# Bio-based construction materials for climate change mitigation: scalability and sustainability

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***Declaration***

The work presented in this thesis has not been submitted for any other degree or professional qualification. The thesis is the result of my own independent work, and I am the author of the entire work: this statement includes published works in collaboration with other authors.

The first half of this thesis (Chapters 1 to 3) includes material written (by me) during the course of the research and published in the following journal articles.

Arehart, J. H., [†] Hart, J., [†] Pomponi, F., & Amico, B. D. (2021). Carbon sequestration and storage in the built environment. *Sustainable Production and Consumption*. *27*, 1047–1063. https://doi.org/10.1016/j.spc.2021.02.028

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# Abstract

Construction products made from timber and other organic materials are understood to contribute to climate change mitigation by causing relatively low greenhouse gas emissions in the supply chain, whilst also storing biogenic carbon within the material itself. A logical progression would therefore be policy support for a steep increase in the penetration of the construction market by such materials. This raises questions that this work aims to resolve, primarily in the context of using more timber – ideally from domestic forests – in UK construction. These questions relate to the scale of any climate change mitigation and the validity and accuracy of methodologies and assumptions used to quantify the impacts, taking account of the uncertainties associated with modelling parameters. This work employs a novel model for exploring the flow of carbon from field and forest to buildings and – eventually – back to the atmosphere or to semi-permanent storage below ground. The model is driven by changes in demand for construction timber, and it incorporates dynamic features and a stochastic approach to the use of uncertain variables. Results suggest that when coupled with a domestic afforestation agenda strong enough to support future demand, an increase in timber use can lead to an increase in terrestrial carbon pools of 47 MtC after 100 years (albeit too late to contribute significantly to 2050 emissions targets). The benefits may be much reduced, however, if the link to afforestation and reforestation is weak, which is likely to be the case if the additional demand is met through imports. Products from faster-growing crops have advantages and should be further developed and deployed as a complementary strategy. In the medium term, more impressive results can be achieved by focussing on timber waste management rather than demand growth, with one reasonable scenario reducing emissions by 14 MtC by 2050.

# Glossary of Abbreviations & Acronyms

| | | |
|---|---|---|
| BAU | - | business as usual |
| BECCS | - | bioenergy with carbon capture and storage |
| BioCarp | - | Biogenic carbon pools model |
| C2C | - | cradle to cradle |
| CCS | - | carbon capture and storage |
| CE | - | circular economy |
| CLT | - | cross-laminated timber |
| $D_f$ | - | displacement factor |
| DOCf | - | degradable organic carbon fraction |
| EC | - | embodied carbon |
| EoL | - | end of life |
| EPD | - | environmental product declarations |
| FOD | - | first order decay |
| GHG | - | greenhouse gases |
| GWP | - | global warming potential |
| HWP | - | harvested wood products |
| ILCD | - | International Reference Life Cycle Data System |
| IRF | - | integrated radiative forcing |
| LCA | - | life cycle assessment |
| LFG | - | landfill gas |
| LULUCF | - | land use, land-use change, and forestry |
| MDF | - | medium-density fibre board |
| NZ2050 | - | net zero by 2050 |
| Ob | - | overbark |
| OSB | - | oriented strand board |
| PADS | - | proactive anticipatory demand scenario |
| PCR | - | product category rules |
| PDF | - | Probability density function |
| PV | - | photovoltaic |
| RHI | - | renewable heat incentive |
| RUDS | - | reactive unplanned domestic scenario |
| $S_f$ | - | substitution factor |
| SIS | - | Scandinavian import scenario |
| SWDS | - | solid waste disposal site |
| Ub | - | underbark |
| VST | - | virgin structural timber |
| WRME | - | wood raw material equivalent |
| YC | - | yield class |

# Contents

# 1 Introduction

*Research questions and key findings. Motivations for investigation of biogenic carbon and background commentary on the broader context.*

## 1.1 Thesis Overview

Bio-based construction materials have long been viewed as having strong environmental credentials, but in recent years the case has increasingly been made for them having a significant role to play in mitigating climate change. Key reasons for this are that such materials are understood to cause lower greenhouse gas emissions in the supply chain, and that the materials themselves represent a store of biogenic carbon that would otherwise exist as carbon dioxide in the atmosphere, contributing to climate change. In the case of wood products, the trees that produce the timber also store biogenic carbon for long periods, which can also be taken into consideration. At the same time, there is increasing awareness of the potential for a wider role for such materials in construction including, for instance, engineered timber used structurally in tall building. As a result, there is an argument for a rapid and substantial increase in the use of such materials in construction. Research that evaluates the climate change mitigation potential of such strategies is patchy.

Although material selection and the role of bio-based materials in buildings to mitigate climate change is – in common with most other climate mitigation strategies – likely to make a minor contribution (whether positive or negative, depending on how well it is understood and implemented), it is vital that this contribution is better understood, so that the collective power of such contributions from many sources can be quantified and harnessed to deliver beneficial change. The purpose of this thesis is, therefore, to offer a deeper understanding of the extent to which bio-based construction materials can offer climate mitigation.

In particular, as wood in its various forms is the incumbent bio-based construction material, used in the greatest quantities and with potential for further growth, this material is the main material of interest to the thesis. The key question is whether scaling up the use of construction timber in the UK (the 'more timber' strategy) above the existing baseline can actually deliver such benefits, and if so what the potential significance of those benefits will be and when they will be delivered.

The working assumption is that this additional timber is provided from UK coniferous forestry, as there is a case to be made that this will be the most sustainable approach. However, as much of the demand for construction timber in the UK is currently met through imports, the possibility that additional demand will also be met through imports is considered in scenario analysis.

### *Research Questions and Methods*

The primary research question is:

> *How much greenhouse gas emissions can be avoided and what associated climate benefit can be achieved over time periods of up to 100 years by coupling an ambitious but realistic increase in construction timber usage in the UK (preferably supplied from domestic forestry) with an afforestation agenda designed to meet future demand*?

Addressing this question (Chapter 6) requires an understanding of other topics – such as forestry, and waste management – which are also discussed in the literature review (Chapters 2 and 3) and included in modelling. The model developed to explore the question (Chapter 4) can also be used to answer other questions, and the complementary issue of the optimal waste management choices for wood already in the product system gives further insight into the main question whilst also demonstrating options for more immediate cuts in GHG emissions. Specifically, this subsidiary research question is:

> *How much greenhouse gas emissions can be avoided and what associated climate benefit can be achieved by making immediate changes to timber waste management priorities in the UK*?

These research questions are investigated with a tool developed specifically for the purpose. The biogenic carbon pools model (BioCarp) analyses the carbon flows, and greenhouse gas emissions, from the atmosphere to forest, and on to construction products and then end-of-life processes. It also models the substitution benefits associated with using timber in construction in place of other – more carbon-intensive – materials that might otherwise be used. Variables used in BioCarp which have a significant degree of uncertainty are represented by probability density functions which are randomly and iteratively sampled, meaning that results of each scenario tested are presented as ranges. Some variables also have dynamic characteristics, in that they change over time, for instance to reflect technology development. The combined climate effect of the changes to the carbon balance and any associated greenhouse gas emissions (especially methane) is modelled with an established dynamic impacts calculation tool.

### *Key Findings*

The research shows (Chapter 6) that an increase in timber use in construction can deliver significant climate change mitigation, but only over the longer term. With demand increasing by 3% per year for 30 years, and met from domestic forests, terrestrial carbon pools can be increased by 47 MtC, but much less can be achieved before 2050, with the pools only reaching 3.4 MtC after 30 years. [1] Owing to the time it takes to gather momentum, and initial carbon losses shortly after harvest, this strategy delivers no climate change benefit (at best) for the first decade, in terms of integrated radiative forcing, and very little by 2050, but thereafter the climate change benefit begins to accelerate rapidly. For these results to apply, an essential condition must also be met, namely that the increase in timber use must be coupled with an afforestation strategy with a commercial forestry component that anticipates long-term timber demand development. This points to the need for the domestic forest and construction product industries to work together on developing products that will use timber that can be grown in the UK in sufficient volume. Results can be further improved with a change to end-of-life management of wood with combustion only permitted if supported by carbon capture and storage. Scenarios investigated where this assumption does not apply, including the possibility of the additional timber being imported, can still deliver strong results in the long term (after 100 years for instance). However, it can take over 40 years for the initial forest carbon debt to be cleared, and 60 years before any benefit in integrated radiative forcing is seen.

In order to deliver significant climate change mitigation before 2050, a better objective is to optimise the management of timber already in the system, rather than growing new demand (modelled in Chapter 5), with one reasonable scenario yielding a cumulative gain of 14 MtC in terrestrial carbon pools by 2050. This can be implemented alongside the more timber strategy, resulting in a balanced approach to short, medium, and long-term climate change mitigation. An additional option would be to focus new demand on products made from faster-growing crops.

## 1.2 Background and context

Buildings have so far played a central role in the unfolding climate crisis. The greenhouse gas (GHG) emissions from the systems required to heat, cool and power buildings, and from the many industries required to construct, maintain, and – in the end – replace buildings amounts to around 40% of all anthropogenic GHG emissions[2] **(GABC, 2021)**. The Paris agreement **(United Nations, 2015)** aims to limit global temperature rises to 1.5°C above pre-industrial levels, but warming is now proceeding at up to 0.20°C per decade and has already exceeded 1°C **(NASA, 2020)**. As such, unless the problem is entirely and successfully devolved to the rapid and total decarbonisation of all energy

[1] All results mentioned in this section are median values.
[2] 37% in 2020, but emissions were down by ~10% in that year because of pandemic-related effects.

systems (not just power), or to geo-engineering **(Sovacool, 2021)** considerable progress must be made in reducing the GHG emissions associated with constructing and operating the built environment. Such progress must take in the full life cycle of buildings, including material selection and processing, building design, operation, and reuse and ultimate disposal of materials.

Some progress has been made in terms of the energy efficiency of new buildings, driven in part by tightening regulations **(European Union, 2018)**. And partial decarbonisation of power grids in many parts of the world has reduced the GHG emissions per unit of electricity used in buildings. For instance, in the UK, the emission factor[3] for grid electricity consumed in the UK has reduced by more than half, from 0.524 to 0.253 $kgCO_2e/kWh$ in the ten years to 2018 **(DBEIS, 2020)**, largely on account of renewable sources of power taking over from coal-fired generators.

Building owners using fossil fuels for heating, however, are in a different position, and cannot wait for decarbonisation of their supply chain to reduce their emissions. Whilst the idea of shifting heating load from the gas to the electricity network has merit, progress has so far been slow despite the considerable subsidy support offered for heat pump installations through the Renewable Heat Incentive (RHI) in the UK. By July 2021, through the Domestic RHI, payments of £392m had been shared between 70,000 households with heat pump installations **(OFGEM, 2021)**, which means that average subsidies of £5500 per installation[4] have not been enough to reach even half of one percent of the addressable market. The mixing of biogas into the UK mains gas network has so far brought the emission factor for mains gas down by less than 0.5% **(DBEIS & DEFRA, 2020)**; and whilst the idea of mixing up to 20% of 'green' hydrogen[5] into the gas main is thought to have sufficient potential to justify a local pilot project stage **(Keele University, 2020)**, there is much work to do before it becomes a reality, and some scientists are highly sceptical of any potential role for hydrogen in a sustainable economy on account of the very low energy return on energy invested for systems involving hydrogen **(Krumdieck, 2019)**.

Whilst decarbonising heating systems is central to climate change mitigation efforts in the UK, globally, the growth in demand for air conditioning, in response to rising temperatures

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[3] Emission factor is taken to mean the GHG emissions per unit of energy, mass, volume or activity: in this case, per kWh of energy consumed.
[4] A figure that will rise, as the RHI provides ongoing quarterly payments as energy is used.
[5] Green hydrogen is hydrogen produced with negligible GHG emissions, typically from water using renewable electricity.

and increasing populations able to afford it is potentially even more important **(Davis & Gertler, 2015)**. Air conditioning fuels a positive feed-back loop in which climate change increases demand for one of its drivers. Even if, ultimately, much of the air conditioning demand can be met with photovoltaic (PV) systems **(Laine et al., 2019)**, the problem of the global warming potential (GWP) of the refrigerants remains. A shift has taken place from 'natural refrigerants' such as ammonia or carbon dioxide (known since the nineteenth century), to refrigerants with both very high ozone depleting potential and GWP (i.e. chlorofluorocarbons – CFCs) to refrigerants with merely very high GWP (hydrochlorofluorocarbons and then hydrofluorocarbons) as a result of the Montreal Protocol. The latter are now to be phased out through the Kigali Amendment to the Montreal Protocol, but this process runs until the year 2047: the whole saga is discussed by **Ciconkov (2018)**.

Whilst improving the energy and carbon standards of new buildings is essential, it is important to understand that only a fraction of new (efficient) buildings are replacing old (inefficient) buildings. Most are simply adding to the global built floor area; and even replacement buildings add to GHG emissions in the short term due to demolition and upfront embodied carbon in the new. It can take decades for improved operational performance to pay off the carbon debt. Thus, the world must face the challenge of retrofitting the existing building stock to operate with much lower GHG emissions, which is especially important in countries (such as the UK) with a mature building stock with high cultural and heritage value and very low stock turnover.

Furthermore, gains in energy efficiency in both new and existing buildings are also partially undermined by the rebound effect which, for instance, allows the benefits of energy efficiency to be taken as improved comfort rather than lower energy consumption and is acknowledged as an important consideration for policy-makers **(Sorrell, 2007)**.

## 1.3 Embodied carbon

Another side-effect of the attention paid to the energy efficiency of new buildings is that the environmental costs – particularly the embodied carbon[6] – of materials incorporated in a building represent an increasingly significant proportion of the building's life cycle environmental burden, and in highly energy efficient buildings embodied carbon typically accounts for around half of whole-life GHG emissions, and up to 90% in extreme cases **(Röck et al., 2020)**. This effect is amplified by the environmental impacts associated with

[6] The embodied carbon of a product or material is here defined as the net sum of the GHG emissions (in carbon dioxide equivalents – $CO_2e$) associated with the processes involved in its production, use and disposal. Operational GHG emissions are excluded and biogenic carbon storage is not accounted for.

the supply of the additional materials that are often required to achieve improved energy efficiency and lower GHG emissions during the operational life of buildings. Examples include the increased volumes of insulation required to reduce heat loss; the inert gases sealed into glazing units and the extra layer(s) of glazing in those units; and the photovoltaic panels used to generate 'zero-carbon' electricity for the building even though no environmental product declaration (EPD)[7] is available for PV system components to enable informed consideration of the environmental costs and benefits in terms of life cycle GHG emissions. There is now more interest in this issue from professional organisations **(RICS, 2017)**, policy-makers **(GLA, 2017)** and from people and organisations interested in signalling their attention to climate change. As such, embodied carbon of materials should be an important consideration in material selection if the information and data on the subject is of sufficient quality.

## 1.4 Bio-based construction materials

One possible strategy is to increase the role of timber in the construction industry, as discussed by **Hart & Pomponi (2020)**, and with other – faster growing – bio-based materials **(Habert, 2021)**.[8] Mineral-based construction materials – metals, cement, glass, etc. – are often characterised as coming from limited or finite resources and requiring significant inputs from fossil fuels in order to process them into the sophisticated products required by the industry **(Berge, 2000)**. Timber, by contrast, is presented as a 'renewable' material with its own integrated renewable fuel supply: production waste can be used to generate heat and power, as can bio-based construction products at the end of their useful lives. Additionally, timber construction products physically embody carbon that, prior to the tree's intervention, existed in the form of atmospheric carbon dioxide. Thus it can be argued that the combined forestry, wood products, and construction systems can play a role in carbon sequestration and storage strategy to mitigate climate change **(Geng et al., 2017)**, acting as a so-called carbon 'sink' **(Churkina et al., 2020)**. Attention is increasingly focussing on the policies needed to make the shift to timber a reality **(Maniak-Huesser et al., 2021)**. Bio-based construction products produced from annual or faster growing crops

[7] An EPD is a standardised approach to quantifying and reporting verified and certified life cycle environmental impacts of a product, in accordance with ISO14025. At least one EPD **(Acciona, 2017)** for a unit of electricity produced by a PV solar farm (in Chile) exists, but there is a clear need for similar information (although normalised by capacity) on PV equipment from different sources.

[8] Bio-based materials and products are here defined as those that contain a significant proportion of material of biological origin - usually plant-based, such as timber, crop-residues, and grasses. The term 'bio-based' and 'biogenic' can be used interchangeably in many contexts, although 'biogenic carbon' accurately describes the carbon in an object that has been sequestered by living matter over human rather than geological timescales. 'Bio-based' is the preferred term when describing products and materials that are largely – but not necessarily totally – produced from such matter.

(straw, hemp or bamboo for instance) have the same property, along with the advantage of transferring carbon from atmosphere to building on a faster timescale.

The idea of using human-made artefacts to mitigate climate change by storing carbon relies on the storage period lasting long enough to have an impact. This might be achieved by such artefacts collectively forming an ever-increasing 'pool' of terrestrial carbon and/or by retaining carbon beyond some critical future date when some as-yet unspecified globally-effective climate change management or mitigation option is in control. Whilst much harvested timber is oxidised and released to the atmosphere within years, there is potential for it to be retained for much longer periods, for instance decades to a millennium for structural timber (Table 1-1). And the discovery of ancient shipwrecks – up to 2400 years old – in the Black Sea demonstrates how, given the right conditions, timber can be preserved to an 'extraordinary' degree: in this case, sitting on the sea bed at depths of up to 2100 m, but – critically – below the 'anoxic boundary' at around 150 m **(Pacheco-Ruiz et al., 2019)**. Equally, timber can survive for extended periods in very dry conditions: **Blanchette (2000)** provides an interesting review of the factors that support survival – or, conversely, degradation – from an archaeological perspective, with examples going back 6000 years.

| TTT Log10(t), years | 0 | 1 | 2 | 3 | 4 + | … | 8 |
|---|---|---|---|---|---|---|---|
| **Cycle** | News | Flatpack | House | Cathedral | Anaerobic | … | Carboniferous |
| **Technosphere** | Paper | Furniture | Timber frame | Oak rafters | Landfill? | … | - |
| **Biosphere** | Straw | SRC | Forestry | Ancient forest | Peat | … | Fossil fuel |

*Table 1-1. Timber turnover timescale (TTT: includes biomass generally) with examples. Durability of human-made artefacts ('technosphere' row), and the timescale for sequestration and storage within the biosphere. SRC is short-rotation coppice. Landfill is awarded a question mark, as engineered, lined and capped landfill sites do not yet have such a long record. The shading is a subjective view of environmental preference gradient (green is preferred) for the accretion of material from the biosphere: i.e. ideally material is taken from the biosphere at TTT 0 to 1 or 2, and transferred to the technosphere on track to achieve TTT of 3+.*

These examples are, of course, far outweighed by similar materials that have not survived: they are the exception rather than the rule. For instance, the 2019 fire in Notre Dame, Paris, which destroyed its 13th century roof structure,[9] caused much consternation **(Gombault, 2020)**, but in reality it is just the latest in a long succession of fires that have over time hit cathedrals such as Chartres, Rouen, Reims, York, and St Pauls in London – in some cases more than once. However, if we can understand more about the conditions

[9] Known as 'the forest', much of this structure – each element made from a separate oak tree – would have existed in the biosphere from around the 9th century.

under which wood can survive for centuries or millennia, then we may be able to manage it for longevity, both in and after its useful life.

