	0.00	0.22	0.00
Deer and Goats	25.00	1.62	0.12

Source: Common Reporting Framework http://www.epa.ie/pubs/reports/air/airemissions/ghg/nir2015/

Table 13. Average LU per Ha by Soil Code

Soil Code	Dairy	Cattle	Sheep
SC1	2.09	1.74	1.62
SC2	2.03	1.48	2.13
SC3	2.00	1.42	1.66
SC4	1.84	1.35	1.25
SC5	1.58	0.73	0.42
SC6		1.14	1.69
Average	2.00	1.45	1.28

Source: C-ForBES

Table 14. Average tCO_2e per hectare by source of emissions by Soil Code

Soil Code	Dairy	Cattle	Sheep	Fuel	Fertiliser	Crops
SC1	1.77	4.00	0.21	0.000207	0.001297	0.000019
SC2	0.96	4.00	0.79	0.000190	0.001071	0.000018
SC3	1.15	3.79	0.56	0.000221	0.000937	0.000008
SC4	1.31	3.58	0.38	0.000219	0.000853	0.000003
SC5	0.36	1.42	0.41	0.000080	0.000303	0.000002
SC6	0.00	2.80	0.68	0.000107	0.000378	0.000001
Average	1.24	3.57	0.45	0.000191	0.000993	0.000012

Source: C-ForBES

Table 15. -C-ForBES Modelling Assumptions

Model component	C-ForBES parameters and assumptions	Comparisons with parameters/assumptions in NIR (2018) and Bateman & Lovett (2000)	Additional notes
Scale of analysis	Per hectare net carbon storage or emissions in a given year.	Bateman & Lovett (2000): Per hectare net carbon storage or emissions in one year. NIR (2018): National Carbon Stock Change (CSC) over time.	
Period of analysis	200 years. To accommodate approx. 4 rotations of SS (dep on YC) to allow for consideration of half-life of Harvested Wood Products (HWP). Replanting is assumed in the year following harvest.	Bateman & Lovett (2000) model extends to 1000 years NIR (2018): Annual CSC 1990 – 2016	
Yield Class (YC)	Yield classes 14-24 are analysed. YC14 is lower bound for eligibility for afforestation grants and subsidies.	NIR (2018) reports across all historic and current YCs. Bateman & Lovett (2000) estimate YC 6 – 24.	
[1] C-ForBES Forest Management Assumptions derived from Teagasc FIVE (see Ryan et al., 2016)	[1a] Tree species: Forest Investment and Valuation Estimator (FIVE): Sitka spruce (SS), which is the most commonly planted conifer in Ireland. To reduce complexity a simplifying assumption is made to model one hectare of pure SS (see [2] and [11]) as cost, growth and price data in FIVE are not as robust for mixtures of broadleaf and conifers. However, productive area is modelled at 85% to account for broadleaf component. In this analysis therefore, carbon sequestration from broadleaves is not modelled.	NIR (2018) estimates CSC for all species planted in Ireland. Bateman & Lovett (2000) model and map C storage for SS and Beech in Wales	"The forest must contain a minimum of 15% broadleaves by area. This can comprise: broadleaves planted in broadleaf GPC plots of minimum width; and/or broadleaves planted as part of the 'at least 10% diverse' requirement for GPC 3; and/or additional broadleaves planted for environmental and landscape reasons". www.teagasc.ie/forestry/Grants
	[1b] Forest Yield: Merchantable Timber Volume (MTV) (Edwards & Christie, 1981)	NIR (2018) is based on a range of models including FORCARB (based on Edwards & Christie, 1981), CARBWARE & Carbon Budget Model (CBM) (NIR, 2019). Bateman & Lovett (2000): MTV from (Edwards and Christie, 1981) and combine with data on carbon storage in Sitka spruce (Cannell and Cape, 1991; Matthews, 1993) to plot thin and no-thin carbon storage curves.	

