End of Project Report

Title: Enhancing and assessing the impact of novel circular economy sectors in the bioeconomy

Project Reference: 18/RDD/235

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1. Summary

The purpose of this project is to assess the impact of novel circular economy sectors within the Irish biomass bioeconomy and issues in relation to scale-up. Quantifying the impact of the application of circular economy principles within the bioeconomy requires a modelling framework that incorporates the multidimensionality of the circular economy including a multisectoral approach focusing on multi-functional outputs and different spatial areas. The project makes use of the Economic, Social, Spatial and Environmental (ESSE) modelling framework to accomplish this task. The economic and environmental aspects of the model are demonstrated using the Bioeconomy Input-Output (BIO) model and its extended version incorporating life-cycle assessment (BIO-LCA). The social and spatial segments of the model are carried out using the Simulation Model of the Irish Local Economy (SMILE).

The project is divided into seven work packages:

1. Horizon Scanning

2. Value Chain Assessment

3. BIO-Economic Impact

4. BIO-LCA: Environmental Impact

5. SMILE: Spatial Impact

6. Institutional Analysis, Innovation System and Knowledge Exchange

7. Project Management

WP 1 – Horizon Scanning

The aim of this WP was to undertake an initial horizon scanning and foresight exercise relating to novel bio-economy novel technologies. A secondary analysis of potential circular bioeconomy sectors was completed using a literature review of novel technologies. The SEAI Bioenergy Supply in Ireland 2015-2035 report provided valuable guidance on current bioeconomy areas within the energy sector. The technologies chosen to be modelled were those with strong land use implications – forestry, grass, willow, and ground mounted solar PV.

A horizon scanning workshop was originally planned to take place early in the project but due to the project's late start and the onset of the Covid-19 pandemic, the workshop was delayed. In December 2021, the workshop held in the RDS as part of the RDS Climate Smart Agriculture Dialogue series. The workshop was livestreamed on YouTube and has over 4,300 views at time of writing. A report was written summarising the main discussion points raised as part of the workshop.

WP 2 – Value Chain Assessment

The objective of this work package was to map the structure of the value chains for potential circular bioeconomy sectors and undertake a qualitative study of the barriers and opportunities in relation to the establishment or expansion of the value chain. A qualitative study was undertaken to analyse the Irish government's previous 2007 Bioenergy Scheme and draw lessons for future production of bioenergy. The study found that the scheme was a significant policy but that due to low take-up, there was an underspend at the farm level. Significantly, there was low take-up across the value chain with limited market opportunities for bioenergy crops, particularly Miscanthus.In conclusion, for such a scheme to work in the future, there needs to be balanced development across the whole value chain.

WP 3 – BIO-Economic Impact & WP 4 - BIO-LCA: Environmental Impact

The objective of WP 3 was to assess the economic and employment impact of the circular bioeconomy novel technology. The objective of WP4 was to assess the environmental impact in terms of GHG emissions of circular bio-economy novel technologies using the environmentally extended input-output model BIO-LCA. As WP 3 and WP 4 are related, they are reported together. The analysis was split into two parts. The first analysed short-run returns in the bioeconomy by comparing the economic returns from alternative land uses. The second examined long-run returns by comparing the economic and environmental results of forest planting to agricultural use.

The use of grass in anaerobic digestion (AD) to generate biogas/biomethane and short rotation coppice (SRC) willow as a solid fuel to generate heat and electricity was compared with alternative land uses – agriculture and solar PV. The private (market and subsidy) and social (GHG) return from substituting renewable energy sources for the existing agricultural land use on 1 ha was considered. The results show that the market return from growing one hectare of grass silage and SRC willow was less than for all the main farming systems. When the social value of the feedstock is considered by putting a monetary value on displaced GHG emissions, producing grass silage and willow biomass can provide a better return to farmers than cattle and sheep farming. Due to the high land rental amounts available to farmers from solar companies, the market and social return is higher for using land to erect solar PV than any of the main agricultural activities. We also use input-output modelling to generate output multipliers for renewable energy technologies: biogas, biomethane, short rotation coppice (SRC) willow, and solar PV. Results from the output multiplier analysis showed that biomethane production has the highest multiplier while solar PV had the lowest.

The Bioeconomy Input-Output (BIO) model was used to examine the economic and environmental impact of policy targets for forest planting and beef production. Two scenarios were considered. Scenario A looked at actual forest planting in the years 2010 to 2020 while Scenario B assumes that forestry planting targets were met in years 2010 to 2020. In order to account for the full value of forestry versus other potential land uses, the scenarios run from 2010 to 2075 with all amounts generated discounted back to 2010 values and presented as net present values (NPVs). In Scenario B, the extra land used for afforestation is reallocated from land used for beef production. The results showed that both beef and afforestation targets could have been reached with overall decrease in GHG emissions and net emissions. Hitting the afforestation target in Scenario B by reallocating 3.7% of land used for beef production in Scenario A also results in greater economic output and higher overall value added.

WP #5 – SMILE: Spatial Impact

This WP aimed to assess the spatial impact of the selected circular bioeconomy novel technologies using the Spatial Model of the Irish Local Economy (SMILE). This involved updating the SMILE model with the latest available data and then incorporating the spatial structure of the value chains modelled in WP 2 and 3. Within SMILE, both the Farm model and the Household model were updated to the most recent years. The commuting part of population model has been calibrated to the 2016 POWSCAR data.

The forestry circular economy has been added to the SMILE model, allowing for spatial distribution for and analysis of economic, environmental and carbon trade-offs. Economic analysis comparing private economic return to afforestation with existing agricultural land use at a townland level shows that areas with lower agricultural productivity have a greater share of farms with a higher return from forestry than agriculture compares with areas with higher productivity, especially the western seaboard and upland areas. However, environmental constraints are significant. 23.7% of agricultural land is environmentally unsuitable for afforestation, with just under half of the environmentally unsuitable land potentially having higher economic returns from forestry than current agriculture. Case study sectors have been added to the SMILE model and analysis will be completed in the near future. The aim of this analysis is to spatially compare returns from renewable energy case studies with returns from agriculture. For grass and SRC willow, soil quality will be used to spatially compare returns with current farming systems.

WP #6 – Institutional Analysis, Innovation System and Knowledge Exchange

The purpose of this WP was to develop an Innovation System from the perspective of delivering circular economy based novel technologies. An Innovation System analysis of the forest circular bioeconomy in Ireland has been carried out by Kilcline et al. (2021), identifying barriers, issues and trade-offs within the forest Innovation System for actors within the system. One trade-off is related to the timing of forest harvest and whether to maximise value for farmers, processors or carbon sequestration. The results show that the optimum time to harvest for carbon is over double that for farmers and processors (98 years vs. 41 years). This poses many issues for policymakers in terms of how to develop a forest sector that satisfies all actors within the system.

2. Outputs from Project

Refereed Publications in International Journals

Geoghegan, C., & O'Donoghue, C. (2022). An analysis of the social and private return to land use change from agriculture to renewable energy production in Ireland. *Journal of Cleaner Production*, *385*, 135698.

Refereed Publications Under Review in International Journals

O'Donoghue, C., Geoghegan, C., Ryan, M, & Kilcline, K. (2022). Incorporating multiple cohorts in a forestry input-output model.

O'Donoghue, C., O'Fatharta, E., Geoghegan, C., & Ryan, M. (2022). Farmland afforestation: Forest optimal rotation ages across discrete optimisation objectives.

O'Donoghue, C., Ryan, M., Styles, D., Lanigan, G., Duffy, C., & Kilcline, K. (2023). Distributional analysis of the social and private return to farm afforestation, accounting for the cost of carbon.

Book Chapters

O'Donoghue, C. (2019). Renewable energy production. In C. O'Donoghue (Ed.), *Unlocking rural potential* (pp. 91-93). Ballintemple: Oak Tree Press.

Media

The RDS. (2021, December 3). *RDS climate smart agriculture series – farm level renewable energy* [Video fie]. Retrieved from https://www.youtube.com/watch?v=yJ9nw0nEbZI

O'Brien, A. (2021, December 9). Anaerobic digestion: Policy and financial support needed. *Agriland*. Retrieved from https://www.agriland.ie/farming-news/anaerobic-digestion-policy-and-financial-support-needed/

An Analysis of the Social and Private Return to Land Use Change from Agriculture to Renewable Energy Production in Ireland

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Abstract

Increased use of renewable energy is a major part of Ireland's plans to reduce greenhouse gas (GHG) emissions, lessen reliance on fossil fuels, and enable sustainable economic growth. Two forms of renewable energy interesting Irish policymakers are bioenergy and solar PV. To scale up these technologies, land will be required to grow bioenergy feedstocks and erect ground mount solar PV arrays. A micro, farm-based approach is utilised to generate private returns from renewable energy production, and agriculture, adjusted for the cost of GHGs. This considers both the cost to society of increasing GHG emissions and the benefit of reducing emissions. This paper compares income that could be derived from renewable energy production with average agricultural income from the main farm systems in Ireland. The private (market and subsidy) and social (GHG) return from substituting renewable energy sources for the existing agricultural land use on 1 ha is considered. We also use input-output modelling to generate output multipliers for renewable energy technologies: biogas, biomethane, short rotation coppice (SRC) willow, and solar PV. The results show that the market return from growing 1 ha of grass silage and SRC willow is less than for all the main farming systems. When the social value of the feedstock is considered by putting a monetary value on displaced GHG emissions, producing grass silage and willow biomass can provide a better return to farmers than cattle and sheep farming. Due to the high land rental amounts available to farmers from solar companies, the market and social return is higher for using land to erect solar PV than any of the main agricultural activities. Results from the output multiplier analysis show that biomethane production has the highest multiplier while solar PV has the lowest.

Introduction

The Renewable Energy Directive (RED) requires the European Union to fulfil at least 32% of its total energy needs with renewable energy by 2030, building on a 20% target realised in 2020 (European Parliament and Council, 2018, 2009). Despite an increased amended target being considered by the EU, many countries are having difficulties achieving their targeted renewable energy shares (European Commission, 2022; Eurostat, 2020). As a result, attention is being focused on how countries can sustainably increase their usage of renewable energy.

One area of interest in the field of renewable energy is bioenergy (Malico et al., 2019; European Commission, 2016). Bioenergy can be produced from a range of biomass feedstocks, each with its own resource availability, production costs, and energy conversion methods (SEAI, 2016). Biomass feedstocks are generally, though not exclusively, derived from land-based resources including forestry, perennial crops, annual crops, and agricultural wastes and by-products.

Currently, Ireland is failing to reach its declared targets in the area of renewable energy with the overall share of renewable energy in 2020 being 13.5%, compared to the target of 16% (SEAI, 2021). In 2020, wind accounted for 59% of total renewable energy consumed, followed by biomass and renewable wastes with 19% and liquid biofuels accounting for 11% (SEAI, 2021). Solar contributed 1% of consumed renewable energy. The Climate Action Plan 2021 pledges to increase renewable energy production from electricity from 29% in 2020 to up to 80% by 2030 and double biomass supply as a fossil fuel substitute (Government of Ireland, 2021). Achieving these ambitions will require an expansion of available renewable energy supplies, most likely requiring an increase in land dedicated to producing these resources.

This paper takes a value chain approach to look at the multi-dimensional impact of the land use decision regarding renewable energy. The value chain framework can cut across the multiple economic sectors involved in assessing the land use decision but can also, using input-output (IO) analysis, quantify how the land use decision affects actors across the value chain. Economically, the land use decision can be assessed at the individual farm level but also at a macro level through the generation of output multipliers. Additionally, the environmental impact can be examined through comparing GHG emissions associated with different land uses across the value chain and deriving a social value for the land use based on differential carbon pricing.

Using Ireland as a case study, a micro, farm-based approach is utilised to generate the private returns from renewable energy production, adjusted for the cost of GHGs. This considers both the cost to society of increasing GHG emissions and the benefit of reducing emissions (Tol, 2018). Two bioenergy feedstocks, grass silage and short rotation coppice (SRC) willow, as well as an alternative renewable energy production land use, ground mount solar photovoltaic (PV), are assessed. The Sustainable Energy Authority of Ireland (SEAI) have estimated that using grassland in excess of livestock requirements, 127,000 ha of grassland could be available for bioenergy production by 2025, rising to 171,000 ha by 2035[1] (SEAI, 2016). The same study estimates a potential 203,000 ha of land could be available for bioenergy crops like SRC willow by 2035[2]. This estimate is based on forecasts of potentially available land, the overall limit on conversion of pasture land imposed by the CAP, and giving priority to additional land for annual crops (SEAI, 2016).

[1] This is estimated to produce about 349 ktoe (14.6 PJ), increasing to 469 ktoe (19.6 PJ) by 2035.
[2] Potentially producing 1,167 ktoe (48.9 PJ).

Previous studies have utilised IO analysis in the context of analysing the production of biomass. Bruckner et al. (2019) develop a multi-region agriculture biomass IO model (FABIO) that traces biomass flows across global supply chains. De la Rúa et al. (2016) analyse miscanthus biomass production in France using a multi-region IO model, estimating direct, indirect, and induced impacts from the growing and processing of the plant. Nishiguchi and Tabata (2016) assess the use of unutilised woody biomass for energy generation in Japan, using IO analysis to calculate the economic, employment and GHG impacts associated with utilising the wood for energy.

The production of land-derived renewable energy feedstocks must consider not just the direct environmental, economic, and spatial advantages and disadvantages of the land use decision but also the upstream and downstream impacts of the decision across the affected value chains. Previous studies examining land use change in relation to renewable energy have looked at individual aspects of the land use decision such as the effect on GHG emissions (Clarke et al., 2019; Czyrnek-Deletre et al., 2016), the potential resource availability (McEniry et al., 2013; Smyth et al., 2009) or the feedstock's role in the economic viability of renewable energy generation (O'Shea et al., 2017).

Given the lack of solar PV utilisation in Ireland, there is a lack of literature surrounding the application of the technology in the country. Most focus has been on the economic viability of installing the technology from a residential (Murphy & McDonnell, 2017; Li et al., 2011) and investor viewpoint (Assereto & Byrne, 2021; Ryan et al., 2016). More attention has been paid internationally to aspects of the land use perspective including social acceptance (Pascaris et al., 2021), spatial socioeconomic context (Delfanti et al., 2016), and shared land use with agriculture (Dinesh & Pearce, 2016).

This paper contributes to the literature by modelling multiple segments of the renewable energy value chain from both an economic and environmental perspective. With growing emphasis on expanding renewable energy production to address climate change and increasing energy security, policymakers must consider how best to achieve these goals in a holistic way. The value chain concept allows us to assess the suitability of various renewable energy technologies in such a holistic manner (Singh et al., 2021; Matsuo & Schmidt, 2019). Given the importance of the land use decision in producing renewable energy feedstocks, this paper also contributes by assessing multiple technologies against the current land use (agriculture), providing insight into which land uses may be optimal for landowners and should be incentivised by policymakers.

Within the context of the Sustainable Development Goals (SDGs), this paper addresses SDG 7 "Ensure access to affordable, reliable, sustainable and modern energy for all" and SDG 13 "Take urgent action to combat climate change and its impacts by regulating emissions and promoting developments in renewable energy". We examine how Ireland can increase its share of renewable energy production in the land use context, as well as build knowledge and capacity to help Ireland meet climate change through alternative land use scenarios.

This paper examines Ireland as a case study country which has numerous potential domestic bioenergy feedstocks, large agricultural resources, and a nascent solar industry (SEAI, 2016). In addition, there is growing policy interest in designing incentives to mitigate agricultural GHG emissions, which account for one third of national emissions. The theoretical framework is presented in the next sections of the paper. The methodology utilised in the estimations is explained in section 3, with results described in section 4. Discussion and conclusions are given in section 5.

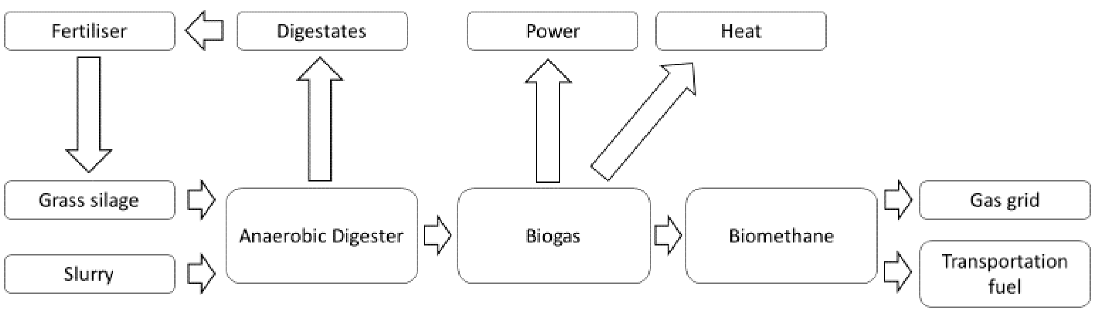
Theoretical Framework: Private and Social Returns from Agriculture and Renewable Energy

There is a wide literature concerning the decision of farmers to use their land for producing renewable energy. Several studies have looked at the economic aspects of energy crops, comparing annual gross margins of conventional crops with an equivalent annualised value for perennial energy crops (Lantz et al., 2014; Gasol et al., 2010; Styles et al., 2008). Other studies have examined non-economic factors affecting the adoption of bioenergy crops including cultural factors, awareness and educational barriers, long-term commitment of land, and perceived risks (Glithero et al., 2013; Bocquého & Jacquet, 2010). The use of a carbon value to subsidise the substitution of agriculture for woody biomass has been studied by O'Donoghue and Ryan (2020).

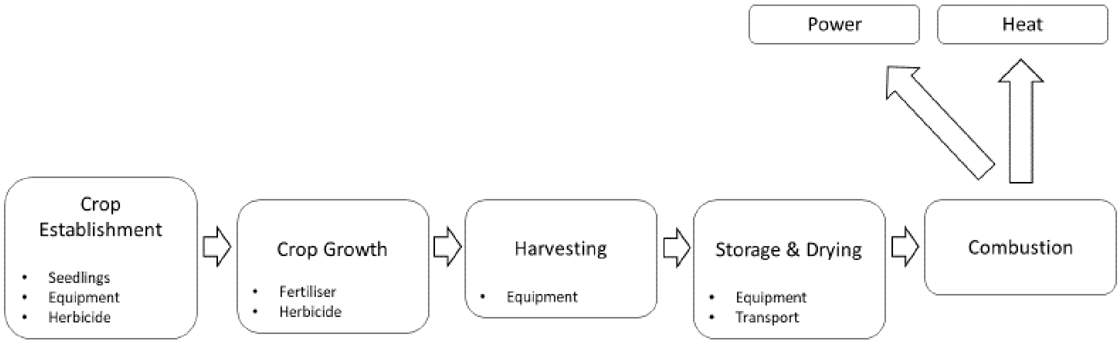
As demand for renewable energy has risen in recent years, the suitability of erecting solar PV on agricultural land has been increasingly studied. Multiple studies have made comparisons between agricultural and potential renewable energy revenues from solar PV including Farja and Maciejczak (2021), Kim et al. (2020), and Gazheli and Di Corato (2013). Attention has also been paid to the potential of agrivoltaic systems, which seek to combine agriculture with renewable energy production from solar PV arrays (Weselek et al., 2019; Dinesh & Pearce, 2016)

We use the concept of the value chain to examine the multifaceted effect of producing renewable energy on Irish farmland. The value chain makes it possible to examine the individual elements of an economic sector but also measure the impact of a sector in its entirety (Kaplinsky & Morris, 2000). With the renewable energy value chain being made up of many components from farmers to processors to energy generation, a value chain approach is appropriate as it examines not just the value chain as a whole but also how each individual component is affected by the production of feedstocks (Yang et al., 2014; Dautzenberg & Hanf, 2008). Value chains for biogas/biomethane production from AD, SRC willow biomass production, and solar PV are shown in Figures 1, 2, and 3. Input-output analysis will be used to quantify the impact of renewable energy production and compare with alternative land uses.

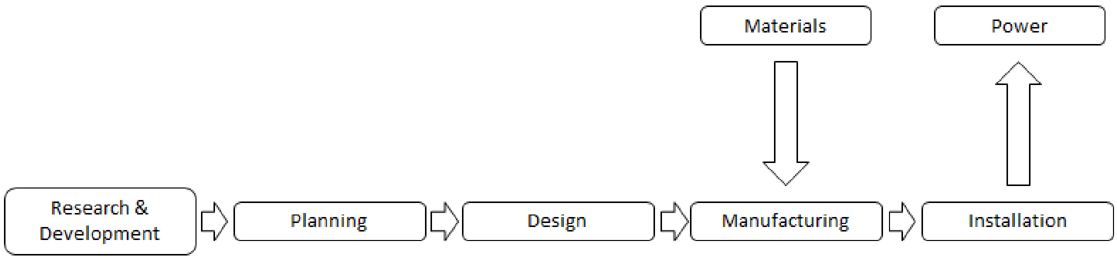
Biogas/Biomethane from Anaerobic Digestion Value Chain



SRC Willow Value Chain



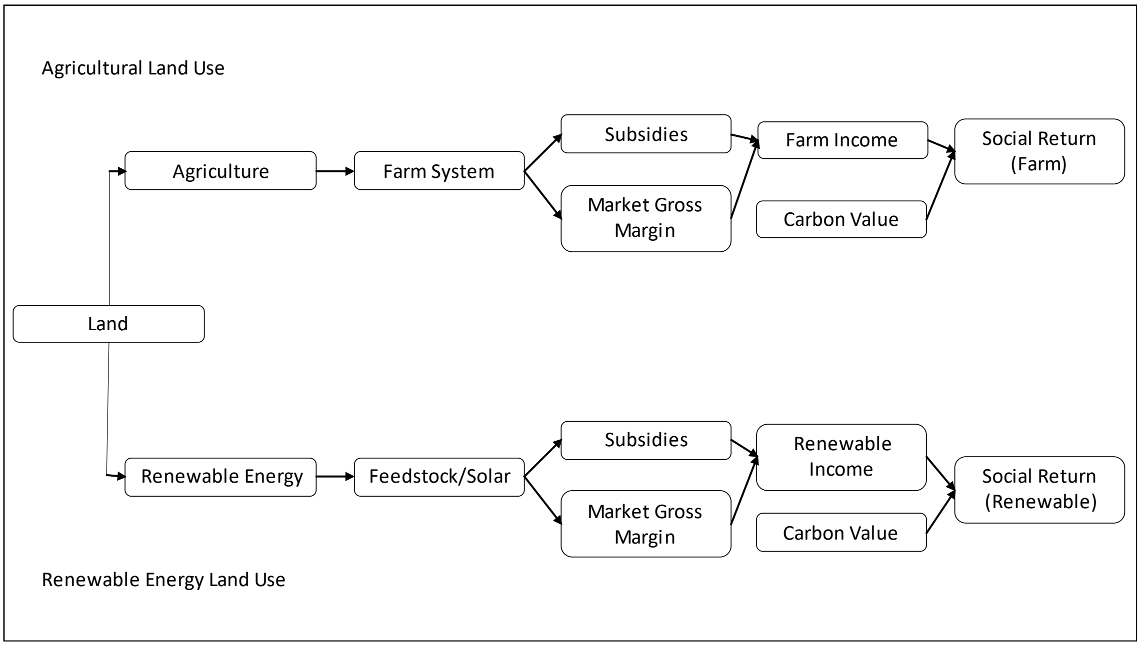
Solar PV Value Chain



We will first look at one segment of the value chain, the use of agricultural land to produce renewable energy. This paper examines the land use decision of substituting renewable energy production for an agricultural enterprise on a per hectare basis. To compare the economic returns from agriculture and renewable energy, we will determine actual incomes at farm level and also the counterfactual income streams that could be derived from using one hectare for renewable energy production. The theoretical framework of the land use decision is described in Figure 4.

The income streams represent the private returns to the specific land uses. To account for the social returns to each land use, the private return to changing land use is expanded to incorporate the social value of carbon (SCC) (Nordhaus, 2017). Benefit-cost analyses to inform policy should incorporate estimates of the marginal value of changes in emissions i.e. the value/cost of global damage caused by the impact of an additional tonne of CO_2 emitted (Smith & Braathen, 2015).

Theoretical Framework of Land Use Decision



The elements of agricultural and renewable energy income to be modelled include market and subsidy income, carbon sequestration, emissions displaced from agriculture, and the future value of carbon. To compare returns from grass silage/SRC willow/solar PV and other land uses, we must define how income is generated for the farmer from the renewable energy sources. The private income ($V_{private}$) generated by the farmer includes market income from feedstock sales (grass/SRC willow), land rental (solar), and farm subsidies (grass/SRC willow), minus the annual agricultural income foregone as a result of the land use change:

$$V_{private} = \sum RenewableEnergyNetMarketIncome\ ha^{-1} + \sum FarmSubsidy\ ha^{-1} - \left(\sum FarmNetMarketIncome\ ha^{-1} + \sum FarmSubsidy\ ha^{-1}\right) \quad (1)$$

The value of carbon sequestered as a result of the land use change and displaced GHG emissions from agriculture are added to the private returns to generate social value (V_{social}):

$$V_{social} = \sum RenewableEnergyIncome\ ha^{-1} + \sum FarmSubsidy\ ha^{-1} - \left(\sum FarmNetMarketIncome\ ha^{-1} + \sum FarmSubsidy\ ha^{-1}\right) + \left(\sum Value\ of\ net\ Carbon\ Sequestered\ ha^{-1}\right) + \left(\sum Value\ of\ GHG\ Emitted\ by\ Displaced\ Agriculture\ ha^{-1}\right) \quad (2)$$

To calculate the social value of carbon, a carbon subsidy is implemented (tonnes (t) of carbon equivalent x carbon value (v)) for the renewable energy sources. For comparison, we calculate the private return in terms of annual equivalised net present value (AE NPV). Thus, the carbon subsidy is calculated similarly:

$$AE(vCO_2e\ ha^{-1}) = AE(v.tCO_2e\ ha^{-1}) = v.AE(tCO_2e\ ha^{-1}) \quad (3)$$

Input-output

Input-output (IO) analysis is used to quantify how the land use decision affects actors across the value chain. Economically, the land use decision can be assessed at the individual farm level but also at a macro level through the generation of output multipliers. IO models statistically describe the interdependencies between different sectors of the economy by analysing the relative relationship between the flow of production inputs and the resultant flow/destination of produced outputs in an economy. The model consists of a system of linear equations, each of which describes the distribution of an industry's product through the economy (Miller & Blair, 2009).

Inputs to an industry's production cycle come from imports, labour, and capital. However, firms also use outputs of other firms, intermediate consumption, as inputs to their production. Outputs are designated to exports and final demand represented by households, governments, and non-profit organisations. Such a framework is useful in analysing knock-on effects of changes in demand, output, employment, gross value added (GVA) and household income (Grealis & O'Donoghue, 2015). The matrix algebra which forms the basis of IO modelling is briefly summarised as follows:

$$X = Y + Ax \tag{4}$$

Where:

X = a column vector of output from each sector of the economy

Y = a column vector of final demand

$A = a_{ij} = \frac{z_{ij}}{x_{ij}}$ = a matrix of input coefficients indicating how many units of inputs from sectors i-j are required to produce one additional unit of output from sectors i-j

z_{ij} = intermediate demand for inputs between sector i and the supply sector j

x_{ij} = total output for sector i

The IO matrix can be reformulated to give the following expression for output for each sector in terms of final demand.

$$X = (I - A)^{-1}Y \tag{5}$$

Where:

I is the identity matrix;

$(I - A)^{-1}$ is defined as the Leontief inverse matrix.