Accordingly, much evidence has been presented to support the notion that carbon stored in timber buildings is an effective climate change mitigation strategy – for instance **Churkina et al. (2020)**, and reviewed in **Arehart et al. (2021)**. Recent evidence from China does appear to suggest that increasing commercial timber production is compatible with carbon sequestration in the context of rapid and large-scale afforestation programmes: **Wang et al. (2020)** calculate that the Chinese land carbon biosphere sink between 2010-16 is equivalent to 45% of anthropogenic emissions over that period. However, the effect will start to dissipate when less land is made available for afforestation.

As well as the carbon storage potential, the processes required to transform the objects provided by nature into a given quantity of construction materials and products are demonstrably less intensive for bio-based materials **(Hill & Dibdiakova, 2016)**. By way of illustration, contrast the process of tending, cutting, drying and sawing a tree to produce dimensional construction timber to that of mining iron ore and putting it through very high temperature processes to transform it into a steel beam. It is widely argued that, as a result, buildings designed and constructed with an increased role for timber have a lower carbon footprint than their 'conventional' counterparts **(Hart et al., 2021)**, and therefore timber can, in some way, be credited with the GHG emissions thereby saved. This is a theme that is critically explored in several parts of this thesis, from the literature review onwards.

Clearly, there is now a body of opinion that contends that a viable way forward for the global construction industry might be to grow construction products rather than extract them. It has not, however, been well demonstrated that this really is likely to be a good plan and, if so, in what contexts. A wide variety of related issues need to be addressed in order to reach a conclusion: for instance, questions about whether sufficient land area is available to grow the products needed; whether the strategy suits countries without a domestic supply of sustainable timber; and the best roles for bio-based materials in terms of climate change mitigation. These are some of the high-level issues explored in this work, before focusing in on the context of the UK and the climate change mitigation potential of timber in construction.

It is generally understood that trees – and therefore afforestation and reforestation – make a huge range of contributions to the viability of all forms of life on the planet. The process of sequestering and storing carbon from the atmosphere is one facet of this, and **Bastin et al. (2019)** are amongst those who point to the great global potential for afforestation /

reforestation to mitigate climate change. **Boysen et al. (2016)** however, argue that terrestrial carbon dioxide removal through photosynthesis cannot prevent large temperature rises without eliminating virtually all natural ecosystems, and that dramatic emissions reductions are still the priority. There is much intellectual ground to explore between these two points of view.

## 1.5 Optimising wood use

If we are to argue for more timber to be used in construction, it is important to evaluate what that means in terms of volumes required, demands placed on existing forest lands and the need for land use change, and whether this is – in fact – the best use to which trees can be put. The demand for paper and pulp continues to increase despite the digital enablement of the paperless society.[10] There is an increasing expectation that trees are harvested for fuel because of the support available for using it (for instance, in the UK, renewable obligations certificates for electricity generation[11] and the domestic and commercial versions of the RHI). And markets may additionally emerge for novel materials arising from innovative treatments of wood, such as densification, which involves the use of some combination of pressure, heat and chemical impregnation to adjust material properties **(Báder et al., 2018)**.

It may be that none of the above uses for timber are to be commended, if and when the optimal place for wood is in a living tree: undoubtedly this will be the case in many contexts, but defining those contexts is far from straightforward. Standing forests provide a store of carbon in trees (above and below ground) and soils which can be much more secure than the fraction of the carbon in the tree stem that survives into the harvested wood product (HWP) derived from it, until such time as the final owner of that product discards it. In other cases, the carbon stored in the forest may be more at risk – especially from fire – than it would be in HWP. The future health of much of the planet's forests may now be beyond our control, with increasing evidence of stress, and forests not regenerating after fire **(Coop et al., 2020; Parks et al., 2019; Stevens-Rumann et al., 2018)**. This would have grave implications for the use of this research, as it would tend to mean that afforestation and reforestation (after harvest) may not perform as expected, and the focus should be on scenarios that take account of this. An alternative reaction to the same information (not advocated here) might be to extract the wood before climate stressors

[10] According to statistics from the Food and Agricultural Organization of the United Nations **(FAO, 2020)**, global production of pulp for paper increased by more than 5% from 2014-2019.
[11] In the UK in 2019, 25 TWh (more than 7% of the total electricity generated, **(DUKES, 2020)**) of electricity was provided by plant biomass, mostly consisting of imported wood pellets.

have a chance to do so. Other reasons not to harvest trees include protection of biodiversity, and the socio-economic value of forests in terms of recreation and cultural heritage and various eco-system services provided, such as their role in soil stability and flood prevention **(Markart et al., 2021)**. **Strassburg et al. (2020)**, for instance, show how biodiversity and climate change mitigation can work together in the restoration of natural environments. They calculate that if 15% of converted crop land and pasture land was restored, then – with biodiversity and carbon sequestration prioritised together in the land selection – an additional 299 $GtCO_2$ of carbon would be sequestered in the long term.

The Committee on Climate Change in the UK **(CCC, 2018)** makes the case for an increase in land area under sustainably managed forestry, which can lead to an increase in the carbon stored in the forests *and* in durable harvested wood products (HWP). This reinforces the view of authors such as **Oliver et al. (2014)** whose analysis demonstrates that some HWP save more carbon than unharvested trees, and biomass is better applied to HWP than to bioenergy. The CCC argues that biomass should only be harvested at a sustainable rate and that such material should be put to the most climate-beneficial use: a *"substantial increase in the use of wood in construction"* is identified as a priority [6, p.15]. Other priorities are for a move away from combustion of biomass for energy except when supported by carbon capture and storage (CCS) – and switching biofuel use from surface transport to aviation. The CCC report notes the importance of construction timber in GHG abatement, in terms of storing carbon and in displacing materials with higher EC. The use of timber in domestic construction in the UK results in over 1 $MtCO_2$ being stored per year in new homes, with potential to increase to 3 $MtCO_2$ per year by 2050, with similar progress in the commercial and industrial sectors through uptake of new types of engineered timber systems **(CCC, 2018)**.

The CCC analysis is based on a continuing shift away from masonry towards timber frame homes, and includes biogenic carbon storage. This is justifiable in the context of the report – which is about identifying the best role for biomass in a low carbon economy – but the question of whether it is justifiable to produce more biomass to meet such needs gets to issues at the centre of this thesis. Furthermore, increased use of engineered timber such as cross-laminated timber (CLT) will result in demand for increased volumes of construction timber per unit of floor area in comparison to more conventional timber frame construction. It is important to explore the environmental costs and benefits of such a shift.

## 1.6 Circular Economy

The concept of a Circular Economy (CE) has also gained traction in recent years, and much of the discussion around the sustainability of material resources is now framed in these terms. Advocates in the UK include Zero Waste Scotland and the Ellen MacArthur Foundation **(EMF, 2015; Pratt & Lenaghan, 2015)**. The sheer number of available definitions of CE is frequently commented on, with one study finding 95 different definitions in a set of 114 **(Kirchherr et al., 2018).** Definitions typically include references to abstract concepts ('an economy that is restorative and regenerative by design' **(EMF & McKinsey, 2015)**); principles such as decoupling of economic growth from resource extraction; and strategies such as implementing a waste hierarchy and extracting as much utility from resources as possible through a cascade of options from design for durability, through to reusability, repairability, and recyclability. It is worth noting that in the much of the CE community, recycling is seen as something of a failure – a last resort – as it involves a loss in 'value' from manufactured goods. Instead, the emphasis should be on durability, maintenance, reuse, remanufacture, and the business models needed to facilitate these strategies. However, some implementations of CE actually emphasise 'recycle' more than 'reduce', thereby subverting it to the cause of unsustainable 'business as usual'. A range of related concepts have been co-opted into CE or further developed including design for disassembly, the sharing economy, product as service, and CE business models to name a few.

CE presents challenges for all types of structural products **(Hart et al., 2019)**, and wood is no exception. Reuse of timber presents particular challenges in that timber construction products are typically a composite of organic (wood) and mineral (adhesives, preservatives and finishes) substances that may be impossible to separate in a controlled way. On occasion, re-use is not feasible, and recycling is uneconomic, and the question arises of whether to store the materials *ad infinitum* (e.g. in landfill), or whether to return the organic component to the atmosphere (through combustion with energy recovery for instance), with the associated release of $CO_2$ and other pollutants. Landfill is anathema to CE-based policy but it does offer the benefit of long-term carbon storage. This raises the question of whether a new resource management system might be developed as an alternative – more warehouse than rubbish dump – in which carbon-rich waste/material is stored for the long-term (whether above or below ground) with the specific aim of securing the carbon, but potentially allowing access as uses and markets for such material arise.

The end goal of reducing GHG emissions to limit climate change is reasonably clear (although vastly more complex than it appears on the surface, given – for instance – the variations in impact across time and space resulting from different emission reduction trajectories). The same cannot be said of CE: is the true purpose of CE economic growth rather than the amelioration of environmental pressures? One of the promises of CE is that it will result in the decoupling of economic growth from resource use, but warning signs exist that the reverse may be true. Reduction of waste, for instance, can reduce costs thereby facilitating the production of cheaper stuff and stimulating an inevitable rebound effect. Another example is the continuing global increase in demand for data services overwhelming the energy efficiency gains associated with Moore's Law. **Zink & Geyer (2017)** have identified a limited set of circumstances where the environmental benefits of CE strategies are likely to exceed the costs of the rebound effects: they include circular products being straight substitutes for existing conventional ones (at a similar price, thereby not stimulating market growth), and such products should target satiable demand. Construction products are, however, quite likely to fit into such categories.

## 1.7 Environmental Goals

If we espouse the view that we should (or should not) use more timber in construction to save the environment, we need to be clear about what we mean.

For instance, we need to be clear about what is intended by the exhortation to 'use more'. Context is important: the type of timber, how and where it is produced, how it is used and re-used are all important parts of the equation. At one extreme, conversion of forest land to pasture or agricultural land, with the extracted timber used as pulp or fuel, results in a rapid and permanent loss of stored carbon and carbon sequestration potential, for negligible gain. Towards the other end of the scale, an increase in demand might be met by restoring previously forested land, and selectively harvesting wood from existing sustainably managed forest, with negligible impact on the carbon stored in that forest. The harvested wood might then be sawn for use in situations where carbon storage times and substitution benefits are optimised.

Also important is what is meant by 'save the environment' or whatever related terms are used (save the planet, halt climate change, reduce biodiversity loss, etc.). For whom (or what) are we saving the environment? For instance, do we favour people, sentient beings, or the biosphere; or those alive now, in the next few generations, or *ad infinitum*? Can we trade off environmental losses in some parts of the world with environmental gains elsewhere? Such are the genuine ethical dilemmas facing, for instance, plans for geo-engineering to mitigate climate change. Global manipulation of forest coverage for this end is itself a form of geo-engineering (albeit one that humanity has much more experience of than options such as ocean fertilisation or solar shielding).

The development of a system to provide more timber and store more carbon in both living trees and in harvested wood products has potential environmental and social costs. Firstly, there are the impacts on the current generation, which may be distributed inequitably. Land being used for other purposes must be reallocated to forestry, and this is most likely to happen in locations where land prices are relatively low. This is a climate justice challenge with echoes of colonialism, as lower-income countries are asked to reconfigure their landscapes and economies to address the needs of other countries and the problems caused by them. If there is an overwhelming focus on mitigating climate change, then by definition this will lead to de-prioritisation of other environmental challenges, such as biodiversity, that the human race also depends upon for its long-term future. The power of the market will be summoned to develop and scale an effective and efficient conveyor to move carbon from the atmosphere, into the forest, with a proportion going onwards to the built environment: at its best this could lead to a durable increase in stored carbon, but there is also the possibility that it will merely increase the anthropogenic hold over the environment whilst also inducing a shift of carbon from the forest back to the atmosphere. The market values what is paid for, and currently there is more money to be made from intensive management of plantations than there is from a more *laissez-faire* approach to the management of forest habitats: this might work well for the carbon balance (although this is not guaranteed), but not for biodiversity.

A further problem is that the carbon storage offered by trees and timber will be viewed as a solution to climate change that gives licence for further fossil fuel extraction: a form of rebound effect. Without a firm global cap on fossil fuel extraction, almost any attempt to mitigate climate change will meet this problem, which suggests a need to prioritise degrowth rather than growth that is slightly less damaging than before **(Keyßer & Lenzen, 2021).**

The 'more timber' strategy is the last link in a chain that moves carbon from secure storage deep in the earth, into the atmosphere, and then into a much less secure store at the earth's surface. Such carbon storage is fraught with risk. For instance, deforestation and climate change in Amazonia are turning the region into a carbon source **(Gatti et al., 2021)**. And in the North American boreal forests, it has been found that the carbon lost directly through wildfires (57.1 Mt/yr for the last 31 years) overwhelms the net ecosystem production of the region (1.9 Mt/yr), which is itself reduced as a result of fire **(Zhao et al.,**

**2021)**. Therefore, tree harvesting might both contribute to the conditions that result in fire, whilst also safely removing timber out of reach of those fires. Additional forest risks come from increasing windstorms and pests and diseases **(Lecina-Diaz et al., 2021)**. With regard to the built environment, without constant growth in timber consumption, HWP stocks may gradually saturate and even go into reverse, as has been modelled for Japan **(Kayo et al., 2014)**. Protecting the stocks that do exist may also be an uphill battle in the face of changing tastes, technologies and demographics, and potentially war, that have seen dramatic rises in building stock turnover in the past.

'More timber' is often promoted as a strategy on the basis that its climate change impacts are less bad than the other options (except the 'build less' option which is rarely referred to). But the granting of rewards for being not quite as bad as expected is potentially hazardous, even if the reward is only the burnishing of green credentials. For instance, buying carbon offsets to compensate for, or 'forgive' GHG emissions has long been likened to sinners buying 'indulgences' from the medieval church, although not without push-back that the metaphor is over-done because of the different motivations **(Dalsgaard, 2021)**. In a sense, claiming substitution benefits seems to go even further, in that unmediated claims of carbon substitution allow significant emissions to be indulged at no cost, if this is done outwith any compliance market or well-codified voluntary system. There is a case for only considering substitution gains in the context of a project or programme that takes – and retires – an approved share of capped regional, national or global GHG emissions budgets.

To bring it back to timber, authors of a great many Life Cycle Assessments[12] compare and contrast the results of their assessments of an interesting case with the counterfactual – an assessment of what would have happened otherwise. But the reader is usually expected to take the choice of counterfactual on trust, and the scope for abuse of that trust is almost unlimited. Examples might include assumptions that wood residues are used as fuel instead of coal in situations where it is already phased out; or instead of being harvested for a particular use, commercial forestry plantations are allowed to return to nature; or that there will be no innovation in future. Investigations into the life cycle impacts of biomass electricity and biofuels, for instance, clearly highlight the significance of the choice of counterfactual land use **(Chilvers & Jeswani, 2017; Stephenson & Mackay, 2014)**.

[12] Life Cycle Assessment is a systematic approach to evaluating the environmental impacts of products and services, discussed in Chapter 2.

Therefore, when reading about counterfactuals, it is important to look out for what might be termed 'counterfictionals'.

A final point is the question of how to incentivise, measure and reward the positive interventions needed. Currently, various overlapping drivers exist. For instance, carbon fluxes associated with land use, land use change and forestry (LULUCF) are part of national accounting of GHG emissions associated with the Kyoto Protocol, as are emissions from fossil fuel use. Therefore, the collective system of forests, HWP, buildings, and fuel substitution is a subset of the emissions reductions incentivised at a national level, although that does not in itself ensure that individual actors within the system see any incentives.

In the context of the UK, it can be seen that incentives to decarbonise different parts of the UK economy have enjoyed varying degrees of success, with impressive results for the electricity network, but less so for heating and transport for instance. The UK government and devolved administrations have also set targets for tree planting, but potentially without sufficient incentives to ensure that those targets will be met: The Climate Change Committee has observed that whilst the UK target is 30,000 ha/yr of afforestation by 2025, only 13,000 ha was achieved in 2019/20, and calls for more urgency **(CCC, 2021)**. Building developers may or may not decide to use timber extensively, but so far there has been little to incentivise this: it has even been possible to get top ratings from the environmental accreditation schemes without paying much heed to embodied carbon, as adequate performance in one category can be traded against excellence in other categories. The picture is beginning to change, however, with embodied carbon being increasingly recognised in professional **(BED, 2020; RICS, 2017)** and planning circles **(Mayor of London, 2020)**. However, whilst the option for prescriptive requirements for building designs and materials exists **(World Green Building Council, 2019)**, the preferred approach is more typically to allow freedom to operate beneath a cap for whole-life or embodied carbon, which means that there will rarely be an automatic preference for timber or other bio-based materials. Furthermore, there is increasing caution about the use of timber in buildings as safety concerns have risen up the agenda in the wake of the Grenfell Tower fire, even though – ironically – the problem in that case related to synthetic materials.

## 1.8 Summary

This chapter has put the topic of the thesis into the context of the role that buildings play in climate change, and – by extension – the role that everybody involved in the design, supply chain, construction, ownership and operation of buildings can play in mitigating climate change. It should be apparent to all, by now, that there is no 'silver bullet' for reducing GHG emissions from buildings, and civilisation cannot afford to wait for one to appear. Therefore, all credible methods for mitigating climate change need to be evaluated (and constantly re-evaluated as conditions develop and change) and implemented accordingly.

Existing and new buildings must be more energy-efficient; energy networks (not just electricity) must be decarbonised; and the construction products industry and its supply chains also need to be decarbonised. Depending on the extent of the almost inevitable failure to rise to such challenges in good time, human civilisation may have to learn to live with less, which might mean any or all of: less floor space; less comfort; lower turnover of 'stuff' (including buildings); lower biodiversity; and a less-habitable climate.