	[1c] Thinning: Marginal Thinning Intensity (MTI[12]) @ 5 year intervals from Edwards & Christie (1981)	NIR (2018)/Bateman & Lovett (2000) also use this static thinning assumption.	
	[1d] Rotation: 'Reduced rotation' = (Age of max MAI[13] – 20%) (Phillips, 1998; Anon, 1977).	NIR (2018) rotation = max MAI from CARBWARE (Black, 2016). Bateman & Lovett (2000) estimates felling year (F) based on age of max NPV for given species, YC and discount rate – see Bateman (1996)	
[2] ForSubs model (Ryan et al. (2014)	Forest subsidy: General Planting Category GPC3 (10% Diverse Conifer, e.g. Sitka spruce and 10% broadleaves) €510/ha/year - paid annually for 15 years for first rotation only		GPC 3 – 10% Diverse Conifer/Broadleaf: Comprises of a mix of Sitka Spruce/Lodgepole pine together with at least 10% Diverse conifer (approved conifer other than SS/LP). Broadleaves adjacent to roads and watercourses may also form part of this 10% www.teagasc.ie/forestry/grants
[3] Costs Teagasc FIVE (Ryan et al., 2016)	Ground preparation, fencing, planting, maintenance, insurance, replanting.		
[4] Timber prices	Coillte (State forestry body) 10 year average timber prices Annual timber price series published annually by Irish Timber Growers Association (ITGA) (see Teagasc, 2019)		
[5] NPV discount rate	The conventional discount rate used for forestry in Ireland is 5% (Clinch, 1999).	Bateman & Lovett (2000) also use a discount rate of 5% for forest NPVs.	
[6] Soil organic matter (SOM)	Analysis assumes land use change on grassland on mineral soil only with no change in SOC. In the sensitivity analysis of planting on mineral or peat soils, the coefficients from NIR (2012) are used.	NIR (2018) assumes (a) no carbon stock change on the planting of forests on mineral soils and (b) a mean organic soil EF of 0.59 t C/ha/year over the first rotation (50 years) as organic soils are not a source following successive rotations (Byrne & Farrell, 2005).	NIR (2018) categorises Irish soils into three major groupings based on soil carbon characteristics. All mineral soils are grouped together, while all organic soils with an organic layer greater than 30 cm are classified as

[12] MTI is defined in gross volume terms as 70 % of YC ($m^3 ha^{-1} yr^{-1}$) (see Ryan et al., 2016).
[13] MAI= Mean Annual Increment (see Ryan et al., 2016)

		Bateman & Lovett (2000) assume long term net gain of soil carbon (50 t C ha-1 on mineral soils) or loss (750 t C ha-1 on peat soils) occurring within 200 years.	peat. Finally, organic soils with an organic layer less than 30 cm are classified as peaty/mineral. CARBINE (Thompson & Matthews, 1989): V Changes in soil carbon are assumed to take place in response to land-use change. Magnitude and changes over time are estimated according to soil type (texture) and major land use category.
[7] Early growth	We use a logistic function to interpolate early growth and the growth in 5 year intervals recorded in Edwards & Christie (1981) models.	NIR (2018) uses a modified expo-linear growth function (Monteith, 2000) to simulate early annual growth. Bateman & Lovett (2000) fitted an S shaped curve to Edwards & Christie (1981) data	
Carbon mass of Sitka spruce (SS)	[8] Basic density 0.387 (NIR, 2012 p 123)	[9] Carbon fraction 0.5 (NIR, 2015 p 123)	
Biomass – above ground	[10] Biomas Expansion Factor (BEF) follows NIR (2018) methodology.	NIR (2015/18). A dynamic BEF is used in this analysis based on species, yield class and growth phase. Ranging from a value of 2 to 1.68 for lower YCs (14 & 16), 3 to 1.68 for YCs 18 & 20, 4 to 1.68 for the most productive YCs (22 & 24). A constant BEF of 1.68 is utilised once stand volume is equal to or greater than 200 $m^3 ha^{-1}$ Bateman & Lovett (2000).estimated functional relationships for livewood. MTV is related to total woody volume (TWV) by allowing for branchwood, roots, etc. (Corbyn et al., 1988; Matthews, 1991).	NIR (2018): Based on the model developed by Dewar and Cannell (1992), (Kilbride et al., 1999) used a static value of 1.3 for all species, age and yield classes, while the 2012 NIR uses a value of 1.64. However, since the allocation of biomass between different forest components is dependent on species, yield class and the growth phase of the forest, current estimates of sink capacity have been revised to use age and species-specific BEF values that include the below ground fraction.
[11] Productive area	85% of the area taken out of agriculture is classified as productive area due to mandatory areas of biodiversity enhancement (ABE), set-back distances for roads, rivers, houses, fencing, unplantable terrain etc., (Ryan et al., 2016).	In scaling up, NIR (2018) applies a 10% area reduction to account for open spaces.	