The Leontief inverse matrix enables an estimation of individual sectoral multipliers, capturing both the direct and indirect macroeconomic effects of potential changes in exogenous demand. The IO model operates under the assumptions of constant returns to scale, constant factor ratio of employment and perfectly elastic factor inputs (Miller & Blair, 2009).

Methodology and Data

Private returns

To derive private returns, the revenues and costs for agriculture and renewable energy must be calculated. We assume that renewable energy will only be produced on a small percentage of the overall farm so farmers will continue to incur agricultural overhead costs after planting. Therefore, feedstock and agricultural overhead costs cancel each other out and a variable costs and income approach to calculating farm income (market gross margin) is adopted. As such, the private returns are defined as the net value of the simulated renewable energy income stream for each farm less the before-tax agricultural market gross margin and direct payments using Teagasc NFS micro-data from 2015[3]. Average yearly per hectare gross margin incomes for the main five Irish agricultural systems are modelled: dairy, cattle rearing, cattle other[4], sheep and tillage.

For grass, it is assumed that all grass grown in a specific year is cut for silage and sold to an AD plant to be processed into biogas. For SRC willow, it is assumed that harvested SRC willow is chipped and sold at the farm-gate biomass price. For solar, land is leased out to a solar energy company who erect a solar PV array on the land and sell the resulting energy to the national grid. The farmer receives per hectare farm subsidies which continue under the new land use for grass and SRC willow but not for solar[5]

For SRC willow, financial and production cycle calculations are based on data from Styles et al. (2008) and Caslin et al. (2015). The willow is assumed to be chipped at harvest, stored and dried on-farm and sold at the farm gate as per scenario C1b in Styles et al. (2008). Direct subsidy payments are absent from the Styles et al. (2008) analysis so are added here to ensure a fairer comparison with agricultural land uses. Subsidy payments are assumed to be the same per hectare as they would be on the rest of the modelled farm system.

For grass, data regarding silage production and cost is derived from McEniry et al. (2011). We assume that grass is cut twice per year and made into silage with a total yield of 41.1 tonnes (9.1 tDM/ha), following scenario 1 from McEniry et al. (2011). Departing from McEniry et al's analysis, we assume the land is owned rather than rented and that the farmer receives subsidies on the land based on their primary farm system. Silage is sold for €30/t, the average estimated silage price in 2015 (Reidy, 2015).

For solar PV, we assume the farmer rents out the land to a solar company so all farmer income from solar is from land rental and all costs related to the solar array are borne by the solar company. The land rental price is €2,400/ha, which is the standard price for land rental for solar purposes (Joint Committee on Agriculture, Food and the Marine, 2022; O'Sullivan, 2022; Teagasc, 2020). It is assumed that no agricultural activity takes place on the land following the installation of the solar panels.

Social returns

Although carbon is also sequestered in grassland, the main advantage replacing livestock agriculture with renewable energy production is the emissions that fail to be generated by the livestock. Although indirect land use change must be considered, this paper looks at the case of a marginal hectare of land, so no current agricultural production is being displaced.

The estimation of GHG emissions from agriculture as defined in by the IPCC (IPCC, 2003) incorporates a number of dimensions:

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[3] Data from 2015 is used so as to match the most recent national input-output tables which are from that year.

[4] Cattle other refers to cattle finishing farms where cattle are fattened up in preparation for slaughter.

[5] We assume the installation of solar PV makes the land no longer fit for agricultural purposes and therefore ineligible for agricultural subsidies.

- methane (CH_4) emissions from enteric fermentation and manure management;
- total direct plus indirect nitrous oxide (N_2O) emissions from various sources for both livestock (manure management, organic and mineral fertiliser application, dung/urine deposition) and tillage (organic and mineral fertiliser application, residue incorporation) activities;
- carbon dioxide (CO_2) from fuel and other uses.

The inventory-based approach to estimating GHG emissions from various farm sub-components involves multiplying an activity (e.g. animal numbers) by an emission factor (e.g. the amount of methane per animal). Animal numbers and age categories on individual farms are used to derive livestock (equivalent) units, where a dairy cow represents one livestock unit per hectare ($\mathrm{LU}ha^{-1}$). The actual agricultural emissions are determined by applying an IPCC inventory model (emission factors x activity data) for each component of the agricultural systems at individual farm level.

The social returns from each land use incorporate the cost of GHG emissions into the previously described private returns. Annual per hectare agricultural emissions for each farm system are taken from Ryan and O'Donoghue (2020). Data for emissions from SRC willow cultivation are sourced from Styles and Jones (2008). For grass production, GHG emissions based on the level of fertiliser application in McEniry et al.[6] are calculated using emissions factors contained in Duffy et al. (2020). It is assumed all fertiliser is slurry derived from animals on the farm and no inorganic fertiliser is used.

The annual sum of carbon sequestered and the avoided agricultural emissions arising from the land use decision provide the average tCO_2e per hectare for each farm system and renewable energy source. In calculating the social return, a carbon subsidy is added based on the avoided agricultural emissions and the quantity of carbon sequestered. The carbon values utilised are the Irish government shadow price of carbon for non-ETS sectors for different years: 2020 (€32), 2030 (€100), 2040 (€163) (DPER, 2019).

Input-output

Disaggregating the chosen Irish renewable energy sectors from the rest of the economy requires detailed information about the sources of inputs and destination of outputs. However, where information on these sectors is available, it is recorded in much broader terms, generally consisting of basic (and non-monetary) information such as gross energy output and biomass produced. Additionally, the renewable energy sector as a whole is at a nascent stage in Ireland, especially within the case study areas of SRC willow, biogas/biomethane and solar PV. As there is limited or no availability of survey information on inputs in the case study sectors, intermediate consumption shares for the chosen renewable energy sectors are apportioned on the basis of case studies from the literature and judgement from industry experts.

For the production of grass silage, data from the Bio-Economy Input-Output (BIO) model is utilised (Grealis & O'Donoghue, 2015). The BIO model disaggregates the Agriculture, Forestry and Fisheries sector of the 2015 Irish national IO tables into a number of sub-sectors. Silage production is one of the disaggregated sub-sectors and its associated cost structure is used in this paper's IO model.

[6] N = 225 kg/ha, P = 30 kg/ha, K = 205 kg/ha

The structure of the biogas and biomethane sectors is constructed using relevant information from the literature and expert opinion. Input shares for the production of biogas and biomethane are derived from Balussou et al. (2012). Expenditure from the biogas and biomethane sectors on feedstocks, plant investment, transportation, maintenance, energy, processing, personnel, and other inputs are assigned to their relevant NACE code source sectors in the IO table, from which new columns are created for biogas and biomethane production. As the bioenergy plants described by Balussou et al. use maize rather than grass silage, data related to the grass silage volume and prices in the Irish context is sourced from SEAI (2017b). We assume that all inputs are sourced domestically with no inputs being imported. New rows detailing the destination of outputs from biogas and biomethane are created. Outputs are assigned to Irish IO sectors on a pro rata basis using data from SEAI energy balances in 2015 (SEAI, 2020).

The input structure of the SRC willow sub-sector is derived using data from Styles et al. (2008) and Caslin et al. (2015). Based on the literature, we assume that inputs relating to establishment of the willow crop, harvesting and drying/storage of the harvested crop is carried out by contractors, so this share of input is assigned to the construction sector. All other inputs are assigned to the relevant NACE code sector.

For the solar PV sector, we look at the impact of ground mount solar PV rather than rooftop solar PV. As a result, the solar PV sector reflects the structure of the ground mount solar PV sector rather than the solar PV sector as a whole. The input structure of the solar PV sector is based on information from SEAI (2017a) and KPMG (2015). The vast majority of inputs consist of costs related to the solar panels themselves, all of which are assumed to be imported from abroad. All other inputs are assigned to the relevant NACE code sector. All output from the solar PV sector is assumed to go to the electricity sector.

Results

The private and social return to using a hectare of land for agriculture and renewable energy is presented in Tables 1 to 3 an. In Table 1, the returns from agriculture and grass silage are shown. Dairy farming has the highest per hectare gross margin, followed by tillage, cattle other, cattle rearing and sheep. The market gross margin from the production of silage is €185 per hectare. This is combined with the subsidy payment associated with the land to achieve the gross margin for silage production. For the average farm, this is €460 but that amount is slightly higher or lower depending on the subsidy payment associated with the farm system.

As shown in Figure 5, the private return to silage production is negative when compared with average per hectare income from all farm systems. The highest differential is with dairy with a €1,520 gap between the two land uses. The smallest difference comes from sheep farming with a €248 differential. The social return incorporates the displaced agricultural emissions and carbon sequestration into the value of the feedstock. When the displaced emissions are valued at €32 per tonne of CO_2e, returns for all agricultural systems are still greater than for silage production. When the value of emissions is increased to €100 per tonne, silage production delivers a higher gross margin than for cattle rearing, cattle other and sheep systems. Increasing the emissions value to €163 per tonne increases the gross margin for silage production over the cattle and sheep systems but still remains less than that of dairy and tillage.

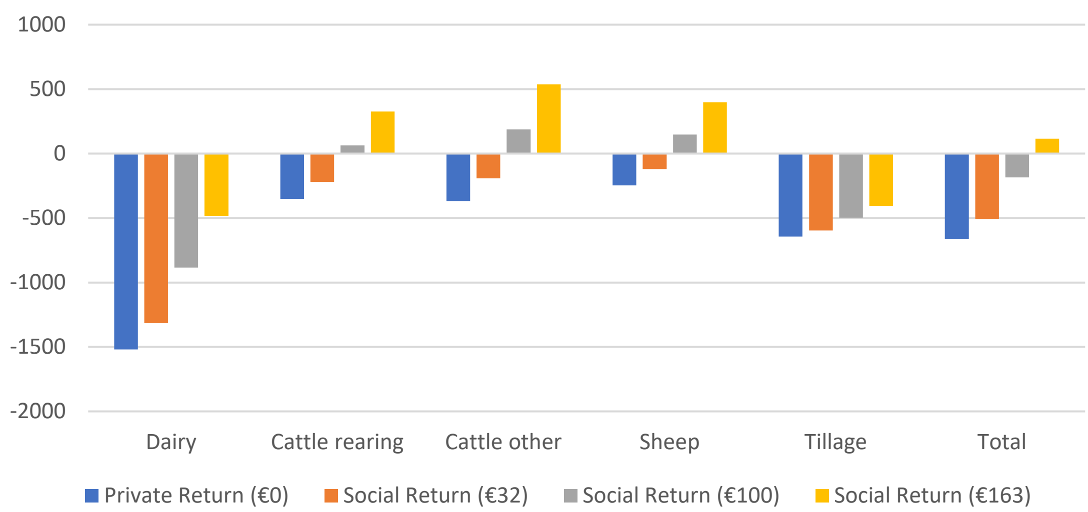
Private and Social Return to Agriculture and Silage in 2015

System	Agriculture	Grass		Agriculture	Grass	Private Return	Social Return		
	A	B	C	D	E	(B+C) - A	(B+C) – A + (D-E)*M		
M	€	€	€	tCO_2e	tCO_2e	0	€32	€100	€163

Dairy	1982	185	277	-7.4	-1.04	-€1520	-€1316	-€884	-€483
Cattle rearing	776	185	239	-5.2	-1.04	-€352	-€219	€64	€326
Cattle other	864	185	310	-6.6	-1.04	-€369	-€191	€187	€537
Sheep	672	185	239	-5.0	-1.04	-€248	-€121	€148	€397
Tillage	1161	185	332	-2.5	-1.04	-€644	-€597	-€498	-€406
Total	1120	185	275	-5.8	-1.04	-€660	-€508	-€184	€116

Note:
A – Market Gross Margin per hectare (Agriculture)
B – Market Gross Margin per hectare (Feedstock)
C – Subsidies per hectare (Feedstock)
D – tCO_2e – Agriculture
E – tCO_2e – Feedstock
M – Monetary carbon value (€ tCO_2e^{-1})

Private and Social Return to Agriculture and Silage in 2015



The private and social return to agricultural and SRC willow production is shown in Table 2. The market gross margin for willow production is higher than for silage with a market gross margin of €245/ha being found for willow versus €185 for silage. However, willow production does emit more GHG emissions per hectare than silage production, so the social return associated with displaced emissions is lower for willow. As with silage production, the private return to willow is lower than for all the agricultural systems (see Figure 6). The social return is greater for willow than for cattle and sheep farming at the €100 and €163 per tonne emissions values but is lower than the social return seen for silage due to the higher relative GHG emissions associated with willow production.

Private and Social Return to Agriculture and SRC Willow in 2015

System	Agriculture	Willow		Agriculture	Willow	Private Return	Social Return		
	A	B	C	D	E	(B+C) - A	(B+C) – A + (D-E)*M		
M	€	€	€	tCO_2e	tCO_2e	0	€32	€100	€163
Dairy	1982	245	277	-7.4	-1.34	-€1460	-€1266	-€854	-€472
Cattle rearing	776	245	239	-5.2	-1.34	-€292	-€168	€94	€337
Cattle other	864	245	310	-6.6	-1.34	-€309	-€141	€217	€548
Sheep	672	245	239	-5.0	-1.34	-€188	-€71	€178	€409
Tillage	1161	245	332	-2.5	-1.34	-€584	-€547	-€468	-€395
Total	1120	245	275	-5.8	-1.34	-€600	-€457	-€154	€127

Private and Social Return to Agriculture and SRC Willow in 2015

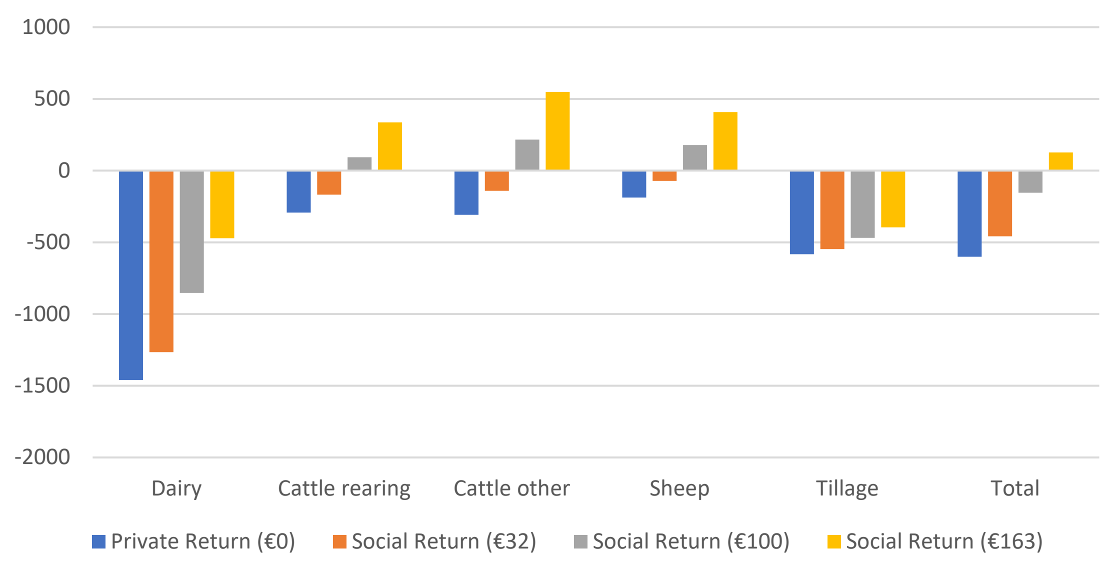
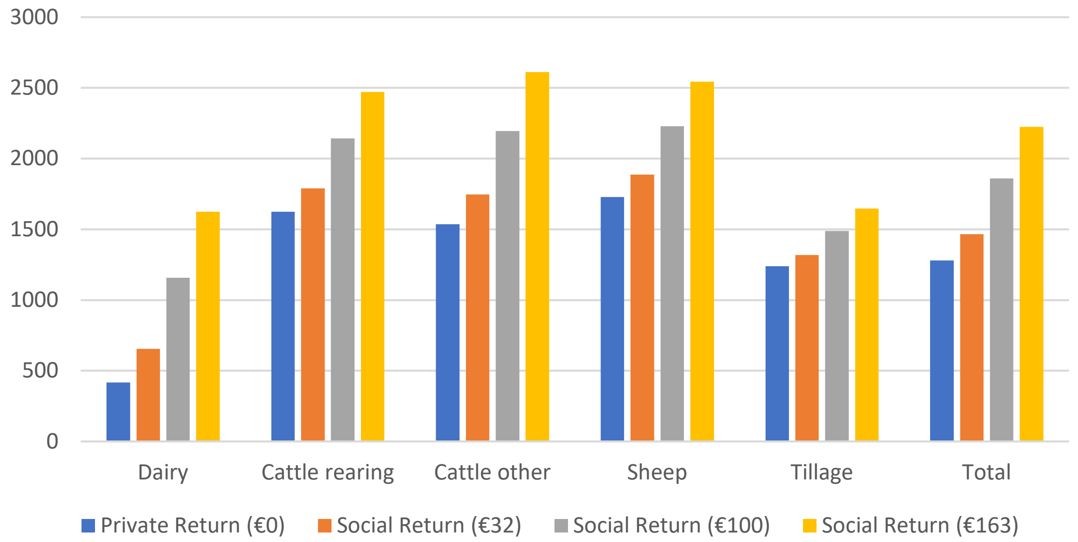


Table 3 details the return to using land for agricultural production and ground mount solar PV. The entire market gross margin accruing to the farmer from solar comes from the rental of land for the solar panels to be erected on, with land rental prices for solar currently far higher than land rental prices for normal agricultural purposes. Additionally, land being used to erect solar panels no longer has animals grazing upon it or fertiliser being spread so GHG emissions from the land drop to zero. The land is no longer being used for agricultural purposes so subsidy payments on the land can no longer be claimed. As seen in Figure 7, given the high rental price of land for solar PV, the private return is higher for solar than for any of the farming systems, with returns to solar only increasing when emissions values are factored in.

Private and Social Return to Agriculture and Solar in 2015

System	Agriculture	Solar		Agriculture	Solar	Private Return	Social Return		
	A	B	C	D	E	(B+C) - A	(B+C) – A + (D-E)*M		
M	€	€	€	tCO_2e	tCO_2e	0	€32	€100	€163
Dairy	1982	2400	0	-7.4	0	€418	€655	€1,158	€1,624
Cattle rearing	776	2400	0	-5.2	0	€1624	€1,790	€2,144	€2,472
Cattle other	864	2400	0	-6.6	0	€1536	€1,747	€2,196	€2,612
Sheep	672	2400	0	-5.0	0	€1728	€1,888	€2,228	€2,543
Tillage	1161	2400	0	-2.5	0	€1239	€1,319	€1,489	€1,647
Total	1120	2400	0	-5.8	0	€1280	€1,466	€1,860	€2,225

Private and Social Return to Agriculture and Solar in 2015



Multiplier Analysis

Following the addition of new columns and rows representing the biogas/biomethane, SRC willow, and solar PV sectors to the 2015 IO tables, output multipliers for the various sectors are calculated. The composition of the output multiplier for the biogas sector is shown in Table 4. The multiplier for the biogas sector is 2.22, mostly driven by inputs from the financial services sector, agriculture (grass silage and pig slurry) and land transport, as well as inter-sectoral activity. Inputs from the financial services sector are mainly determined by debt repayments associated with the capital expenditure costs of building biogas plants.

Composition of Biogas Multiplier – Primary Input Sectors

Sector		Sector	
Biogas	1.00	Computer programming, consulting and information services activities	0.02
Financial services activities, except insurance and pension funding	0.40	Electricity	0.02
Silage	0.31	Legal and accounting activities	0.02
Land transport	0.12	Activities auxiliary to financial services and insurance activities	0.02
Pigs	0.09	Manufacturing n.e.s.	0.02
Electricity, gas, steam, and air conditioning supply	0.05	Other	0.12
Repair and installation of machinery and equipment	0.03		
Biogas output multiplier	2.22		

Table 5 shows the structure of the output multiplier associated with the biomethane sector. The output multiplier is primarily driven by inter-industry activity, feedstock input from grass silage, the financial services sector, the electricity and gas sector and land transport. Combined with smaller inputs from other sectors, the overall output multiplier from the sector is 2.42.

Composition of Biomethane Multiplier – Primary Input Sectors

Sector		Sector	
Biomethane	1.29	Land transport	0.08
Silage	0.54	Repair and installation of machinery and equipment	0.04
Financial services activities, except insurance and pension funding	0.18	Electricity	0.02
Electricity, gas, steam, and air conditioning supply	0.16	Other	0.09
Biomethane output multiplier	2.42		

The structure of the output multiplier associated with SRC willow is shown in Table 6. Construction and electricity are the primary drivers of the sector's output multiplier with the overall multiplier for SRC willow being 2.31.

Composition of SRC Willow Multiplier – Primary Input Sectors

Sector		Sector	
SRC Willow	1.00	Insurance, reinsurance and pension funding, except compulsory social security	0.03
Construction	0.50	Financial services activities, except insurance and pension funding	0.02
Electricity	0.12	Legal and accounting activities	0.02
Electricity, gas, steam, and air conditioning supply	0.10	Advertising, other professional, scientific, technical and veterinary activities	0.02
Wholesale trade	0.06	Administrative and support service activities	0.02
Chemicals and chemical products	0.05	Crude oil	0.02
Mining, quarrying and extraction	0.05	Telecommunications	0.02
Repair and installation of machinery and equipment	0.04	Other non-metallic mineral products	0.02
Activities of membership organisations	0.03	Other	0.19
SRC willow output multiplier	2.31		

Table 7 shows the configuration of the solar PV output multiplier. Given that nearly 80% of inputs to the sector are imported, the overall multiplier for the sector is lower that for biogas/biomethane and willow at 1.29. The main sectors driving the output multiplier are repair and installation of machinery and equipment, architecture and engineering activities, and construction.

Composition of Solar Multiplier – Primary Input Sectors

Sector		Sector	
Solar	1.00	Construction	0.05
Repair and installation of machinery and equipment	0.09	Real estate activities	0.01
Architectural and engineering activities, technical testing and analysis	0.07	Other	0.05
Solar output multiplier	1.29		

Discussion and Conclusions

Renewable energy has the potential to reduce Ireland's level of GHG emissions, as well as to contribute towards meeting EU targets. The production of many renewable energy sources relies upon a sustainable supply of organic feedstocks. The production of organic feedstocks for bioenergy purposes, as well as renewable energy sources such as solar PV, relies upon the land use decision of individual landowners. For bioenergy, the land use decision takes place within the context of a biomass value chain which encompasses the growth and supply of the biomass feedstock but also its processing into energy and the subsequent usage of that energy. The structure of the value chain has macroeconomic impacts which can be measured using input-output analysis.

Regarding the land use decision, the results show that the market return from growing one hectare of bioenergy feedstocks, in this case grass silage and SRC willow, is less than for all the main farming systems in Ireland. However, when the social value of the feedstock is considered by putting a monetary value on displaced GHG emissions, producing grass silage and willow biomass can provide a better return to farmers than cattle and sheep farming. This is the case when the social value of a tonne of $CO2e$ is valued as low as €100/t. Due to the land rental amounts available to farmers from solar companies, the market and social return is higher for using land to erect solar PV than any of the main agricultural activities.

By incorporating the social cost of GHG emissions into the returns to land use, growing feedstocks for bioenergy can provide better returns for farmers than cattle or sheep farming. Currently, cattle and sheep farms make up 68% of the total number of Irish farms (CSO, 2018), with 69% of agricultural land being used by these farm systems (Geoghegan & O'Donoghue, 2018). Despite this, average incomes for cattle and sheep farms are much lower than for dairy and tillage enterprises, with direct subsidy payments generally providing over 100% of farm income (Dillon et al., 2019). Providing payment for the social costs of carbon therefore may not only help to reduce emissions from the agricultural sector but may also provide a more economically sustainable income for Irish farmers.

Nevertheless, a scheme to provide such payments would have to be designed properly by policymakers. A previous Bioenergy Scheme was established in 2007 by the Irish government to incentivise farmers to grow SRC willow and another energy crop, miscanthus. Between 2007 and 2013, over 900 hectares of willow were planted (DCENR, 2014). The scheme was ended in 2015 due to the lack of a market for the feedstocks, showing the necessity of a holistic, value chain centred approach. However, the scheme did show that farmers are willing to plant willow and have the ability to manage the crop (Lindegaard et al., 2016).

From an international perspective, the results show the need for varying levels for subsidisation for different renewable energy technologies. For governments hoping for farmers to convert land from agriculture to renewable energy production, it will be necessary to subsidise these new technologies so that they are economically competitive with agriculture. Going beyond straight income comparisons, the experience of the 2007 Bioenergy Scheme highlights the role of risk in farmers' decision making, which may require a risk premium be added to subsidy payments to convince farmers to switch from agriculture (Anand et al., 2019; Reise et al., 2012).

The output multipliers show the potential of the bioenergy value chain to contribute significantly to economic activity in Ireland. This is particularly important for rural regions that would supply the primary inputs across the value chain as these areas have struggled to maintain economic activity and employment in recent years (O'Donoghue et al., 2017). Additionally, the development of the bioenergy sector in Ireland could reduce dependence on imported fossil fuels for energy generation and help to meet EU climate change targets.

The IO analysis in this work has been entirely demand led, with a focus on evaluating new value chains, especially at the farm level. Therefore, the multipliers produced here reflect demand-induced changes upstream of the renewable energy sectors. Possible future work may look at the downstream, supply-induced effects of these sectors, using a Ghosh supply-driven model. The supply-side approach captures the effects of input factors such as capital and labour, rather than the output-centred approach of Leontief's model. Incorporating supply-side approaches into environmentally extended IO models can help illustrate the effect of input factors on environmental emissions (Sajid et al., 2021). Recent work has looked at both supply and demand chains in order to assess environmental impact (Sajid et al., 2022; 2019).

The increased use of renewable energy from sources like solar PV may raise practical issues for Irish policymakers. Solar PV power plants do not have the built-in inertia of turbine-powered plants, so they have no reserve energy that can be switched on if a generator fails or a power line is cut. This can lead to a rapid drop in frequency and destabilisation of the grid. Eirgrid, the State-owned electric power transmission operator, plan to invest heavily in building capacity to accommodate new sources of renewable energy in the coming years (Eirgrid, 2021). However, given the timescales involved, it may not be feasible to upgrade the grid so as to meet future renewable energy targets e.g. 80 per cent renewable electricity share by 2030.