### *Thesis Structure*

From this point, the thesis moves on to a review of how the climate change impacts of bio-based construction materials are assessed in the literature (Chapter 2). In Chapter 3, in line with identified research questions, the emphasis is on the information and data sources needed to support the modelling of carbon fluxes and associated climate change impacts, taking UK waste management, construction and forestry into consideration. The model developed and used to evaluate these scenarios is described in Chapter 4. Subsequent chapters present the results of such modelling, with Chapter 5 focussing on how timber waste management might be optimised to mitigate climate change, and Chapter 6 taking a wider view (forest carbon included) of the potential climate change benefits of scaling up the use of timber in UK construction. Finally, conclusions drawn from the results are presented in Chapter 7, along with lessons for government and industry about the potential for contributing to climate change mitigation objectives by deploying bio-based construction materials and the associated biogenic carbon, and how that contribution might be optimised.

# 2 Bio-based Construction Materials: Quantifying Impacts

*A literature review into the environmental assessment of materials and buildings with biogenic carbon content, particularly with respect to climate impacts.*

## 2.1 Introduction

In this chapter current thinking and practice relating to the assessment of the embodied carbon (EC) of bio-based or biogenic construction materials is discussed. The most often-used method for approaching this – life cycle assessment (LCA) – has many layers of complexity which are only increased by the need to consider (or the choice to ignore) the role of biogenic carbon storage in mitigating climate change alongside the role of Greenhouse Gas (GHG) emissions in contributing to it. Additionally, results of LCAs for single products or buildings may not be valid when scaled up to cover, for instance, a national construction or infrastructure programme or policy, so need to be considered alongside other methods. The use of LCA is therefore discussed in this chapter, together with its application to buildings and construction products with biogenic carbon content. The limitations of LCA are also highlighted, along with methods to provide a more accurate account of biogenic carbon within the LCA framework. From section 2.6, the discussion moves away from the bottom-up view of the environmental impact of products and buildings offered by LCA to a top-down view, in order to consider how carbon flows through the construction system and – more broadly – the technosphere and the biosphere.

## 2.2 Life Cycle Assessment and Buildings

Environmental impacts of buildings and construction products can be considered from a top-down viewpoint, in which energy flows and material volumes are mapped across an economy, a geography, or an industrial sector for instance, and the impacts of these fluxes are assessed. But the environmental impacts associated with a single building, for instance, can be assessed from the bottom upwards, by developing an inventory of all the components of the building and assessing the life cycle impacts individually and as a collection. This approach – LCA – can help building design teams working to minimize environmental impact for instance, but both viewpoints are needed when considering the future of a much larger entity such as a sector or country.

LCA is the accepted method for scientific investigation and reporting of environmental aspects of materials, products, and even buildings, and is the subject of this section. LCA is underpinned by accumulated scientific understanding of the interactions that substances have with the environment. As such, LCA can be used to assess a suite of impact categories, such as eutrophication, acidification, ozone depletion, and ozone formation.

However, as the focus of this work is on the relationship between material selection and GHG emissions and climate change, the only environmental aspect considered here is Global Warming Potential (GWP), usually (in line with convention) but not exclusively GWP100, i.e. GWP with a 100-year time horizon **(IPCC, 2014)**.

LCA requires the curation of an inventory of resource flows required for the creation of a given quantity of a material or product and then assesses the impacts. Suppliers to the construction industry provide LCA information in the form of Environmental Product Declarations (EPD) based on Product Category Rules (PCR) in accordance with the International Standard ISO 14025 **(BRE, 2014; ISO, 2010)**. When comparing LCA results of different options to inform design and specification decisions, a suitable functional unit for assessment and comparison must be defined. In some cases, this might simply be a fixed volume or mass of material: for instance, it can be appropriate to compare results for $1m^3$ of dried coniferous sawn wood from different species or sources if they have broadly similar properties. On the other hand, a given volume of wood will fulfil a different role to that performed by the same volume of concrete, so more context is needed to form a comparison. Therefore, a functional unit such as $1m^2$ of external wall with a given performance specification is needed, or – for a whole-building assessment – $1m^2$ of floor space, and a comparison can be made by constructing a life cycle inventory of all of the products and materials involved. In this way, EPD provide background data for LCA studies of buildings which can be carried out in line with EN 15978 **(BSI, 2011a)**. Such LCAs divide their analysis and results into separate life cycle categories, as illustrated in Figure 2.1.

A whole-life assessment covers all the stages shown in Figure 2.1. There are often justifications for performing a more limited assessment: for instance, if a comparison between different options is undertaken and equivalence is assumed for the use stage. And by definition, an embodied carbon assessment excludes B6 and B7. Much published research also excludes the end-of-life stage, only offering cradle-to-grave (A1-A3) or cradle-to-handover (A1-A5) for instance.

Other common terms for different scopes of assessment in construction include:

- Cradle-to-site (A1-A4) which additionally includes transport to the building site.
- Cradle to practical completion (A1-A5), which additionally includes construction processes on site, and the life cycle of materials consumed on site but not

incorporated into the building (e.g. formwork and offcuts). Construction site worker transport is optionally included.

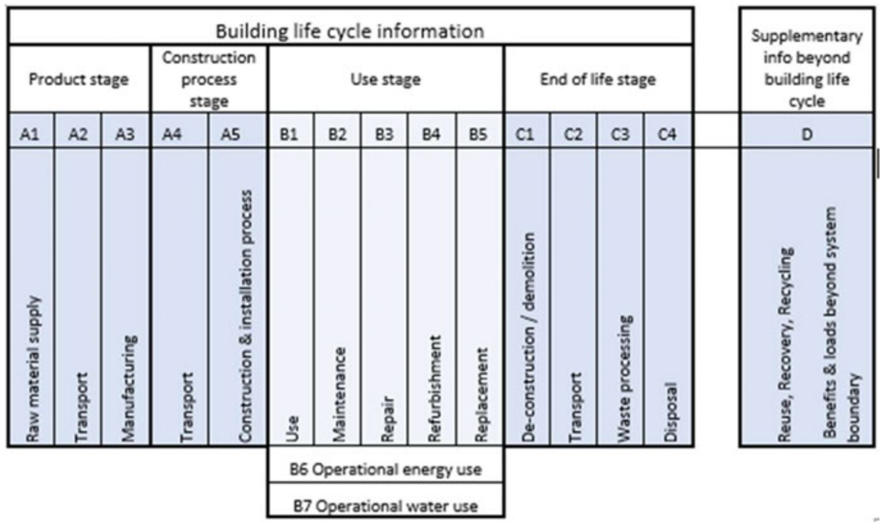


*Figure 2.1. Life cycle stages classification, as defined in EN 15978* ***(BSI, 2011a)*** *and used throughout this article. The stages that are covered by research in this thesis are shaded, although in a lighter shade for B1-B5, as – although these are not explicitly excluded – equivalence is usually assumed between timber buildings and their comparators.*

Module D is reported separately, and takes account of the environmental benefits and loads associated with subsequent use of materials at the end of life. For instance, in the case of timber extracted from a building, this could include the energy substitution benefits associated with thermal energy recovery, or the life cycle impacts avoided in the next product system by recycling or reusing this timber rather than virgin timber.

The full cradle-to-grave assessment (including module D) should not be confused with cradle-to-cradle (C2C). C2C is a different concept **(Bjorn & Hauschild, 2017)**, more a design philosophy associated with circular economy (CE) than an analytical method. 'Ordinary' cradle-to-grave LCA does, however, take account of some C2C/CE priorities giving credit where appropriate. For instance, using recyclate rather than virgin raw materials can lead to lower emissions at the product stage; and embedding priorities like design for adaptability can lead to better results across the piece, by prolonging the expected life of the building.

## 2.3 Static and Dynamic LCA Methods – Timing and Biogenic Carbon

For short-lived mineral or synthetic products, most GHG emissions occur during the months leading up to the product creation and potentially cease after its disposal, with the direct consequences of those emissions (i.e. raised concentration of the GHG in the atmosphere and associated climate forcing) decreasing over time. For long-lived products such as buildings, with a biogenic element, time is a more important variable, as relevant carbon fluxes take place in the forest decades in advance of construction, on the same land after construction, and from the eventual destruction or decay or combustion of the HWP. These nuances are not evaluated by conventional LCA studies, the majority of which – including EPD – are 'static LCAs', although this term has only been needed in recent years to distinguish them from the results of LCAs using dynamic techniques (dynamic LCA). In a static LCA, all emissions that occur within the study period (i.e. prior to the LCA endpoint) are summed and treated as one, as if they occur on the same day.

Dynamic LCA has been developed to capture and analyse the timing of carbon emissions and sinks in the life cycle inventory **(Levasseur et al., 2013)**[13]. The technique is potentially applicable to any situation in which GHG emissions occur over a period of time, but was described initially in the context of wood products, to include in the assessment the period of carbon sequestration occurring in the forest, and the temporary storage of the carbon in the assessed product. Dynamic LCA is discussed in more detail in section 2.3.3, but first, a discussion of the handling of biogenic carbon storage in static LCA is warranted.

### 2.3.1 Biogenic carbon storage in static LCA

EN15804 **(BSI, 2021)** does not permit either temporary or permanent biogenic carbon storage to be factored into an assessment, and even that fraction of biogenic carbon that is deemed to survive indefinitely in landfill is modelled as 'emitted to nature'. However, this does not prevent researchers from examining the role of carbon storage in mitigating climate change, through the medium of LCA. However, attention has been drawn to the lack of clarity in a high proportion of studies, around whether and how biogenic carbon is assessed in them **(Andersen et al., 2021)**. In static LCAs of bio-based products, there are several options for assessing (or not) and reporting biogenic carbon storage. These can be characterised as follows:

- The -1/+1 approach[14], in which biogenic carbon is 'checked in' to the system at stage A and recorded as a credit (a negative emission) and 'checked out' at stage C. **Arehart et al. (2021)** found that in 48 LCA studies including relevant search terms (a set that required some reference to 'biogenic carbon', 'carbon storage' or similar), 15 reported results using this approach (in some cases, amongst other approaches).

[13] It is also worth being aware that the term 'dynamic LCA' has been used previously in different ways, for instance to capture the dynamic nature of technology development **(Pehnt, 2006)**.
[14] This is shorthand for -1 $kgCO_2e/kgCO_2$ / +1 $kgCO_2e/kgCO_2$.

- 0/0 approach, in which biogenic carbon is excluded from the assessment altogether. Again, this was commonly observed in the **Arehart et al. (2021)** review (18 out of 48 cases). Ideally any LCA reporting biogenic carbon emissions would present results using the 0/0 approach alongside, so the role of biogenic carbon can be considered in context.
- 0/+1, means no credit being taken for the biogenic carbon sequestration despite a cost being applied to its emission. This is rarely seen in assessments, but see below.
- The difficult to justify -1/0 approach, in which biogenic carbon is checked in at stage A but is never checked out. This might be because the analysis goes no further than the product or construction stage. Or it might be because the product has a projected lifespan extending beyond the LCA endpoint: for instance a building expected to last for over 100 years. As many as 16 studies in the **Arehart et al. (2021)** review used this approach.
- A final, non-binary option, is -x/+y, which comes in a number of varieties. The most straightforward is where x=1 and y is variable, as in the case of the ILCD/PAS2050 methodology **(BSI, 2011b; JRC, 2010)**, where biogenic carbon is checked in, but the amount checked out is discounted (linearly) by 1% for every year that the biogenic carbon emission is delayed as a result of it surviving in situ, or in landfill (up to a limit of 100 years). Thus, for instance, in the case of combustion of a bio-based product after an 80-year life, the balance would be -1/+0.2, resulting in a net credit.

A justification for 0/0 and -1/+1 (which amount to the same thing in a static assessment) is that when the timber comes from a sustainably managed forest an assumption of carbon neutrality is appropriate: carbon extracted from the forest with the harvest is replaced by an equivalent uptake of carbon from the atmosphere in the forest. ISO 21930:2017 **(ISO, 2017)**, which provides the product category rules (PCR) for construction product EPD, allows the sustainability of the forest to be demonstrated by either one of two ways:

1. Providing evidence that the wood comes from forests certified to applicable sustainability standards, such as Forest Stewardship Council, the Sustainable Forestry Initiative, or the Programme for the Endorsement of Forest Certification.
2. UNFCCC[15] national reporting demonstrates a stable or increasing stock of forest carbon. This is a condition that many jurisdictions currently meet. In the UK, for

[15] United Nations Framework Convention on Climate Change

instance, the National Atmospheric Emissions Inventory **(BEIS, 2019)** shows forest land operating as a sink of around 5 MtC per annum in the last 20 years.

If neither condition is met, then the implication is that the situation should be assessed as 0/+1. In reality though, the system will not be binary and there may be a continuum between sustainable and non-sustainable, and any value between -1 and 0 for carbon uptake might be appropriate given the circumstances. And furthermore, it has been suggested that values outside this range might be justified **(Johnson, 2009)**: if carbon stock is accruing in the forest, then a larger carbon sink might be assumed (i.e. < -1 uptake, +1 at end of life, leading to a net result of < 0). For instance, the UK might be defined as one landscape for this purpose. With UK forestry currently operating consistently as a net sink of around 5 MtC/yr, and production statistics **(FAO, 2020)** showing that the UK has produced an average of 10.2 million $m^3$ of roundwood per year over the last five years (holding approximately 2 MtC per year), the characterisation factor for the carbon sink effect is – 5MtC/2MtC which is equivalent to – 2.5 $kgCO_2e/kgCO_2$. This, however, would imply that timber harvested in the UK is actually *better* than carbon neutral, but sadly matters are not quite so straightforward. Put simply, there is a problem of cause and effect: if we take a tonne of carbon out of the forest, it would take some convoluted reasoning to argue that this would lead to 2.5 tonnes of carbon being sequestered from the atmosphere. Market enthusiasts argue that extra demand for wood will result in increased incentives to optimise forest management for productivity, as well as to increase interest in land-use change **(Jefferies & Tracy, 2017; Tian et al., 2018)**. However, there can be no guarantee that this will be the case: forests have a range of valued functions in addition to timber production, and the gradual increase in carbon stock could be an indication of slack in the market. In this case, increased extraction might not lead to changes in land use and management, and will therefore result in *reductions* in carbon stock, suggesting a characterisation factor for the carbon sink effect of -1 at the best. Consideration of the average productivity of the wider forest landscape is also central to the concept of Ecosystem Carbon Cost metric used by **Head et al. (2019, 2020)**. This metric is the average net flux from forest to atmosphere per quantity of wood harvested per year.

Another variant of -x/+y is GWPbio (the net effect of -x/+y), put forward by Cherubini and colleagues **(Cherubini et al., 2011; Guest et al., 2013)**. This metric was developed to explore GHG emissions from bioenergy, but extended to use of biogenic materials in long-lived products. $CO_2$ emission pulses are given a weighting of -1 to +1 depending on the residence time of solid carbon. This is governed by a combination of rotation time for the crop and storage time within the product. For storage ($\tau$) and rotation ($r$) periods of up to 100 years, calculated GWPbio values can range from -0.99 ($r$ = 1 year, $\tau$ = 100 years – for example an annual crop resulting in a structural material) to +0.44 (100 years and zero respectively – for example a slow-growing timber crop used as biofuel), over a time horizon of 100 years **(Guest et al., 2013)**. Use of this method has been illustrated in an assessment of timber cladding on buildings **(Tellnes et al., 2014)**, with forest rotations of 30-100 years depending on species. The 60-year lifetime assumed for the product is sufficient to ensure at least a small reduction in whole-life EC in all cases, and a strong reduction for the short-rotation scenarios. However, in all cases the net change with respect to the 0/0 assessment is less than when the ILCD method is used, providing evidence that the ILCD method does not offer a conservative approach to biogenic carbon accounting. **Vogtländer et al. (2014)** have highlighted that this effectively pushes part of the impact of delayed emissions to a period beyond the assessment period, therefore allowing it to be ignored. As an illustration of the point, take the hypothetical example of a timber product incinerated after 60 years, and assessed for GWP100 over a 100-year period (itself an arbitrary point in time) and compared to the same product burned today. The emission occurring after 60 years will only have 40 years of the assessment period left to make its mark, and so the reported GWP of the delayed emission can be reduced by 60% compared to an equivalent emission at the start of year one. This curve-straightening approximation adopted by PAS2050 results in a systematic understatement of the impact of the later emission, as illustrated in Figure 2.2 (label 'E'), and therefore an overstatement of the benefit of the temporary carbon storage.

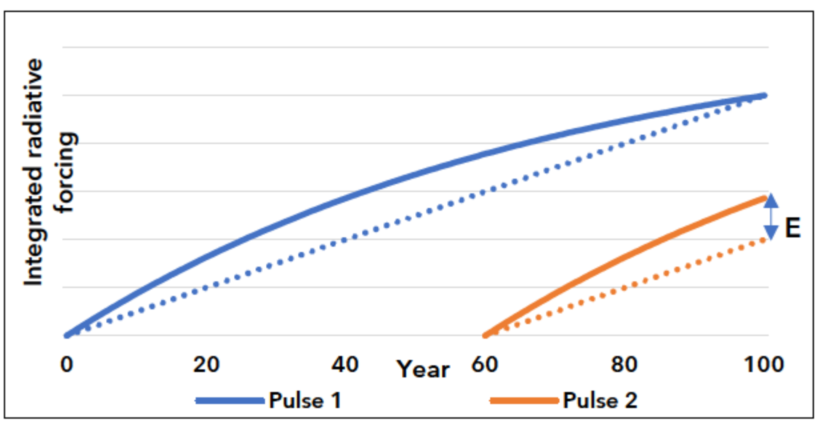


*Figure 2.2. Impact as cumulative radiative forcing, resulting from equivalent pulse emissions of carbon dioxide at year zero (pulse 1) and year sixty (pulse 2), with the linear emissions proposed by PAS2050 shown as dotted lines as a proxy for impact. E indicates the underestimation of the impact of the pulse 2 emissions after 100 years, when using the linear approximation.*

In common with other methods, the question of how to deal with allocation is an addressable challenge. When timber is harvested for durable products, which might have a negative GWPbio, a significant proportion of the harvest is much more rapidly oxidized when it is pulped or burned as fuel for instance (with a positive GWPbio). It can be argued that economic allocation is more appropriate than the default mass allocation option, and therefore that the durable timber should take some of the GWPbio burden of the short-lived component. This does, however, require additional steps and LCAs of buildings and bio-based products generally do not refer to allocation.

### 2.3.2 GWP time horizon

By convention, and for ease of communication, the climate change impact of a single pulse emission of GHG is distilled down to a single figure, usually GWP100 in the context of building LCA. GWP100 is a mid-point indicator – as it represents the cumulative radiative forcing over the subsequent 100 years, relative to the radiative forcing caused by a pulse emission of 1kg of $CO_2$ **(Balcombe et al., 2018)**. There is complex relationship between this and the physical endpoints in the system such as temperature change or sea level rise. GWP100 allows emissions consisting of different gases with different interactions with infrared radiation and with different decay rates in the atmosphere to be described in terms of a single figure of kilograms of carbon dioxide equivalents (kg$CO_2$e). But the choice of time horizon is, in a sense, arbitrary, and it affects the result. For instance, a pulse emission of methane-containing landfill gas is quantified differently depending on whether the focus is on instantaneous radiative forcing, cumulative radiative forcing over 20 years (GWP20), or over a hundred years (GWP100). GWP20 for methane is approximately three times as high as its GWP100 value, for instance.[16] Opting for GWP20 focuses attention on the radiative forcing over the next twenty years, and the impacts on climate that will follow up to around mid-century, but at the cost of drawing attention away from the focus needed on the emissions reductions required to stabilise climate in the longer term. **Balcombe et al. (2018)** review other climate metrics that are available and make recommendations on their application, which include transparency, reporting the impacts associated with different gases separately in some circumstances, and using dynamic approaches and end-point metrics for assessment of long-term decarbonisation pathways.