Biomass – below ground	[12] Ratio of below ground to above ground biomass: 0.2 Country specific ratio (NIR, 2015/2018)		
DOM	Litter [13] L_{LF} represents the transfer of carbon from the above ground pool to the litter pool. It is simulated using derived leaf/needle biomass (LB) and the foliage turnover rates (Ft) from Tobin et al. (2007): $$L_{LF} = LB \times F_t$$ The F_t rate is assumed to be 6.7 years for conifer crops and 1 year for broadleaf crops (Tobin et al., 2007). Needle biomass is calculated according to the equation defined in Annex 3.4.A.4 of NIR (2018): $$LB = 0.025 \times AB + 0.089 \times exp(-0.003 \times AB)$$ The litterfall L_{LF} is assumed to decompose at a rate of 14% per year (NIR (2018) p 222).	NIR 2018 – p197: Biomass carbon losses from the above ground biomass pool are calculated based on harvest (Ltimber), harvest residue (LHR), litter fall (LLF), above ground losses due to mortality (Lmort(AB)) and fire (Lfire): Ltimber is calculated based on the above ground biomass removed from harvest, LHR includes the harvest residue representing all stems and branches with a DBH less than 7cm and litter left on site after timber is removed LLF reflects the transfer of carbon from the AB pool to the litter pool NIR (2018): Equation from NIR (Tobin et al. 2007) (needle turn- over is 6.7 years for conifers and annually for broadleaves Calculation of matter from equation in NIR 2018 Decomposition = 14% decline/yr p222 NIR 2018 Bateman & Lovett (2000): litter is not modelled	CARBINE (Thompson & Matthews, 1989): Litter is not modelled
	Deadwood [14] Inflow is 1.6% Decomposition rate is 14% decline /year (Carbware)	NIR (2018) Mortality: Growth, harvest and mortality derived from Edwards & Christie (1981) described by Black et al. (2012). Net deadwood stock changes (□CDW) are derived from carbon inputs associated with timber extraction residue (Ltr), timber from mortality (Mtimber), dead roots from mortality (Lmort(BB)), roots from harvest (LHRroot) and carbon loss due to decomposition of the new and previously existing deadwood pool (DDW): Biomass carbon losses from the below ground biomass pool are calculated as the sum of losses due to death of roots after harvest (LHRroot), natural mortality of roots (Lmort(BB)) and root death following fire (Lfire).	

		Bateman & Lovett (2000): Assume 5 year oxidation of deadwood	
[15] Harvest losses	We assume differential harvest losses for each harvest as per Teagasc FIVE. Ist Thin – 14% loss of Merchantable Timber Volume (MTV) 2nd Thin - 12% loss of MTV Subsequent Thin - 9% Clearfell/Final harvest – 5%	NIR (2018) assumes static harvest losses of 4%	Morison et al. (2012) include HL in HWP as they may not be immediately oxidised
Wood fuel oxidation [16]	34% In the 2017 Wood Flow report, 34% of forest biomass is used as wood fuel (Knaggs & O'Driscoll, 2017).		
HWP allocations [17]	ForBES follows Pingoud & Wagner (2006) model (2006 IPCC guidelines) Assumes the same allocation of wood to the value chain from thinnings and from final harvest (scenario analysis to examine alternative scenario). Allocation of SW, WBP and PPB to HWP SW: 52%, WBP 48%. PPB: zero (no longer any paper production).	Pingoud & Wagner (2006) model (2006 IPCC guidelines)	The UK CARBINE model (Thompson & Matthews, 1989) allocates MTV to HWP pool (long-lived sawnwood, short-lived sawnwood, particleboard and paper), with remainder to waste.
Saw-milling losses [18]	SW: 50% WBP: 41%		
IPCC conversion factor [19]	Conversion factor from C to CO_2: 3.67	3.67	A cost of USD 1 per tonne of carbon dioxide is equivalent to a cost of USD 3.67 per tonne of carbon. OECD
Carbon Valuation [20]	Carbon values applied as per Irish Government shadow price of carbon for 2019, 2020, 2030, 2040. Future carbon sequestration and emissions are discounted at 5%. Scenario analysis of impact of discount rates analysed in O'Donoghue et al. (forthcoming). Discount rates: 0-7%)	NIR (2018) does not discount future carbon sequestration or emissions. Bateman & Lovett (2000) discount carbon at 5% and include scenario analysis for discount rates of 2 – 12%	
Annual Equivalised (AE) NPV per hectare [21]	$\text{AE} = \dfrac{r.NPV}{1-(1+r)^{-n}}$ Assume no agricultural income in year of planting		