Renewable energy from solar PV has yet to develop as an industry in Ireland, despite predictions in the 2010s that such development was imminent (O'Sullivan, 2022). At the time, the government was not subsidising solar PV development, especially not large-scale installations. This has changed with the introduction of the Renewable Energy Support Scheme (RESS) in 2020. From a farming landowner's point of view, issues still remain despite the potential revenues available from leasing land for solar PV. These include higher income tax payments compared to leasing out land for agriculture, potential problems related to taxes after inheritance, and the length of lease, which can be 25 years and longer (Staines Law et al., 2020).

Wind energy is currently the largest and cheapest renewable energy resource in Ireland (SEAI, 2021). While it will play an important role in the ongoing expansion of renewable energy production, it is not addressed in this paper for a number of reasons. First, this research examines the land use decision faced by farmers regarding renewable energy. Installation of wind turbines does not require a change in land use for landowners, with farmers continuing to be able to graze livestock and plant crops where turbines are located (Helldin et al., 2012). Second, future wind energy development is likely to be concentrated offshore rather than onshore (Eirgrid, 2021). Third, onshore wind turbines require quite specific locations in order to maximise output, such as an exposed site, absence of trees/houses, nearby high voltage grid connection, and being a non-protected habitat (Caslin, 2020). As a result, it is much more site specific than the renewable energy sources examined in this paper.

Increased use of land for renewable energy production has been questioned on the grounds of harming food security and having ecological spillover effects (Hastik et al., 2015; Escobar et al., 2009). McEniry et al. (2013) show that with better grassland management, alternative uses for grassland biomass such as AD would not compete with traditional dairy, beef, and sheep production systems. SEAI (2016) have also shown the prospect of grassland in excess of livestock requirements for grass silage and crops for bioenergy. The erection of solar PV arrays also does not preclude agricultural activity on the land, with the ability of sheep to graze on the land around the arrays (Dower, 2018). Fertiliser use in the production of grass and SRC willow must also be considered, due to the energy used to manufacture synthetic fertilisers and the potential for acidification and eutrophication from biological fertilisers (Beausang et al. 2021; Murphy et al., 2013

One possible limitation of this paper is that analysis of the land decision is based on currently unused land rather than land currently used for agriculture. Substituting agricultural activity for the production of renewable energy would have implications for farming intensity and overall agricultural output. Maintaining or increasing output, as is current government policy, where land is given over to producing renewable energy would require more intense production techniques on the remaining farmland. Greater intensity may require increased stocking rates, higher use of chemical fertilisers and more utilisation of animal feed. As a result, the avoidance of GHG emissions through the alternative land use may be negated by the increased intensity of production. However, such effects are beyond the scope of this paper.

Due to the nascent level of the bioenergy sector in Ireland, especially in the bioenergy value chains analysed here, this analysis relies heavily on data from the literature in order to add new rows and columns to the most recent IO tables. As a result, these sectors, were they to come to fruition, may not have an identical input structure to that used here and the resultant multipliers could be slightly different in reality. However, this analysis, based on the available data, shows that the potential for renewable energy value chains to contribute to the Irish economy is significant and demands further analysis.

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Incorporating Multiple Cohorts in a Forestry Input-Output Model[7]

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Abstract

This paper describes a forestry input-output model that disaggregates the Irish forestry sector in order to analyse the economic contribution of the various sub-sectors of the forest economy. With increased attention being given to forestry as a driver of rural development, as well as a shifting policy context, it is important to determine how forestry interacts with the overall economy. Disaggregation allows the intertemporal nature of the sector to be considered, as well as dealing with aggregation bias arising from the heterogeneous sub-sectors characteristic of forestry. The results show that the forestry sector generates a relatively small amount of value added with most the value added being produced by the contracting sector when trees are planted and harvested. The harvested wood products (HWP) sector is also disaggregated. Finally, the disaggregated input-output table is used to analyse the impact of two different land use situations related to the forestry sector.

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Introduction

The forestry sector is moving from its traditional role as a supplier of raw material to the wood products industry to an engine of the green economy (Hetemäki, 2014; Ollikainen, 2014). The multifunctional nature of the forestry sector including carbon sequestration in the fight against climate change, as a supplier of renewable fuels to replace fossil fuel energy and as an amenity for recreation and tourism are now recognised as important aspects of forestry's economic and environmental roles (Pelli et al., 2017; Gren & Carlsson, 2013; Stupak et al., 2007). However, the traditional economic role of forestry as a supplier of timber continues to be important in many countries, especially as a driver of employment and economic activity in rural areas (McEwan et al., 2020; Pelkki & Sherman, 2020).

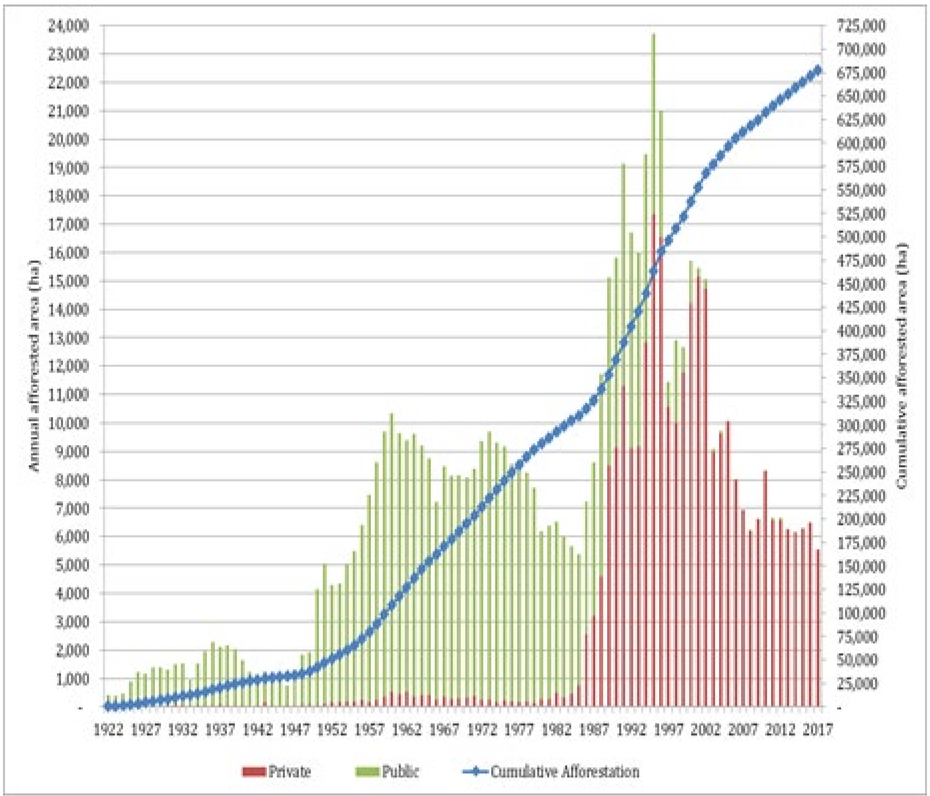
There is a need to assess the value of forestry in a way that can take account of the many roles the forestry sector fulfils for different stakeholders. One such methodology is input-output (IO) modelling which can be used to examine the flows between different industrial sectors of an economy. IO models can be extended to trace environmental loads across different economic sectors through the use of environmentally extended input-output (EEIO) models (Piñero et al., 2015). IO modelling can also be combined with life cycle assessment (LCA) to form an IO-LCA to examine product chain impacts of goods and services in an economic system (Chang et al., 2014).

A difficulty that arises when examining the contribution of a sector like forestry to a national economy using IO analysis is the long timeframe over which the forestry sector operates (Thomson & Psaltopoulos, 2005; Eiser & Roberts, 2002). IO analysis examines the flow of goods within a country's economy at a single point in time. Forestry works over multiple time frames depending on the species being cultivated, the forest's yield class, forest management decisions and government policy. Forestry is also a heterogeneous industrial sector, involving the growth and harvesting of forests on one hand and the processing and sale of wood products on the other (Bösch et al., 2015). Additional aspects of forestry such as ecosystem services, carbon sequestration and amenity values may also need to be considered (Bullock et al., 2016).

This paper disaggregates the forestry sector within the IO framework in order to address the intertemporal nature of forestry and the wide range of forest-based industries, using Ireland as a case study. The Irish national forest estate began the 20th century accounting for only 1% of total land use but increased steadily through the century. From 1949 to 1988, 322,000 hectares were planted, leading to a national forest estate of 465,000 hectares (see Figure 1). The 1980s saw the State, with financial assistance from the European Union, introduce forestry grant schemes incentivising afforestation by private landowners. Private afforestation increased rapidly with an estimated 20,000 landowners afforesting land for the first time between 1980 and 2012 (DAFM, 2014). In 2017, 50.8% of forests were in State ownership, with the rest being in private hands (DAFM, 2018). Conifers occupy of forested land while broadleaved species cover 29% with the most prevalent tree species type is the Sitka spruce (*Piceasitchensis* (Bong.) Carr.).

Current government policy is to increase the level of afforested land by 15,000 ha each year, but current planting rates come in well below this level with only 4,025 ha being planted in 2018 (DAFM, 2019). This comes at a time when the ecosystem services provided by forests are being increasingly valued by policy makers (European Commission, 2021). The explicit role of afforestation in moving towards carbon neutrality and greenhouse gas mitigation with policy makers now looking at ways to mitigate greenhouse gas production as agricultural production is increased in response to increasing global demands for food. In Ireland, the agri-food sector is responding by significantly expanding dairy and beef production (DAFM, 2015).

1. Annual State and private afforestation 1922-2017



Source: Forest Statistics Ireland 2019

In Ireland, Ní Dhubaháin et al. (2009) used an IO approach to assess the value of the forest sector to the national economy. The study showed that in 2003, the forestry sector had a direct output of €255 million while the wood products sector produced a direct output of €975 million. Output multipliers and employment multipliers were also calculated for the forestry sector as well as for three sub-sectors of the wood products sector: panel board manufacturing; sawmilling and manufacture of other wood products.

This paper builds on previous work in several ways. First, the economic impact of the forestry sector is updated using the most recently available IO table for Ireland, which is for the year 2015. Second, the IO accounts for the forestry and wood products sectors are disaggregated into a number of sub-sectors. Third, the forestry sector is disaggregated to account for multiple forest cohorts. This reflects the multiple options related to forest planting and harvesting in Ireland. Finally, two policy scenarios are modelled to illustrate the capabilities of the disaggregated sector.

Therefore, this paper aims to produce an IO model of the Irish forestry sector by disaggregating the sector from national IO accounts and using bioeconomic modelling to address the intertemporal aspects of the sector. The paper is laid out as follows. Section 2 provides a theoretical framework for the paper and a review of the literature in the area. Section 3 addresses the use of IO methodology. Section 4 presents the data used to construct the model while Sections 5 and 6 show the results of the disaggregation of the IO model. Section 7 presents a scenario analysis based on a policy example while Section 8 draws some conclusion from the paper as a whole.

Theoretical Framework and Literature Review

Input-output analysis has been a popular methodology for assessing the contribution of the forestry sector to the wider economy. IO models statistically describe the interdependencies between different sectors of the economy by analysing the relative relationship between the flow of production inputs and the resultant flow/destination of produced outputs in an economy. The model consists of a system of linear equations, each of which describes the distribution of an industry's product through the economy (Miller & Blair, 2009).

In the United States, the IMPLAN model system has been frequently used at the state and regional level to estimate the economic contribution of the forestry sector (Cox & Munn, 2001; Henderson & Munn, 2012; Brandeis & Hodges, 2015). In the Nordic countries, Rimmler et al. (2000) used IO analysis to determine the effect of alternative timber cutting scenarios in Finland on gross output, household income and employment while Johansen et al. (2017) used a regional IO model to calculate output and value added multipliers for different industries within the forestry sector. McGregor and McNicoll (1992) used a modified IO model to assess the forward and backward linkages of the forestry industry in the United Kingdom while Psaltopoulos and Thomson (1993) used regional IO tables to estimate multipliers for forestry industries in rural Scotland.

From an Irish perspective, Ní Dhubáin et al. (2009) assessed the value of forestry to the Irish economy using an IO approach. The study used the IO table for Ireland, compiled by the Central Statistics Office (CSO), which combines the forestry sector with agriculture and fisheries. The forestry sector is separated from agriculture and fisheries and is then further divided into forestry and wood product sub-sectors. The results show that in 2003, the forestry sub-sector produced a direct output of €255 million. A Type 1 output multiplier of 1.25 and a Type 2 multiplier of 1.9 were identified through the IO analysis. The wood products sub-sector in 2003 produced an output of €975 million with Type 1 output multipliers of between 1.2 and 1.31 and Type 2 multipliers of 1.61 and 1.72 being identified, depending on the type of wood products sector.

One issue cited in the literature regarding the use of IO modelling in the forestry sector is related to the "snapshot" nature of IO analysis (Roberts et al., 1999; Eiser & Roberts, 2002). A difficulty in interpreting the results of a forestry IO model arises from the fact that forestry is characterised by an extremely long production cycle, ranging from at least 35 years in the case of conifers to over 100 years for broadleaves. In contrast, IO analysis is based on a "snapshot" picture of the economy, indicating the flows that take place during a period of a single year. This may be problematic when analysing the impact of, for example, a marginal increase in the area of a particular forest type due to the fact that the marginal impact arising from harvesting the new woodland area will not be realised for at least 30 years following the planting. Additionally, unless forest planting is separated from other forest activities in the IO model, the resulting multipliers will relate to the whole production process (Roberts et al., 1999).

Another issue related to the use of IO modelling of the forestry sector is sector aggregation bias (Piñero et al., 2015). In general, IO models are used to get an approximation of how much change in total output is induced by variations in final demand. From a practitioner's perspective, aggregation bias occurs when different resolutions or combinations of sectors in IO models give dissimilar total outputs per economic sector. When sectors with different technology are collapsed together, the outcome might be different than if such aggregation had not taken place. Aggregation bias in IO models is particularly problematic for biomass extractive sectors such as agriculture, forestry, fishing and mining, which are frequently grouped together into a single sector in IO models (de Koning et al., 2015; Piñero et al., 2015). In order to deal with aggregation bias, it has been suggested that disaggregation of IO data, even if based on few real data points, is preferable to the aggregation of environmentally sensitive sectors into a single IO classification in determining accurate IO multipliers (Lenzen, 2011).

Methodology

The IO approach is comprehensively described in the research literature. It is a linear modelling framework that was first developed by Leontief in the 1930s (Hendrickson, et al. 1998). The production in an economy is described as a cyclical system in which inputs are used to produce outputs, which in turn can be used as inputs to other processing systems. These help to analyse interdependencies that exist in an economy and trace input requirements through a product life cycle (Grealis & O'Donoghue, 2015).

Inputs to the production cycle come from imports, labour and capital. However, firms also use outputs of other firms – intermediate consumption – as inputs to their production. Outputs are designated to exports and final demand represented by households, governments and non-profit organisation. Such a framework is useful in analysing the knock-on effects of changes in demand, output, employment, gross value added (GVA) and household income. In the context of policy decision-making in relation to the allocation of limited economic resources, IO analysis enables one to target investments where combined benefits are the greatest.

The IO tables can be divided into two main elements: the transaction matrix, which describes the flows from sector *i* to sector *j,* and the final demand (de la Rúa & Lechón, 2016). Intermediate goods and services are further processed by other sectors. The production cost components in an IO table are presented in columns, accounting for the resources consumed from other sectors to get a specific production in each sector, while the rows describe the distribution of one sector's production among the other sectors.

Therefore, the total output from each sector is defined by:

$$x_i = z_{i1} + z_{i2} + \cdots + z_{in} + y_i \tag{1}$$

This equation will be set for all sectors included in the IO table. We can use the matrix notation to describe the different elements of the equations system as:

$$x = \begin{bmatrix} x_1 \\ \cdots \\ x_n \end{bmatrix}; \; Z = \begin{bmatrix} z_{11} & \cdots & z_{1n} \\ \vdots & \ddots & \vdots \\ z_{n1} & \cdots & z_{nn} \end{bmatrix}; y = \begin{bmatrix} y_1 \\ \cdots \\ y_n \end{bmatrix} \tag{2}$$

where *x* is a vector that expresses the total output, *Z* is the IO matrix and *y* is the final demand vector.

The IO matrix can be expressed with the technical coefficients, which represent the ratio of input to output. This is the amount required by one sector from other sector to produce one monetary unit of output. The technical coefficients are shoen as:

$$a_{ij} = z_{i1}/x_j \tag{3}$$

The technical coefficients can be expressed as a matrix, the A matrix. We can substitute z_{ij} in Equation (1) for the technical coefficients and the total output is defined by the following matrix equation:

$$x = Ax + y \tag{4}$$

Reorganizing Equation (4), we get the following expression:

$$x = (I - A)^{-1}y \tag{5}$$

where $(I - A)^{-1}$ is the Leontief inverse matrix. This enables an estimation of individual sectoral multipliers, capturing both the direct and indirect macroeconomic effects of potential changes in exogenous demand. The IO model operates under the assumptions of constant returns to scale, constant factor ratio of employment and perfectly elastic factor inputs (Miller & Blair, 2009).

Data

In order to undertake an IO analysis of the forestry sector, it is necessary to disaggregate the forestry sector from the rest of the economy. The national IO tables, which are generated every five years by the Central Statistics Office (CSO), combine the forestry sector with the agriculture and fisheries sectors to form an aggregated Agriculture, Forestry and Fisheries sector. The national accounts do provide limited data for the forestry sector on an individual basis in the form of the sector's output, value added, and intermediate consumption. This paper uses figures generated by the Bio-Economy Input Output Model (BIO). BIO uses numerous datasets to disaggregate bioeconomic sectors within the Irish economy including agriculture (O'Donoghue et al., 2019) and aquaculture (Grealis et al., 2017). Further information on the creation of BIO is provided in Grealis and O'Donoghue (2015).

The destination and final uses of outputs from the forestry sector are described in Table 1. The largest output destinations are within the forestry sector itself and the wood processing sector, accounting for €164 million, with €97 million of outputs going to construction. The total inter-industry outputs are €429 million. In terms of final uses, gross fixed capital formation (GFCF) and change in stocks are derived based on the existing share of forestry output of total outputs from the Agriculture, Forestry and Fisheries sector in the CSO's national IO tables. Ttotal sectoral outputs are €417 million.

Allocation of Output within BIO Model (€ millions)

	Forestry	Source
Wood Processing	164	Use of Agriculture, Forestry and Fisheries in Wood and wood products, except furniture, CSO 2015 IO Table
Construction	101	Use of Agriculture, Forestry and Fisheries in Construction, CSO 2015 IO Table
Within Primary Forestry Sector	164	Unexplained Outputs from Primary Forestry
Total Inter-Industry	429	
Final Uses		
Final Demand	0	Final Demand assumed from Wood Sector
Non-Profit Institutions Serving Households (NPISH)	0	Final Demand assumed from Wood Sector

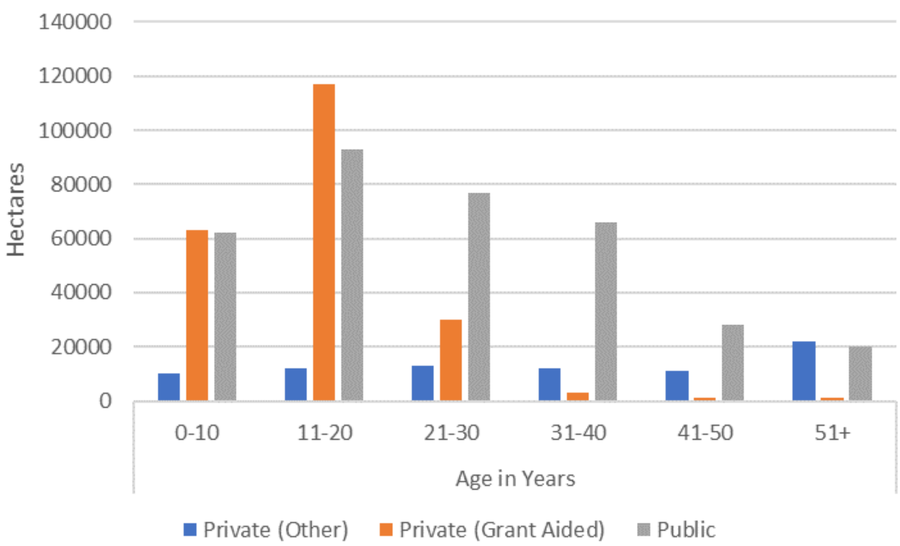
Government Consumption Plus Transfers	0	Final Demand assumed from Wood Sector
Gross Fixed Capital Formation (GFCF)	5	Using existing share of Output times GFCF for AFF
Change in Stocks	-17	Using existing share of Output times total Change in Stocks for AFF
Exports	0	Exports assumed from Wood Sector
Total Final Uses	-12	
Total Outputs	417	

Source: CSO Output and Value Added (Euro Million) by Activity, Industry Sector NACE, 2015; BIO

The data in Table 1 only provide a "snapshot" of the forestry sector in the year 2015 and fails to account for the intertemporal aspects of forestry associated with multiple forest cohorts. These cohorts reflect the multiple options related to forest planting and harvesting in Ireland such as tree species (conifer or broadleaf), thin or no thin, and optimal point of clearfell harvest. To account for these different cohorts, information from the Teagasc Forest Bio-Economic System (ForBES) is used by the BIO model to adjust the IO model as necessary[9].

In Figure 2, we see the total forest area currently planted by age and ownership. In terms of age, the largest land afforested area is in the 11-20 year range, which should currently be at the stage of first thinning. The majority of land in this age cohort is privately planted, unlike older forest stands, which are largely publicly owned with the exception of those in the 51+ age range. Given the pattern of forest planting in Ireland, it is obvious from Figure 2 that there are multiple age cohorts amongst the currently existing forest area.

2. Area Planted by Age and Ownership



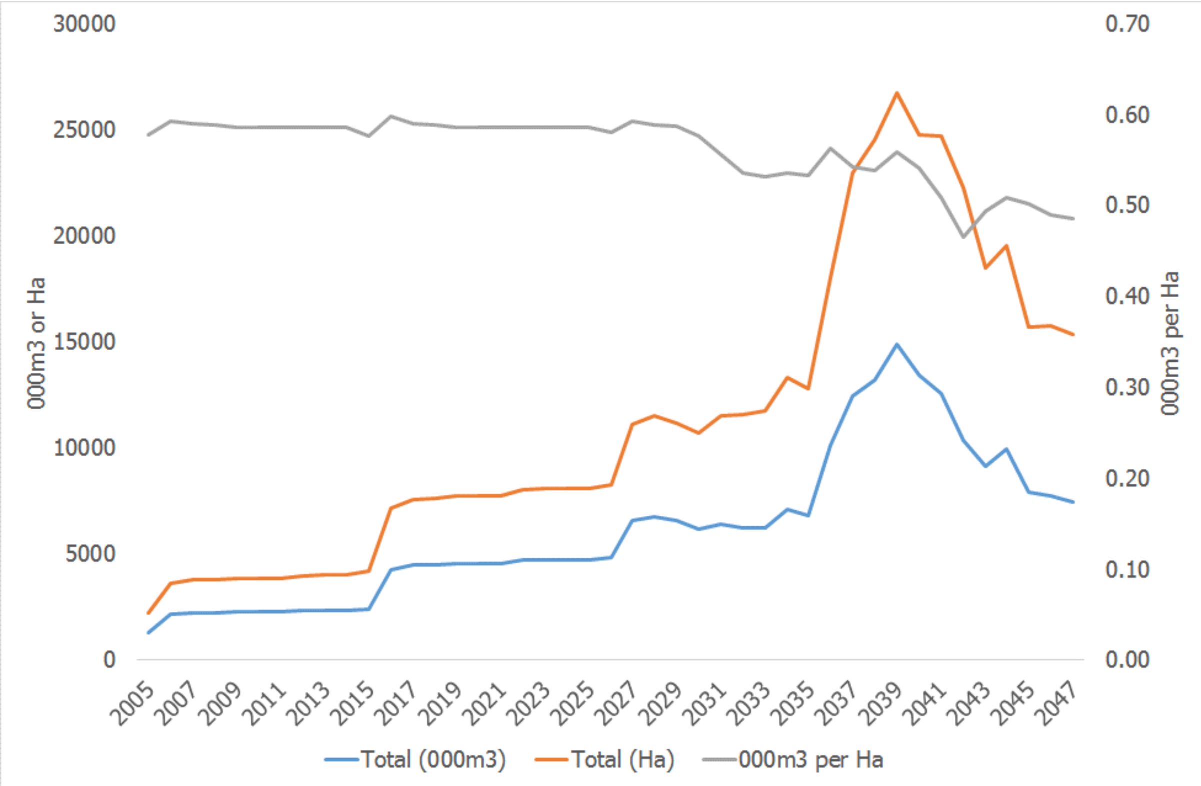
Source: National Forest Inventory

The uneven nature of forestry estate shown in Figure 2, as well as falling planting rates in recent years will have future consequences for timber production. The ForBES model's roundwood production forecast up to the year 2047 is shown in Figure 3. As forest stands continue to mature, roundwood supply will continue to increase, peaking in the year 2039 with net realisable volume (NRV) reaching 14.89 million m^3 or 26,694 hectares. Given current

[9] The forestry management parameters used by the ForBES model are available in the Annex.

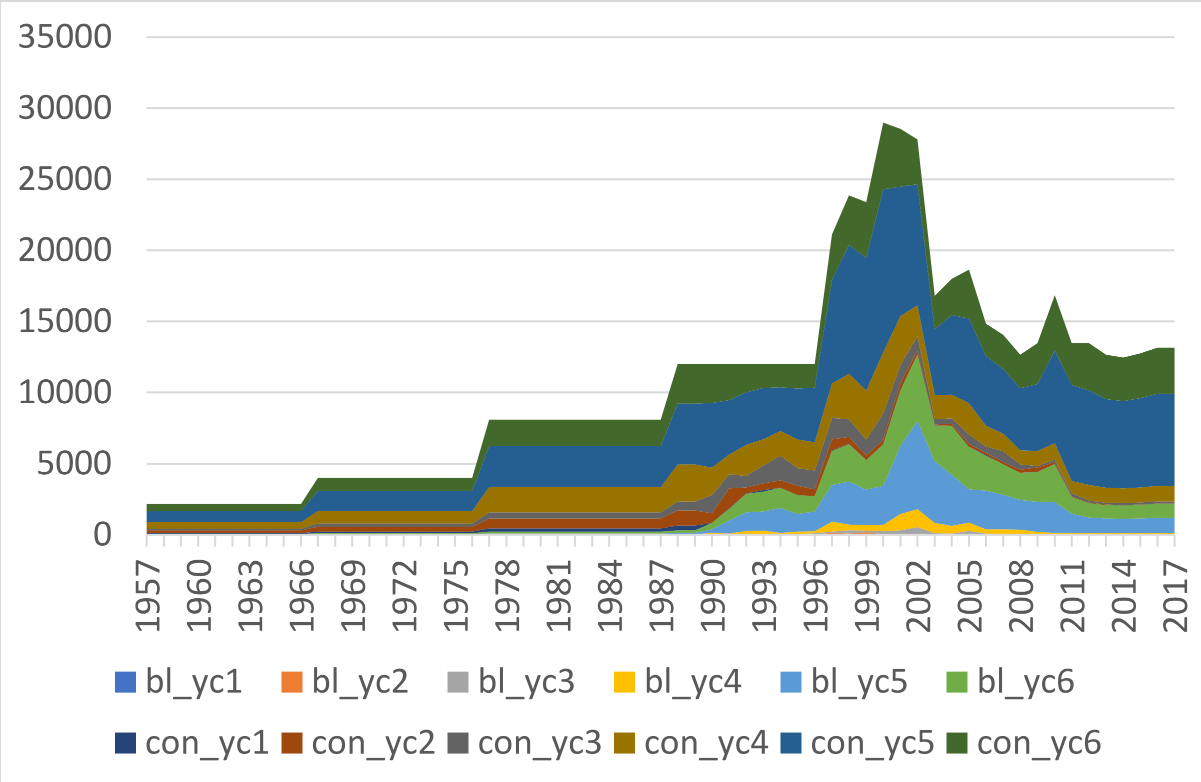
planting rates, roundwood supply will fall after 2039, with 7.44 million m^3 being supplied in the year 2047.