The GWP time horizon should not be confused with the LCA endpoint even though – in practice – they are frequently both set at 100 years.

[16] Methane – $CH_4$ – has a GWP100 value of 28 (or 34 including indirect – but less certain – climate-carbon feedbacks). Because methane has a shorter residence time in the atmosphere, its effect measured over shorter timescales is higher: GWP20 for methane is 84, or 86 including feedbacks **(Myhre et al., 2013).**

### 2.3.3 Dynamic LCA

'Dynamic LCA' includes the timing of all biogenic carbon fluxes in the life cycle inventory, and allows for increased storage of carbon in soils to be factored into the assessment **(Levasseur et al., 2013)**. The method involves compiling an inventory of all GHG fluxes deemed to be in scope, providing an account of the flux of each gas in each year. The impact on climate change is then analysed in terms of radiative forcing and – potentially – atmospheric temperature change. The method allows impacts to be viewed easily over any chosen timeframe, showing how, for instance, long-term benefits can be at the cost of short-term increases in radiative forcing. Results can be significantly impacted by the choice of whether the tree attributed to a product is the tree harvested or the tree planted to replace the one that is harvested. Or, put another way, does the sequestration precede the emissions associated with production ('growth' model – preferable in terms of results) or follow on from the production ('regrowth' model – less advantageous)? Although the growth model follows the actual biogenic carbon physically incorporated in the product, the link to the product and the actual sequestration of that carbon is not convincing. A more realistic approach would be to assume that the various agents responsible for the production of – for instance – a timber house can most strongly influence the processes that happen around the time of production, and should not take credit for events (planting of trees) that occurred decades earlier. **Fouquet et al. (2015)** make the same choice, for related reasons, in their assessment of a house design in alternative timber and concrete configurations using dynamic as well as conventional LCA. The conventional approach – leaving out biogenic carbon and module D – shows the timber option to have at least 10% lower EC than the concrete options. Using the dynamic approach, the advantage for timber becomes considerably more marked ten years or so into the life of the building (when carbon sequestration by the new trees is starting to make an impression) and projected centuries into the future.

Whilst there is some consensus that storage of biogenic may have a role to play in mitigating climate change, there is less consensus on how this role should be evaluated, and so current thinking tends towards leaving this out of LCA. The assessment of biogenic carbon remains a field where consensus is yet to be reached since different methodological choices and assumptions lead to opposite conclusions and incorrect or inaccurate assessments of biogenic carbon can be the cause for missed opportunities as well as inefficient or counterproductive strategies **(Breton et al., 2018)**.

## 2.4 Timber in Buildings – LCA Results

The relative environmental merits of concrete and steel structural systems have been debated for at least twenty years **(Jonsson et al., 1998)**. Engineered timber is now an increasingly recognised option for building structures, with examples of up to 14 storeys already realised, 24 storeys under construction, and even taller buildings planned **(CTBUH, 2017; Teshnizi et al., 2018)**. As a consequence, timber buildings and materials increasingly feature in such LCA work.

Several approaches have been adopted for investigating the EC of buildings with different characteristics. These range from like-for-like comparisons of pairs of individual buildings that differ only in the aspect of interest (e.g. the choice of material for the structure), to sweeping searches for benchmarks and trends from large samples of buildings differing in function, location, scale, and very often in the methodologies and scopes of the studies.

### 2.4.1 Static LCA

A body of recent work has provided insight into the embodied carbon of buildings and building structures, exploring a range of building types, life cycle stages (for instance, cradle-to-gate or cradle-to-grave), and scopes (for instance structural frame, or whole building). **Simonen et al. (2017)** identified 384 $kgCO_2e/m^2$ of floor area as a median value for embodied carbon across all types of building covered in their meta-analysis. There is evidence in the literature that the selection of timber as a construction material is a useful first step in targeting lower embodied carbon in buildings than this. In their cradle-to-gate analysis of structural frames, **De Wolf et al. (2016)** reported that timber frames had the lowest median value (~200 $kgCO_2e/m^2$) compared to the steel and concrete systems (at ~350-380 $kgCO_2e/m^2$). The ranges, however, are wide and overlapping, partly because of the variety of building types studied.

A meta-analysis of non-residential, single, whole-building LCA studies found that in eight out of eight studies, wood frames achieved lower GWP than concrete, and in five out of six cases wood was better than steel. The exception here was a steel design credited with high optimisation and durability **(Saade et al., 2019)**. A more recent meta-analysis of 79 papers and over 200 scenarios, found that timber buildings assessed by attributional LCA techniques had an average of 3.9 $kgCO2e/m^2$ p.a. over fifty years (so 195 $kgCO_2e/m^2$ when harmonised with the above examples), with this result disaggregated by LCA method and building type **(Andersen et al., 2021)**.

Where a timber building LCA is presented alongside a non-timber comparator, the result is sometimes reported as a 'substitution factor' as shown in Table 2-1, and if not, the substitution factor ($S_f$) can be easily derived as shown in Eq 2—1,

$$S_f = \frac{EC_{comp} - EC_{timber}}{EC_{comp}} \qquad \textit{Eq 2—1}$$

where $EC_{timber}$ is the embodied carbon of the timber building and $EC_{comp}$ is the embodied carbon of the comparator building. Unless carbon-negative assumptions are permitted, a perfect value for $S_f$ is 1, whilst a negative value implies that the timber option is poorer than the comparator.

Several studies have found significantly lower values for the EC in timber buildings, timber structural systems and timber components than in their steel, concrete or masonry counterparts, without including biogenic carbon content in the account. For instance, with regard to structural systems, **Hart et al. (2021)** identified a median value for EC of 119 $kgCO_2e/m^2$ for engineered timber as compared to 185 and 228 for concrete and steel respectively. And **Skullestad et al. (2016)** compared cradle-to-gate impacts for timber and reinforced concrete alternatives for four structures up to 21 storeys: results were in the range 111-121 $kgCO_2e/m^2$ for mid-rise concrete structures and 26-40 $kgCO_2e/m^2$ for timber. In this case the strikingly low values are at least partly attributable to the low emission factor of the Nordic electricity mix (0.139 $kgCO_2e/kWh$)[17], and sensitivity analysis showed a closing of the gap between timber and concrete if a higher emission factor is assumed.

Another metric often used to explore the value delivered by bio-based products is the displacement factor $D_f$, which quantifies the substitution benefit per quantity of carbon embedded in HWP. This is presented in terms of tonnes of GHG emissions avoided per tonne of carbon in the timber itself.

$$D_f = \frac{EC_{comp} - EC_{timber}}{MC_{timber} - MC_{comp}} \qquad \textit{Eq 2—2}$$

Where $MC_{timber}$ is the mass of carbon in the timber building, $MC_{comp}$ is the mass of carbon in the comparator building (which might include some timber), and EC is embodied

[17] Although five years later, it is now clear that the UK's own grid emission factor is well on the way to reaching such low levels.

carbon expressed in terms of carbon, not carbon dioxide, so $D_f$ has the dimensionless units of kgC/kgC. The utility of this measure is discussed in section 2.9.

Substitution factors and displacement factors that could be calculated from the studies discovered in the systematic search employed by this author and colleagues in **Arehart et al. (2021)** – along with the results from the author's own study are shown in Table 2-1.

| Context | $S_f$ | $D_f$ | Reference / Note |
|---|---|---|---|
| Hybrid CLT (mid-rise, non-residential) building versus reinforced concrete option – USA | 0.265 | 0.51 to 0.80 | **Pierobon et al. (2019)** |
| Bio-maximised building block versus concrete alternative – Sweden | ~0.48 | - | **Peñaloza et al. (2016)** |
| Wood frame multi-story residential building compared to steel and concrete - Canada | 0.22 to 0.38 | - | **Padilla-Rivera et al. (2018)** |
| Timber residential buildings versus mineral comparators - Germany | 0.09 to 0.56 | - | **Hafner & Schäfer (2018)** *Smaller buildings performed better than large* |
| Timber-maximised house designs versus standard alternatives - Australia | ~0.3 to 0.4 | - | **Ximenes & Grant (2013)** |
| Straw bale house (UK) versus comparator with masonry walls | 0.20 | 0.51 | **Sodagar et al. (2011)** |
| Mid-rise engineered timber structural frame versus concrete and steel alternatives (UK) | 0.36 to 0.48 | 0.51 to 0.85 | **Hart et al. (2021)** |
| Cross Laminated Timber (CLT) / reinforced concrete floor slabs, controlling for span length and load bearing capacity | 0.74 | 0.71 | **Hassan et al. (2019)** *Better than average values for shorter spans.* |
| Timber roofing systems versus steel (South Africa) | 0.92 | 0.95 | **Crafford et al. (2017)** |
| Structural sawn timber versus steel | 0.44 | 1.52 | **Bolin & Smith (2011)** |
| Agglomerated cork insulation compared to EPS | -0.48 | -0.49 | **Sierra-Pérez et al. (2016)** |
| Wood fibre insulation compared to mineral wool | 0.18 | 0.07 | **Densley Tingley et al. (2015)** |

*Table 2-1. Substitution factors ($S_f$) and Displacement factors ($D_f$) for studies that compare wood buildings or systems with typical counterparts.*

In most cases the values in Table 2-1 have been derived from tables and plots in the identified references in order to harmonise the results, so they are based on similar scopes (cradle-to-grave, excluding operational carbon, module D, and biogenic carbon from the embodied carbon numbers). The relevant quantity of biogenic carbon is not always readily discernible, in which case no attempt is made to calculate or estimate $D_f$. In all but one of the cases presented in the table, a positive substitution factor is calculated, meaning that a static LCA has shown that the bio-based option is preferred for embodied carbon.

**Purnell (2012)** investigated the EC of structural materials as a function of their load capacity. The importance of material selection for EC was found to be dependent on the context and general conclusions were drawn that timber should be preferred for very light duty columns and longer, light duty beams, whilst other cases needed more careful assessment. In many cases, it was found that the lowest EC values were achieved with reinforced concrete made from a mix optimised for low EC (C50/60 with 40% of cement substituted by pulverized fuel ash). However, the implications of using heavier columns and beams on the elements beneath them was not part of the analysis. Although concrete has lower EC than timber per unit of mass **(Pomponi & Moncaster, 2018)**, in an analysis by **Moncaster et al. (2018)** timber emerged consistently as being the lowest carbon option in practice.

The great variability in the embodied carbon coefficients (which indicate GHG emissions per unit of material for a defined life cycle stage) that might be ascribed to materials has been highlighted by **Pomponi and Moncaster (2018)**. Within the product stage, values for concrete are consistently lower than those for timber and steel in terms of $kgCO_2e/kg$ of material. But an analysis of a real building aimed at understanding the impact of methods on results **(Moncaster et al., 2018)** returned a different picture, with timber consistently being the option with the lowest embodied carbon out of the systems compared (CLT, reinforced concrete frame, steel frame, and load-bearing masonry).

The use phase—stage B—is usually pared down or neglected altogether in building LCAs **(Pomponi & Moncaster, 2016)**. This is because design for identical performance and maintenance is tacitly assumed and also because studies explicitly focusing on EC will exclude operational carbon emissions by definition. Poor data availability on maintenance and repair regimes and on the probability of major refurbishment is also a factor limiting the scope of many studies. Modules B2-B5 – maintenance, repair, replacement and refurbishment – are important aspects in a building life cycle, which can involve several iterations of redecoration, renovation, and even extension. In general, such interventions would either be too unpredictable to model, or would not impact significantly on the building structure (the focus of many studies), which will have a design life that matches the design life of the building **(Helland, 2013)**. Differing routine maintenance needs for different materials is a challenge to this approach, with **Caruso et al. (2017)** drawing attention to the additional maintenance needs of glulam structures. However in this case, maintenance needs are highly dependent on the specification of the glulam itself (timber species and initial treatment), on design details, and on exposure to moisture and ultraviolet light. Accordingly, it is not feasible to identify a generic maintenance regime, so it is typically left out of scope **(Lolli et al., 2019; Robertson et al., 2012)**.

For buildings and construction products in general, stage C impacts are typically reported as having relatively low values compared to those for the product stage (A1–3). **Pomponi & Moncaster (2018)** found examples in the academic literature whereby the stage C impact is approximated with a small, fixed percentage of the EC of the product stage. The situation with timber does not justify this approach, however: even if the -0/+0 approach to biogenic carbon analysis is adopted, the methane emissions from landfill (if landfill is an option for end-of-life) may be a significant proportion of the life cycle emissions. The ecoinvent database, for instance, indicates C4 embodied carbon coefficients relating to landfill that are very close in value to those for the product stage (e.g., for CLT A1–A3: 0.55 kg$CO_2$e/kg of CLT; C4: 0.54 kg$CO_2$e/kg).

### 2.4.2 Dynamic LCA Examples

An assessment of apartment block designs in Sweden, with concrete and CLT options – plus another with a further increased bio-based content – compared dynamic and static LCA results **(Peñaloza et al., 2016)**. As with nearly all cases reported in Table 2-1, the bio-based options are clearly preferred in a static LCA situation (i.e. $S_f > 0$). The results from dynamic assessment are more nuanced, with the results heavily dependent on time horizon. In the medium-to-long term, the concrete option is always the poorer option, but for time horizons of up to around 30 years, it outperforms the enhanced bio-based option in most of the experimental setups (the exception being the 'growth' scenario, in which trees are assumed to be planted and grown to serve the project rather than planted to replace those used in the project). With a time horizon of 100 years, however, the dynamic account is significantly more favourable than the static in most experimental setups.

Further studies have shown favourable results in all scenarios tested, using dynamic LCA methods for contexts as diverse as a small road bridge in Sweden (functionally equivalent concrete and timber design alternatives) and an enhanced HWP-use scenario across the Swedish building stock **(Peñaloza et al. 2018a, 2018b)**.

**Fouquet et al. (2015)** use a case study of a timber frame house compared to masonry/cast concrete alternatives. Under the conventional static approach – leaving out biogenic carbon and module D –the timber option has at least 10% lower EC than the alternatives. Using the dynamic approach, the advantage for timber becomes considerably more marked ten years or so into the life of the building (when carbon sequestration by the new trees is starting to make an impression) and projected centuries into the future.

In a comparison of different wall construction technologies **Pittau et al. (2018)** deploy dynamic LCA to highlight the advantages of storing fast-growing bio-based materials (straw and hemp) in the wall constructions. In general, the timber option out-performed the masonry / cast concrete options, but – in turn – this was easily out-performed by the constructions involving hemp and straw on all timescales (so long as straw is not ultimately landfilled).

## 2.5 Further LCA Limitations, Inconsistencies and Challenges

Despite the underpinning scientific approach, LCA does have a range of limitations, both in general terms, and when applied to buildings in particular, and again when applied to bio-based materials. These themes are introduced below, with aspects related to the assessment of biogenic carbon developed and explored through the thesis.

### 2.5.1 Forward-looking analysis and system change

LCA is useful for answering questions about the environmental impact of a product that is, in a sense, already 'in the system,' but assessment of future products and services is less secure, as their introduction may alter the system in which the products are assessed. For example, an established brick manufacturer wanting to determine the environmental effects associated with their existing product can get information from an LCA that might help them to improve the environmental profile of the product by adjusting the recipe. In, for instance, using less energy than they otherwise would, it might be argued that they have altered the wider geo-techno-economic system by making more fossil fuel available to others through the process of carbon leakage. However, it might be argued that the energy efficiency gain is already 'priced in' to the system as progress on energy efficiency is expected in principle, so we cannot say that the change in recipe has altered the system, and if it has the impact will be infinitesimal.

By contrast, the question explored in this thesis is not so easily answered with LCA. Firstly, the study is forward-looking, and secondly, the change proposed is widespread, potentially involving government policy and incentives, as well as numerous growers, manufacturers and products. In other words, it is looking at a system change rather than a tweak to a recipe. Although the focus is on the UK context, the prospect of global replication also needs consideration. Put simply, even if a 'perfect' LCA could be undertaken on wood products using currently applicable data, the results could no longer apply in a world in which timber is widely seen as the default structural material, because the geo-techno-economic system would have been altered so much to get to that point. The changes to the planet resulting from land use change would be visible to the naked eye from space, and the increased commercial use of timber would have a range of effects on the energy and agricultural industries to name two.

Whilst continuing improvements in energy efficiency, as discussed in the brick example, are a given (even if total energy consumption does continue to rise), the same cannot be said of increased timber usage. Scenarios can be presented (as they are in this thesis) in which timber is used much more in construction, but as a construction material, timber has been through previous periods of expansion and contraction in different regions. Timber is also not universally accepted as a solution for mid-rise and high-rise buildings. The Grenfell Tower fire in London, whilst not implicating timber in any way, has resulted in a heightened awareness of risk in the UK industry: in such circumstances contractors are known to err towards the more conservative approach of working with familiar methods. Even where it is demonstrated that timber is as safe as any other material, it may still carry the perception of risk, and that perception in itself is a risk to the industry.

This means that attempts to scale results from the product or building level at which studies are often carried out up to the level of the strategic impact assessment needed to influence policy development are not guaranteed to be valid. For instance, **Hafner & Rueter (2018)** scale up the results of building-level LCAs to test the potential impact of increased timber use by the construction industry in Germany: one scenario tested involves timber residential buildings increasing their market share from 16% to 55% across the country as soon as 2030. This is a reasonable place to begin, but such studies rarely include assessment of how the scaling up affects the inputs to the building-level assessment.

A range of consequential impacts can arise both within and outside the system. As one example of each: increased demand for construction timber may result in geographically longer supply chains and therefore an increased impact from transport; and it might cause a price increase, pushing existing users of timber products – whether sawn wood, woodchip, or pulp – out of the market and towards products with different life cycle GHG emissions.

### 2.5.2 Uncertainty and Variability

It can be particularly difficult to apply LCA meaningfully to buildings, as these are complex assemblies of many different materials and products with different expected life-times, maintenance needs and disposal pathways. For a full cradle-to-grave assessment, data on all such aspects must be included in the model. Inadequate data exists in a useable form on the likely lifetime, maintenance needs and disposal routes for different types of buildings and their constituents, and that which does exist is likely to be more pertinent to buildings constructed decades ago than the buildings being constructed now.