3. Timber Forecast



In addition to the multiple age cohorts existing in the Irish forest estate, there is a great deal of variability in terms of the species and yield class of Irish forests. This can be seen in Figure 4 which reports the historical trend in hectares planted from 1957 to 2017. For broadleaf species, the yield classes range from yield class 4 (bl_yc1) to yield class 14 (bl_ycl6) while for conifers, the yield classes range from yield class 14 (con_yc1) to yield class 24 (con_yc6). The wide range of yield classes observed indicates variability in rotation lengths and timber accumulation over time.

4. Historical Trend in Planting by Species and Yield Class



Results (Forestry Production)

In order to begin to disaggregate the forestry sector within the IO tables, we first look at the forest production sub-sector of the forestry sector. The ForBES model is used to generate net revenue figures for forest production over a single rotation. The model generates fixed costs, clearfell costs, clearfell revenue and finally net revenue on a per hectare basis for a conifer (Sitka spruce) and broadleaf (ash) species. All results are reported when thinning does or does not take place and across a range of yield classes. The results are shown in Table 2. On a per hectare basis, Sitka spruce produces higher net revenues than ash. Net revenue is higher when Sitka spruce forests are unthinned rather than thinned but the opposite is the case for ash forests with thinned forests generating more net revenue per hectare than unthinned ones. As would be expected, net revenue increase as yield class increases.

Net Revenue Per Hectare by Yield Class, Thinning and Species

Total Costs	Fixed Costs		Clearfell Costs		Clearfell Revenue		Net Revenue	
Thin	N	Y	N	Y	N	Y	N	Y
Sitka								
YC 14	1855	1925	1113	685	22260	13708	19292	11098
YC16	1715	1960	1247	831	24944	16625	21982	13834
YC 18	1575	1960	1260	986	25209	19718	22374	16772
YC 20	1505	1820	1339	981	26775	19620	23931	16819
YC 22	1435	1680	1397	956	27938	19119	25106	16483
YC 24	1400	1575	1525	945	30493	18902	27569	16381
Total Costs	Fixed Costs		Clearfell Costs		Clearfell Revenue		Net Revenue	
Thin	N	Y	N	Y	N	Y	N	Y
Ash								
YC 4	1995	2485	121	181	2428	3621	312	955
YC 6	1820	2240	194	299	3871	5978	1857	3439
YC 8	1645	2065	497	391	9936	7825	7794	5369
YC 10	1645	2065	626	512	12525	10246	10254	7669
YC 12	1470	1820	695	524	13907	10481	11742	8137

In Table 3, a collapsed version of the national IO tables is shown with only four industries: forestry; wood and wood products: construction; and insurance, reinsurance and pension funding. These are the only industries with which the forestry sub-sector is significantly connected. In the original national IO tables, €100.7 million of forestry's outputs go to the construction sector. The construction sector is reallocated to the wood and wood products sector as the forestry outputs that are used in construction are initially part of the wood and wood products sector. No fertiliser is produced domestically in Ireland so all must be imported so the fertiliser and other imports sectors make up total imports.

Collapsed Forestry and Wood Products IO Table without Construction (€million)

IO	Forestry	Wood and Wood Products	Construction	Other Intermediate	Total inter-industry	Total final demand	Total outputs
Forestry	164	264		0	429	-12	417
Wood and Wood Products	0	74	284	157	515	500	1015
Construction	0	29	2967	2723	5719	11766	17486
Insurance, reinsurance and pension funding	2	3	19	3021	3045	11507	14552
Other Intermediate	215	206	4317	86685	91423	94198	506450
Total intermediate consumption	381	576	7588	92587	101031	117959	539820
Fertiliser	12	0	0	206	218	0	218
Other Imports	0	242	4877	177589	182466		
Imports	12	242	4877	177795	182684		
Product taxes less subsidies	1	4	232	5335	5572		
Total consumption at purchasers' prices	395	821	12697	275475	289288		
Compensation of Employees	5	128	2655	75638	78426		
Operating surplus, net	101	14	1914	109022	111051		
CFC	6	49	195	59867	60117		
Non-product taxes less subsidies	-91	3	25	1000	936		
Value Added	21	194	4789	245527	250531		
Total inputs	417	1015	17486	521002	539819		

1. *Cost Base*

In order to determine the input structure associated with the multiple cohorts of the disaggregated forest growth sector, it is necessary to calculate the cost structure associated with growing forests across their rotation[10]. The costs are spread across different age cohorts with establishment costs and the establishment grant occurring in Year 0, forest premia being paid to farmers by the State over the first 20 years, and clearfell costs occurring from the 36-40 age range onwards. The largest cost is the establishment of the forest stand, but this is offset by an establishment grant that is provided to forest planters by the Irish state. Establishment, thinning and clearfell costs will require the hiring of contractors from the construction sector. The cost structure associated with a broadleaf forest rotation is also taken into account in the forest growth input structure.

Contractor Costs

With contractor costs for the establishment, thinning and clearfell of the forest stand making up a high percentage of costs in the forest sector, the input structure of the construction sector is used to allocate these contracting costs within the forestry sector. First, the construction sector's input structure is adjusted to derive construction costs without materials as the contracting work carried out within the forestry sector uses materials from the forestry sector itself. As a result, materials-based sectors such as wood and wood products, rubber and plastic products, other non-metallic material products, and fabricated metal products are removed from the construction input structure and the sector is rebalanced accordingly.

The total contracting costs for the forestry sector in 2015 are shown in Table 4. Total establishment costs are €61.66 million, harvesting costs are €7.38 million and total thinning costs are €580,000. Total contracting costs are €69.62 million.

Forestry Contracting Costs (€ millions)

	Conifer	Broadleaf	Total
Establishment Costs (same as grant)	37.73	23.93	61.66
Harvesting Costs	7.33	0.05	7.38
Thinning Costs	0.56	0.02	0.58
Total	45.61	24.01	69.62

The allocation of contracting costs within the forestry sector is shown in Table 5. The components of contracting costs are derived from the shares of inputs into the adjusted construction sector where material inputs are removed. For example, intermediate consumption makes up 68.9% of total inputs for the construction (less materials) sector so the same share of total inputs is applied to the contracting share of forestry, giving an intermediate consumption of €48 million for contracting within forestry.

Contracting Sector and Construction

	Construction	Construction (less materials)	Contracting
Intermediate Consumption	7588	6305	48.00
Imports	4877	226	1.72
Product taxes less subsidies	232	120	0.91
Compensation of Employees	2655	1391	10.59
Operating surplus, net	1914	1002	7.63
Consumption of fixed capital	195	102	0.78

[10] Full cost structure is available in the Annex.

Forestry Growth	0	0	0.00
Non-product taxes less subsidies	25	0	0.00
Value Added	4789	2495	19.00
Total	17486	9145	69.62

2. *Value of Biomass*

Within the forestry sector, there are two sources of wood biomass to be valued: biomass in the trees that have been harvested and biomass in trees yet to be harvested. At sales prices, the total value of clearfelled biomass is €147.65 million but adjusting for value added and other intermediate consumption, the producer value of the biomass is €132.63 million. The majority of clearfell value occurs in the 35 to 39 year old range of forests, accounting for €105.32 million. Thinnings are valued at €2.09 million at producer prices.[11]

The accumulation of biomass over time and the different biomass cohorts that exist in Ireland currently is shown in Table 6. The horizontal axis displays the current five-year age range of the forest stand while the vertical axis shows the value of how much biomass has been added in previous five-year periods. For example, conifer forests that are currently 15-20 years old (age range 15) added €0.44 million of biomass in their first 5 years, €6.05 million of biomass in years 5 to 10 and €49.68 million of biomass in years 10 to 15. In a static IO table, only biomass that is harvested will be counted as it only becomes something of economic value once it is sold in the marketplace. However, as shown here, there is a great deal of accumulated biomass that is yet to be harvested which needs to be accounted for in the IO framework if the forestry sector is to be properly valued.

Conifer Biomass Accumulation by Cohort and Time of Growth (€ millions)

	Conifer										
	Current Age of Forestry										
	0	5	10	15	20	25	30	35	40	45	50
Age of Forestry at time of growth											
0		0.28	0.37	0.44	0.23	0.26	0.20	0.18	0.10	0.09	0.05
5			5.00	6.05	3.11	3.47	2.69	2.42	1.33	1.21	0.72
10				49.68	26.21	28.64	22.31	20.05	11.02	10.00	5.92
15					39.07	42.98	33.56	30.16	16.57	15.04	8.91
20						50.12	39.13	35.16	19.32	17.53	10.39
25							41.86	37.62	20.67	18.76	11.11
30								37.00	20.33	18.45	10.93
35									17.90	16.24	9.62
40										3.92	2.32
45											0.59
50											

In Table 7, we see the flow of forestry value across the different forest age cohorts. This allows us to define the values present in the rows and columns of the disaggregated forestry sector in the IO tables. The destination of value is generally represented in the forest biomass, which is shown in the Change in Inventories column. When clearfell occurs (from the 35-year age range onward), the value realized from the clearfell sales is shown in the Clearfell column. The sources of value are generally made up of the forestry growth that takes place within each five-year age range, seen in the Forestry Growth column. Much of the non-growth value generated is offset by forestry subsidies. For example, contractor costs (Contractor) produced by the need to hire contactors to plant the forest are offset by establishment grants (Product Taxes less Subsidies) that are paid by the State. Operating surplus (Operating Surplus, Net) generated in the first 20 years of forestry growth is offset by forestry premiums (Non-product Taxes less Subsidies) paid by the State to forest owners.

[11] The full breakdown of biomass value is available in the Annex.

Origin and Destination of Forestry Value

	Destination						Source											
Age Group	Clearfell	Thinning	Clearfell Losses	Thinning Losses	Change in inventories	Total	Contractor	Construction	Fertiliser	Biomass from the Past	Product taxes less subsidies	Compensation of Employees	Operating surplus, net	Consumption of fixed capital	Forestry Growth	Non-product taxes less subsidies	Value Added	Total
0.00					0.38	0.38	61.66				-61.66		31.80		0.34	-31.80	0.34	0.34
5.00					5.13	5.13		3.22					46.89		1.91	-46.89	1.91	5.13
10.00					59.25	59.25		3.22	0.92				46.90		55.11	-46.90	55.11	59.25
15.00					84.39	84.39		6.68					9.86		77.71	-9.86	77.71	84.39
20.00					52.86	52.86		1.84							51.02		51.02	52.86
25.00					54.49	54.49		1.95							52.54		52.54	54.49
30.00					41.61	41.61		1.43							40.18		40.18	41.61
35.00	94.60		5.54		32.90	133.05		1.19		99.58					31.71		31.71	132.48
40.00	29.25		1.72		4.36	35.33		0.21		30.86					4.15		4.15	35.22
45.00	7.82		0.47		1.01	9.29		0.06		8.38					0.95		0.95	9.38
50.00	0.93		0.05		0.05	1.03		0.01		0.98					0.04		0.04	1.03
55.00	0.03				0.05	0.09				0.03								0.03

3. Value Added

Disaggregating the forestry sector allows us to allocate value in a more accurate way than before the disaggregation took place. Previously, the national IO tables told us that the forestry sector generated €21.33 of value added in 2015. This figure can now be further broken down, with value being allocated to the contracting, clearfelling, and thinning sub-sectors. The majority of value is allocated to the contracting sub-sector, accounting for €13.53 million of value added. It is assumed that the contractor share of value added is the same as construction. €7.64 million of value added is allocated to the clearfelling sub-sector while €170,000 of value added is allocated to the thinning sub-sector.

The results show that there is a relatively small amount of value added by the forest growing sector and that the majority of value is allocated to the contracting sector. The value added that accrues to the growers of trees through clearfelling and the production of thinnings is small. Most of the value flowing through the sector is offset by subsidy payments. Currently, value is not added to the timber produced until it reaches the processing sector but this may change if the carbon sequestered in the trees is explicitly valued in the marketplace in the future (O'Donoghue & Ryan, 2020).

Results (Harvested Wood Products)

The next aspect of the disaggregation of the forestry sector in Ireland deals with the Harvested Wood Products (HWP) sector. In the national IO tables, this sector is represented by Wood and Wood Products, except Furniture. The data used to disaggregate the sector is provided by Knaggs and O'Driscoll in their COFORD Connects series "Woodflow and forest-based biomass energy use on the island of Ireland". In Table 8, the supply and use of wood fibre in Ireland is shown. The majority of roundwood is supplied by Coillte[12], although private sector supply is trending upwards. The other major fibre sources beyond roundwood are sawmill residues, wood-based panel residues and post-consumer recovered wood (PCRW). The majority of wood fibre is currently used in sawmilling and the manufacture of wood-based panels, although an increasing amount is being employed for energy use.

Supply and Use of Wood Fibre in 2015 (000 m3 OB RWE)

Roundwood Supply	2015
Commercial softwood Imports less exports	40
Coillte	2380
Private sector	646
TOTAL	3066
Fibre source	
Roundwood	3063
Sawmill residues	949
Wood-based panel residues	114
Residue imports	47
Harvest residues	60
Post-consumer recovered wood (PCRW)	300
TOTAL	4533
Uses	

[12] Coillte is the State-owned national forestry company

Sawmilling	1867
Round stake	169
Wood-based panels	1370
Wood for energy use by the power generation and forest products sector	796
Other uses Horticultural bark mulch	30
Wood chip for heating	114
Export of forest product residues	36
Pellet manufacture	151
Other uses including shavings and animal bedding	0
TOTAL	4533

Source: Knaggs & O'Driscoll (2016)

Unlike the forest growth sector, the HWP sector does import some inputs from other countries. The majority of wood fibre imported is used for energy purposes. In 2015, 119,000 m^3 of wood fibre was imported for energy purposes when net exports are accounted for. Sawdust accounted for 45,000 m^3 of imports, while wood chip accounted for 3,000 m^3. Other wood fibre imports amounted to 40,000 m^3.

4. *Price Calculations*

Although the data from Knaggs and O'Driscoll provides information about the volume of wood biomass used by the HWP sector, data about the how that volume translates into monetary amounts is missing. As the IO tables describe the monetary flows between different sectors, such information is required. In order to impute prices, the volume amounts from Knaggs and O'Driscoll are divided into forestry sub-sectors. These sub-sectors are forestry, wood, wood products, biomass, other sectors, imports/exports and timber fuel. Volumes from the wood flow summary are then assigned to the appropriate sub-sector. These volumes are then divided by their equivalent value in the aggregated IO table in order to determine the price per cubic metre (€/m3) of product in each sub-sector. These prices are then checked by multiplying them by wood flow summary volumes to see if they match the IO aggregate figures.

Price Imputation

IO Aggregate (€million)	Wood	Wood Products	Biomass	Other Sector	Exports	Timber Fuel
Forestry	151					
Wood	26	48	27	441	157	
Wood Products				51	292	
Biomass			28	38		
Other Sector						29
Imports	157	85				
Wood Flow Summary ('000 m^3)	Wood	Wood Products	Biomass	Other Sector	Exports	Timber Fuel
Forestry	3419					
Wood	7029		1459	1834	429	
Wood Products						
Biomass			447	615		
Other Sector						844
Imports	445					
Price (€/m^3)	Wood	Wood Products	Biomass	Other Sector	Exports	Timber Fuel
Forestry	0.0441					
Wood	0.0037		0.01823	0.24063	0.3657	
Wood Products						

Biomass			0.06239	0.06239		
Other Sector						0.03445
Imports	0.3519	1.00000				

Check	Wood	Wood Products	Biomass	Other Sector	Exports	Timber Fuel
Forestry	150.9					
Wood	25.8	0.0	26.6	441.3	156.9	
Wood Products						
Biomass			27.9	38.4		
Other Sector						29.1
Imports	156.6					

Results (Scenario Analysis)

In order to illustrate the capabilities of the disaggregated forestry IO sector, we will use scenario analysis to analyse the impact of two different land use situations related to the forestry sector. The scenarios are related to separate policy targets set out by the Irish government in recent years, namely, a yearly afforestation target and an output target for the beef sector. In 2011, the Irish government pledged in their Programme for Government to implement an annual 14,700 hectare afforestation programme (Department of the Taoiseach, 2011). In 2010, the Food Harvest 2020 (FH2020) policy document laid out a target of a 20% increase in the value of beef output by the year 2020, using the average of the years 2007 to 2009 as a baseline (DAFF, 2010).

Scenario A reflects what occurred in reality in the years 2010 to 2020 (see Table 10). The beef target was achieved by the year 2020, with a 43% increase in the value of beef output being realised. The afforestation target was not reached with forest planting actually declining over the period 2010 to 2020. Scenario B looks at what would have happened had the forestry target been met each year from 2010 to 2020 while also achieving the beef target.

Beef Output and Forestry Planted from 2007 to 2020

Year	2007-2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Beef Output (€m)	582	587	736	753	700	763	847	819	849	816	777	835
Forest (ha)	6615	8314	6653	6652	6252	6156	6293	6500	5536	4025	3550	2434

We use the disaggregated IO tables to analyse Scenario A and Scenario B in terms of overall economic output, value added and net greenhouse gas (GHG) emissions. The disaggregated forestry sector allows us to account for the life-cycle nature of forestry in a way that is not possible with an aggregated forestry sector. In order to account for the full value of forestry versus other potential land uses, the scenarios run from 2010 to 2075 with all amounts generated discounted back to 2010 values and presented as net present values (NPVs). In Scenario B, the extra land used for afforestation is reallocated from land used for beef production. This decision is taken due to cattle farming having the largest land share amongst farm systems in Ireland, higher quality, more productive land being concentrated on dairy and tillage farms, and the difficult financial conditions currently affecting many beef farmers in Ireland currently (Geoghegan & O’Donoghue, 2018). To simplify our analysis and to concentrate on the effects of a single forestry rotation, it is assumed that no new afforestation takes place after 2020, and that beef and milk production levels also remain unchanged after that year.

The results of our analysis are shown in Table 11. Results are presented as the NPV of the change in output, value added, and GHGs that would result from each scenario relative to the average of 2007 to 2009. In terms of overall economic output, a €164.7 million increase in output is produced by the forestry and beef sectors in Scenario B compared with the €123 million increase in Scenario A relative to the 2007 to 2009 average. Over the scenario period, €61.1 million extra is generated in terms of value added in Scenario B compared with Scenario A. Regarding GHG emissions, Scenario A sees yearly average emissions staying almost completely flat relative to 2007-2009 while GHG emissions fall by 124.3 ktCO2e relative to the baseline in Scenario B. For carbon sequestration, we see 276.8ktCO2e extra being sequestered over what would take place in Scenario A. When emissions and sequestration effects are combined, a greater reduction in net emissions takes place in Scenario B with a fall of 2013 ktCO2E compared with 1612ktCO2e in Scenario A.

Changes in Output, Value Added an GHG Emissions Resulting from Scenarios A and B Relative to the 2007-2009 Average

	Scenario A	Scenario B	Change (Average per year)
Output (€m)	123.0	164.7	41.7
Value Added (€m)	18.9	80.0	61.1
GHG Emissions (ktCO2e)	0.2	-124.3	-124.5
Carbon Sequestration (ktCo2e)	1611.8	1888.6	276.8
Net Emissions	-1611.6	-2012.9	-401.3

The results show that both beef and afforestation targets could have been reached while realising an overall decrease in GHG emissions and a larger overall decrease in net emissions. Hitting the afforestation target in Scenario B by reallocating 3.7% of land used for beef production in Scenario A also results in greater economic output and higher overall value added. These results suggest that agricultural intensification can exist in accordance with GHG emissions reduction goals.

Discussion and Conclusions

This paper describes a forestry input-output model that disaggregates the Irish forestry sector in order to analyse the economic contribution of the various sub-sectors of the forest economy. With increased attention being given to forestry as a driver of rural development, as well as a shifting policy context, it is important to determine how forestry interacts with the overall economy. Although forestry is represented in the national IO tables, it is aggregated with two other economic sectors: agriculture and fisheries. In order to more fully understand how forestry contributes to the national economy, it is necessary to disaggregate forestry not just from agriculture and fisheries but also to further disaggregate the sector itself.

Disaggregation is necessary for two reasons. First, forestry operates in long time frames, sometimes over 100 years. In contrast, IO modelling describes the flows within an economy at a single point in time. To fully capture the economic contribution of a sector like forestry, the intertemporal aspects of the sector must be taken into account. Such an exercise is easier to carry out in a disaggregated model. Second, aggregation bias can occur when an economic sector is analysed at too high a level of aggregation. Heterogeneity within the sub-sectors of the sector can lead to misleading estimates, especially when generating economic multipliers. Such heterogeneity is a clear feature of the forestry sector due to the multiple cohorts that exist in terms of forest age, species type, yield class, and forest management decision.

We first disaggregate the forestry sector i.e. the forest planting sector. It is found that the sector generates a relatively small amount of value added, with most of the value added being produced by the contracting sector when trees are planted and harvested. The rest of the value added is generated through clearfelling and thinning, although the value generated by clearfelling far outweighs that of thinning. The harvested wood products (HWP) sector is also disaggregated. Scenario analysis using the disaggregated IO model shows the ability of the disaggregated tables to give insight into policy matters. The scenario analysis shows the ability of forestry to be used by policymakers to enable economic growth and address environmental goals in the context of agricultural intensification.

The disaggregation of the forestry sector within the IO tables gives a more detailed view of the sector and accounts for the intertemporal nature of forestry in a novel manner. This will provide researchers and policymakers with greater information with which to study the future of the sector and make forestry-related policy decisions. While issues related to the intertemporal nature of forestry and the "snapshot" nature of IO analysis have been previously noted (Roberts et al., 1999; Eiser & Roberts, 2002), this paper directly addresses the multiple age cohorts that exist within the forest sector and incorporates them into the newly disaggregated IO tables. Such disaggregation leads to greater granularity, which can reveal useful information that would otherwise stay undetected if the temporal aspects were not modelled (Su & Ang, 2022). This particularly the case in a country like Ireland which does not have an historic tradition of farm forestry so is experiencing a changing national forest age structure.

The findings from the scenario analysis are an example of how disaggregating the forestry sector within the IO tables can be used to analyse the effect of policy decisions in an intertemporal manner across the value chain. With issues relating to greenhouse gas emissions and climate change becoming a more significant part of government decision making, policy making will increasingly have to factor in long time horizons and trade-offs between different economic sectors. The Irish government's Climate Action Plan 2021 identifies afforestation of new land as a key part of reaching climate neutrality by the year 2050 (Government of Ireland, 2021). Work such as this will allow the full impact of such afforestation to be assessed as land uses are changed and new forests mature. More detailed IO models will only help in the process of formulating future policy. Future steps would be to generate multipliers for the forestry sub-sectors of interest to policymakers and to further address environmental aspects of forestry by explicitly valuing the carbon sequestration provided by forests.

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Farmland Afforestation: Forest Optimal Rotation Ages across Discrete Optimisation Objectives

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Abstract

Forestry is increasingly seen as having a multiplicity of functions, some of which may lead to tensions between the preferences of different forestry stakeholders. One expression of these differing preferences is in the area of optimal rotation length. This paper models the optimal rotation lengths of different aspects of the forestry innovation system – the forest owner, the timber processor, the carbon sequestered, and the social value. A microsimulation framework is utilised to simulate the impact of alternative rotation lengths on different outcomes including timber volume, processor value of harvested timber, forest owner incomes, carbon emissions and incomes adjusted for a carbon value. The main conclusion is that the optimal rotation lengths vary quite significantly depending upon the stakeholder. We assess the impact of a forest owner harvesting their timber at a time that was financially optimal for them. The processor would have the smallest difference in their annual equivalised net present value (AENPV) from the forest owner. The biggest difference is with the optimal rotation length for carbon, with the AENPV of sequestered carbon being about 50% higher than for the forest owner's optimal rotation length. The differential for timber lies between the processor and the carbon. As a combination of private and carbon goals, the social return also lies between the processor and carbon amounts, the differential increasing with higher carbon values, reflecting an increasing importance of carbon. Additional survey results find that the financial optimum for a forest owner at a 5% discount rate differs substantially from the actual rotation length. These differing preferences for rotation length reflect the need for a systems-based approach to forestry policy, considering the needs of all the system's stakeholders.

Key words Optimal Rotation Length; Forest Economics; Microsimulation; Innovation System

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Introduction

Forest carbon uptake, including maintaining or enhancing the carbon sink and the substitution of wood for other emissions-intensive materials, will be crucial to compensate for human-caused greenhouse gas (GHG) emissions in the coming years. To become carbon neutral by 2050, the European Union (EU) net sink from forest land will need to increase to about -450 Mt CO_2e yr^{-1} by 2050 (European Commission, 2020). Forest management practices will play a key part in whether such an increase in net carbon sink will occur (Pilli et al., 2022).

One important aspect of forest management is the timing of the harvesting decision. Forest economics has long been concerned with optimising harvesting strategies (Tassone et al., 2004; Olschewski & Benítez, 2010), with particular importance ascribed to the optimal rotation length (ORL) (Pearse, 1967; Roberge et al., 2016). Rotation age analysis has generally been driven by commercial concerns, namely improving the net present value (NPV) of timber-related cash flows for forest owners (Saraev et al., 2019). However, this perspective ignores non-timber assets and the multi-functionality of forests (Cubbage et al., 2007; Kumar & Kant, 2007), while also neglecting stakeholders across the forest value chain who may require alternative rotation lengths to achieve their distinct economic aims (Amacher et al., 2004; Loureiro & Arcos, 2012). This draws into question the prudence of basing rotation ages solely on timber stand NPV. The central hypothesis of this paper is that idiosyncratic stakeholder needs and optimisation objectives manifest in differing harvesting schedules. Relevant stakeholders comprise the public who derive utility from optimising forest ecosystem services, producers of harvested wood products (HWPs) reliant upon the supply and quality of timber resources, and forest owners concerned with optimising private returns.

The literature on ORL analysis typically revolves around the application of Faustmann-based formulas to identify the rotation length which maximises the land expectation value (LEV) of a forest stand according to timber production, where the ORL is positively related to changes in planting costs and inversely related to the changes in timber prices and discount rates (Saraev et al., 2019). Hartmann (1976) pioneered the inclusion of non-timber values in harvesting decisions by incorporating recreational values in the rotation age formula, indicating that such intangible services may serve to lengthen rotations.

Recently, research has sought to identify socially optimum rotation ages by integrating non-market ecosystem services into harvesting models based on biodiversity objectives (Koskella et al., 2007; Nghiem, 2014; Felton et al., 2017; Saraev et al., 2019) with findings indicating the promotion of wildlife generally necessitates longer rotations beyond the Faustmann optimum. Economically, payments for ecosystem services can internalise the social value of forests in a forest owner's business decisions and incentivise extended rotations (MacPherson et al., 2017).

Forest carbon sequestration functionality is now assuming precedence in rotation length discourse. Biomass carbon storage simulations have demonstrated the positive relationship between rotation ages and forest carbon stocks (Kaipanen et al., 2004; Zanchi et al., 2014). Forest carbon storage increases over a rotation before levelling out if the stand is kept long enough to permit substantial natural decay (Boman et al., 2010). Joint production problems seeking to balance timber production alongside carbon sequestration similarly indicate extended rotations beyond business-as-usual scenarios (Pohjola & Valsta, 2007; Cao et al., 2010).

Under private ownership where timber value takes precedence, non-timber services such as carbon sequestration are externalities which may cause socially sub-optimal management decisions (Van Kooten et al., 1995; Tassone et al., 2004). Carbon pricing is advocated as a means to economically rationalise the lengthening of Faustmann rotations towards a socially optimum rotation age (Romero et al., 1998; Ekholm, 2016). Simulation models generally indicate that carbon pricing regimes discourage harvesting and lengthen optimal rotations (Kula & Gunalay, 2012; Lutz et al., 2016; West et al., 2019), or promote near cessation of felling particularly when assuming immediate carbon release upon harvesting and/or higher carbon prices (Cao et al., 2010; Ekholm, 2016). However, NPV may be maximised by shorter rotations relative to timber-based projects if forest owners are remunerated for stored carbon but are not taxed upon harvesting, akin to assuming no carbon release (Nghiem, 2014).

Using Ireland as a case study, this paper constructs a rotation age framework that models clear-felling schedules according to stakeholder utility objectives. In light of Irish government policy that incentivises farmers to carry out woodland planting (McDonagh et al., 2010), these rotation issues are analysed within the framework of farm afforestation projects. This study identifies optimal harvesting ages by maximising the NPV of stakeholder returns with respect to farmer income based on timber production, social returns in relation to forest carbon sequestration and social costs of carbon from specific grades of timber. We utilise a forest bio-economic model (ForBES) calibrated to function across multiple management scenarios and assimilate Irish forest data relating to an uneven-aged Sitka spruce (Picea sitchesis (Bong.) Carr.) plantation to produce a microsimulation system.

The Irish State is proactively seeking to increase the country's forest cover for both economic development and public welfare reasons (Upton, 2014). Since the mid-1980s, national and EU financial aid packages have sought to encourage afforestation among private landowners (Howley et al., 2015), while government policy envisages that the vast majority of future woodland planting will be fulfilled by farmers converting agricultural land (Duesberg et al., 2014). This raises challenges for the planning of harvesting strategies, considering the general lack of forest management expertise among farmers (Rois-Díaz et al., 2018).

There is also increasing demand for forest services beyond the supply of timber (Bonsu et al., 2019). Forestry and forest products will be the single largest land-based climate mitigation measure available to Ireland, while forest creation is the single most effective land-use change for meeting Ireland's carbon emission targets (Lanigan et al., 2019). In the Climate Action Plan 2021, a target has been set to achieve 8,000 hectares of newly planted forest per annum, but afforestation rates are currently well below this target with just 2,016 hectares of newly planted forest in 2021 (Government of Ireland, 2021). Low planting rates in recent years have been identified as a future risk in terms of the ability of Irish forests continuing to act as a significant carbon sink (EPA, 2020).

Increasing attention is also being paid to the ecosystem services provided by forestry in relation to habitat, water quality, moderation of run-off, recreation, and amenity (Bullock et al., 2016). There is an ever-increasing demand for activity-based pursuits in Ireland's forests and significant opportunities also exist around their potential use as centres of learning and in outdoor education. For instance, visits to Coillte[13] forests increased by nearly 40% during the Covid-19 pandemic, with 2.2 million people visiting Coillte's top 50 forests between March

[13] Coillte is a commercial forestry company, owned by the State, that manages approximately 7% of the country's land.

and December 2020 (DAFM, 2022). Given these multiple perspectives on the utility associated with forests, it is worthwhile assessing rotation solutions from various points-of-view.

The remainder of the paper is laid out as follows. Section 2 provides a theoretical framework for how optimal rotation lengths and net present values are calculated. Section 3 covers the methodology utilised in the paper while Section 4 outlines the results of the paper. The discussion and conclusions are presented in Section 5.

Theoretical Framework

The practical challenge presented within this paper involves analysing the optimal rotation age for a farm afforestation project that maximises expected net returns from an infinite number of rotations. This involves devising an amended Faustmann formula that allows NPV calculations to be modified according to different stakeholder objectives, namely (i) private returns from timber production (ii) wood product market returns from serviceable timber; or (iii) social returns from joint production. The study's conceptual foundation embodies ecological principles – forest stand development, carbon sequestration, and forest product yields – in conjunction with accounting rubrics for estimating forest cash flows emanating from timber production, social costs of carbon, and wood product markets.

Forest Stand NPV Maximisation

In a basic forest enterprise, net cash flows are based on revenue from timber harvests and costs from forest management outlays (planting, maintenance, logging, etc.). Akin to the microeconomic view of the firm where organisations maximize profit by functioning at that level where marginal costs increase to equal marginal revenues, the financially optimum rotation period is the forest age where incremental (marginal) cost equates to incremental (marginal) revenue (Pearse, 1967). Harvesting before or after this period yields sub-optimal returns (Ryan et al., 2016).

The Faustmann solution that maximises the NPV of private returns across a perpetual cycle of rotations for timber production is defined by the ORL T as below (adapted from West et al., 2019):

$$\boldsymbol{Max}\ NPV\{T\} = (pw.vw(T) - K)\frac{1}{e^{rT-1}} \quad (1)$$

where the optimal rotation is a function of the price of wood pw, the merchantable wood volume vw at time T, management costs K, and the discount rate r.

When deducing whether a forest stand is financially mature for harvesting, the Faustmann model does not integrate forest owner-specific traits such as subjective time preferences into its specifications (Tahvonen & Salo, 1999). Higher discount rates can be used to account for the time preference of forest owners requiring more immediate returns from plantations. Opportunity costs from alternative investments foregone can also be used to adjust discount rates. In the case of farmers transitioning from agriculture, this may be approximated from aggregated rates of return from alternative land uses, which for this study encompass dairy farming or livestock management.

Faustmann formulas can be tailored to incorporate afforestation subsidies as additional income given their deterministic nature. In Ireland, these comprise grants to landowners covering 100% of forest establishment cost alongside fifteen annual forest premium payments (Duffy et al., 2020). Meanwhile, carbon pricing regimes can be included in calculations, based on annual payments afforded for each additional increment of carbon stored or a lump sum at the commencement or conclusion of harvest cycles. It is important to note that alternative payment specifications will affect rotation lengths differently (Chladná, 2007).

Annual compensation instalments linked to incremental increases of forest carbon stocks represent a useful remuneration mechanism given their scientifically indexed nature (related to biomass carbon storage modelling) and the economic incentivisation of providing a regular flow of income akin to agriculture. An appropriate price of carbon may be derived from the social cost of carbon literature which estimates the potentially negative environmental impacts of CO2 emissions (Nordhaus, 2014; Tol, 2008). Given the above, the optimal rotation for afforested farmland based on joint production for social returns can be described as follows (adapted from West et al., 2019):

$$\boldsymbol{Max}\, NPV\{T_T\} = [pw.vw(T_T) - K + S + pc\theta\left(\frac{\int_0^{T_T} w(t)dt}{T_T} - \frac{\int_0^{T} w(t)dt}{T}\right)]\frac{1}{e^{rT_T - 1}} \quad (2)$$

where $vw(T_T)$ is the merchantable wood volume at the new optimal harvest time T_T , S is subsidised income, pc is the imbursement awarded per ton of additional average CO2 stored by lengthening the rotation length to T_T, θ denotes the conversion of merchantable wood volume to tons of CO2, and T expresses the basic optimal rotation age without carbon pricing as per Eq. (1). The NPV is then maximized vis-á-vis the extended rotation length, T_T.

Methodology

ForBES Model

This paper utilises the Forest Bio-Economic Model (ForBES), developed by Teagasc, Ireland's Agriculture and Food Development Authority. While only 51% of the national forest estate is composed of Sitka spruce (SS) and many plantations are characterised by a mixture of species (DAFM, 2018), to simplify computations the forest stands under analysis are assumed to constitute 100% SS. The model relies upon UK Forestry Commission yield tables (Edwards & Christie, 1981) to approximate total merchantable timber volume (MTV), while a logistic function is applied to interpolate fledgling tree growth and subsequent growth in five-year intervals with reference to specifications by Edwards and Christie. MTV estimations accordingly underpin the calculations of total forest biomass and concomitant carbon stocks. The model caters for thin and no thin scenarios, with thinning regimes characterised by the marginal thinning intensity (MTI) or the maximum rate of volume removal that does not hinder cumulative volume production (Ryan et al., 2016). MTI is denoted by 70 percent of yield class per hectare per year ($m^{3/}$ha/year). The model uses yield classes (YC) 14 to 24, with YC14 being the minimum grade eligible for afforestation subsidies in Ireland (DAFM, 2017).

Given the overlap of certain functions such as the wood product carbon liberation curve from a prior harvest having not declined to zero before the subsequent rotation is felled and a second such curve begins, it is necessary to analyse a relatively long time period (Bateman & Lovett, 2000). Therefore, the analysis considers a 150-year planning horizon which allows a series of rotations to be modelled. As described later, the optimal rotation age for each management objective is identified by maximising forest stand NPV on a per hectare basis standardised to annual equivalent values.

In order to calculate social returns, the total carbon sequestration value of a SS plantation must be determined with regard to carbon pools in biomass, dead organic matter (DOM), soil, and wood products (including substitution effects)[14]. Biomass carbon is based on the merchantable volume of trees and ecological parameters such as basic wood density, carbon fraction, and biomass expansion factor provided by Ireland's National Inventory Report (NIR) (Duffy et al., 2018). DOM comprises litter and deadwood carbon pools. Litter pools are based on NIR coefficients for estimating leaf/needle biomass. Conifer foliage decomposition rates and litter decomposition rates are also included. Deadwood carbon pools are supplied by timber from tree mortality and harvest losses coupled with residual roots post-harvest. Due to environmental regulations, it is assumed that forests are planted on mineral soil conditions only. The carbon liberation timeline in wood product pools is dependent upon the nature of the product and its associated lifespan, in tandem with its end uses, whether being recycled, burnt, or stored in landfill (IPCC, 2006). Harvested timber is assigned into production lines in the following ratios: 52% to sawn wood, 48% to wood-based panels, and 0% to paper/paperboard given paper production is currently not operational nationally (Duffy et al., 2018). Carbon liberation schedules in product categories are estimated in line with NIR calculations that follow methodology developed by Pingoud and Wagner (2006).

Combining the carbon pools, total forest carbon stock is expressed as tCO2e/ha/year using IPCC conversion factors. Carbon stocks are then priced according to the average social costs of carbon (SCC) provided by Tol (2008), which estimated an SCC ($/tC) of $148 (€129), $122 (€106), and $50 (€44) under discount rates of 0%, 1%, and 3% respectively. This pricing methodology represents a relatively prudent approach given the considerable ambiguity surrounding the economics of climate change (Ackerman & Stanton, 2012). The NPV optimisation solution for social returns involves maximising the carbon sequestration values based on SCC while including the returns that accrue from incidental timber production provided by thinning and harvest operations over the course of a rotation.

Optimal Rotation for Private Returns

The analysis calculates private returns based on timber production and forest subsidies with the MTV modelling approach. Timber from thinning and clear felling is assumed to be sold standing and forecast revenues are based upon MTV yields coupled with annual price series data generated by the State's forest agency Coillte[15]. Forest subsidy income is estimated with reference to the ForSubs model (Ryan et al., 2014) which is calibrated according to the policy rubrics of national afforestation subsidy schemes. Forest management outlays are derived from Teagasc's Forest Investment Valuation Estimator (FIVE)[16] decision support tool.

While opportunity costs from alternative land uses may be accounted for by modifying the discount rate, ForBES instead integrates agricultural income foregone as an annual cost across the forest rotation. Data on expenditure and income across the three farming systems of interest – dairy, beef, and sheep – are provided by the Teagasc National Farm Survey (NFS) (Dillon et al., 2018). To integrate agricultural returns as an opportunity cost into rotation calculations they must be defined and estimated on a corresponding level with forest returns. Cash flow is denoted in annualised per hectare terms and is calculated on the basis of agricultural productivity with regard to soil quality commensurate with yield class classifications. In line

[14] See Annex for full set of equations used to calculate carbon pools.
[15] Teagasc (2020) Contracted Standing Sales (€/m3) by average tree size and per year https://www.teagasc.ie/crops/forestry/advice/markets/timberprices/#Annual%20timber%20prices
[16] Teagasc FIVE – specifies establishment and reforestation costs.

with Farrelly et al. (2011), SS yield classes are matched with corresponding NFS soil classes used to grade a site's productivity[17].

The appropriate discount rate in forestry is a matter for debate (Brukas et al., 2001), while Irish governmental guidelines provide no specific rate when analysing forest investments (Phillips et al., 2013). This paper follows Ryan et al. (2016) in applying a benchmark rate of 5% comprising an interest rate of 3% and a risk premium of 2%.

Forest investment projects are asymmetric in that costs are typically incurred early in the planning horizon while proceeds are deferred until timber becomes merchantable (Ryan et al., 2016). Meanwhile, rotations geared towards different management objectives such as carbon sequestration or wood product markets may entail contrasting planning horizons and cost-revenue schedules. To permit cross-comparison of investments with discrete time horizons, it is necessary to convert NPV to an annuity value that approximates cash flows for a defined period (Meadows et al., 2005). The NPV for all rotation cycles is thus standardised in terms of an equivalent annual value (EAV):

$$EAV = \frac{r(NPV)}{1-(1+r)^{-t}} \tag{3}$$

where r denotes the discount rate and t denotes the rotation period.

Results

In the results section, we attempt to understand the factors that influence the ORL of different stakeholders for wood and wood products. We consider a number of dimensions of the wood innovation system. These include

- The bio-physical volume of wood for timber mobilisation – relating to the volume of merchantable timber
- The processor – the market price of the timber purchased
- The producer – the profit a forest owner will receive for their harvest
- The externality – the carbon sequestration associated with planting and harvesting of wood and its use in differential wood products
- The social value – incorporating the private value to the processor and the value of the carbon sequestered in forests and stored in harvested wood products
- The substitution effect – the social value incorporating carbon emissions saved by substituting other fossil fuel uses for HWPs in fuel and materials
- The consumers – private and social value

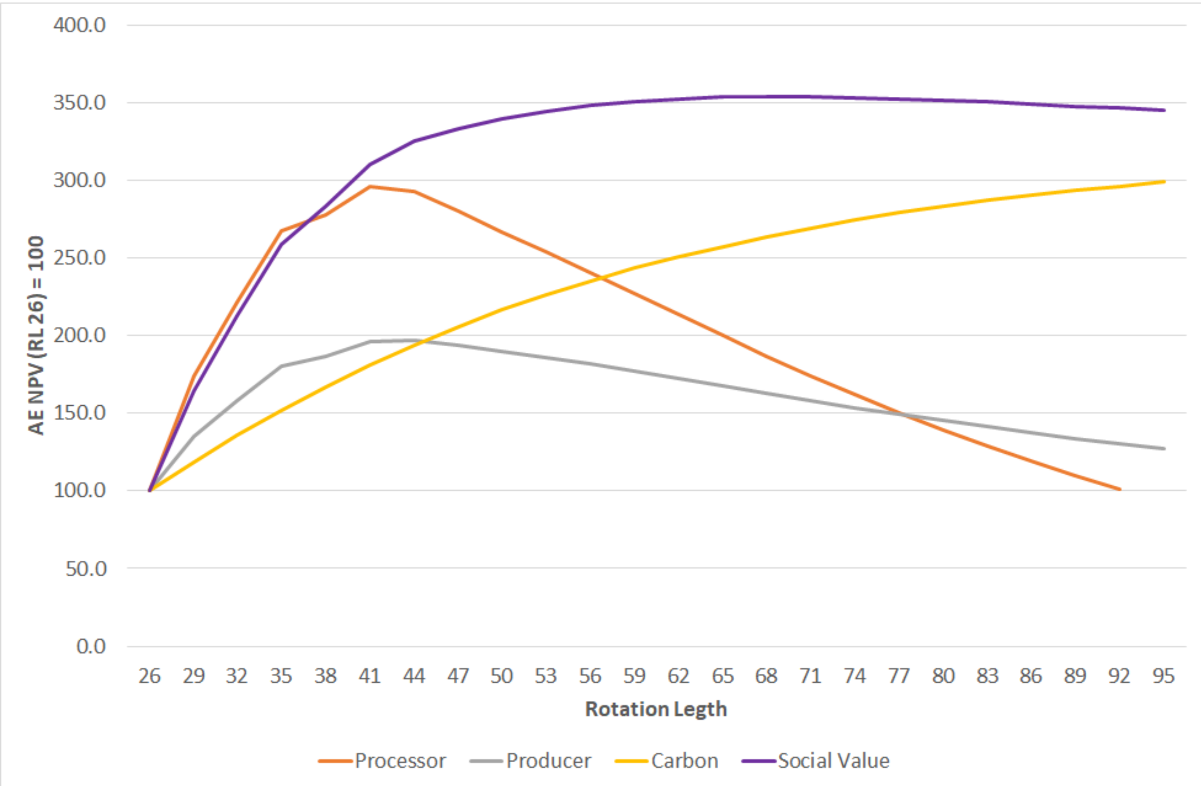
Each of these members of the forest innovation system has different needs, costs, and benefits. These differential factors discounted over time will result in different ORLs to optimise their specific preferences.

To identify the ORLs for different stakeholders, it is necessary to simulate the optima for each stakeholder. We utilise the ForBES bio-economic model to identify the average equivalent annual value (EAV) of the relevant optimisation measure for different stakeholders. Figure 1 highlights the trend in the EAV over different rotation lengths. As different control values have different scales, we rescale so that the value for a 26-year rotation length is set to 100. The curves assume a SS forest on yield class 20. The values utilise a discount rate of 5%. We note

[17] Under this arrangement, YC 24 is commensurate with farm soils (SC1) appropriate for a broad array of uses, while the most marginal farm soils (SC6) equate to SS YC (14).

that the trends for different stakeholders in the forest innovation system are quite different. The bio-physical curve for timber rises and then falls rapidly as the growth rate of the forest declines. Reflecting the combination of the price size curve and the merchantable timber volume, the processor curve grows at a similar rate to the bio-physical timber curve, but then falls rapidly as the price and volume gain from longer rotations declines. The curve faced by the forest owner or producer rises more slowly and then declines, reflecting the influence of subsidies. The bio-physical curve for carbon and the social value. based upon a carbon value of €32, exhibit a similar pattern, reflecting the fact that carbon sequestration is incorporated into both calculations. There is a higher growth rate for social value as the EAV incorporates carbon stored in harvested wood products, while the carbon curve represents the return to the producer, which grows more slowly.

Trends in the Annual Equivalent Net Present Value by Rotation Length (rotation length, 26 years = 100)



Note: The curves represent the EAV for a Sitka spruce on land with yield class 20.

Table 1 details the ORL for the different stakeholders using various yield classes and for two separate management choices (thin and no-thin). Unsurprisingly, the ORL declines as the yield class improves, reflecting the faster growth rate of the trees on better soils. In general, the ORL for thinned forests is slightly lower than for unthinned forests, demonstrating the higher growth rate achieved through thinning. It should be noted that the total timber biomass and associated carbon sequestration for thinned forests is lower than that of unthinned forests. For lower yield classes, the ORLs for both processor and producer are identical to the optimal length for timber growth. However, for higher yield classes, the ORL is in general lower for producers and processors.

For carbon, we find that the ORL is much higher than for timber, producer, or processor, reflecting the stored carbon in harvested wood products. Incorporating both the carbon and private values in the social value, we find a lower ORL than for the carbon optimum, albeit longer than for private returns. The higher the value of carbon, the longer the ORL, given the higher weight placed on the ecosystem service provided by the carbon sequestration.

Optimal Rotation Length by Yield Class (Thin and No Thin)

	Timber	Processor	Producer	Carbon	Social Value (20)	Social Value (32)	Social Value (100)	Social Value (163)
No Thin								
14	48	48	48	99	63	84	99	99
16	47	47	47	98	59	77	98	98
18	43	43	43	100	55	73	100	100
20	44	41	41	98	53	71	98	98
22	42	39	42	99	48	63	99	99
24	41	35	35	98	50	62	98	98
Thin								
14	47	41	47	98	62	77	98	98
16	45	42	42	99	57	72	99	99
18	42	39	39	99	57	69	99	99
20	41	35	41	98	44	68	98	98
22	40	34	34	100	43	58	100	100
24	40	31	37	97	40	55	97	97

Note: Discount Rate – 5%

Using a common discount rate when comparing different stakeholders may not be appropriate. A landowner planting a forest wishing to have a return during their lifetime may have a higher discount rate than an institutional investor investing in a timber mill or plantation[18]. Table 2 reports the outcome of a survey of average rotation lengths by forest contractors used to underpin timber forecast assumptions (COFORD, 2021)[19]. The surveyed rotation lengths are much shorter than the modelled ORL at a 5% discount rate, varying by six to eleven years, with lower yield classes typically having a higher gap than higher yield classes. The discount rate that would equivalise the ORL with actual rotation lengths for most yield classes is 14 - 15% except for yield class 14, where we compare an open-ended yield class of 14 or lower with an actual yield class of 14.

Discount Rate where rotation length is the equivalent to the surveyed Rotation Lengths

YC	Producer ORL	Actual Rotation Length (Survey)	EDR	dFR
14	51	42.6	10	-8
16	47	36	15	-11
18	43	36	14	-7
20	44	32	15	-9
22	42	32	15	-7
24	41	32	15	-6

[18] A prospectus for investing in a forestry fund indicated a rotation length of 30 years, a far shorter optimal length than this paper's analysis, indicating a higher discount rate for private investors https://www.irishtimes.com/business/difficult-seeing-wood-from-trees-in-funds-1.218720

[19] See http://www.coford.ie/media/coford/content/CofordAllIrelandRoundwoodBookREVISED150721.pdf

Note: Unthinned Sitka Spruce forests are assumed; ORL - Optimal Rotation Length; EDR - Equivalent Discount Rate; dFR - Difference from Financial Rotation

In Table 3 we look at the sensitivity of different discount rates in relation to the ORL of different stakeholders. Setting the discount rate at zero increases the optimal rotation length as a higher weight is placed on future gains relative to the present. Economic ORLs (processor, producer, and social returns) are higher than the bio-physical ORLs (timber, carbon) as the harvest return is later in the life cycle. Setting the discount rate at 4% or 15% makes little difference to the bio-physical ORLs, compared with the economic ORLs, with the higher discount rates for the processor and social returns compressing the range of ORLs between high and low yield classes.

Optimal Rotation Length by Yield Class and Discount Rate (No Thin)

Stakeholder	Timber			Processor			Producer			Carbon			Social Return @€32		
Discount rate	0	4	15	0	4	15	0	4	15	0	4	15	0	4	15
Yield class															
14	54	36	36	98	48	32	99	51	39	41	111	107	98	81	53
16	53	35	32	99	47	33	98	47	35	45	107	108	99	74	48
18	55	31	28	99	43	33	100	43	34	33	103	108	99	70	45
20	50	32	26	98	41	29	98	44	32	29	101	104	98	68	41
22	51	30	24	61	42	25	99	42	33	28	99	100	61	63	40
24	50	29	23	58	35	22	98	41	29	28	98	97	58	59	37

Stakeholders preferring different ORLs means that there may be variations in optimal value if a rotation is optimised according to the goal of another stakeholder. In Table 4, we calculate differences in stakeholder optima when first, the producer ORL is implemented and second, the preferred ORL reported in the producer survey is implemented. The results show the difference in annual equivalised net present value (AENPV) that would result if the producer's ORL is implemented.

For Table 4(a), we find that the processor would have the smallest difference in their AENPV, which is unsurprising given that processor payments for timber from the producer form a large part of the producer's financial return. The biggest difference is in carbon with the AENPV of sequestered carbon being about 50% higher for carbon's ORL. The differential for timber lies between the ORL for the processor and for carbon. As a combination of private and carbon goals, the social return also lies between the two, the differential growing with higher carbon values. When we compare outcomes with the rotation length described in the survey in Table 4(b), we find that the gap narrows for timber, but that for every other stakeholder, the gap widens significantly due to the earlier harvesting.

Proportional Difference Between Stakeholder Optimum and Forest Owner Optimum in Terms of Annual Equivalised NPV

YC	Timber	Processor	Producer	Carbon	Social Value (20)	Social Value (32)	Social Value (100)	Social Value (163)
(a) Using Producer Financial ORL								
14	0.162	0.000		0.559	0.000	0.074	0.144	0.385
16	0.180	0.000		0.547	0.000	0.038	0.102	0.351
18	0.195	0.039		0.536	0.001	0.045	0.100	0.339
20	0.343	0.109		0.401	0.002	0.006	0.043	0.239
22	0.225	0.061		0.489	0.003	0.023	0.065	0.288
24	0.223	0.071		0.493	0.004	0.024	0.063	0.283
(b) Using Producer Preferred ORL from Survey								
14	0.048	0.150	0.088	0.809	0.336	0.396	0.640	0.701
16	0.000	0.460	0.269	1.194	0.792	0.804	1.027	1.088

18	0.028	0.143	0.105	0.938	0.316	0.385	0.688	0.776
20	0.000	0.339	0.248	1.235	0.604	0.662	0.976	1.067
22	0.013	0.126	0.121	1.050	0.311	0.389	0.733	0.841
24	0.014	0.090	0.084	1.062	0.264	0.345	0.705	0.825

Discussion and Conclusions

This paper models the optimal rotation lengths (ORLs) of different actors in the forest innovation system. A key lesson is that the ORL varies for different stakeholders. While it is well known that the ORL for carbon is higher than that for timber (Price & Willis, 2011), This paper highlights additional differences that occur amongst other stakeholders. With the potential for a misalignment of preferences, it may be difficult to align the overall value chain around particular goals.

This paper utilises a simulation framework, employing forest bio-economic and policy models, together with farm micro-data, to produce a microsimulation system. The modelling framework simulates the impact of alternative rotation lengths on different outcomes including timber volume, processor value of harvested timber, forest owner incomes, carbon emissions and incomes adjusted for a carbon value.