Sources of uncertainty in LCA (Figure 2.3) include the historic data used to derive embodied carbon coefficients (accuracy, completeness, and geographical relevance); uncertainties about future events (e.g. in-use and end-of-life emissions); and uncertainty in system boundaries and methods of measurement **(Gantner et al., 2018)**. This last point is crucial when coalescing data from a variety of sources with varying methodologies and degrees of translucency, which is a particular problem with building LCAs. Although international standards such as EN 15978 **(BSI, 2011a)** support consistency in principle, in some contexts it is not always clear where the boundaries should be drawn. **Emami et al. (2019)** investigated the significance of the database selection for the analysis of embodied environmental impacts of a residential building: for this they compared results of analyses using ecoinvent with SimaPro software, and GaBi software-database. For some impact categories the differences between the results are stark: more than an order of magnitude for marine eutrophication, for instance, although in the case of climate change the level of consistency is better (±15%).

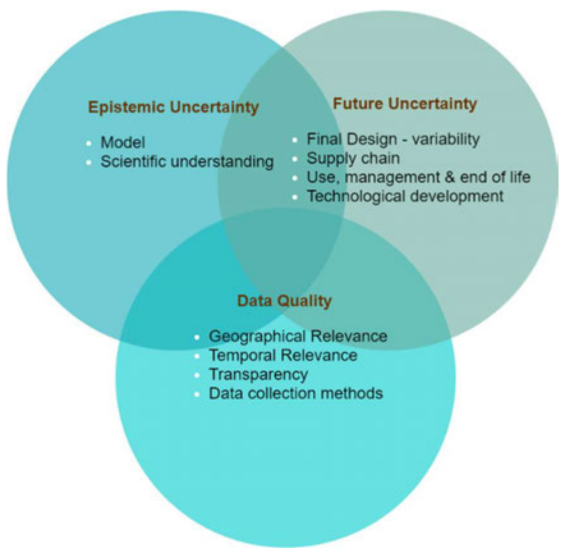


*Figure 2.3. Sources of uncertainty illustrated so as to emphasise how the different aspects overlap and reinforce each other.*

Future uncertainty is particularly important for buildings given their individual uniqueness (often) and their long lives, meaning that many of the impacts require individual modelling and will occur decades into the future. With some exceptions, LCAs underpinning EPD must now cover the product stage (A1-A3) the end-of-life stage (C1-C4), and Module D **(BSI, 2021)**. This ensures a reasonably complete picture (at least, as far as practicability and harmonisation between a wide range of product types allows: note, for instance, that maintenance and repair is not included in the minimum requirement), but at the cost of uncertainty in relation to assumptions around the future stage C. A further problem is that impacts far into the future are modelled with reference to today's technologies and practices **(BSI, 2021)**, which potentially introduces systematic bias into results for stage C and D (overstating emissions at C and overstating benefits at D, as a result of improving technology).

Epistemic uncertainty relates to things we could in theory understand, but are not yet able to **(Spiegelhalter & Riesch, 2011; Van Der Bles et al., 2019)**. This overlaps with data quality issues, but it also encompasses issues of scientific understanding of – for instance – the effects that substances have on the environment. This also includes model uncertainty: we may not always be focusing on the right target and in the right way, a point that is illustrated well by, for instance, the discussion of biogenic carbon in section 2.

A failure to engage with the topic of uncertainty and to communicate this in the results of a study imbues studies with an unjustified sense of finality, as noted by **Pomponi et al. (2017)**. The authors demonstrate that using Monte Carlo techniques to repeatedly sample small datasets is greatly to be preferred to using single values, whilst being at no real disadvantage compared to more arduous processes involving the compilation of large numbers of data points to produce more accurate distributions for sampling.

### 2.5.3 Handling of End-of-Life Issues

A general problem with the assessment of end-of-life impacts of bio-based construction products is (as alluded to in section 2.5.2) uncertainty around the treatment methods that will be in use when the time comes, and what the associated inventories will be. Currently the main options for such products – in 'waste hierarchy' order – are reuse, recycle, energy recovery, and landfill. These have benefits and loads that are picked up in different ways in LCA, or neglected altogether. To fully understand the life cycle impacts of timber in construction, detailed consideration of the end-of-life scenarios is required, and this is particularly important at the strategy level, as governments for instance can actually develop policy that supports or cuts off end-of-life options, whereas life cycle assessors can only attempt to identify plausible scenarios. **Morris et al. (2021)**, for instance, found that different modelling choices led to different conclusions about whether glulam should be preferred to functionally equivalent steel. A limited selection of scenarios is typically offered in LCA, without justifying the assumptions behind them or the relative likelihood of those scenarios being applicable. Examples include future landfill availability; biodegradation and landfill gas production rates in the local conditions; energy recovery efficiency; and carbon intensity of future counterfactuals (e.g. electricity produced from a grid that will be largely decarbonised when EoL is reached). These issues need more consideration in order to inform policy development, and the scenarios covered in this thesis take some of these factors into account.

### *Reuse and recycling*

It is sometimes possible for complete timber products and assemblies to be salvaged for direct reuse in a different building with minimal alteration, and the stored carbon continues to be stored. An LCA of the product in isolation might in theory cover multiple use phases, but in the context of its use in a building, it would have to be checked out of the system after its initial use phase, taking the stored carbon with it. The stored carbon continues to be stored in the next product system in reality, but to no overall effect in terms of the module D assessment, as the carbon is also checked out of the next system. Recycling of structural timber typically involves processing the material to lower-grade, shorter-life products such as particleboard. Again, stored carbon continues to be stored, but probably for a shorter period than for reuse as structural timber.

### *Energy recovery*

When waste timber is burned, the stored carbon is returned to the atmosphere, mainly as $CO_2$ (which is only debited from the LCA account at C3 if a credit for biogenic carbon was applied at the product stage), along with much smaller but still relevant quantities of methane and nitrous oxide. On the other side of the equation the heat and/or power generated offsets the carbon that would otherwise have been emitted in supplying energy to the user of the energy generated, earning credits in module D. However, the grid emission factor for GHG in the UK is on a downward slope, whilst the efficiency of burning biomass barely changes. Accordingly, the substitution benefit in the future (when materials specified now reach end of life) might be minimal.

Faith in the -1/+1 model of biogenic carbon (section 2) allows the calculation of a very low emission factor for biomass energy – biogenic carbon neutral overall, with the emission factor predominantly arising from processing and shipping the fuel rather than burning it. Many researchers, however, question the low-carbon credentials of energy from wood in almost any context. For instance, it has been noted that even in comparison to burning coal, burning wood results in a carbon debt that takes between 44 and 104 years to pay back, and that projected growth in wood harvest bioenergy will increase carbon dioxide in the atmosphere over many decades **(Sterman et al., 2018)**. **Vass & Elofsson (2016)**, who also take a less accepting view of the carbon neutrality of biomass, come out in favour of increasing sequestration in European forestry (at the expense of bioenergy and HWP) as the most cost-effective way for the sector to support EU climate policy. A Massachusetts Department of Energy Resources report **(Walker et al., 2010)**, notes that burning biomass results in higher immediate GHG emissions than burning fossil fuel – in effect creating a carbon debt that can take decades to pay off through forest regrowth (nearly a century, in fact, if the comparison is made with an efficient form of fossil fuel electricity generation, i.e. combined cycle gas turbines). It has also been shown that compared to a 'business as usual' scenario of a managed beech forest and natural gas for fuel, changing to an energy crop (poplar) was approximately neutral with respect to climate impact **(Taeroe et al., 2017)**. **Brack et al. (2021)** argue that the subsidies applied in the UK to the combustion of imported biomass actually undermine the 2015 Paris Agreement and increase the likelihood of tipping points being crossed, because it takes so long for the carbon stored in the harvested forests to be re-sequestered. **Millward-Hopkins & Purnell (2019)** also question the carbon-neutral credentials of virgin waste wood combustion (and much of the imported wood pellets is presented by the industry as being a by-product if not actually a waste), pointing out that reforestation does not automatically follow the combustion of waste wood. They observe that the biofuel market is not necessarily in and of itself a driver strong enough to influence the rate of reforestation and afforestation in the supply chain. Accordingly, to meet climate change objectives, the wood should remain in the living tree where possible, or otherwise stored in solid form if not.

In the UK, the sustainability criteria for future electricity generation 'Contracts for Difference' have been tightened to a level (supply chain emissions of 29 $gCO_2e/kWh$) that – unless relaxed under lobbying pressure – renders new contracts for imported wood pellets across the Atlantic unlikely given the distances involved. Even these sustainability criteria, however, take it as read that the forest itself is a carbon neutral system. The UK Forestry Standard **(Forestry Commission, 2017)** accepts the benefits of wood fuel, but with important caveats: for instance the wider range of benefits that conventional woodland can bring is typically missed by focussing on short-rotation forestry. And generally the removal of forest brash and stumps is not countenanced without a risk assessment that shows the carbon cost of extraction (factoring in soil carbon oxidation) is outweighed by the displacement benefits. Sustainable forestry certification schemes such as the Forestry Stewardship Council and the Programme for the Endorsement of Forest Certification (FSC and PEFC) tacitly support biomass combustion by certifying responsibly sourced biomass, but this is not sufficient for the Sustainable Biomass Program (the SBP, formerly the SB Partnership, a consortium of major biomass users including Drax, Suez and others) which needs more carbon-rich data to satisfy its reporting needs (e.g. supply chain fossil emissions). The SBP's own certification scheme includes a requirement that regional carbon stocks should be maintained over the medium to longer term **(Sustainable Biomass Partnership, 2015).** This, however, allows forest carbon levels to remain where they are, falling behind where they otherwise would be: a requirement for a significant rate of increase forest carbon might be a better objective, and one that – at the national level – much of Europe is meeting.

### *Landfill*

When timber is disposed of in a managed landfill, a high proportion of the carbon – especially in the lignin – is stored for the long term but the decomposition of the cellulose and hemi-cellulose that does take place produces landfill gas (LFG), which is typically around 50% methane, a powerful GHG. Biogenic carbon is checked out of the product system and 'emitted to nature' **(BSI, 2021)**: in the reality this is different to emissions to the atmosphere, but whether assessors recognise this in their work depends upon the standard they conform to.

### *End of Life in EPD*

In an analysis of publicly available LCAs for glulam and CLT,[18] **Hart & Pomponi (2020)** noted results encompassing wide ranges for each treatment method, with significant overlap between the ranges for different methods: this is at odds with the straightforward assumptions about the waste hierarchy. This is related both to dependence on context (especially with regard to the counterfactual assumed in energy recovery – module D), scientific uncertainty (especially regarding landfill gas production – relevant to both C4 and D), and modelling assumptions regarding the checking in and out of biogenic carbon (with one LCA checking all the biogenic carbon out at C4, and another only checking out the part that is eventually degraded and emitted to the atmosphere: not now allowed in EN 15804).

Furthermore, the apparent rewards (module D) for recycling and reuse are less than for energy recovery with a favourable counterfactual (e.g. a grid mix still heavily reliant on fossil fuel). This is because the benefit of extending the carbon storage through reuse and recycling is not captured which is correct technically[19], although not (in the view of the author) in principle.

[18] Mostly EPD: **(Cross Timber Systems Ltd, 2017; Egoin, 2018; Stora Enso & Divsion Wood Products, 2016; Studiengemeinschaft Holzleimbau e.V., 2017; Wood for Good & PE International, 2013; Wood Solutions, 2017)**

[19] If the -1/+1 system is applied rigorously, then this also applies to the subsequent product system.

### 2.5.4 Greenhouse gas fluxes in forestry

In general, LCAs of forest products account for the inputs to the forestry process for instance fertilizer, if relevant, and diesel used in the planting, management and harvest of the forest. However, even when biogenic carbon is included in the LCA (section 2), this is typically limited to the carbon physically embedded in the wood harvested,[20] with no thought given to soil carbon and non-GHG emissions from forest processes.

The PCR for wood-based construction products, EN 16485:2014 (**BSI, 2014b)**, draws a line between the technosphere and the ecosphere – a 'system boundary with nature.' On the technical side of the boundary lie the anthropogenic inputs and processes leading to the production of the timber; the sequestration of carbon from the atmosphere into the crop can be included, as this is a physical property of the product. But all other forestry processes are understood to be part of nature, even in a commercial monoculture. Therefore other GHG fluxes between the atmosphere, the trees, the soil and the groundwater associated with the growth and decay of trees are outside the scope of the PCR and of standard LCA practice. This simplification is both contestable and potentially justifiable. On one hand, the GHG fluxes in a commercial forestry stand are different to those that would occur if the land were put to different use. To the extent that increased demand for construction timber supports either new afforestation or future rotations of existing forestry, then the product ought to account for its contribution to the net change in emissions. On the other hand, there may not be sufficient data available to characterise the full range of GHG fluxes in the range of possible contexts (soil type, tree species, climate, stand age and rotation number, to name but a few variables) to make inclusion in LCA generally practicable.

#### *Forest Methane and Nitrous Oxide*

The UK Government's reporting on land use and forestry emissions **(Ricardo Energy and Environment, 2021)** in accordance with the relevant Guidelines for National Greenhouse Gas Inventories **(Pingoud et al., 2006)**, provide a top-down annual perspective. Figures for 2019 show forest land acting as a net carbon sink of 17.2 $MtCO_2e$, together with an associated increase of 2.0 $MtCO_2e$ in the national stock of harvested timber. However, non-$CO_2$ GHG emissions such as nitrous oxide and methane emissions from wildfire, direct nitrous oxide emissions associated with land-use change, and drainage and rewetting of organic soils show combined emissions of 0.85 $MtCO_2e$. This is a sharp

[20] Or in some interpretations, the equivalent quantity of carbon that will become embedded in the tree that replaces the one harvested.

increase on emissions reported in earlier years, following a correction to the methodology, and it offsets more than 40% of the reported HWP stock change.

The Read Report **(Read et al., 2009)** on the role of UK forests in combatting climate change reviews data on GHG fluxes in a range of forestry contexts. Figures from the few studies focusing on methane and nitrous oxide emissions from forest soils span a wide range, from a small sink right up to emissions of 1450 $kgCO_2e\ ha^{-1}yr^{-1}$ for standing forest, with one study (dating back to 1990, and possibly an outlier) reporting a much higher figure for nitrous oxide in the harvest year. Analysis of UK forestry statistics **(Forest Research, 2018)**, shows that an average hectare of commercial conifer plantation can produce enough wood for approximately 4.5 $m^3$ of sawn wood product per year; and it has been estimated that at least 4 $m^3ha^{-1}yr^{-1}$ can be produced from a Norway Spruce plantation **(Ramage et al., 2017)**, which would include around 750 kg of carbon. In the context of the LCA figures presented in construction product EPDs, figures of additional hundreds of $kgCO_2e\ ha^{-1}yr^{-1}$ arising from non-$CO_2$ GHG emissions would be potentially troubling. In general, nitrous oxide emissions are higher from relatively warm soils with a low carbon to nitrogen ratio and high application of nitrogen fertiliser **(Read et al., 2009)**, which is therefore not an issue for most upland forestry in the UK, for instance, but it may be a problem for much of the forestry in Europe supplying the UK market.

Additional indirect effects of forestry such as aerosols and changes to albedo are also generally not included in assessments, although their role is potentially significant **(Røyne et al., 2016).**

### *Forest soil carbon*

Another source of emissions relates to loss of soil carbon after any forestry rotation when land is drained for afforestation, or the decision is taken to remove the stumps, roots and brash for use as fuel. Meta-analysis has found a correlation between harvest and loss of soil carbon, albeit with further research required because of the many uncertainties around the effect of soil type, location, depth of sample, etc. **(James & Harrison, 2016).** With regard to afforestation of moorland organic soils, soil carbon lost over the first rotation can amount to around 450 $tCO_2$/ha – a loss not recovered until at least half way into the second rotation even when non-merchantable parts of the tree below and above ground are included in the account **(Read et al., 2009)**. However, a more recent review concluded that – despite the vast areas drained and afforested in the second half of the twentieth century – evidence surrounding the effects of afforestation on peat bog carbon balance in the UK is still very weak. Furthermore, such evidence is often based on research from other parts of the world, and it is therefore still too early to say whether afforestation of UK peatlands promotes or mitigates climate change **(Sloan et al., 2018)**.

Stump harvesting increases the soil carbon loss further as it involves disturbance to a depth of about one metre which, in organic soils, results in oxidation and mineralisation of soil carbon: the Read report **(Read et al., 2009)** identifies a range of 14 – 20 $tCO_2$ $ha^{-1}yr^{-1}$ for the first few years – possibly at least ten years – of the subsequent rotation. According to **Law et al. (2018)**, the utilisation of harvest residues, in general, for bioenergy results in increased emissions. **Ortiz et al. (2016)**, however, suggest that at the landscape level the impact of stump harvesting on soil carbon has been over-stated, with losses peaking at less than 4 $tCO_2ha^{-1}yr^{-1}$ a few years after the harvest. Their conclusion that stump bioenergy delivers climate change mitigation is undermined by the fact that it takes 12-28 years to repay the initial carbon debt, and this with natural gas identified as the counterfactual: a typical example of such comparisons being made with fossil fuel scenarios and not with the low carbon energy sources that are expected to dominate supply in forthcoming decades. This means that the analysis is skewed towards finding in favour of biomass burning. **Gustavsson et al. (2015)** also come out in favour of stump harvesting, although in cases where the comparator is not a coal-based system the evidence presented is equivocal: in some cases cumulative radiative forcing is higher for many years, or even decades, before benefits are realized. In their review of the environmental impacts of stump harvesting, **Walmsley & Godbold (2010)** indicate rewards of an additional 100 to 250 MWh per hectare for the harvest of stumps for bioenergy. The corresponding substitution benefit in relation to natural gas use would be a one-time benefit of approximately 20 to 50 $tCO_2e$, which would need to be set against the soil organic carbon losses referred to above. The actor who should take responsibility for such emissions is open to question – it might be the biomass energy facility that benefits from using the roots, the forester who elected to provide them when they could have been left in the ground, or the wood product industry which demanded the harvest that made the roots available. Current accounting responsibility rests with the forest itself, it being 'part of nature', but this does not appear to be a satisfactory division.

Although the UK Forestry Commission identifies soils with less than 5 cm of peat as being 'low risk' **(Moffat et al., 2011)**, the fact is that a large proportion of the UK's 'grasslands' are in fact peatlands: approximately 25% of the land area of Scotland (the UK's prime forestry landscape) is classified as having a peat depth of greater than 50 cm **(Evans et al., 2017)**.