The fundamental conclusion is that the ORLs vary quite significantly, depending upon the stakeholder. We assess the impact of a forest owner harvesting their timber at a time that was financially optimal for them. We found that that the processor would have the smallest difference in their annual equivalised net present value (AENPV), while the biggest difference in is carbon with the AENPV of sequestered carbon being about 50% higher than for forest owners. However, the financial optimum for a forest owner at a discount rate differs substantially from the actual rotation length on the basis of a survey. When we instead compare outcomes with this actual rotation length described, we find that the AENPV gap narrows for timber, but that for each other stakeholder, the gap widens significantly due to the earlier harvesting.

The discount rate used for cost benefit analysis in public sector projects in Ireland is 4%, which is much lower than the implied discount rate of forest owners (O'Callaghan & Prior, 2017). A policy maker or NGO concerned about long-run, climate related damage may have a lower discount rate than a market discount rate (Dasgupta et al., 2000; Goulder & Williams, 2012). This can be compared to the implicit social welfare equivalent consumption discount rate (Goulder & Williams, 2012), which varies from 1.4% (Stern, 2007) to 4.3% (Nordhaus, 2007). Indeed, the discount rate may also decline over time (Groom et al., 2005; Price et al., 2020).

This paper has highlighted substantially different harvesting optimisations for various stakeholders in the forest innovation system. This poses an important issue for new and existing timber value chains, where forest owners prefer to harvest much earlier than processors would like. Indeed, the preferred harvesting time is earlier than would be financially optimal for them. This reflects the long-term nature of forestry, meaning that forest owners may wish to harvest earlier in order to get a return during their lifetime (COFORD, 2018). 81% of forests planted since the 1980s have been established by farmers (DAFM, 2020), and given the age profile of the Irish farming population, it is reasonable to assume that many of those who planted in the 1980s and 90s are now likely past retirement age. Given that the average age of farmers has only continued to increase since that time, farmers planting forests now may be likely to want to harvest at an even earlier point in future years.

Earlier harvesting by forest owners also causes issues for processors. Processors prefer sawlogs to be cut at particular dimensions, with processing equipment and machinery calibrated to operate using sawlogs of particular standard sizes. Earlier harvesting by forest owners causes a shortage of supply for larger sawlogs as the harvested trees will not have had time to grow to the desired size (COFORD, 2018). Additionally, earlier harvesting causes an oversupply of smaller sawlogs, which are not suitable for processing into many wood products.

Maximising carbon sequestration requires much longer rotation lengths (c. 100 years) than those preferred by forest owners or processors (c. 40 years). Therefore, there is an obvious tension between achieving environmental goals and satisfying the goals of producers and processors. Large-scale afforestation is a key part of government strategy to reduce net GHG emissions and achieve net zero emissions by the year 2050 (Government of Ireland, 2021). As a result, incentives may be required to encourage landowners to plant forests with a view to optimising the amount of carbon sequestered rather than the value of the trees as timber.

To achieve a level of sequestration that would enable a net zero position by 2050, large-scale afforestation would have to begin almost immediately. However, afforestation levels in recent years have continued to decline, falling well below government targets (DAFM, 2022). Therefore, it may be time to reconsider the current Afforestation Scheme which concentrates on incentivising individual landowners, especially farmers, to plant. Achieving the ambitious afforestation targets needed to accomplish the net zero target for 2050 requires a scale of response which may be beyond the scope of thousands of private landowners acting individually, especially if they are operating on relatively short time horizons (Hoogstra & Schanz, 2009). As a result, it may be necessary for government itself, a State-sponsored body such as Coillte, or institutional investors, to undertake the afforestation required, bringing to bear the economies of scale required to accomplish such a task. Unlike individual actors, the State or long-term investors can operate with a much longer timeline and are not required to recoup investments in the shorter time frames that may motivate private landowners (Verhoeven et al., 2019). Transaction costs are also reduced by a consolidated approach, where instead of thousands of individual landowners being required to afforest in a very short time frame, a small number of actors are making the afforestation decision (Galik et al., 2009).

Ensuring the amount of forest land required for increased carbon sinks at both the Irish and EU level will require careful consideration of how new land is afforested but also how existing forests are managed (Yousefpour et al., 2017). The adjustment of rotation lengths away from historic norms may be a significant part of these updated forest management regimes (Kaipainen et al., 2004). With a departure away from purely commercial concerns, the forest innovation system will need to adjust to a new policy environment where climate change mitigation and adaptation, biodiversity, conservation, and sustainable use of forest resources are equally important (European Commission, 2021).

Accommodating the varied interests of actors within the innovation system will require cooperation between the various actors and leadership from formal institutions within the system. Kilcline et al. (2021) identify numerous issues relating to the Irish forest sector's innovation system, especially in relation to how network actors interact. The authors find that while formal institutions within the network acknowledge the need to foster a culture of collaboration and networks within the sector (through incentives to support owner and discussion groups), those interviewed as part of the research felt that the same culture of inclusiveness and co-learning was not evident in institutional communications with private small-scale forest owners, value chain actors and new emerging societal stakeholders. A lack of leadership and capacity among formal institutions and traditional forest stakeholders to meaningfully engage with emerging stakeholders and coalitions who value other ecosystem goods ahead of wood production was also evident. As a result, a great deal of work is required by system actors in order to coordinate actions within the sector and align forestry with other land use policies.

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Distributional Analysis of the Social and Private Return to Farm Afforestation, Accounting for the Cost of Carbon[20]

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Abstract

This paper examines the private (market and subsidy) and social (carbon) return from substituting conifer forest for the existing agricultural land use on one hectare across a national population of farms. A farm and forestry microsimulation framework is utilised, in tandem with a forest carbon bioeconomic model to model the average private and social returns incorporating forest carbon sequestration and displaced agricultural emissions and, across all farms systems over a 200 year life-cycle. The results show that replacing one hectare of agriculture with forestry results in net annual emissions reduction of 19.2 tCO_2e ha^{-1} (average annual equivalised net present value of carbon). However, these average values mask considerable heterogeneity as the farm distribution analysis shows that replacing agriculture with forestry on the most productive land would result in a greater reduction of net carbon emissions than planting the poorest land. This arises primarily as (a) carbon sequestration in livewood is greater on productive land and (b) substituted agricultural activity per hectare, and associated methane and nitrous oxide emissions are greater. Looking at private returns, only 32% of farms have higher returns from forestry than from agriculture. Substituting a carbon subsidy for an afforestation subsidy, at a low value €32), the social return for forestry mainly exceeds that of agriculture on the poorer soils. At a value of €100, the social return becomes positive and invariant to soil class, whilst for the highest value (€129), the social return is largest on more productive soils, clearly illustrating the substantial differential impact across farms, of different carbon values on the social return to planting.

Keywords: carbon sequestration, farm distributional analysis, land use change

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

Introduction

Historically, the focus of national forest programmes has been the production of timber products, with other non-provisioning objectives seen as complementary. However, the more recent emphasis on delivery of regulating ecosystem services such as climate change mitigation, has resulted in a greater policy focus on the social and environmental benefits of forests (Lawrence, 2020). In their ability to store (sequester) carbon in biomass and soils, forests not only generate a private market return but also generate a positive externality or benefit for society. However, greenhouse gas (GHG) emissions from industry, transport and agriculture give rise to negative externalities, incurring long-term costs for society as a whole.

Where the planting of new forests (afforestation) displaces emissions from superseded agricultural enterprises there can be an additive net environmental benefit. This will depend on the typology, extent and location of displaced agricultural activity, as well as the system boundaries applied when evaluating GHG emissions. While there is considerable literature that examines the private returns to afforestation, the modelling of carbon benefit in the land use change from agriculture to forestry is complex and has thus received little attention in the literature. This paper addresses a gap in this literature by valuing both the private and social implications of the land use change.

Agriculture comprises approximately 14% of global GHG[22] emissions (Tang et al., 2016). The generation of carbon sinks through land-use change based afforestation is a key mitigation strategy to achieving medium term net GHG stabilisation (Richards and Stokes, 2004, IPCC 2014). However, achieving greater farm afforestation depends largely on the relative market and subsidy returns to forestry and agriculture, but also on differential incentives that take account of the heterogeneity across the farm distribution (Ryan et al., 2018).

Studies that explicitly consider agriculture-forestry land use trade-offs tend to be undertaken at quite an aggregate level (Plantinga, 1997). While some models incorporate significant farm and spatial heterogeneity at a regional scale (Adams et al., 2011), other studies look in fine detail at agriculture and forestry under alternative policy regimes, but do not capture inter-farm heterogeneity (Nelson & Matzek, 2016).

Much of the focus on forest carbon in the literature is either on the loss of carbon due to deforestation (Paterson & Bryan, 2012) or the reporting of carbon stock change in forests, for national and international (IPCC[23] and UNFCCC[24]) reporting. There is also a literature that focuses on modelling carbon sequestration (Morison et al., 2012; Tobin et al., 2007) and the impact of forest management regimes on carbon sequestration (Sedjo, 2001; van Kooten et al., 2009; Im et al., 2007; Yemshanov et al., 2015; Duffy et al. 2020). Economic studies also consider the impact of the value of carbon, the length of the period under analysis and the impact of discount rate (Bateman & Lovett, 2000; Tol, 2018), while Duffy et al. (2020) use a

[22] Carbon dioxide (CO_2) is the most significant of the greenhouse gases, accounting for about two thirds of the total climate impact. Other agricultural greenhouse gases include methane (CH_4) and nitrous oxide (N_2O). The global warming potential (GWP) of CH_4 and N_2O are expressed in terms of carbon dioxide equivalents (CO_2e) where CO_2 has a value of 1, CH_4 has a value of 25 and N_2O a value of 298 (IPCC, 2014)
[23] Intergovernmental Panel on Climate Change
[24] United Nations Framework Convention on Climate Change

simplified measure of SCC to demonstrate the value of the net GHG abatement potential of a change in land use from agriculture to forestry using average agricultural and forest incomes.

However, these studies, while modelling differential returns, ignore and thus under-estimate significant heterogeneity in the returns both between and within different farm typologies. As farm afforestation decisions are made at farm level rather than at landscape level, this intra-area, inter-farm variability is critical in terms of elucidating the potential returns to the uptake of afforestation on agricultural land. In addition, the importance of knowing the location of individuals on a budget constraint is important in designing efficient incentives that can be targeted at specific points in the distribution (Moffitt, 1990). This gap in the literature is largely due to the complexity and micro-data requirements of modelling land use change at individual farm level.

In this paper we take a micro (farm) based approach to generate the private returns from farm afforestation, adjusted for the cost of carbon, which takes into account both the cost to society of increasing carbon emissions and the benefit of reducing emissions (Tol, 2018). The analysis here extends a farm distributional analysis on the private return to afforestation (Ryan et al., 2018), by incorporating the social return to a marginal land use change on a nationally representative sample of livestock farms in Ireland. Private forest returns are simulated using a bespoke forest carbon model based on forest yield models, while the agricultural economic and activity data are provided by the Teagasc National Farm Survey[25] (NFS) micro-data. Social returns (forest carbon storage and agricultural GHG emissions) are calculated using an inventory-based approach, as the product of activity data (for agriculture and forestry on a per hectare basis) and greenhouse gas (GHG) emission/land-usefactors. The value/cost of carbon benefits/emissions to society are then estimated by applying a range of carbon values (per tonne CO_2e).

This paper examines Ireland as a case study country where, despite having dramatically expanded forest cover over the last 30 years, incentivised annual afforestation still fails to meet national targets. In addition, there is growing policy interest in designing incentives to mitigate agricultural GHG emissions, which account for one third of national emissions and are driven by production for export. The Irish context and theoretical framework are presented in the next sections of the paper. The methodology utilised in the estimations is presented in section 4, with results and discussion presented in section 5. Conclusions are presented in section 6.

Context

Ireland faces a particular challenge in relation to meeting greenhouse gas (GHG) reduction targets. On the one hand, the agricultural sector is characterised by a heavy reliance on grass-fed ruminant livestock, which means that agriculture is responsible for almost one third of national GHG emissions[26] (NIR, 2018). EU GHG reduction targets for Ireland require a reduction of 20% by 2020 with a 30% reduction by 2030 (relative to 2005 levels). However, an increase in animal numbers since the removal of the dairy quota in 2015 has seen an increase in emissions (Donnellan et al. 2018), with further increases expected.

[25] The Teagasc NFS collects detailed information from a representative (by farm system and size) sample of farms in Ireland annually and collates these data as the Irish input to the EU Farm Accountancy Data Network (FADN). While the Teagasc NFS represents 90% of productivity, it doesn't represent small farms (<€8,000 standard output).

[26] The ruminant digestive system generates methane through the process of enteric fermentation, while the application of chemical fertiliser and organic manure (animal slurry/dung/urine) to land generates nitrous oxide (and ammonia) emissions. As ammonia is not a greenhouse gas it is not considered here.

On the other hand, over the last 100 years, Ireland has undergone a significant land-use change, expanding forest cover from 1% to 11%, with a government target to achieve 18% forest cover by 2046 (DAFM, 2015a). Irish forests are dominated by fast-growing even-aged conifer species, particularly Sitka spruce (SS) (*Picea sitchensis* (Bong.) Carr.), which is highly productive (in terms of both timber and carbon) on heavy mineral soils that are marginal for agricultural production (Farrelly, 2010). Dewar & Cannell (1992) reported that in the medium term (50 years), fast-growing conifers achieve the greatest sequestration, while in the long term (100 years), oak and beech (with above average growth rates) can store as much carbon as conifer forests.

The primary incentive for afforestation is the payment of an establishment grant and an annual subsidy for 15 years to compensate for the loss of agricultural income. Research undertaken by Ryan & O'Donoghue (2016) showed that whilst soil type, agricultural market income and level of subsidies all have an impact on uptake of afforestation, 84% of Irish farmers surveyed would not consider planting in the future. A meta-analysis of the reluctance to plant forests across Europe undertaken by Lawrence & Dandy (2014) revealed insufficient financial incentives, the permanence of the planting decision and a reluctance to plant 'good' land as common reasons for the reluctance to plant. However, with the mean annual carbon sink of Irish forests estimated at 2.1Mt CO_2e (valued at €97.4 million) (Lanigan et al., 2018), and a potential land pool of 1.3 m ha of grassland which is marginal for agricultural production but suitable for forestry (Farrelly & Gallagher, 2015), the impact of accounting for carbon in the return to the land use change warrants investigation at individual farm level.

Theoretical Framework: private and social returns from forests and livestock

Existing micro-economic studies of farm afforestation focus on different measures to assess the return from the afforestation of farmland. Adams et al. (1996) examined the per hectare Net Present Value (NPV) of forest and agriculture returns, while others have incorporated the average agricultural opportunity cost at farm system level (e.g. Herbohn et al., 2009), or analysed trends in the potential net farm afforestation returns over time (Upton et al., 2013). Here, we investigate the impact of substituting a Sitka spruce (SS) forest for an agricultural enterprise on a per hectare basis. This requires the determination of actual incomes at farm level and the subsequent simulation of counterfactual income streams across the farm distribution for the replacement of one hectare of livestock production (at the average farm gross margin) with one hectare of SS (of equivalent soil productivity). As microsimulation is increasingly used for analyses requiring the use of counterfactual data (Zander et al., 2007), this analysis utilises an over-arching microsimulation framework to incorporate outputs from a forest bioeconomic model and NFS micro-data to incorporate the cost of carbon in the private and social return to farm afforestation[27].

The components of agricultural and forest income to be modelled include:

- market and subsidy components of the replaced agricultural income
- forest market and subsidy income
- forest carbon sequestration and the emissions displaced from the superseded livestock enterprise
- future value of carbon.

[27] For the purposes of clarity and simplicity in this paper, we refer the reader to O'Donoghue & Ryan (forthcoming), which describes in greater detail the microsimulation model and the modelling assumptions used in the generation of the private and social income streams.

Forest income streams

Afforestation is an activity in which an upfront investment takes a long time to realise a return, given the length of time between planting and harvesting and the legal requirement to replant harvested forests (ISB, 2014). As a result, comparing relative returns from forestry and other land uses requires a net present value (NPV) framework, which presents the future return from forestry in present day values. The NPV of private forest income includes forest market income and forest subsidies, less the annual agricultural income foregone from the market and subsidies and is defined as:

$$NPV_{Private} = \sum_{t=0}^{n} \frac{ForestNetMarketIncome\ \ ha_j^{-1}}{(1+r)^t} + \sum_{t=0}^{n_j} \frac{ForestSubsidy\ \ ha_j^{-1}}{(1+r)^t} - \left(\sum_{t=0}^{n_j} \frac{FarmNetMarketIncome\ \ ha_j^{-1}}{(1+r)^t} + \sum_{t=0}^{n_j} \frac{FarmSubsidy\ \ ha_j^{-1}}{(1+r)^t}\right) \tag{1}$$

The value of forest carbon sequestered and displaced GHG emissions are included with the private returns as follows:

$$NPV_{Social} = \sum_{t=0}^{n} \frac{ForestNetMarketIncome\ \ ha_j^{-1}}{(1+r)^t} + \sum_{t=0}^{n_j} \frac{ForestSubsidy\ \ ha_j^{-1}}{(1+r)^t} - \left(\sum_{t=0}^{n_j} \frac{FarmNetMarketIncome\text{[28]}\ ha_j^{-1}}{(1+r)^t} + \sum_{t=0}^{n_j} \frac{FarmSubsidy\ \ ha_j^{-1}}{(1+r)^t}\right) + \left(\sum_{t=0}^{n} \frac{Value\ of\ net\ Carbon\ Sequestered\ by\ Forests\ \ ha_j^{-1}}{(1+r)^t}\right) + \left(\sum_{t=0}^{n} \frac{Value\ of\ GHG\ Emitted\ by\ Displaced\ Agriculture\ \ ha_j^{-1}}{(1+r)^t}\right) \tag{2}$$

As long-term forest NPVs are not directly comparable with annual agricultural incomes, the forest NPV is annualised to express it on the same basis as agricultural returns (Herbohn et al., 2009). The annual equivalised (AE) NPV is calculated on a per hectare basis as:

$$AE = \frac{r.NPV}{1-(1+r)^{-n}} \tag{3}$$

Forest carbon sequestration

Biogenic carbon sequestration describes the natural and anthropogenic mechanisms that enable the storage (sequestration) of carbon in oceans, soil, vegetation geological sinks (Sundquist et al., 2008). Over a forest rotation, there are gains (sequestration) and losses (emissions) of carbon through carbon fluxes (respiration/photosynthesis) between various terrestrial carbon pools and the atmosphere. Several carbon pools make up the total carbon stock that exists in forests and in harvested wood products (UNFCCC, 2014), namely:

- soil organic carbon (SOC)
- above-ground biomass (>7cm) and below ground (roots >5cm))
- dead organic matter (DOM) litter (decaying needles/leaves, branches <7cm diameter) and dead wood (above/below-ground >7cm)
- harvested wood products (HWP) (wood products in current use from domestic harvests)[29].

[29] 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 importance of these pools varies significantly between different forest types and there is considerable biophysical complexity (with consequent uncertainty) involved in the estimation and reporting of the carbon sequestered by these pools across the scientific and national accounting literatures.

Soils are the principal terrestrial store of carbon, which is influenced by soil type, climate and land use/land management. Forest ecosystems contain over half the global stock of SOC (Gross & Harrison, 2019) and it is estimated that soils and vegetation comprise 69 and 31% of forest carbon respectively (Dixon et al., 1994), while choice of species, forest productivity, management regime and rotation length all affect both the the forest carbon stock and the rate of carbon accumulation. The physical process of carbon sequestration takes significant time; Bateman & Lovett (2000) note that 95% of soil carbon change (peat and non-peat soils) is achieved within 200 years. Forests established on mineral soils can accumulate carbon gradually over time, while forests on peat soils incur carbon losses due to oxidation of carbon stocks following drainage.

The growth stage of a forest has a large impact on the rate of carbon sequestration. Saplings initially sequester little carbon but this increases rapidly to a peak in annual growth increment, before either ending abruptly at harvest or gradually declining as canopy closure occurs and forests reach a 'steady state' (or close to a steady state) where sequestration is balanced by emissions from tree mortality. As carbon accumulation largely follows annual volume increments (Morison et al., 2012), forests with high yield classes (YC)[30] have fast growth rates and greater carbon storage potential.

Forest biomass consists largely of merchantable timber volume (MTV) which is derived using forest yield models (Edwards & Christie, 1981). However, many other components of forests such as branches, tree-tops, roots (livewood) and dead trees and leaf/needle litter also contain carbon. The carbon in this additional biomass is estimated by applying a biomass expansion factor (BEF) to the timber volume. The carbon sequestered in live biomass is a function of timber volume, the density of the timber, BEF and its carbon fraction (Dewar & Cannell, 1992). Carbon mass per cubic metre in fresh wood is equivalent to about half of its basic density, expressed as tonnes of dry matter per m^3. However, there can be considerable variation in basic density between different species and growth rates i.e. a slow growing broadleaf such as Oak typically has a value of 0.56 compared to a value of 0.33 for Sitka spruce (Morison et al., 2012). Over time as the science has evolved, there has been considerable refining of the biomass expansion factors used in national carbon reporting While a static BEF was historically used for SS in Ireland (Lowe et al., 2000), BEFs have developed to reflect differential growth stages and yield classes (Tobin et al., 2007).

As it is important to differentiate between above and below ground biomass when estimating carbon in harvested wood products, a ratio of above to below-ground biomass of 0.2 is also applied to account for root biomass. In general, conifer species have a smaller root mass than broadleaves. Carbon estimations on a per hectare basis also need to account for unproductive forest areas to avoid over-estimating timber volume.

Dead organic matter (DOM) including leaf/needle litter and deadwood (arising from tree mortality) decomposes with a loss of carbon. The decomposition of litter in particular,

[30] YC: average annual timber volume production (measured in cubic metres per hectare ($m^3\ ha^{-1}$) and increasing with soil productivity.

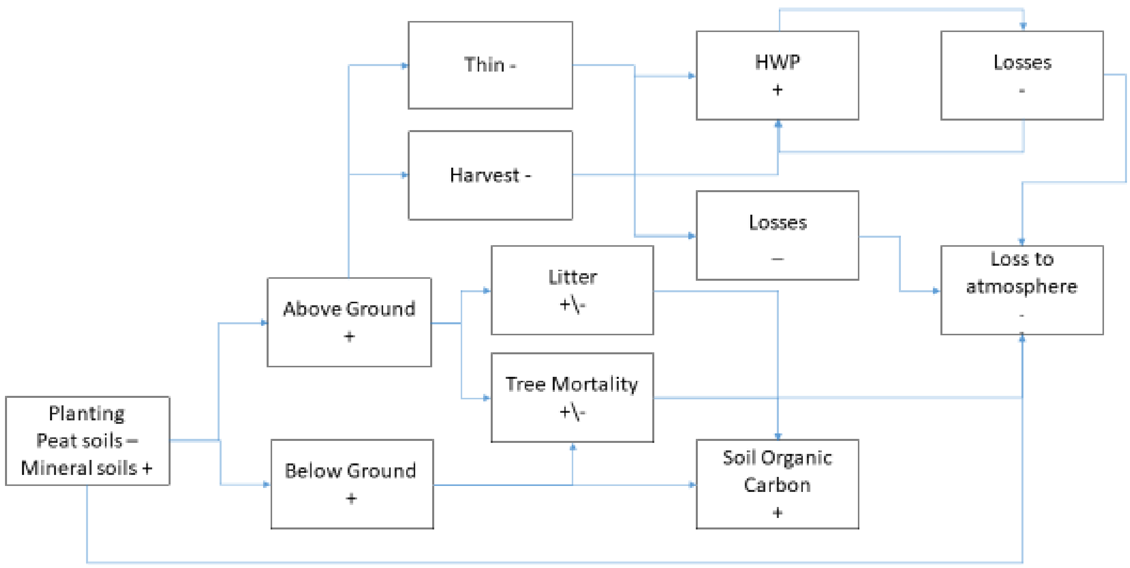
contributes indirectly to the soil organic carbon pool over long time-periods, however there is considerable uncertainty in relation to the mechanisms (Gross & Harrison, 2019).

As a general rule, Morison et al. (2012) report that while greater livewood sequestration can be achieved by thinning forests to promote growth, total carbon stock is maximised in unthinned forests, as the harvesting of thinned trees releases stored carbon to the atmosphere as CO_2 through the oxidation of wood biomass and deadwood over time (Bateman & Lovett, 2000). In addition, timber and biomass losses are incurred as a result of the harvesting process. These and other timber removals that end up as wood-fuel and bedding are oxidised, with a loss of carbon to the atmosphere. While there are advantages in the substitution of renewable wood biomass for non-renewable (fossil) fuels, this is beyond the land use change focus of this analysis.

Harvested wood products (HWP)

There can be considerable accumulation of carbon in HWP until the carbon is eventuallyliberated as products decay or enter waste streams. Further carbon can be oxidised during timber processing. HWP categories include sawn-wood (SW) which has the longest half-life[31], wood-based panels (WBP) with an intermediate half-life and paper and paper and paperboard (PPB) with a very short half-life. Dewar & Cannell (1992) suggest that the lifetime of HWP is similar to the rotation length i.e. approx. 40 years for conifers and 100-150 years for broadleaves. Figure 1 presents the carbon flow and indicative gains and losses for the forest carbon pools.

Forest Carbon Flow chart



Source: Authors
Note: + and - denote carbon storage and loss respectively

[31] Half-life (HL) is defined by the IPCC (2006) as the number of years it takes to lose one-half of the material currently in the pool. The HL of the SW and WBP categories is 35 and 25 years respectively. PPB is assumed to be oxidised.

Combining the estimations for the six forest carbon pools, the resulting total carbon (tC) represents the total annual carbon sequestration/loss for one hectare of SS forest and is converted from carbon to CO_2 equivalent (CO_2e). [32]

Agricultural incomes

Firstly, agricultural cost and revenue streams are calculated for each of six agricultural systems (dairy, cattle rearing, cattle other, sheep, tillage and mixed livestock) across six soil classes in order to generate agricultural farm gross margins in a given year. As afforestation generally takes place on a small proportion of farms (DAFM, 2015b), farmers continue to incur agricultural overhead costs after planting, thus forest and farm overhead costs cancel each other out and a variable costs and income approach to calculating farm income (market gross margin) is adopted. Combining subsidies with market gross margin, total gross margin is derived at individual farm level.

Agricultural emissions

The estimation of GHG emissions from agriculture (displaced on planting) as defined in by IPCC (IPCC 2013) incorporate a number of dimensions:

- methane (CH_4) $e$missions from enteric fermentation and manure management
- total direct plus indirect nitrous oxide (N_2O) emissions from various sources for both livestock (manure management, organic and mineral fertiliser application, dung/urine deposition) and tillage (organic and mineral fertiliser application, residue incorporation) activities
- carbon dioxide (CO_2) from fuel and other uses.

Of these, methane accounted for almost two thirds of Irish agricultural emissions in 2018, with N_2O comprising the bulk of the remainder (Duffy et al. 2020). The inventory-based approach to estimating GHG emissions from various farm sub-components (animal, housing, field etc) involves multiplying an activity (eg. animal numbers, amount of fertiliser) by an emission factor (eg. the amount of methane per animal or amount of N_2O per unit N applied). Animal numbers and age categories on individual farms are used to derive livestock (equivalent) units, where a dairy cow represents one livestock unit per hectare (LUha^{-1}). Thus, farms with a higher livestock density have greater total absolute livestock emissions. Howevermore highly stocked farms may have lower emissions intensities (emissions on a 'per kilo of product' basis) due to the fact that the increased rate of production can be higher than the rate of emission increase upon intensification (Lynch, 2019).

The actual agricultural emissions are derived by applying an IPCC inventory model (emission factors x activity data) for each component of the livestock (dairy, cattle and sheep systems) at individual farm level. The difference between the net total forest carbon sequestered and the displaced agricultural emissions gives the total abatement for one hectare of land converted from livestock agriculture to SS forest.

Valuation of Carbon

At the global scale, economic and policy discussions on climate change focus on the social cost of carbon (SCC) (Nordhaus, 2017). Thus, benefit-cost analyses to inform policy should incorporate estimates of the marginal value of changes in emissions i.e. the value/cost of global

[32] One tonne of carbon is equivalent to 3.67 tonnes of carbon dioxide.

damage caused by the impact of an additional tonne of CO_2 emitted at a particular time (Smith & Braathen, 2015). However, this calculation involves considerable complexity in the inclusion of the impacts of emissions, to the inclusion of economic damages from climate change (Nordhaus, 2017). In addition, valuing carbon is a difficult process due to the associated uncertainty, technical challenges and ethical issues (NESC, 2018), such as the definition of baseline emissions and lack of consensus around discounting.

Discounting determines how we weight the value today of costs or benefits in the future i.e. the weight to be given to future impacts rather than the valuation of those impacts (Kula & Evans, 2010). While there is a human preference for money now, the influential Stern report (Stern et al., 2007) argue that this is not relevant if weighting society's interests across future generations. The choice of discount rate has a dramatic effect on the estimated social cost of carbon, with some economists arguing that lower discount rates are more appropriate for longer time-frames. For example, 2017 estimates of US SCC change from $10 at 5% discount rate to $50 at 2.5%[33].

To calculate the social value of carbon in planting one hectare, we substitute a carbon subsidy (tonnes (t) of carbon equivalent x carbon value (v)) for the afforestation subsidy. For comparison purposes, we calculate the private return in terms of AE NPV. Thus, the carbon subsidy is calculated similarly:

$$AE(vCO_2e\ ha^{-1}) = AE(v.tCO_2e\ ha^{-1}) = v.AE(tCO_2e\ ha^{-1}) \quad (4)$$

Methodology and Data

While the impact of GHG emissions on global climate may persist for centuries, the cost of farm-level climate-change mitigation measures are immediate. In this analysis we value the social return from a farm-level externality, by broadening the private return to planting to incorporate the social value of carbon[34] in replacing one hectare of agriculture with a hectare of forest, on similar soils. The O'Donoghue & Ryan (2019) static forestry-microsimulation framework is enhanced to expand the Teagasc ForBES forest bioeconomic model (Ryan et al., 2016) to include forest carbon (C-ForBES), in order to calculate the private (financial) and social (carbon) returns from substituting one hectare of livestock agriculture with Sitka spruce forest[35] on individual farms in 2015.

As planting on undrained peats is no longer grant-aided, we assume planting on mineral sites (formerly grassland). As this results in little change in soil organic carbon and is difficult to estimate (Byrne & Black, 2001), the model assumes no change in SOC.

The relative productivity of agriculture and forest productivity is approximated using a categorisation developed by Farrelly (2011) that assigns SS Yield Class (YC) to the Teagasc NFS Soil Classes (SC) that represent the dominant soils on farms. In order to calculate the carbon sequestration for one hectare of forest (displacing livestock), NIR (2018) carbon emission factors for each carbon pool are applied to forest life-cycle data and farm-level activity (see table 8 - Appendix). For a detailed description of the methodology and modelling

[33] https://www.carbonbrief.org/qa-social-cost-carbon
[34] In this analysis, we focus only on the carbon externality, ignoring water quality, biodiversity, recreation and landscape externalities.
[35] For simplicity, this analysis models a forest hectare of 100% SS and ignores the required broadleaf/biodiversity component, due to inadequate data.

assumptions employed in this paper, the reader is directed to the companion methodological paper (O'Donoghue & Ryan, 2020).

The 'private return' is defined as the net value of the simulated annualised forest income stream for each farm (taking into account the annual agricultural opportunity cost) less the agricultural market gross margin (excluding overhead costs) and direct payments, using Teagasc NFS micro-data for 2015. The components of private forest returns, (costs, timber revenues and subsidies) are generated by the Teagasc C-ForBES and ForSubs models (Ryan et al., 2014), using the forest management assumptions detailed in table 7 – Appendix (for further detail see Ryan et al. (2016)). Forest merchantable timber volume (MTV) (from age of first thinning) is derived from Edwards & Christie (1981) static forest yield models and provides the volume inflow per hectare for both the private forest return and the forest carbon estimations. The conventional discount rate used for forestry in Ireland is 5% (Clinch, 1999), while the Department of Finance recommends a rate of 4% for discounting long term Public Private Partnerships. The calculation of the AE NPV for agricultural and forest incomes utilises a discount rate of 5%.

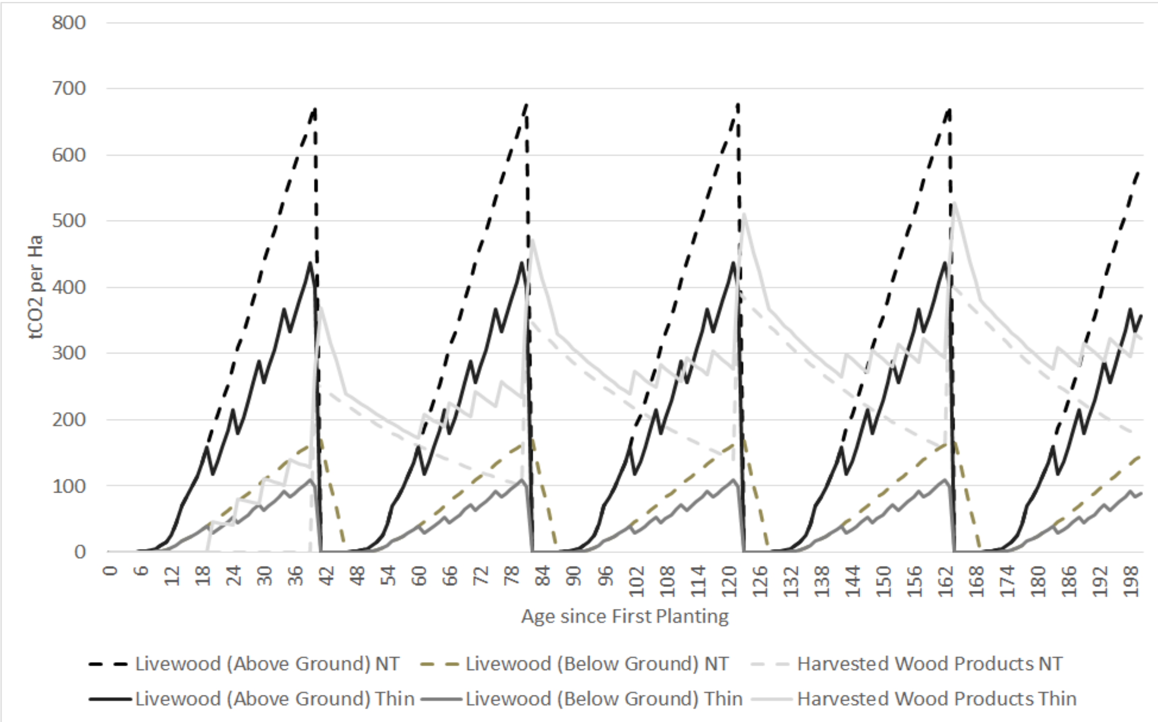
The sum of the forest carbon sequestered and the avoided agriculture emissions arising from the land use change is then discounted and annualised to produce the average discounted $tCO_2e\ ha^{-1}$ for each farm system and soil code/SS yield class. In calculating the social return, the forest subsidy is replaced with a carbon subsidy. The carbon subsidy is calculated by applying carbon values to the quantity of carbon sequestered. The carbon values utilised are the Irish government shadow price of carbon for non-ETS sectors for different years i.e. 2020 (€32), 2030 (€100), 2040 (€163) (DPER, 2019)[36]. Taking the long-term impacts of carbon losses (GHGs) and storage (long half-life HWPs) into account, this analysis is undertaken over 200 years to span multiple SS rotations and includes sensitivity analyses of the impact of carbon price and discount rate.

Results

Forest Carbon Sequestration

Carbon sequestration in livewood and HWP for Yield Class 20 thinned (rotation: 44 years) and un-thinned (rotation: 41 years) forests is reported in Figure 2.

Livewood and HWP Carbon Sequestration ($tCO_2e\ ha^{-1}$) for Thinned (44 years) and Unthinned (NT) (41 years) SS Forests (YC20)



The greatest increase occurs in above-ground biomass reflecting the smaller proportion of root biomass. On thinning, above-ground biomass decreases but increases again rapidly due to the reduction in inter-tree competition for light, moisture and nutrients. While each thinning/harvest timber losses, cumulative carbon in the HWP pool increases as all harvests are assumed to have the same share of final end-use. There is an immediate decline in HWP carbon relative to livewood, as a third is oxidised as woodfuel (Knaggs & O'Driscoll, 2017). While in practice, a greater share of wood energy may come from earlier thinnings, reducing the carbon flow to HWP, thinning practices vary and data are scarce. In general, unthinned forests sequester more carbon in livewood as thinned forests incur thinning losses, while thinned forests have the potential to sequester additional carbon in HWP. At final harvest, the above-ground biomass declines to zero, while the below-ground biomass decomposes, adding to the DOM pool over time.

Average private and social returns

Forest carbon sequestration and agricultural emissions by soil code/yield class are presented in Table 1, which shows greater average discounted carbon sequestration for higher yield classes reflecting more productive soils, with greater carbon sequestration in unthinned forests over the life-cycle.

For agriculture, there is a similar pattern albeit with negative values indicating carbon emissions. More productive soils carry higher livestock densities with consequently higher emissions. Typically, dairy farms on productive soils have higher livestock densities and fertiliser usage than other sectors, generating higher absolute emissions, (however the most intensive dairy farms may produce lower emission intensities (Lynch et al., 2019). The relationship is not however as clear for tillage farms, as the primary source of emissions is N_2O

from fertiliser application and tillage farms on less productive soils also carry livestock, potentially resulting in higher emissions.

5. **Comparison of Average Annual Equivalised Forest Carbon Sequestration and Agricultural Emissions ($tCO_2e\ ha^{-1}$) across all farm systems over 200 years (5% discount rate, 2015 planting year)**

	Forestry		Agricultural systems					
Soil code/ Yield Class/	No Thin	Thin	All Agriculture	Specialist Dairy	Cattle Rearing	Cattle Other	Sheep	Tillage
SC1/24	14.9	10.8	-6.5	-7.9	-6.1	-7.4	-6.6	-2.0
SC2/24	14.9	10.8	-6.3	-7.6	-6.1	-6.7	-6.6	-3.4
SC3/20	11.8	9.0	-5.6	-7.2	-5.0	-6.2	-4.8	-2.5
SC4/20	11.8	9.0	-5.6	-6.4	-4.6	-6.2	-5.2	-2.8
SC5/18	10.8	9.1	-3.5	-6.3	-3.3	-3.9	-2.2	-
SC6/14	7.8	5.4	-3.9	-	-3.2	-2.7	-5.5	-
Weighted Average	13.4	10.0	-5.8	-7.4	-5.2	-6.6	-5.0	-2.5

Table 2 reports the components of the private and social return per hectare assuming a proportionate per hectare displacement of agriculture with forestry across all farm systems. Columns A, B, C and D respectively report the average market gross margin and subsidies for agriculture and forestry, across all farm systems (focusing on unthinned forests to reduce complexity). Market gross margins for both agriculture and forestry are more correlated with soil type than with subsidies, which have a more redistributive focus. Looking at columns E and F, the average carbon sequestered per hectare on the most productive soil codes is more than double the agricultural emissions displaced across the farm systems[37], while for SC5, this ratio is greater than three.

6. **Components of the Average Annual Equivalised Private and Social Return to planting one hectare of forest (2015) over 200 year life-cycle at 5% discount rate**

Soil Code/ Yield Class	Agriculture		Forestry		Agriculture	Forestry No Thin	Private Return	Social Return[38]		
	A	B	C	D	E	F	C +D - (A+B)	C – (A+B) + (E+F)*M		
M	€	€	€	€	tCO_2e	tCO_2e	0	€32	€100	€163
SC1/YC24	1200	366	224	306	-6.5	14.9	-1036	-364	1065	2389
SC2YC/24	792	388	224	306	-6.3	14.9	-651	16	1434	2747
SC3/YC20	803	342	154	302	-5.6	11.8	-690	-142	1022	2101
SC4/YC20	731	351	154	302	-5.6	11.8	-627	-79	1085	2164
SC5/YC18	356	314	124	300	-3.5	10.8	-246	204	1161	2047
SC6/YC14	258	326	52	298	-3.9	7.8	-234	133	915	1639
Total	878	359	155	296	-5.8	13.4	-407	1106	2298	2298

Note:
A – Market Gross Margin per hectare (Agriculture)
B – Subsidies per hectare (Agriculture)
C – Market Gross Margin per hectare (Forestry) (No Thin)

[37] The farm system represents the dominant enterprise.
[38] In calculating the social return, the forest subsidy is replaced with a carbon subsidy based on the carbon sequestered at the shadow price of carbon for non-ETS sectors for different years i.e. 2020 (€32), 2030 (€100), 2040 (€163) (DPER, 2019).

D – Subsidies per hectare (Forestry) (applied for 15 years only)
E – tCO_2e – Agriculture
F – tCO_2e – Forestry
M – Monetary carbon value (€ tCO_2e^{-1})

The implication here is that the mean (discounted) net emissions reduction of replacing one hectare of agriculture with forestry would be 19.2 tCO_2e ha^{-1} over the life-cycle. However, this assumes acomplete displacement of agriculture across the afforested area and that no intensification would occur on the remaining agricultural area. Replacing agriculture with SS on better soils achieves greater emissions reduction, with poorer soils achieving lower reductions, due to the lower forest sequestration and displaced agricultural emissions – assuming that agricultural production is not displaced to other farms within Ireland.

From a monetary perspective, the private return to forestry across all farm systems, is on average lower than that for agriculture in the current policy (forest subsidy) environment. However, the situation changes when the annual forest subsidy is replaced with a carbon subsidy based on Irish government carbon prices. At the 2020 carbon value of €32 tCO_2e^{-1}), the social return for forestry mainly exceeds that of agriculture on the poorer soils. At a value of €100, the social return is positive and invariant to soil class, whilst for the highest value (€129), the social return is largest on more productive soils, clearly illustrating the substantial impact of different carbon values on the social return to planting.

Distributional returns

However, these mean values mask a wide and heterogenous distribution both withn and between different farm typologies. Table 3 presents the distributional assumptions in relation to private returns and social returns. On average 32.4% of all farms have positive private returns to planting. A carbon value of €32 tCO_2e^{-1} delvers a positive social return is 39.2% of farms, while for a value of €100, it rises to 52.8% and a carbon value of €163 results in over 90% of farms having positive social returns.

7. Share of Farms with a positive Private and Social return to Forestry

	Private Return	Social Return		
Forest Subsidy	1	0	0	0
Carbon Value (€ tCO_2e^{-1})	0	32	100	163
Proportion of Positive returns	0.324	0.392	0.528	0.932

Looking at this in more detail across farm systems in table 4, the average share of positive private and social return after planting varies substantially as only 2.4% of dairy farms have a positive private return, compared with over 40% for sheep and cattle systems and almost 30% for tillage systems. At a carbon value of €32 tCO_2e^{-1}, over half of cattle and sheep farms have a positive social return, while just over a third of tillage farmers and only 2% of dairy farms have positive social returns. For higher values the share of non-dairy farms with a higher social return from planting increases. At a carbon value of €163 tCO_2e^{-1} only 1% of cattle rearing and 20% of dairy farms don't have positive social returns to planting.

8. Share of Farms by Farm System where Annual Equivalised NPV (2015) for Forestry is greater than for Agriculture at different carbon values

Carbon value (€ tCO_2e^{-1})	0	32	100	163
Specialist Dairy	0.024	0.020	0.077	0.799
Beef (Rearing)	0.574	0.699	0.813	0.992

Beef (Other)	0.462	0.592	0.820	1.000
Sheep	0.444	0.506	0.752	1.000
Tillage	0.310	0.356	0.412	1.000
Total	0.323	0.392	0.528	0.932

The discount rate also has a large impact on the social return to planting as small changes result in large changes in the social return. The analysis thus far assumes a 5% discount rate. Table 5 highlights the sensitivity of the results to the discount rate utilised (at different carbon values). In relation to the private return, using a discount rate of 4% decreases the share of farms with greater returns from forestry than from agriculture, from 34% and 83% for SC1 and SC5 respectively, to 24% and 68% at 5%. As the discount rate increases, the share of farms with greater forest income decreases, while as the discount rate reduces, the share increases.

In relation to social return, the share of farms with where the annual equivalised NPV is higher for forestry, increases with carbon value and for lower discount rates. At a carbon value of €32 and 4% discount rate, 85% of farms have a higher social return from forestry. The share is lowest for dairy farms, albeit high at 77.5% for discount rates higher than 5%, the share for dairy drops below 70%. For carbon values of €100 or higher, virtually all farms at all discount rates, bar the top 2% of dairy farms have a higher social return from forestry.

9. Proportion of Farms with greater returns from Forestry than from Agriculture (Annual Equivalised NPV-2015) for a range of Discount Rates

Discount Rate	0	1	2	3	4	5	6	7
Private Return								
Soil Code (SC)								
1	1.000	0.993	0.806	0.532	0.341	0.238	0.180	0.131
2	1.000	1.000	0.928	0.780	0.514	0.321	0.173	0.113
3	1.000	0.993	0.831	0.710	0.493	0.389	0.260	0.220
4	1.000	1.000	0.887	0.696	0.538	0.390	0.242	0.208
5	1.000	1.000	0.923	0.894	0.806	0.683	0.633	0.574
6	1.000	1.000	1.000	1.000	0.838	0.682	0.682	0.319
Total	1.000	0.996	0.862	0.681	0.485	0.353	0.248	0.198
Carbon Value (€):	32							
Soil Code								
1	1.000	0.998	0.982	0.895	0.776	0.685	0.603	0.560
2	1.000	1.000	1.000	0.979	0.940	0.911	0.870	0.825
3	1.000	1.000	0.985	0.910	0.834	0.774	0.750	0.731
4	1.000	1.000	1.000	0.957	0.880	0.819	0.788	0.755
5	1.000	1.000	1.000	0.979	0.923	0.919	0.912	0.903
6	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Total	1.000	0.999	0.991	0.935	0.854	0.796	0.749	0.715
Carbon Value:	100							
Soil Code								
1	1.000	0.998	0.998	0.998	0.993	0.979	0.963	0.922
2	1.000	1.000	1.000	1.000	1.000	0.997	0.991	0.980
3	1.000	1.000	1.000	1.000	1.000	0.985	0.968	0.945
4	1.000	1.000	1.000	1.000	1.000	1.000	0.981	0.947
5	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
6	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Total	1.000	0.999	0.999	0.999	0.998	0.990	0.977	0.950
Carbon Value:	163							
Soil Code								
1	1.000	1.000	0.998	0.998	0.998	0.995	0.991	0.988
2	1.000	1.000	1.000	1.000	1.000	1.000	1.000	0.997

3	1.000	1.000	1.000	1.000	1.000	1.000	0.992	0.992
4	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
5	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
6	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Total	1.000	1.000	0.999	0.999	0.999	0.998	0.995	0.994

Carbon displacement capacity of SS forest

Finally, we examine how many hectares of livestock production could be displaced by one hectare of forest, to provide an indication of forest to livestock (grassland) ratio in a 'carbon neutral' scenario. Relating the livestock units per hectare on different soil codes to the corresponding forest yield classes, Table 6 reports the average carbon displacement capacity of one hectare of SS in relation to livestock density. For simplicity, values are reported for no thin (NT) forests for a land use change, with carbon sequestration from forestry and the displacement of GHG emissions from agriculture. One hectare of forestry replaces 4.82 livestock units on average, but with the number of livestock units displaced varying from 2.79 for the poorest soil types to 5.38 on the best soil types.

10. Areas of livestock production that could be offset by one hectare of SS at different livestock densities ($LUha^{-1}$), accounting for land use change by soil code

Soil Code/YC	Stocking Rate $LUha^{-1}$	$LUha^{-1}$ Forestry No thin	$LUha^{-1}$ Forestry Thin
SC 1/YC 24	1.57	5.38	3.85
SC 2	1.52	5.28	3.77
SC 3	1.33	4.32	3.34
SC 4	1.36	4.40	3.41
SC 5	0.87	3.67	2.90
SC 6	0.90	2.79	2.21
Average	1.41	4.82	3.57

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 the majority of tillage farms have livestock and tillage land is primarily used for animal feed, this is a reasonable assumption.

Discussion and Conclusions

This paper undertakes a distributional analysis of the private and social returns from planting one hectare of Sitka spruce to replace the current agricultural enterprise on each farm in a national population, based on agricultural micro-data, forest yield and bioeconomic models and national inventory report coefficients.

Examining averages across all farm systems over the life-cycle, the private return for forestry is lower than for agriculture. However, the lower private return to forestry is reversed when the existing forest subsidy is replaced by a carbon subsidy, so that the social return to forestry (displacing agriculture) is higher on many farms, even at low carbon values. For increasing carbon values of €32, €100 and €163 tCO_2e^{-1}, the share of all farms with a positive social return increases from 39.2% to 52.8% and over 90% respectively.

However, these average values mask considerable heterogeneity as when we look at the distribution of farms, the average share of positive private and social return varies substantially across farm systems with only 2.4% of dairy farms having a positive private return, compared with over 40% for sheep and cattle systems and about 30% for tillage systems. In particular, we see that the proportion of farms with positive social return is higher for cattle than for dairy farms, due to differences in private returns, even if the net sequestration is higher on dairy farms.

It is clear that the discount rate applied has a considerable impact on both private and social returns. In this regard, Kula & Evans (2010) advocate differential discount rates for forest and carbon returns.

It should be noted that these results represent a partial equilibrium approach, whereas in a general equilibrium approach, agricultural returns would be greater at higher carbon values, thus reducing the relative social return of forestry. However, it is likely that the qualitative conclusions would still hold, but that the gap would be different if there were endogenous prices.

On average, the analysis shows that replacing agriculture with forestry on comparable soils, would result in a reduction of net carbon emissions of 21.4 and 11.7 $tCO_2e\ ha^{-1}$ annually on the most productive and poorest soils respectively. In terms of livestock emissions, one hectare of forestry can offset between 2.79 LUha^{-1} on less productive soils to 5.38 LUha^{-1} on the best soils.

While thinned forests store more carbon, there is potential to increase sequestration in thinned forests by using timber in longer half-life products, particularly where such products are substituted for carbon intensive steel, aluminium and concrete and where timber products are recycled, closing the carbon cycle (Sathre and O'Connor, 2010). This highlights the need to extend the current work from a farm scale to a wider value chain scale.

This analysis extends the agriculture-forestry land use change literature by modelling differential returns incorporating significant inter-farm returns from agriculture, improving our knowledge of differential financial drivers of the 'attractiveness' of afforestation across the farm distribution. It also extends the literature on the micro-economics of farm afforestation by incorporating both private and social returns.

The modelling approach adopted here allows for the incorporation of carbon in a robust but stylistic assessment of the land-use change from agriculture to forestry. However, the analysis is limited by available data and the flexibility to consider different emerging management practices. In this context, a cross-national working group to standardise assumptions and parameters for land use change analysis would be useful. National emission reductions are based on the assumption that afforestation displaces agricultural production and associated emissions, outside of Ireland – reducing the €14.5 billion of annual agri-food exports from Ireland (Bord Bia, 2019). It has been suggested that this could lead to emissions 'leakage' internationally if livestock production is displaced to regions with higher emissions intensities. Sharma et al. (2018) showed that whilst afforestation is the most effective use of marginal agricultural land to reduce national environmental impacts, livestock production is the most effective option to reduce international impacts. However, such analyses are heavily dependent on assumptions about the environmental-intensity of global marginal production, and a recent review has suggested that such leakage effects are likely to be small (Emmet-booth et al., 2019). There is a need for high-level analysis of the wider, potentially cascading displacement effects associated with afforestation, to include aforementioned substitution of materials and fuels with wood and international displacement of livestock production in the context of future demand projections.

In conclusion, these results clearly show the importance of taking individual farms' differential positions into account in highlighting the variability of farm system and soil productivity impacts on the returns to planting. The incorporation of carbon value in the returns could change the attractiveness of forestry as an option for many farmers (depending on the value of carbon and how this is likely to change over time), and provides insights for the design of effective policy incentives to promote afforestation as an agricultural emissions mitigation option.

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Appendix

11. Assumptions/EFs used in carbon estimations

Component	Assumptions/Data inflow
Soil Organic Carbon	Assume planting on mineral sites only as little current planting on peats, therefore no change in SOC, however lower carbon sequestration from forests planted on peats can be expected given soil carbon losses due to drainage of peats.
Live biomass	Annual growth prior to first thinning is interpolated using a logistic function. Total forest carbon is calculated using timber volume per hectare (Edwards & Christie (1981), basic density value of 0.0387 (NIR, 2018), SS carbon fraction 0.5 (NIR, 2018) and dynamic biomass expansion factors (BEF) from Black et al. (2004) and NIR (2018) varying with age classes from 1.68 to 4, reverting to a BEF of 1.68 once stand volume exceeds 200 $m^3 ha^{-1}$. Productive area is 85% (Ryan et al., 2016). Above and below-ground biomass differentiated using 0.2 root ratio (NIR, 2018).
Dead Organic Matter DOM	Leaf/needle litter modelled according to NIR (2018) algorithms, applying Tobin et al. (2006) foliage rates and NIR (2018) decomposition rate. Deadwood modelling assumes mortality rate of 1.6% yr^{-1} (NIR, 2018) with decomposition rate of 14% yr^{-1} (Black, 2016).
Losses (applied to MTV)	Harvest losses: Thin: (first: 14%, second: 12%, subsequent: 9%) Final harvest: 5% (Ryan et al., 2016). Energy losses: 34% allocated to woodfuel (Knaggs & O'Driscoll, 2017)
Harvested Wood Products	Inflow to HWP modelled as MTV less losses. Allocation of harvested timber to sawnwood, wood-based panels and paper taken from NIR (2018) are 52%, 48% and zero respectively (no longer any paper production). Liberation of carbon from HWP over time is modelled according to Pingoud & Wagner (2006) as used in NIR (2018).
Total forest carbon	Combining the forest carbon pools, total forest carbon is expressed as $tCO_2e\ ha^{-1}\ yr^{-1}$ using IPCC conversion factor (3.67).