### 2.5.5 Competition for resources

Although LCA can be used to investigate direct and indirect land use change, its capacity to offer a complete quantitative assessment, including socio-economic aspects of land use is debatable **(De Rosa, 2018)**. Furthermore, the methods discussed above have little to say on the capacity of the biological cycle to deliver the materials required for the construction industry, given competing demands on the land, such as biodiversity, recreation, agriculture and urbanisation. The construction sector is not the only one looking towards the biological cycle to reduce its carbon footprint. Others include biomass power generation, biofuels for transport including aviation, and bioplastics where once the primary focus was on biodegradable plastics but growth is now driven by the like-for-like substitution of petroleum-based products with biogenic equivalents **(Colwill et al., 2012)**. **Smith et al. (2010)** have reviewed competition for land at a global level, with the pressure on forests to give way to food production clearly evident. On the other hand, **Rounsevell & Reay (2009)** indicate a decrease in land area dedicated to food production and therefore offer a more sanguine picture about the possibility of some parts of the world contributing additional timber to the markets. Aside from food production, **Bruckner et al. (2019)** find that two thirds of the land area required to support the EU's consumption of non-food biomass are located in other parts of the world, notably China, the US and Indonesia.

**Hildebrandt et al. (2017)** have attempted to quantify the substitution benefits associated with an increase in the use of construction timber across Europe. Under their most optimistic scenario tested, the use of construction timber in Europe rises to over 30 million tonnes per annum by 2030 (approximately double the 2015 level). An increase in demand at this scale could have significant implications for net importers of construction timber, such as the United Kingdom, and for land use in general. The 2030 scenario in this study **(Hildebrandt et al., 2017)** requires additional land of approximately 7 Mha which is an area equivalent to the areas that are forested and available for wood supply in Austria, Latvia and the UK combined **(European Commission., 2003)**. **Kalt (2018)** focuses on timber supply and demand in Austria, and the possibility of a doubling of domestic demand in construction timber, and a tripling in the stock of carbon stored in construction by 2100. Austria's net exports of timber would still outweigh the extra domestic demand by a wide margin under such a scenario, which does provide reassurance about the likelihood of continuing supply from Austria and similar countries, but not necessarily the ability to meet increased demand internationally. **Ramage et al. (2017)** imply that in Europe, at least, this level of ambition would be justifiable: they estimate that no more than 30% of Europe's *existing* forest area would be needed to keep the population of the continent housed in timber. But **Allwood (2018)** states that global competition for biomass makes the general replacement of steel with timber unlikely, and **Ceccherini et al. (2020)** note that increased harvest rates across Europe are becoming evident, impacting the role of the forest in Europe's climate mitigation efforts. **Göswein et al. (2021)** have found that there is sufficient forest and cropland in Europe to support a radical increase in bio-based construction across the continent, but the article notes Ceccherini's warning, and identifies fast-growing crops as the best option – in particular wheat straw, as this involves no change of land use (in contrast to, for instance, hemp).

A study that combines a construction-related LCA with a look at the capacity of the local industry to deliver the material required – cork insulation – does not however go on to investigate the consequences of increasing the demand and supply of the material in the long run **(J. Sierra-Pérez et al., 2018)**. Such consequences might include land use change and raised prices pushing existing cork users towards synthetic products. This is where consequential LCA can help, and – it has been argued – might even be essential to socially responsible decision-making **(Weidema et al., 2018)**. **Searchinger et al. (2008)** have noted that whilst tightly drawn LCAs of bioethanol (from fast-growing crops) as a substitute for petrol produce favourable results, this is no longer the case if one includes land use change in the analysis. They found that price signals were sent to farmers either locally or in distant countries, to produce either the energy crop itself or other crops that had increased in price as a result of the market distortion, encouraging them to entrain what until then had been marginal land (e.g. grassland or primary forestry), thereby releasing soil carbon. It is worth noting, however, that consequential LCA can also be affected by the methodological choices made: one case study (spruce grown in the south of Sweden) demonstrated that eight different sets of methodological choices, gave widely divergent results **(De Rosa et al., 2018)**.

Competition for land and natural resources will remain fierce in the coming decades given current and projected levels of consumption, urbanisation and population growth. Future studies aiming to quantify the potential for, and benefits of, a global uptake of timber in construction should bear these competing demands in mind and work within realistic assumptions on resource constraints.

## 2.6 Material Flow Analysis

Despite the limitations and anomalies discussed above, there appears to be a consensus that LCA can provide useful information at the project level, for instance to help a building design team to choose between steel and engineered timber building structures.

However, attempts to scale results to inform high level policy development may be misguided. The embodied carbon associated with – for example – a million new homes cannot be reliably estimated by scaling up from an LCA of a representative development of a hundred homes, even though this approach is sometimes used (some examples are included in section 2.7.2 below). It is also clear that most current LCA practice (i.e. static LCA) does not successfully and consistently address the whole system: particularly the upstream impacts associated with forestry, and the temporary storage of carbon in harvested wood products (HWP).

The scope for durable harvested wood products (HWP) to mitigate climate change by storing carbon over long time periods has interested researchers approaching the topic from a variety of perspectives, including biogenic carbon in LCA, cascading strategies to extend the life of HWP, and the global and regional potentials for HWP carbon storage, in some cases linking this to carbon storage in the forest. The underpinning idea is that wood is approximately 50% carbon by oven dry mass, and that a growing technosphere (primarily buildings and landfill sites in this context) might add to stocks of carbon in stored HWP at a higher rate than stocks are removed through oxidation processes. Although this may result in carbon losses from the forest carbon pool (partially compensated by regrowth), in the right set of circumstances the net effect might be an overall increase in carbon stored in the combined forest-HWP system. The following sections consider this method, and associated research findings.

## 2.7 Carbon pools – the forest-HWP system

To assess the climate impact of policy to, for instance, increase annual production and consumption of construction timber by however many thousands of tonnes, an assessment is needed of what this would mean for carbon stocks in existing and as-yet unplanted forest and in HWP, as well as any substitution benefits (or offsets) calculated through LCA.

The approach for this combines a top-down material flow analysis (with a focus on carbon), coupled with information from LCAs. Figure 2.4 illustrates the flow of carbon in a closed system such as a region that is largely self-sufficient in timber. Carbon enters the system from the atmosphere through the process of photosynthesis, building up the store of forest carbon in the process (the forest carbon 'pool'). Some of that carbon is then physically transferred to the in-use HWP carbon pool and then to the landfill carbon pool, with losses to the atmosphere at all stages.

Literature on the subject is informed by the national accounting and reporting of carbon stocks in forests (and all other stocks and flows related to Land Use, Land Use Change and Forestry – LULUCF), and in HWP to the UNFCCC.

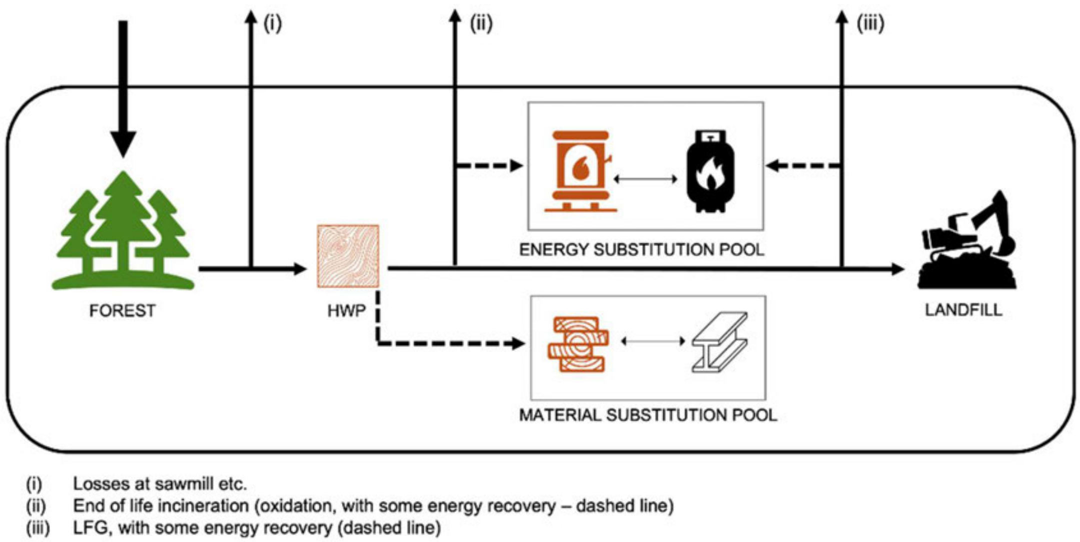


*Figure 2.4. The carbon pool system of forest, HWP, landfill and substitution pools. The solid lines indicate the physical flux of carbon from the atmosphere (photosynthesis), between pools, and back to the atmosphere. The dashed lines represent the contribution of the relevant processes (e.g. use of HWP instead of concrete, or landfill gas energy utilisation) to the 'virtual' substitution pools.*[21]

### 2.7.1 HWP Carbon – National Accounting

The IPCC has issued guidance and subsequent revisions on the reporting of carbon fluxes in forestry and HWP in national accounts **(Hiraishi et al., 2013; Pingoud et al., 2006)**, which has resulted in extensive literature comparing approaches and results. Of the three approaches detailed in the IPCC guidance (IPCC 2006), only the stock change approach can accurately reflect changes in stocks of all HWP in a given country or region, irrespective of the location of the forest. In the production approach (prescribed for national reporting by the IPCC 2013 guidance, and therefore the approach adopted by much subsequent literature) imported HWP is not of interest, but the storage of exported timber overseas is.

Although HWP stocks include the less durable paper and paperboard category, the transient nature of these products limits their contribution to the HWP stock, and it is reasonable to expect a significant proportion of HWP being attributable to long-lived products and construction. In their systematic review, **Arehart et al. (2021)** identified and

[21] This figure has also been used in **Arehart et al. (2021)**

normalised (by population) the HWP carbon stored in various countries, and also globally. In terms of how the wood carbon is stored, in the case of China, 76% of the identified carbon stocks are found to be in wood-based panels and sawn wood, 10% are in solid waste disposal sites (SWDS, or landfill), with the remaining 13% in short-lived products such as paper **(Zhang et al., 2019)**. Information that could in theory be used to segment the long-lived products category is provided by **Churkina et al. (2010)** and **Negro & Bergman (2019)**, who provide metrics for carbon stored in furniture in per capita or per floor space metrics.

On a per capita basis, the annual net increase in the HWP carbon pool varies from negligible (Japan in recent years) to more than 50 kg, and the cumulative HWP storage is typically equivalent to almost (sometimes exceeding) one year of energy-related GHG emissions. The distillation of future scenarios down to just one or two numbers of course conceals many important insights. For instance, **Kayo et al. (2014, 2015)** noted that wood promotion is required to prevent HWP carbon stocks in Japan from declining on account of decreasing HWP volume availability, although there is a possibility of increasing carbon storage in roundwood products in 2050 by 262% (2013 baseline), mostly in buildings. **Pilli et al. (2015)**, projected a decrease in carbon storage in the EU by 2030 under a 'constant harvest scenario', but storage can be kept at approximately the historical level by following an increased harvest scenario. This illustrates how HWP stocks can start to saturate over a relatively short period without aggressive HWP promotion initiatives.

### 2.7.2 HWP carbon – building stock scenarios

In addition to the interest in total quantities of HWP carbon stored within national borders, several articles have reported on the quantity stored in specific building stocks, either presently, in the recent past, or in future scenarios.

Several authors assess the carbon storage potential of future scenarios under aggressive adoption of carbon-storing materials at global and national levels. Results from these are summarised and normalised in Table 2-2. Additionally, **Peñaloza et al. (2018a)** analysed scenarios for new construction in Sweden over the next century, and found a total cumulative difference between scenarios of 2 MtC, including both substitution and storage effects. **Nygaard et al. (2019)** find that increasing timber in construction can make a significant contribution to 2015-30 de-carbonisation targets for Oslo and the surrounding area, although the contribution of the storage effect is secondary to their calculated substitution effects. The relative significance of this storage compared to population and wider GHG emissions varies significantly between studies, with some studies reporting

| REGION | ANNUAL CARBON STORAGE | | | | CUMULATIVE CARBON STORAGE | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Year | MtC $yr^{-1}$ | tC $yr^{-1}$ $cap^{-1}$ | Share of annual $CO_2$ emissions | Period | MtC | tC $cap^{-1}$ | Share of annual $CO_2$ emissions | Notes | Refs |
| Global | | | | | to 2015 | 6700 | 0.908 | 76% | (i) | **(Churkina, 2016)** |
| Global | 2050 | 680 | 0.0699 | 5.8% | 2020-2050 | 20390 | 2.094 | 175% | (ii) | **(Churkina et al., 2020)** |
| USA | | | | | to 2000 | 900 | 3.195 | 78% | (iii) | **(Churkina et al., 2010)** |
| USA | | | | | to 2060 | 33.8 | 0.089 | 2% | (iv) | **(Nepal et al., 2016)** |
| Philippines | | | | | ~2015-2060 | 8.7 | 0.060 | 18% | (v) | **(Zea Escamilla et al., 2016)** |
| EU-28 | | | | | 2018-2100 | 76.6 | 0.149 | 9% | (vi) | **(Pittau et al., 2019)** |
| Austria | | | | | 2015-2100 | 2.6 to 23.2 | 0.28 to 2.54 | 15 to 133% | (vii) | **(Kalt, 2018)** |
| Germany | Ave 2015-30 | 0.26-0.44 | 0.003 to 0.005 | 0.13 to 0.22% | | | | | (viii) | **(Hafner & Rueter, 2018)** |
| EU-28 | 2045 | 4.9 | 0.0095 | 0.6% | | | | | (ix) | **(Brunet-Navarro et al., 2017)** |
| Switzerland | | | | | 2016-2216 | 9.5 to 16 | 0.97 to 1.63 | 85 to 142% | (ix) | **(Mehr et al., 2018)** |
| Germany | ~2020 | 0.55 | 0.0066 | 0.3% | | | | | (ix) | **(Budzinski et al., 2020)** |

(i) carbon stored in urban areas
(ii) mid-rise timber frame buildings, 2020-2050, aggressive adoption scenario
(iii) snapshot of buildings and furniture in conterminous United States (note, includes an allowance for 300 kg of furniture per person, and is exceeded by the 2100 Mt of organic carbon stored in SWDS)
(iv) the additional carbon stored by adopting a high wood scenario compared to BAU
(v) Bamboo residential housing scenario after 45 years
(vi) Opportunity for storing carbon in wall retrofits, I-joists and straw
(vii) Residential construction - variation depends mainly on wood construction share of market
(viii) Residential buildings - reference and high timber use scenarios
(ix) Increase in cascading compared to reference scenario

*Table 2-2. Carbon physically stored in various construction-related situations (converted from $CO_{2}e$ in some cases). Population data and projections as far as 2050 from* **worldometers.info (2020)** *and national $CO_2$ emissions from energy consumption from* **IEA, (2020)**. *This table has also been used in* ***Arehart et al. (2021)***

2-3 tC/capita of realised or potentially additional storage. Other studies report their scenarios delivering much smaller benefits, with cumulative storage amounting to less than 20% of annual emissions: in other words, the benefits of carbon storage accumulated over a century, in some cases, is exceeded by emissions from energy consumption in around two months.

A number of different methods are employed in these studies. For instance, **Kalt (2018)** reports on the application of a dynamic stock model to the residential building sector (shell only) in Austria. With a significant increase in the timber share of the market, Austrian buildings can triple the carbon stored by 2100, without Austria losing its status as a net exporter of timber. **Hafner & Rueter (2018)** scale building LCAs to the national level (Germany) according to scenarios reflecting possible construction rates and penetration of the construction market by timber. Additionally, in their building stock model for Switzerland, **Heeren & Hellweg (2019)** use various construction, demolition and refurbishment scenarios to anticipate the volumetric flows of materials into and out of the residential construction industry. One of the scenarios refers to a doubling in the probability of timber use in construction, and in this scenario, by 2035, wood demand will be 0.8 Mt/y out of a total material demand of 12.2 Mt/y, and wood demolition removals will be 0.4 Mt/y out of a total of 5 Mt/y. Population dynamics point to decreasing material demand but increasing demolition waste over time. The total stock of wood increases from 31 Mt in 2015 to 46 Mt in 2055 (out of a total material stock of approximately 1330 Mt).

One study on MFA of wood in the construction sector in Europe includes historical data on the recent growth of the markets for several types of engineered timber in the years to 2014 which, along with construction scenarios, inform projections through to 2030 **(Hildebrandt et al., 2017)**. The reported figure of 46 $MtCO_2e$ p.a. carbon storage in HWP by 2030 is, however, virtual carbon in the substitution pool.

While there are many other studies that focus upon HWP at a regional scale, there is a missing link between the HWP and their use as a construction material. In order to really tackle this question, further primary research may be needed into understanding and quantifying the roles played by different product categories in buildings (e.g. structure, envelope or fit-out), and the different rates at which stocks of material in these roles are turned over in different regions, without relying on defaults.

## 2.8 HWP carbon plus forest carbon

This section summarises results from articles that look at stored carbon in the HWP-forest system, by region.

A Canadian study **(Chen et al., 2014)** is a reminder that past performance is not necessarily a guide to the future. In the 110-year study period to 2010, 7510 MtC (net) was stored in Canadian forests, with an additional 849 MtC accumulating in HWP. However, the increase in forest carbon is related to disturbance in the 19$^{th}$ Century and will not be repeated in the current period, therefore future opportunities are said to be in substitution benefits, so using timber more wisely should be emphasised, rather than using more timber. Focusing on Washington State, an overall carbon sink is reported when forests and HWP are considered together **(Ganguly et al., 2020)**. By contrast, **Nunery & Keeton (2010)** find that the best scenario for stocks of carbon in forests and HWP in the USA is no harvest. Thus, any intervention leads to a decline in overall stored carbon, with clear-cut harvest providing the worst outcome, with an average stock reduction of 85 tC/ha over the 160-year simulation period. Viewed from an alternative perspective, shifting from a clear-cut management system to individual tree selection increases carbon stocks by 41 tC/ha.

Moving to Scandinavia, **Soimakallio et al. (2016)** found that carbon sequestration in the forest exceeded the direct emissions from timber use and fossil fuel use in its processing, by 3.6 MtC. However, if the comparison is made with a reference system in which no harvesting takes place, then life cycle emissions averaging 15.1 MtC/yr are calculated. They conclude that it is unlikely that increased wood utilisation can contribute to significant emissions reduction target due to the net loss of carbon sink in the forest. In a retrospective analysis of data from HWP and forests in Finland, Sweden and Norway, 1960-2015 the three countries are found to transition from current sources to sinks between 2000 and 2014, but on a cumulative basis it takes until 2020 to 2045 to enter carbon negative territory **(Iordan et al., 2018)**.

**Knauf (2016)** uses a wood market balance to follow the flows of carbon in the German forestry and HWP. The climate change mitigation contribution of Germany's forest sector amounts to as much as 15% of Germany's total GHG emissions. The increase in the HWP stock accounts for only a small part – 6% - of the sector's contribution, with the remainder coming from substitution effects and an increase in the stock of standing timber. **Pittau et al. (2019)** assess the potential of retrofit of buildings across the EU as a carbon sink. Their analysis picks up on the case for rapid action, i.e. by incorporating fast growing crops into building facades.

In contrast to the many studies on intensive management in boreal and temperate forests, **Alice-Guier et al. (2020)** studied the carbon balance of selective logging in Costa Rica. They found that 0.443 tC per hectare of forest per 15-year cycle was stored in the resulting construction products: 0.030 tC/ha/yr. Whilst the total harvest is significantly larger, forest growth appears to exceed extraction overall as a result of growth rates increasing after thinning

## 2.9 Substitution pools

Analyses of carbon pools are typically founded on quantification of carbon flux and storage in a system comprising forest, buildings or HWP more broadly, and solid waste disposal sites (SWDS, or landfill), but they are frequently extended to include an ever-increasing 'virtual' pool of substitution benefits (fluxes into these pools indicated by dotted lines in Figure 2.4). Energy substitution in this context is the substitution of fossil fuels by gaining useful work from the combustion of end-of-life wood, or landfill gas. Material substitution benefits are the life cycle GHG emissions avoided by choosing HWP rather than, say, concrete or steel. Some authors express any advantage that timber construction has over mineral alternatives as a displacement factor $D_f$ (Eq **2—2**), providing an alternative perspective to that given by the substitution factor (Eq 2—1 ). $D_f$ is useful for incorporating virtual pools into the carbon pool system (a carbon-following MFA), as it quantifies the substitution benefit per quantity of carbon embedded in HWP.

Amongst those who have assessed the forestry-HWP system for carbon, there is a tendency to rely on substitution benefits (i.e. that timber has lower associated emissions than steel and concrete) to make the case for timber. Some authors argue that the substitution benefits are permanent, in contrast to the physical carbon pool, which is destined for eventual release back into the environment, with 86% of sequestered carbon lost within a century according to **Ingerson (2011)** . However, Harmon and colleagues **(Harmon, 2019; Harmon et al., 2009)** label substitution as (only) a theoretical carbon pool that is over-estimated by some authors by an order of magnitude, and with a built-in double-counting mechanism that is initiated when the timber is itself replaced. Therefore this virtual pool does not provide the promised ever-increasing climate change mitigation contribution, and certainly not when projected decades into the future; furthermore, the process of 'leakage' means any gains are not permanent. There are further facets to this discussion picked up in Chapters 3 and 4, but one simple point is that as energy networks and industry continue to decarbonise, the real displacement and substitution factors will decrease: this is already occurring in many regions, but models tend to assume constant displacement or substitution factors: this overestimates the substitution benefits. Examples of research that do account for this factor include **Peñaloza, et al. (2018a)** and **Kalt (2018)**.

In their much-referenced meta-analysis of displacement factors, **Sathre & O'Connor (2010)** find an average $D_f$ of 2.1 kgC/kgC, with most of the 21 studies coming between 1.0 and 3.0; and **Geng et al. (2017)** find a $D_f$ range from 0.25 to 5.6[22] in studies dating between 1993 and 2016, including reference to a study comparing wood framed buildings to steel and concrete alternatives, for which the range was 0.9 to 2.2 **(Lippke et al., 2004)**. The wide ranges reported in meta-analyses are a reflection of the different contexts in which timber is compared to other construction materials, and also to differences in the scope of the assessments. **Geng et al. (2017)** also includes a discussion of forestry for bioenergy, and points out that when taking into account the reduced forest carbon pool that results from the increased harvest, the benefits from fossil fuel substitution are not always sufficient to justify the emission of biogenic carbon at the point of combustion. **Nepal et al. (2016)** apply a $D_f$ of 1.68 to the analysis of scaling up of non-residential construction in the USA: when the boundary is extended to include changes in the forest and HWP carbon, the average $D_f$ increases to 2.03. It is worth noting that much of the source material for $D_f$ values is decades old: as manufacturing gradually decouples from GHG emissions, $D_f$ values should decrease over time, and a more recent study **(Smyth et al., 2017)** does indeed report a $D_f$ of 0.54 for sawn wood, and 0.45 for panels. This is much more in line with the results from various studies presented in Table 2-1: in all but one case in this table, $D_f < 1$.

A question regarding displacement factors is whether performance levels can be defined from first principles rather than – for instance – by dividing a ranked list into percentiles. On this basis, the most obvious thresholds are $D_f = 0$ and $D_f = 1$. A negative $D_f$ implies that the introduction of biogenic carbon is automatically damaging from a climate change perspective. For a positive $D_f$, the climate change impact very much depends on the length of service of the biogenic component. $D_f > 1$ suggests that even for a temporary building with a very short service life, and the stored carbon emitted to the atmosphere within a few

[22] The figure of 5.6 in this study is an outlier, and the relevant scenario relies on the substitution benefits of energy recovered at end of life.

years, the substitution benefit is more than enough to offset the rapid transfer of carbon from forest to building and on to the atmosphere even when a carbon neutral assumption does not apply (i.e. 0/+1 rather than -1/+1). This does not, however, take account of the carbon in co-products associated with the harvest: if these are secondary products, there is a case for including them in the carbon account, in which case the threshold for a 'safe' $D_f$ would be around 2 (based on half of the harvested carbon surviving into the building).

There may be far-reaching consequences about the decision on whether or not to include substitution benefits, as studies that rely on the substitution benefits tend to favour more intensive forestry management practices, greater inputs and higher rotation rates and all the environmental impacts that go with these. **Gustavsson et al. (2017)**, for instance, assess three scenarios for Swedish forestry: business as usual; enhanced set-aside protected forest (resulting in more standing biomass); and more intensive forest management (using fast-growing species, fertilization, and greater extraction of brash and thinnings). Their conclusion was that the intensive option is best from a climate perspective, with an average benefit of 31 $MtCO_2e$ per year, over the next 100 years. However, if one subtracts their figures for fossil fuel (coal) displacement and building substitution benefits, the figure would be just 2.4 $MtCO_2e$ per year. Their analysis relates to the Swedish forest industry supplying construction timber for up to 400,000 apartments per year[23], substituting for the same number of apartments based on a reinforced concrete structural frame. To put the claim into context, the 100-year cumulative benefit of 3100 $MtCO_2e$ is more than 8% of the current annual global emission **(Le Quéré et al., 2016)**.

**Oliver et al. (2014)** also rely heavily on substitution benefits to reach their conclusion that increased use of wood products in buildings and infrastructure could result in global reductions in $CO_2e$ emissions of 14 – 31%. **Lippke et al. (2011)** illustrate the supposed benefits of product substitution over a long period, with accumulated substitution and displacement benefits reaching around 500 tonnes of carbon per hectare after 160 years (and, in another case, around 1100 tonnes after 300 years). This dwarfs both the forest carbon (which peaks at around 160 tC/ha at the end of each rotation) and the HWP carbon pool itself.

**Law et al. (2018)** also include substitution benefits in their analysis, but they make different assumptions and reach different conclusions. Referring to forestry in Oregon, they conclude that rotation periods should be increased, to facilitate greater carbon storage

[23] This is beyond Sweden, of course, where the market size is much smaller, a reversal of the UK situation, where perhaps 200,000 homes per year might be constructed with enhanced quantities of *imported* timber.

in the forest, and harvest residues should not be utilised for bioenergy. One factor contributing to this conclusion is what the authors take to be more realistic assumptions about the rate of loss of stored carbon from buildings, associated with their lifespan. Another important consideration is that reducing harvest and increasing rotation periods can yield results in the short term, as the carbon losses associated with harvest and processing are reduced, and therefore the strategy can make a contribution to climate mitigation prior to 2050.

Research relating to the Pacific NW clearly lays out the areas of uncertainty and controversy surrounding on-site and off-site forest carbon pools, and notes that results frequently hinge on the inclusion or exclusion of substitution benefits, and the associated assumptions about product lifetimes **(Fain et al., 2018)**. The authors review the various approaches taken by others, which include, on one hand, accumulating the benefits of substitution almost *ad infinitum* and, on the other, defining a more limited period for the calculation (70 years in the example given).

An alternative to using 'off-the-shelf' displacement factors is to invert the viewpoint and investigate the $D_f$ required to achieve certain goals. In scenario analysis of carbon pools related to 1 $m^3$ of harvested wood, a $D_f$ as high as 2.9 is required to offset the overall emissions when natural gas is the energy source for the material substituted **(Butarbutar et al., 2016)**. Another study finds that to justify a 33% increased harvest of timber in Finland, a $D_f$ of 2.4 is needed. However, they report that the average $D_f$ is likely to be below 1.1, which presents a serious challenge to increased harvesting in Finland study **(Seppälä et al., 2019)**.

**Chen et al. (2018a)** assume a generous displacement factor of 2.43 to underpin their more optimistic conclusions about the benefits of increased harvest in Canada. In particular, they argue that better targeting of forest products towards long-lived HWP allows the carbon debt of increased harvest rates in the Ontario province to be repaid within 20 years, and – at the end of the simulation in 2100 – an extra 187.9 MtC of carbon pooled. For Canada overall, it will take from zero to 84 years to repay the carbon pool losses from harvest (84 years is business as usual, zero years when there is a dramatic shift towards structural panel manufacture as these have the best $D_f$). A sensitivity analysis using a low-end estimate of 0.68 tC/tC for $D_f$ resulted in the minimum time to carbon sequestration parity for structural panels being 75 years, not zero **(Chen et al., 2018b)**.

**Werner et al. (2010)** consider the forest, HWP and substitution pools under different scenarios in Switzerland, concluding that use of wood in long-lived construction products is its best role in climate change mitigation. **Braun et al. (2016)**, calculate a climate change mitigation efficiency (CCME) metric for timber use in Austria, in the range of 0.61 to 0.68 $tCO_2e/m^3$ of wood used (averages between 2025 and 2100), depending upon the scenario. Physical and virtual carbon pools are considered here, but energy substitution is the dominant force, as wood is calculated to substitute a mix of fossil fuels throughout the period of the study.

## 2.10 Conclusions

LCAs of buildings and structures consistently support the view that use of timber results in a building with lower EC. However, LCA needs support from other methods in order to assess the merits of a strategic shift towards the use of more bio-based materials in construction. Within the scope of the LCA itself there are difficult questions about the potential impacts at end of life that are rarely answered, and factors potentially leading to changes in GHG emissions are neglected by LCA altogether. A significant body of research already exists into the many factors that should contribute to such an analysis, but there is still much that can be learned by adopting and adapting methods that have been used in the past and applying them to the question of the role for bio-based materials – especially timber – in UK construction and climate change mitigation. As such, the research in this thesis adopts the material flow analysis approach discussed from section 2.6 onwards, but it is also informed by the body of LCA studies on timber in construction in order to evaluate substitution benefits. Furthermore, the research also adopts methods used in dynamic LCA (section 2.3.3) to explore the impact of the timings of the fluxes of carbon between the various carbon pools.

Chapter 3 continues the review of relevant literature, but with a specific focus on evidence needed to construct and populate (with data) the model described in Chapter 4.

# 3 UK perspectives – scenario development

*Identification and interpretation of information and data sources used to populate the modelling in this thesis. UK waste management, construction and forest industries and trade. Evaluation of energy and material substitution.*

## 3.1 Introduction and Industry Overview

The purpose of this chapter is to identify and evaluate the information and data sources that can help to answer the research questions by facilitating model initiation and the development of realistic scenarios.

The overarching objective that the modelling and research builds towards is the question of:

> *How much greenhouse gas emissions can be avoided and what associated climate benefit can be achieved over time periods of up to 100 years by coupling an ambitious but realistic increase in construction timber usage in the UK (preferably supplied from domestic forestry) with an afforestation agenda designed to meet future demand?*

The secondary question is:

> *How much greenhouse gas emissions can be avoided and what associated climate benefit can be achieved by making immediate changes to timber waste management priorities in the UK?*

Answering these questions involves the investigation and quantification of anthropogenic carbon pools and the climate change mitigation effect associated with using more timber and with different waste management practices. Achieving this requires data and scientific knowledge on a range of topics, including construction industry demand scenarios, the longevity of construction products, the fate of biogenic carbon at the end of product life, and many aspects of forestry in the countries which supply the UK's construction timber including the UK itself. The Biogenic Carbon Pools model (BioCarp) developed for this research is described in full in Chapter 4, and the modelling results in subsequent chapters. Other tools have been developed to investigate carbon balance in the forestry and HWP system. In the UK in particular, the CARBINE carbon accounting model **(Forest Research, n.d.-b)** was developed to assess carbon stocks in forest and HWP at the stand, forest and national level. This is discussed further, alongside the discussion of the model developed for this work, in section 4.1.

Themes explored in this chapter relate to inflows and outflows to each of the carbon pools considered in the model, and the assumptions and data required. Whilst a wide range of topics must be considered, ranging from tree growth rates to waste management to future industrial decarbonisation, such discussions fit into one or more of the following categories, which is the order in which the topics are approached in this chapter.

- The in-use HWP carbon pool, including construction and end-of-life scenarios
- The stored underground HWP carbon pool, comprising
- The landfill carbon pool, and
- The carbon capture and storage (CCS) carbon pool
- Emissions to atmosphere from landfill gas and from energy recovery from both solid waste and landfill gas
- The forest carbon pool.
- Material and energy substitution pools.

### 3.1.1 UK Forestry Overview

After centuries of decline in forest cover, only 5% of the UK's land area was forested at the end of the first world war. The Forestry Commission was established in 1919 to reverse this decline through a combination of state planting and providing incentives for planting on private land. Since then, woodland cover has extended to over 13% of land area, and is still increasing (at an average of 13,200 hectares per year between 1998 and 2020, for instance) **(Forest Research, 2020; Forestry Commission, 2017)**.

With the primary emphasis being on timber production initially, fast-growing non-native species of conifer were investigated and tested (and these days, hardwoods only constitute around 5% of the harvest), with Sitka spruce widely preferred to native species such as Scots pine. As a result, uplands in the north of the UK – especially Scotland – have been widely planted with Sitka spruce. The benefits of such planting have been contested, for instance where carbon-rich organic soils (e.g. deep peat) have been degraded and biodiverse habitats reduced to extensive monocultures. However, since around the 1970s, wider objectives for forestry projects have been increasingly recognised and adopted, including recreational amenity, landscape, regulating water flows, limiting soil erosion, and climate change mitigation **(Forestry Commission, 2017)**. More than 80% of UK timber produced – products from all state forestry and most private forestry – is now covered by sustainability certification (Forestry Stewardship Council or Programme for the Endorsement of Forest Certification - FSC/PEFC) **(Forest Research, 2020)**.

Climate change mitigation is now a significant driver of government policy in relation to forestry, and the forest carbon stocks have increased by an average of 4.1 MtC per year since 1990 (excluding the change in soil carbon stock, which is attributed to the change in forest area). The Woodland Carbon Code is available for those making claims about carbon sequestration and storage in UK projects. Woodland Carbon Code projects must exhibit permanent land use change (to woodland cover), and there are a number of environmental safeguards. Carbon leakage into areas outside the project boundary must be avoided, and additionality has to be proved: i.e. that the afforestation project would not be a rewarding investment without the Code **(Woodland Carbon Code, 2022)**. 32,000 hectares had been validated by March 2021 **(Forest Research, 2020)**.

### 3.1.2 Production and Trade

More than half of UK softwood production is delivered to sawmills, with the most significant alternatives being wood fuel and wood-based panels; pulp and paper mills take a relatively small share. Although the number is declining as smaller mills close, there were still 141 sawmills processing softwood in the UK in 2019. Of these, the 12 largest mills (more than 100,000 m3 per year) process nearly two thirds of the UK's softwood. The main markets for sawn softwood are (in volume order), fencing, construction and packaging/pallets, although for the Scottish mills, construction is the leading market.

Ultimately, just over 3 hm3 (hectometres cubed – i.e. million cubic metres) of sawn wood and a similar volume of panels are produced per year (the latter including a significant volume of recycled wood). Considerably higher volumes are imported, including 7 hm3/yr of sawn wood, 3.7 hm3 of boards and nearly 9 Mt of wood pellets (much of this from across the Atlantic, for the Drax power station), whilst exports are small in comparison (the strongest category by volume being recovered paper) **(Forest Research, 2020)**.

### 3.1.3 Waste, Recycling and Recovery

Strong markets exist for end-of-life wood, as long as it is not designated hazardous waste because of contamination with toxic preservatives (telegraph poles, railway sleepers etc.). Board manufacturers have an insatiable demand for end-of-life wood, and about a quarter of the input to wood-based panels made in the UK is recycled fibre (and much higher proportions can be used in particleboard), with the bulk of the rest obtained from sawmills or directly from UK roundwood. However, there is strong competition for this material from the energy sector, and the market for wood fuel from post-consumer wood has grown more than fourfold between 2010 and 2019, to 2.39 Mt **(Forest Research, 2020)**.

More detailed perspectives on timber usage, trade and waste management in the UK are included throughout this chapter.

## 3.2 The In-Use Harvested Wood Product Pool

The BioCarp model is used in this thesis to quantify in-use HWP in different contexts, and therefore different sets of data and assumptions are needed to support the modelling. In particular, for the analysis in Chapter 5 the material of interest is the HWP that already exists and is now in use but coming to the end of its life. Whereas, in Chapter 6 the material of interest is HWP that has not yet been produced, and indeed – when looking far enough ahead – will not yet even exist as forest carbon.

### 3.2.1 Waste management in the UK

Detailed national data on the recovery and recycling of wood from construction and demolition waste in the UK is scarce, but a number of publications over the last decade coalesce around a figure of 4 to 5 Mt of wood waste produced annually **(Defra, 2012; Pöyry, 2009)**, with around half of the material coming from construction and demolition. It has been estimated that in 2011, 50% of the 10 Mt of wood consumed in the UK was by the construction industry, and 51% of the 4.1 Mt of wood waste arisings came from construction (25%) and demolition (26%), with variation in the latter linked to the fortunes of the building refurbishment industry **(WRAP, 2011)**.