12. Agricultural Carbon Emission Factors and GHG carbon dioxide equivalent conversion factors)

	Energy CO_2	Energy CH_4	Energy N_2O	Agri CH_4	Agri N_2O	Transport	tCO_2per Animal Number	tCO_2 per LTU	KtCO_2 per €m
Dairy				0.113	0.000		2.863	2.863	
Cattle				0.047	0.000		1.216	2.767	
Sheep				0.006	0.000		0.149	0.667	
Fuel	0.004	0.00000041	0.00000005						0.004
Fert					0.022				6.680
Crops					0.000				0.088
CO_2 equivalent conversion factor	1	25	298	25	298				

Source: EPA CRF2010_2016submission v2.xlsx

Case Study of Bioenergy Crops in Ireland

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"Bottom line - if it was a financially viable crop you would have an industry"

Introduction

The world population continues to increase which is leading to an increasing demand for food, fuel and fibre (Ajewole and Larwanou, 2019). It is accepted that these demands must be met more sustainably, with benefits for people, production and the planet (Rose et al., 2021). However, climate change presents one of the greatest challenges to the sustainable intensification of agriculture as well as being considered one of the gravest challenges to ever face humankind (Unruh, 2002). In light of this, the Paris Agreement (UNFCC, 2015) and the United Nations Sustainable Development Goals (UN, 2020) for example, highlight the need for urgent action and ambitious targets have been set e.g. to keep the increase in average global temperatures below 1.5C above pre-industrial levels (UNFCC, 2015).

From an agricultural perspective there are many developed and emerging technologies/innovations that if adopted can play a role in reducing greenhouse gas (GHG) emissions (Klerkx and Rose, 2020; Mulkerrins et al., 2021 under review). Bioenergy production is one such innovation (O'Connor et al., 2021). Bioenergy, similar to energy produced by wind, solar, geothermal tidal, wave and hydroelectricity, is considered a renewable energy source that is generated from the burning of biomass fuel (SEAI, 2021). Biomass fuel sources include wood residue, farm waste, energy crops and organic wastes. Biomass fuel produced from these sources can take a number of forms including; biogas and biomethane, solid fuel (e.g. wood pellets and wood chips), liquid fuel for transport (SEAI, 2021) all of which can be used as substitutes for fossil fuels for energy production (Chiodi et al., 2015; SEAI, 2019).

This paper aims to explore Irelands experience with energy crops as a case study to inform future policy in both Ireland and internationally in relation to bioenergy. The following sections will discuss the contextual background and Irelands recent history with energy crops.

2. Contextual background

2.1 Agricultural context

Pasture-based ruminant livestock production is the dominant form of agriculture in Ireland (O'Mara et al., 2021). Internationally the grazing of ruminant livestock is in decline (Reijs et al., 2013; Shortall, 2018), however, countries like Ireland and New Zealand are an exception to this (Roche et al., 2017). Irelands temperate climate allows for large quantities of grass to be grown (O'Donovan et al 2011) which makes it the cheapest feed available for farmers (Finneran et al., 2011). Therefore, unsurprisingly, over 90% of the land area is grassland and used for livestock production (CSO, 2016) which contributes positively to the 'green' brand image of the agrifood sector (DCCAE, 2019). Despite high levels of environmental efficiency (Leip et al., 2010) agriculture in Ireland accounts for 33% of all GHG emissions (Lanigan et al., 2018). This makes the contribution of agriculture in Ireland to national emissions the highest in the EU, much higher than the 10% average in other member states (Fellmann et al., 2018).

Ireland aims to reduce GHG emissions by 30% by 2030 and supports the net zero target by 2050 (DCCAE, 2019). Paradoxically, in line with national plans (DAFM, 2015), the Irish dairy industry has went through a rapid phase of expansion since the abolition of milk quotas in April 2015 (Ramsbottom et al., 2020). A further increase in dairy cow numbers is expected in the period up to and including 2027 (Teagasc, 2020). Recent research has shown that replacing dairy farms with forestry would have positive benefits in terms of carbon reduction (Duffy et al., 2020). Considering that dairy farming is typically associated with better land types (Duffy et al., 2020) and is consistently the most profitable farming enterprise (Donnellan et al., 2020) it is unlikely that such a land use change would be realised. With scope for Irish farmers to improve grass utilisation (Hanrahan et al., 2018) an alternative and potentially more attractive proposition for farmers would be to use this additional grass grown and/or their livestock slurry for anaerobic digestion (AD) while maintaining the same livestock numbers (SEAI, 2017).

Smyth et al., (2011) argue that countries with temperate climates like Ireland have a strong potential to produce grass biomethane. AD has the potential to make a substantial contribution to the primary energy supply in Ireland by 2050 (Schulte et al., 2013), however, to do could require ~900 AD plants (SEAI, 2017). Despite the potential benefits to farmers, food processors, the local community, environment and government (Caslin, 2018), the AD industry in Ireland is in its infancy (Goulding and Power, 2013; Lanigan et al., 2018; EPA, 2020). It is argued that for AD to prosper will depend on favourable national policy and financial incentives (EPA, 2020) as relatively long payback periods may deter potential investors (O'Connor et al., 2020). The Irish governments Climate Action Plan indicates that such supports and incentives will be provided through a rollout of the Support Scheme for Renewable Heat (DCCAE, 2019). Considering Ireland has experience with incentivising bioenergy crop production, in particular Miscanthus and Willow, we hypothesise that there may be relevant insights and learning from that experience for future bioenergy policies.

2.2 Miscanthus and Willow

Miscanthus x giganteus (hereafter Miscanthus) is a relatively high yielding perennial, woody rhizomatous grass species originating from south-east Asia with an estimated lifespan of 10-15 years (Jones and Walsh, 2007; Clifton-Brown et al., 2008). Miscanthus is now common in many other parts of the world including Europe and once mature can be harvested annually when the crop has senesced (Lewandowski *et al*., 2000). Spring time is recommended for harvesting as the crop will have a lower moisture content (Bradshaw et al., 2010) which makes the crop easier to store while also increasing the calorific value (Caslin et al., 2015A). From an Irish perspective most miscanthus was supplied to the Bord na Móna power plant in Edenderry, Co. Offaly (Caslin et al., 2015A).

Willow

The Irish Department of Agriculture, Food and the Marine (DAFM) started administering a Bioenergy Scheme in 2007 which provided grant-aid to support the establishment of miscanthus and willow. This scheme operated as an exchequer funded pilot scheme until 2009 and from 2010 to..2013? was an EU co-funded measure under the Rural Development Programme. By the end of 2013 the scheme had grant-aided the establishment of over 2,400ha of miscanthus and over 900ha of willow (Teagasc, 2014). However, as positive as that may appear it is also estimated by Teagasc (the Irish Agriculture and Food Development Authority) that in 2013 500ha of miscanthus was removed (TSECDG, 2014). A *"Miscanthus Best Practice Guidelines"* document published in February 2015 highlighted that a market supply chain *"had failed to materialise in any significant way"* and farmers were advised that unless they were in close proximity to the power plant in Ednederry that it was not economical to transport the fuel (Caslin et al., 2015A).. At the time of writing the authors estimated that a quarter of the miscanthus that had been sown had been removed due to a lack of a market and there were concerns over the €7 million that had been invested being wasted if the remaining crop were to be removed (Caslin et al., 2015A). Presumably due to these challenges when the DAFM launched a third phase of the Bioenergy scheme in 2015 only Willow was being grant-aided (REF).

Nonetheless, energy mitigation measures, which include specific reference to the use of short rotation coppice and miscanthus, have been identified as having potential to abate GHG emissions in Irish Agriculture between 2021 and 2030 (Lanigan et al., 2019). A recent roadmap produced by the DAFM *"Ag Climatise - A Roadmap towards Climate Neutrality"* sets out to develop a climate neutral food system that is compatible with the temperature goals set out in the Paris Agreement. Renewable energy is an important aspect of this roadmap and the agriculture sector is cited as having a key role in the provision of bioenergy feedstocks for the production of biogas/biomethane to help decarbonise the transport and heat sectors in particular. The document cites policy developments such as the Renewable Electricity Support Scheme (RESS), the Support Scheme for Renewable Heat (SSRH) and the Biofuels Obligation Scheme as being key drivers of bioenergy feedstock use (DAFM, 2020).

Considering the relatively low uptake and success of the bioenergy schemes of the past, and the aspirations that bioenergy will still play a role in Ireland and Irish agricultures efforts in relation to our climate change obligations, the aim of this paper is to learn from past experiences to inform successful bioenergy policies in the future. The utility of the study will not be limited to bioenergy as insights will be applicable to other sectoral and/or technology interventions.

3. Analytical approach

3.1 Technology adoption

In terms of behaviour change and technology adoption the literature suggests that although profitability is important, it is often not the primary motivating factor for farmers (Vanclay, 2004; Weersink and Fulton, 2020; Mulkerrins et al., 2021 under review). There is a large body of research on the wide range of factors that influence a farmers decision to adopt, not to adopt or indeed to discontinue the use of particular technologies have been identified (Rogers, 2003; Leeuwis, 2004; Kelly et al., 2016; Daxini et al., 2018; Montes de Oca Munguia & Llewellyn, 2020; Mulkerrins et al., 2021 under review). These factors can be linked to characteristics of the innovation, the adopter or the context of the adopter (Montes de Oca Munguia and Llewellyn, 2020; Pannell and Zilberman, 2020). Farmers often have legitimate reasons for non-adoption (Vanclay, 2004) and it is recognised that decision making in relation to the adoption of agricultural innovations is not an event but rather it is a socio-cultural process (Vanclay, 2004; McAloon, et al., 2017; Pannell & Zilberman, 2020).

Due to the scale and complexity of the problems facing agriculture the linear “top-down” transfer of technologies is deemed unsuitable (Vanclay, 2004; Wood et al., 2014). This model assumed that new technologies developed by scientists and promoted by extension agents (i.e. agricultural advisors) would be adopted by farmers (Leeuwis, 2004; Edquist, 2013). In contrast, the innovation systems (IS) approach has become increasingly popular and as it acknowledges that technological change in agriculture is a complex process of interactions between heterogenous actors (Davis et al., 2008).

3.2 Innovation systems

The World Bank defines IS as *“a network of organisations, enterprises and individuals focused on bringing new products, new processes, and new forms of organisation into social and economic use, together with the institutions and policies that affect their behaviour and performance”* (Hall et al., 2006, p.18). IS are an important determinant of technological change (Hekkert et al., 2007) and therefore agricultural innovation systems (AIS) have become a popular approach to understand and facilitate agricultural innovation (Turner et al., 2016; Klerkx and Begemann, 2020). AIS thinking can be can applied at a national or sectoral level or to a particular technology (Lundvall, 1992; Lamprinopoulou et al., 2014; Kilkline et al., 2021). Evidently, different IS approaches can be employed depending on the focus of the study.

Mulkerrins et al., 2021 (under review) investigated the factors that influenced Irish farmer’s engagement with 6-week calving rate, a key performance indicator for pasture-based dairy systems. Similarly, Augestenborg et al. 2012 surveyed Irish farmer’s opinions on energy crop production. Both papers provide policy recommendations for the respective enterprises, however, Mulkerrins et al. 2021 (under review) also suggest the merit for future research in relation to agricultural technologies and their adoption to take an IS perspective. Such perspectives in an Irish context are limited (Kilkline et al., 2021) and to the best of the authors knowledge there is no published research on such perspectives in relation to bioenergy crops in Ireland. We propose to employ a technological innovation system (TIS) framework, which, as the name suggests, are centred on a technology and has been previously applied to understand the development of renewable energy technologies (Huang et al., 2016; Edsand, 2017; De Oliveira and Negro, 2019; Furtado et al., 2020). We hypothesise that there are lessons to be learned from Irelands experience with energy crops. Ireland has previously provided incentives to encourage farmers to produce biomass for bioenergy.

Theoretical Framework

Origins of the Literature

Key Theoretical Concepts

Methodology

- Propose to collect data using semi-structured interviews or BNIM (to be confirmed)

- Data also available from Stephen Robbs Master's thesis that may be relevant and it is unpublished thus far?

What methodologies are used to analyse the questions

- Data or information collection

What are the main uses of the methodology

What are the main findings of papers that use the methodology

Discussion and Conclusions

2.2. Anaerobic Digestion in Ireland

Smyth et al., (2011) argue that countries with temperate climates like Ireland have a strong potential to produce grass biomethane. AD has the potential to make a substantial contribution to the primary energy supply in Ireland by 2050 (Schulte et al., 2013), however, to do could require ~900 AD plants (SEAI, 2017). Despite the potential benefits to farmers, food processors, the local community, environment and government (Caslin, 2018), the AD industry in Ireland is in its infancy (Goulding and Power, 2013; Lanigan et al., 2018; EPA, 2020). It is argued that for AD to prosper will depend on favourable national policy and financial incentives (EPA, 2020) as relatively long payback periods may deter potential investors (O'Connor et al., 2020). The Irish governments Climate Action Plan indicates that such supports and incentives will be provided through a rollout of the Support Scheme for Renewable Heat (DCCAE, 2019).

Recent research points towards small-scale anaerobic digestion (SSAD) as a promising, economically sustainable technology that commercial Irish dairy farmers with >100 cows could implement to mitigate GHG emissions (O'Connor et al., 2020). Although the potential profitability from the AD of slurries, grass and grass silage may appear attractive (Lanigan et al., 2018; DCCAE, 2019) a number of non-financial barriers to the development of AD have been identified and must be considered for the successful diffusion of AD. These barriers can be summarised under the headings of; feedstock supply, technology and infrastructure, regulatory/financial and behavioural (SEAI, 2017; Scott and Blanchard, 2020). However, according to O'Connor et al., (2020), there is a need for a greater understanding of Irish farmers perceptions of AD, the characteristics of potential adopters, adoption rates and barriers to adoption.

Thoughts on usefulness of methodology

Limitations of the methodology

Ways in which the methodology can be used in your thesis.

References

Contacts in the Teagasc/AFBI Guidline Documents

https://www.teagasc.ie/media/website/publications/2011/Short_Rotation_Coppice_Best_Practice_Guidelines.pdf
https://www.teagasc.ie/media/website/publications/2011/Miscanthus_Best_Practice_Guidelines.pdf

Miscanthus	Willow
Teagasc – Barry Caslin/John Finnan (RIP)	Teagasc – Barry Caslin/John Finnan (RIP)
AFBI – Chris Johnson (NI)	AFBI – Allistair McCracken, Chris Johnson and Linda Walsh (NI)
DAFF Biofuels Policy Unit Portlaoise	DAFM – Danielle Coll, Yvonne Alford
CAFRE Loughry Campus (NI)	CAFRE Nigel Moore Greenmount
SEAI – Pearse Buckly	SEAI – Mathew Clancy
Department of Agriculture and Rural Develoment (UK)	Department of Agriculture and Rural Develoment (UK)
Quinns of Baltinglass, Wicklow	Plant Breeders in Sweden and UK X4
Bord na Móna Energy – Tracy Leogue	Bord na Móna Energy – Tracy Leogue
Biomass Energy Northern Ireland - Tyrone	Suppliers of planting material Agrimann (BM) Ltd – Kells, Meath Billy Kelly Nurseries Flynns of Mullingar Seed Technology
Biotricity Rhode, Offaly	Hegan Biomass – Alan Hegan – supplier of planting material and planting/harvesting equipment
McMahon Eco Fuel Manufactures LTD Limerick	Harvesting equipment in ROI – Bailey Agri and Biofuel (Direct Chip), Portlaoise Teefast Ltd Kells, Meath Joe Bermingham Tuallamore (Direct Chip)
IFA – Blueball, Dublin (know Joe Healy from Galway Grazers group meetings)	Chipping Services Ballynoe Agri Services – Conna, Cork Greengrove- Mark Hanly Strokestown, Roscommon Clare Woodchip Limited Feakle Contessa Trading James Fitzgerald Castlerea Ecwood Energy Malachy McCann, Donegal
Irish Bioenergy Association	

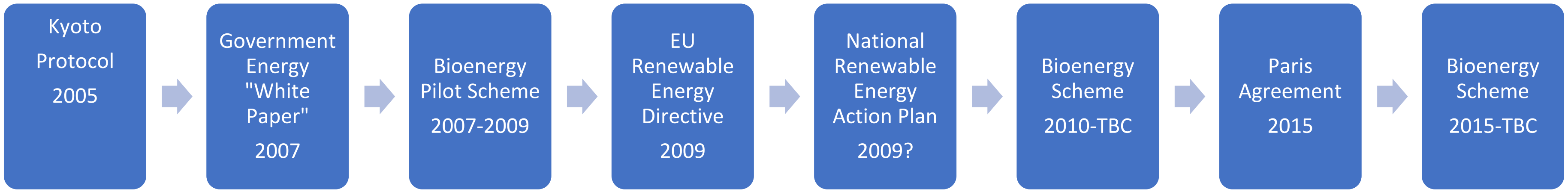

Fig.1 Initial draft timeline of relevant policies to energy crop production in Ireland

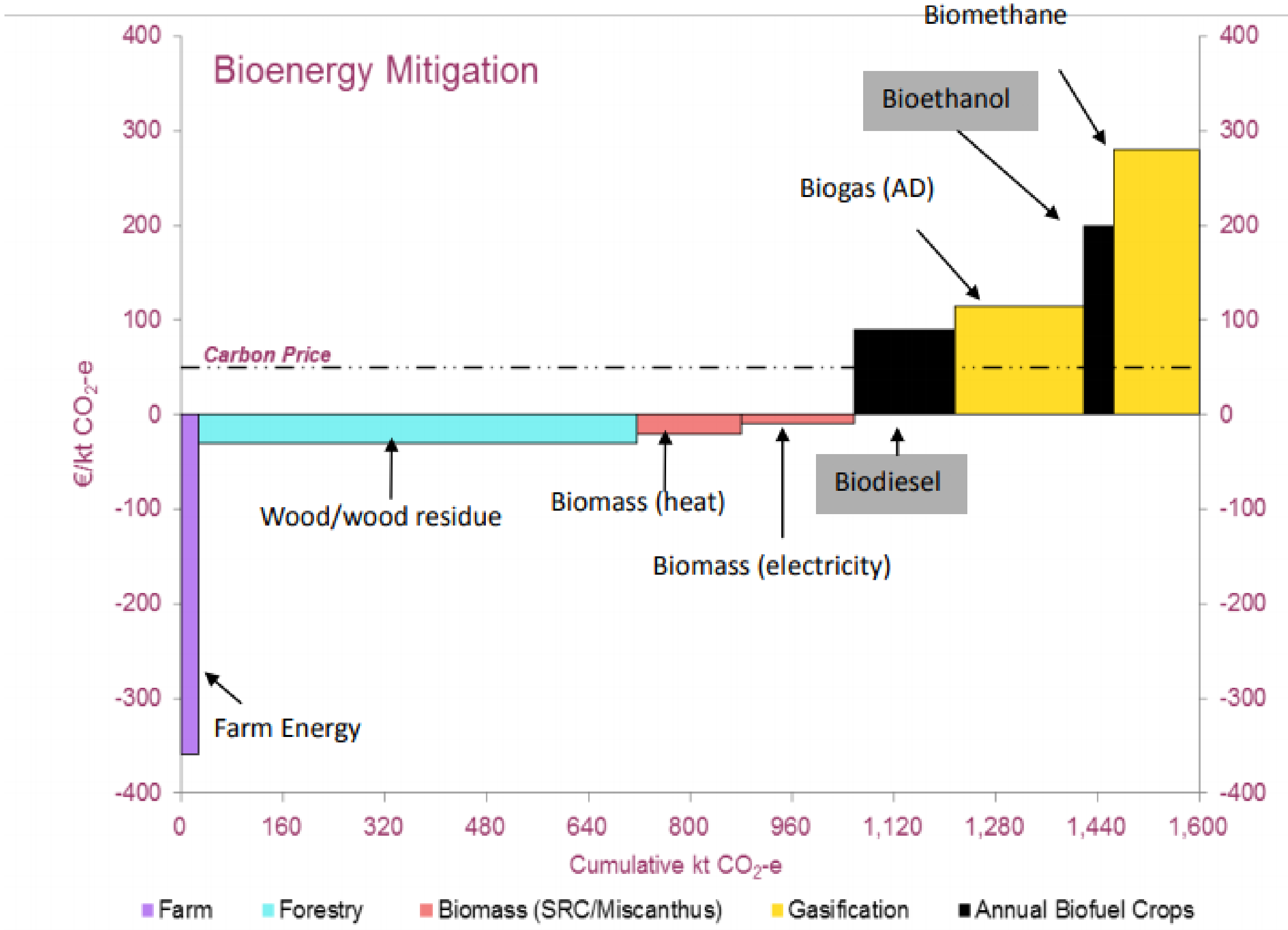

Figure 3.3: Marginal Abatement Cost Curve for agriculture for 2021-2030 for bioenergy produced in the agriculture and forestry sectors. Values are based on linear uptake of measures between 2021-2030 and represent the mean yearly abatement over this period (Abbreviations: AD = Anaerobic digestion, SRC = Short Rotation Coppice, OSR = spring/winter oilseed rape). Dashed line indicates Carbon cost of €50 per tonne CO_2. *Note: Bioethanol/biodiesel does not meet RED II sustainability criteria at present.*

23. Renewable Energy Production

One of the other targets in the 20:20:20 climate change strategy is an increase in the amount of renewable share of energy consumption by 20 per cent. Irish national policy is for 40 per cent of electricity consumption to come from renewable sources by 2020. The share has risen over time from four per cent in 1990 to about 20 per cent in 2013. Eighty per of renewable electricity comes from wind. In terms of wider energy usage, electricity accounts for about 60 per cent, with the remainder coming from heating and transport, with bioenergy being important.

Ireland has many advantages for renewable energy production from abundant wind, wave and tidal power. In some cases the economics do not yet stack up, particularly on the marine side, albeit with a closing gap with non-renewable energy. For land based wind, policy incentives have enabled the development of a large wind-power industry in Ireland.

However one of the challenges for renewables is that the main gains may not coincide with where the costs lie; wind turbines and other renewable resources impose a cost on the local environment, whether it be through potential for noise, landscape impairment or additional local traffic.

Often the main gains are land price sales or rent, plus employment in construction and then employment for ongoing maintenance. Typically the equipment and many of the services used in production are sourced elsewhere. Once the facility is operational, most of the incomes will accrue to the investors. Therefore there may be a disconnect between the winners and losers.

A former student, Niall Farrell in his PhD found that once you factor in the cost of higher electricity to subsidise renewable energy and also factor in where the employment and gains would be, it was in fact urban areas that have the highest net gain from marine renewable energy deployment.

When an economic beneficiary does not bear the full cost of their activity, this is known as an externality. Because of this negative externality, we see many protests in relation to the development of wind farms locally. Therefore in order to mitigate externalities and generate support for these facilities, it is necessary for the generator of the externality to face the cost of this, but also for those impacted to be compensated. This however is a difficult challenge. How do you compensate people in a neighbourhood for noise or congestion?

One option is to give or sell discounted shares in the development to locals, so that they benefit from the incomes generated. An alternative is for local communities to form cooperatives to develop renewable energy. Eight farm families established a wind farm, costing €22 million in 2002 in Killala. Rather than leasing their land to a large utility, whereby the economic return would accrue to a corporation distant to the area, they developed the project themselves to maximise the local dividend.

However this has been a very significant undertaking for a community group, without the resources that large utilities have behind them, with significant risks and challenges. This model however has the potential to increase the acceptance of wind-farms and maximise the economic returns to the areas in which the electricity is generated.

Deriving energy from biomass is another source of renewable energy, where crops are grown, harvested, dried and burnt, either for heating or electricity or combined heat and power (CHP), producing a renewable source of energy. In Ireland a bio-energy sector was incentivised during the last Common Agricultural Policy. The aim was for farmers to grow willow and miscanthus plants to be burnt in peat power stations, replacing a proportion of the non-renewable peat with renewable biomass.

However the policy was not very successful and the scheme was under-subscribed. While the production of bioenergy is technically feasible and it is economically advantageous for many farmers to produce it, a number of problems arose across the value chain which limited success.

Producing a good or service for use by consumers, goes beyond the production at the farm gate. The value chain is the process from start to finish, that results in the consumer product, in this case energy for electricity or heating.

One of the main issues that arose is that the markets for biomass were not fully developed. After drying, harvested biomass is converted into pellets and then burnt in special boilers for heating or in the case of electricity. For farmers to produce, they need to be confident that the market exists, while for domestic users, in order to invest in these boilers, they need to be confident that they will have a regular supply of pellets. Given the cost of transport, these markets need to exist locally and not just nationally.

In terms of electricity production, the issue was one of price and behaviour. At the price paid by electricity producers, it is worthwhile only for cattle farmers to produce bio-mass crops. Given the risk associated with a new crop and the price paid relative to conventional crops, it was not worthwhile for cereal farmers to convert. Yet cereal farmers with experience and equipment used for planting and harvesting are more likely to plant if the price was right. Cattle farmers are inexperienced in this area. Thus price and behaviour limited the uptake for electricity production.

Another source of biomass for energy production is a by-product of forestry, namely forest thinnings. Forests are thinned to create space for more productive trees to grow to their full potential. They are a by-product of improved forestry management, but can be processed and burnt as biomass.

Like in willow based biomass, value chain issues also arise. However a novel solution was developed in County Clare. There was a chicken and egg situation, where there was no existing supply of wood chip to supply biomass boilers, therefore there were no boilers and no demand for wood chips from farm forest owners.

Teagasc and the Clare Local Development Company, developed the County Clare Wood Energy Project to deal with value chain problems. Recognising value chain problems, the two institutions, developed a project linking potential producers and potential consumers of wood chip. For example the project worked with the County Council when they were building a new Council headquarters in Ennis to facilitate the installation of a wood chip boiler and a supply of locally grown and processed wood chip.

In achieving a functioning wood chip value chain, there were a number of benefits for the producer. These included a return to farmers from thinnings, rather than treating it as a waste product. Thinning improves the quality of the main forestry crop and thus the return from final felling. It helps to establish stable long term fuel supply contracts that can help to diversify farm income in a tax-free way.

From a community point of view, it helped to stimulate local employment and the economy as 75-80 per cent of expenditure on wood biomass is retained within the local economy versus oil at 8.5 per cent. It also helps to reduce dependence on imported fuels and benefits environment as wood chip is “carbon neutral”.

Another issue faced by the value chain is that many of the producers are small scale and the fragmented nature of forest plantations makes it difficult to achieve economies of scale. Given the high cost of transporting machinery, it might not be commercially viable to bring in harvesters for plots of less than 5 hectares. However, using GIS along with forest age class and species data, the Clare Wood Energy project was able to identify clusters of forests of similar age, To exploit this and to facilitate thinning, the project developed a web based shop window to link producers with contractors to assist in the development of clusters for thinning, which in turn led to the development of the Clare Timber Producers Group.

It is not enough therefore to have the idea or the potential for an idea to work. It is necessary to have the supporting environment, whether it be the community buy-in or the development of a value chain, for an idea to bear fruit.