Most recently, the UK waste wood market in 2019 has been estimated to be 3.98 Mt, from an available pool of approximately 4.5 Mt **(Wood Recyclers Association, 2020)**. The balance of 0.52 Mt might be assumed to be landfilled, for instance as part of unsegregated construction and demolition waste, but differing conclusions have been reached about this, from the same data source. Analyses of data from the Environment Agency's Waste Data Interrogator for the previous year (2018) show – on one hand – that 0.97 Mt of wood was landfilled in England alone **(DEFRA, 2020)** and – on the other hand – that landfill represents less than 1% of the wood reprocessing market (so ~0.045 Mt) with speculation about the destination of the remaining balance **(Trada, 2020)**. The destinations of the 4.5Mt of wood waste are shown in Table 3-1. According to PAS111:2012 **(BSI, 2012)**, the animal bedding component must come exclusively from Grade A waste wood (typically uncontaminated and untreated packaging and offcuts), whilst the board manufacturers will use a mix of Grade A and B waste wood (which can include some solid wood C&D waste along with associated contaminants). The board manufacturers can tolerate only a small proportion of former panel products in their feedstock, which means that the majority of such material is destined for Grade C, which means biomass fuel.

| Fate | Share EoL timber (%) | Mass (Mt) |
|---|---|---|
| **Panel Board** | 22% | 1.00 |
| **Animal bedding, equine surfaces, other recycling** | 7% | 0.32 |
| **Biomass** | 55% | 2.47 |
| **Export** | 4% | 0.20 |
| **Unaccounted** | 12% | 0.52 |
| **Total** | 100% | 4.50 |

*Table 3-1. Fate of UK end-of-life timber in 2019* **(Wood Recyclers Association, 2020)**.

Initiatives and investments inspired by the circular economy may significantly increase the rates of reuse and recycling of timber products when they reach end of (first) life. There are multiple barriers to the reuse of any construction products at significant scale, as discussed for instance by **Hart et al. (2019)**, but these may be overcome given time and the incentives and investment required. General principles for design for deconstruction and reuse, and specific novel examples are given in an InFutURe Wood report **(Cristescu et al., 2020)**, and significant increases in reuse from the current low level must be possible, although it is difficult to be precise about the realisable potential.

One option for reuse that demands more attention is for used solid wood – and more generally wood of mixed quality and origin – to be remanufactured into CLT panels **(Llana et al., 2020)**. It might even be argued that this is a form of upcycling, as something often regarded as a premium product usually made with virgin wood is instead created from elements of a simpler end-of-life product.

A landfill tax of £7/tonne on active waste was introduced in 1996 (see **Seely (2009)** for the tax history) and after a bedding-in period, was cautiously and then steeply escalated, reaching a current level of £96.70 per tonne **(HMRC, 2021)**. This has provided an increasing incentive to find alternatives to landfilling waste wood, and it is reasonable to expect that such policy drivers will continue to further diminish the share of wood going to landfill.

### 3.2.2 Wood product lifetime

Whilst information about current waste management practice is useful for modelling the relative merits of different waste recycling, recovery and disposal options, much more information is needed in order to understand the timing of such processes for wood currently entering the in-use HWP stock, which is relevant to the modelling and results presented in Chapter 6.

For each category of wood product defined in the model (for instance, virgin structural timber), a function is required to define the rate at which it is removed from use, to enter the end-of-life management processes of re-use, recycling, recovery and disposal. There are different methods for approaching the question.

IPCC guidance on national GHG reporting provides a steer regarding the handling of HWP lifetimes in a number of product categories **(Rüter et al., 2019, Pingoud et al., 2006)**. The approach follows a straightforward first order decay (FOD) assumption in which the rate at which material exits a given pool at a given time relates to the size of that pool at that time. The decay constant (k) for a given pool such as structural timber implies a half-life ($t½$) for material in that pool, such that

$$t½ = \ln(2)/k \qquad \textit{Eq 3—1}$$

The IPCC guidance detailed above refers to an earlier version **(Penman et al., 2003)** for suggested half-lives of different HWP product categories, which are themselves drawn from what, at the time, was recently published literature. The half-lives of materials for which IPCC has allocated defaults are shown in Table 3-2. Note that these figures can be understood as the time period from the point of manufacture to the point at which half of the material has been disposed of (irrespective of whether the material is recycled). Therefore a full accounting of carbon from forest to atmosphere should also allow for the period from forest to product (for instance, a year to allow for drying time and progress through the supply chain), recycling, and partial storage in landfill. Of the product categories listed in Table 3-2, the most recycled is paper: in the UK it has been calculated that 79% of packaging paper and board is recycled (which makes up 51% of the UK's 11.3 Mt annual consumption of paper), and 69% of all paper[24] used in the UK is recycled **(CPI, 2019; Defra, 2020)**. From this information it might be deduced that the half-life of the fibres across multiple paper and paperboard product cycles is at least 3.2 times as long as the half-life of the individual product.[25] However, this is not necessarily the case, as packaging products are the most likely destination for recyclate – irrespective of the fibre quality – and packaging products have a relatively high average turnover rate. Therefore the average half-life of all paper fibres is likely to be well within the given range.

In their article on the place of recycling in this discussion, **Brunet-Navarro et al. (2017)** draw on these half-lives for their modelling, but they apply them as mean life-times of normal distributions (the standard deviation set at one third of the mean) whereas the mean lifetime derived from FOD principles would be given by Eq 3—2.

[24] Includes more than 1 Mt of tissue paper, which is not recycled.
[25] i.e. 1/(1 – 0.69), plus a time allowance for logistics.

$$\tau = \frac{1}{k} = \frac{t½}{\ln(2)} \qquad \text{Eq 3—2}$$

In modelling of the lifetime of the carbon secured in HWP in these product categories, the following recycling rates are assumed: sawn wood 30%; panels[26] 10%; and paper/paperboard 70% **(Brunet-Navarro et al., 2017)**.

| HWP Category | IPCC Default half-life (years) | Range of half-lives reported. | Note |
|---|---|---|---|
| **Saw wood** | 35 | 18 – 50 | (i) |
| **Structural panels** | 30 | 30 – 50 | (i)* |
| **Non-structural panels** | 20 | 20 – 23 | (ii) |
| **Paper** | 2 | 0.5 - 7 | (iii) |

*Table 3-2. Summary half-life figures from* **(Penman et al., 2003)**. *Notes. (i) compounded with other categories in some cases, *the IPCC default category also includes veneer and plywood. (ii) Particleboard referenced specifically in one case. (iii) Various sub-categories define the ranges (e.g. newsprint, household & sanitary paper = 0.5); data from one sub-sub-category (20% of printing and writing paper = 10 years) left out of the table as this has a small and diminishing share of the market. Additional notes: packing wood is also identified in one study (t½ = 3 years); wood fuel is not mentioned, but a reasonable assumption would be to use the same figure as for single-use papers (t½ = 0.5 years) as utilisation and disposal occur simultaneously in both cases; and finally, the appendix states that for the categories of 'solidwood' and paper, an uncertainty range of +/- 50% might be assumed.*

The assumption of FOD is, of course, a modelling assumption that must be an over-simplification of what happens in reality, but defensible on the grounds that whilst more sophisticated decay functions may by be more credible at an intuitive level, they may be equally lacking in evidence to support them. As part of their justification for the consideration of alternatives, **Pingoud & Wagner (2006)** assert that "*it is obvious that the decay from old age classes of a certain wood product is higher than from young age classes*". This is itself an over-simplification, and it is easy to identify counter-examples of wood products in the built environment for which the reverse is true, for instance structural timber of sufficient antiquity for heritage conservation to be important, compared to identical timber products in a more modern setting.

One theoretical possibility would be to use annual data returns reporting the mass of construction timber waste coming from demolition and refurbishment projects, and project forward by adjusting for activity. As discussed above, the data required for this approach is absent.

Another top-down approach is to look at demolition rates of buildings and estimate the quantity of wood released by each demolition. If the wood content and demolition rates of different archetypes could be defined, then this would be a rich source of information. The reality, however, is that demolition data is also scarce: in the UK it is difficult to get much

[26] Structural and non-structural panels treated as a single category with a lifetime of 25 years.

further than a crude estimate based on the difference between the new housing completions and the growth in number of dwellings in a given year. But such growth may partially be driven by subdivision of larger units, as evidenced by the otherwise paradoxical growth (by 212,000) in the number of dwellings built before 1919, between 2008 and 2017 **(UK Government, 2018)**; on the other hand, the average area of pre-1919 buildings increases slightly over the same period, suggesting that extension of old buildings outstrips subdivision. Or that data gathering is not sufficiently consistent from one year to the next to pick up such nuances.

Demolition rates in the UK are certainly very low currently and can only be responsible for a very small fraction of the waste timber entering the system. For instance, in England in 2019-20, a net increase of 243,770 new dwellings was reported **(MHCLG, 2020)**, with the number of new builds (220,600) out-pacing demolitions (9020) by nearly 25:1. The remaining part of the net increase (32,190 dwellings) is partly the result of subdivision of larger properties into flats, but is primarily attributed to change of use, whereby commercial buildings are repurposed towards dwelling accommodation (very often through the application of permitted development rights). Going back to 2006-7, the average is only a little higher at 13,800 demolitions per year. This demolition rate amounts to only 0.06% of the housing stock. Thus, the odds against a randomly selected dwelling being demolished in a given year appear to be around 1700:1, or 6E-04. The gradual decline in demolition rates can partly be explained by the fact that much of the poorer quality stock has already gone, firstly through war (WWII), then through slum clearances up to around 1980; and subsequently there has been a stronger emphasis on building conversions and refurbishment **(Wilson & Barton, 2021)**.

The reported demolition rate in Scotland is higher, with an average of nearly 4000 per year since 1990 **(Scottish Government, 2019a)**, which is equivalent to about 1.5E-03 of the existing stock **(Scottish Government, 2019b)**. This Scottish demolitions data is caveated with the advice that it should be understood as a minimum level of demolition because of the different and sometimes incomplete data collection methodologies employed by local authorities. It might be surmised that similar issues underlie the lower rates reported in England statistics above.

Even if good demolition data were available, it would still be very difficult to model future demolition rates. In the last two generations there has been significant variation in response to different drivers (as alluded to above). The current low demolition rates cannot continue indefinitely, as this would involve the lifetime of buildings designed for lifetimes of 60 years or so to extend for millennia. New drivers will emerge that will make existing buildings undesirable or uninhabitable, and therefore more susceptible to demolition. These might include:

- Impacts of climate change, such as flooding, coastal erosion, overheating
- A requirement to eliminate operational GHG emissions from all homes, especially with regard to heating and air conditioning
- Demographic changes
- Changes in taste and what society expects and requires from housing
- War, revolution, pandemic.

Whilst some of this might reasonably be modelled, it becomes increasingly speculative as one progresses down the list.

Another interesting variable is the link between a building's age and the probability of it being demolished in a given year (the hazard rate). **Zhou et al. (2020)** argue for the use of an age-specific hazard rate to describe the probability of demolition, and find the model useful for understanding building stock turnover in rapidly urbanising regions undergoing high turnover and city rebuilding, such as China. This makes sense, but in the UK the underlying research and data required to support such an approach is lacking. For instance, is an older building necessarily at higher risk of demolition than a newer one, because of an increasing backlog of maintenance tasks; or is it less likely to be demolished because it has already proved its resilience, it has been more solidly built, to more generous space standards, and without including dangerously flammable products that deceitfully navigated through the testing and certification system **(Sharkey, 2021)**? A data series on the conservation listing of buildings might in theory be used as a proxy to answer part of the question, but primary research is really needed to determine the age distribution of buildings currently being demolished, and relate this to the age distribution of the existing stock.

**Miatto et al. (2017)** compare demolition rates in three cities (including, in the UK, Salford) with various curves, and finds there is no universal answer as to which curve fits best. In Japanese cities with relatively high stock turnover, data can be best fitted to the lognormal distribution.[27] In cities that have been subject to some form of cataclysm over the study period (such as Salford's slum clearances), such a conclusion no longer applies. Indeed, it is

[27] Other distributions have been used: for instance Zhou et al. opt for a Weibull distribution **(Zhou et al., 2019)**, but **(Brunet-Navarro et al., 2017)** point out that validation of these is generally not possible, given the dearth of data.

arguable that the estimation of hazard rate is much more important than the selection of probability distribution.

The data and information quoted above for Scotland and England are too vague for an accurate statement of hazard rate: the values discussed above respectively, are 1.5E-03 $yr^{-1}$ and 6E-04 $yr^{-1}$, but for the reasons given, they should be regarded as minimum hazard rates, and that significantly higher probabilities should also be tested in a scenario analysis.

In dynamic building stock modelling of eleven European countries, the paucity of data for such analysis in GB is noted **(Sandberg et al., 2016)**: the study constructed the following picture of domestic building stock in GB. After construction, demolition in the following 40 years is zero,[28] then demolition at a constant rate until 10% of the stock remains, which is retained for the very long term. Dwellings have an average lifetime of 175 years,[29] and the demolition rate for GB is given as 0.003 $yr^{-1}$ (rising to 0.004 $yr^{-1}$ in 2030 and 2050). This is low even by European standards: the mean for the eleven countries in the study is just over 0.005 $yr^{-1}$.

A further consideration is that a 'deep retrofit' for improved energy performance and resilience may be regarded as taking a building back to a 'good as new' state. In terms of the end product, this is functionally equivalent to demolition and rebuild at the same address, but in terms of process (particularly embodied carbon, and construction and demolition waste) large differences should be expected. That being said, significant quantities of material – including HWP – are likely to be removed from a building during deep retrofit (and refurbishment in general) and as such the current low demolition rate disguises the much higher rate at which waste HWP is generated. Accordingly, the modelling in this work draws on the simple hazard rates for the HWP (not the buildings themselves) implied by the data in Table 3-2. An alternative to the FOD approach is also used in sensitivity testing.

### 3.2.3 Construction Timber Utilisation in the UK

Sections 3.2.1 and 3.2.2 have focused on flows out of the in-use HWP pool. Here the focus is on the influx to the pool, with the main point of interest being the potential increase that might follow market and policy developments.

[28] 'Negligible' would be a more realistic word than 'zero' here, albeit lacking the precision required for modelling.

[29] 'Average' is not defined. It is here taken to be either the mean or the median of the 90% that are eventually demolished.

The Forestry Commission's Timber Utilisation Statistics **(Timbertrends, 2015)** provides data and insights into the supply and use of construction timber in the UK from 2002 to 2014, especially when reviewed alongside the latest statistics on timber production **(Forest Research, 2020)**, which cover periods of five, ten, or more years up to 2019.

Sawn soft wood consumption in the construction sector has ranged between approximately 4.8 and 7.5 $hm^3/yr$ (hectometres cubed – i.e. million cubic metres - per year), with the trend following the fortunes of the construction industry. The figure for 2014, at the end of the series, is 5.95 $hm^3$ **(Timbertrends, 2015)**, and the construction industry consumed 63% of the total sawn wood consumed in the five years to 2014, with pallets, packaging and fencing taking most of the remainder. The origin of this timber is discussed in section 3.5.

Consumption of wood-based panel products in the UK can be extracted from the FAO database **(FAO, 2020)**. Over the five years to 2020, the average consumption has been 6.39 $hm^3$ (range 6.10 – 6.63), with an even split between production and net imports. The proportion of this material used in the construction industry is not specified. The breakdown of these figures is shown in Table **3-3**.

| Category | | Volume ($hm^3$) | Totals ($hm^3$) |
|---|---|---|---|
| **Sawn wood** | Coniferous sawn wood | 9.96 | 9.96 |
| **Wood-based panels** | Plywood | 1.34 | 6.39 |
| | Particleboard | 2.82 | |
| | OSB | 0.70 | |
| | Hardboard | 0.11 | |
| | MDF & HDF | 1.36 | |
| | Other | 0.05 | |

*Table 3-3. Average annual consumption of coniferous sawn wood and wood-based panels in the UK 2016-2020, in million cubic meters.*

### *Timber used in UK housing*

Application of the model discussed in Chapter 4 requires the choice of a realistic scenario for future increases in the use of construction timber, which in turn requires an understanding of the market as it is, along with recent trends and possible drivers of future change.

The proportion of construction sector sawn wood used in new housing, 2010-2014 averaged 8.4% (range approximately 6 – 10%). In 2014 for instance, new home construction used 0.555 $hm^3$of domestic timber, 75% of which was imported[30]. The average volume of timber used in housing starts increased from 2.87 $m^3$ per dwelling in

[30] The rest of the construction industry – e.g. non-domestic buildings and renovations – used 5.395 $hm^3$ of sawn wood in 2014, of which 85% was imported.

2009 to 3.39 $m^3$ in 2014 (or around 20 $kg/m^2$). This trend was driven primarily by an increase in the use of timber frame construction, which more than offsets timber's losses in terms of concrete ground floors and lighter forms of timber joists etc. **(Timbertrends, 2015)**.

If the 2014 figure of 3.39 $m^3$ of sawn softwood per dwelling was concentrated entirely on timber frame buildings, when timber frame had 25% of the UK market **(STA, 2017)**, then the average content per timber frame dwelling would be 13.6 $m^3$, an average of 0.19 $m^3$ of sawn wood per $m^2$ of floor area based on the average new home being approximately 70 $m^2$ **(LABC Warranty, 2019)**. In reality, the quantity per average home should be significantly less, as much of this sawn wood is also used in masonry construction systems. On the other hand, this figure does not include wood-based panels, and Table 3-3 shows that for every 3 $m^3$ of sawn wood used in the UK, approximately 2 $m^3$ of wood-based panels are also used.

Taking a bottom-up view, it is worth bearing in mind that the only universal aspect of a timber frame building is the existence of a timber frame attached to some form of structural sheathing (which might or might not include timber products). Therefore, the embodied carbon of a timber frame building is highly variable, even when normalised by floor area. Similarly, the embodied carbon of the masonry alternative is not set in stone. Furthermore, the masonry alternative may or may not include significant volumes of sawn wood or wood-based panels in the specification. Therefore any attempt to report a global displacement factor for wood in new housing must be heavily caveated.

In their analysis of the carbon abatement potential of wood in UK construction, **Spear et al. (2019)** analyse eight housing archetypes, comparing embodied carbon (A1-A3, sequestration excluded) for the whole building with that for a masonry comparator (Table 3-4). At first sight, the additional carbon sequestered by timber frame buildings appears low, but this is because the comparator buildings already have significant quantities of timber and sequestered carbon in joists and trusses. A significantly higher displacement factor is hinted at for timber cladding compared to brick cladding, but not followed through because of unresolved questions about product lifetime.

An assessment of a 4154 $m^2$ multi-storey, multi-family residence in Hackney that uses CLT floors, roof, external walls and party walls **(Darby et al., 2013)** provides results similar to those quoted in Table 3-4: 174 kg of CLT per $m^2$ and a displacement factor of 0.84 (cradle to grave, with sequestered carbon re-emitted at end of life). In their assessment of a lighter glulam/CLT frame[31], with a mass of 80 kg/m$^2$, **Hart et al. (2021)** suggest an average displacement factor of 0.51 with respect to a reinforced concrete frame, and 0.85 with respect to a steel system, but with considerable variation and uncertainty arising respectively from design details and emission factors.

| | Area [m$^2$] | Reduced EE [t$CO_2$e] | Additional Sequestered carbon [t$CO_2$e] | Normalised additional wood mass [kg/m$^2$] | Displacement factor [kgC/kgC] |
|---|---|---|---|---|---|
| **Bungalow** | 58.5 | 1.7 | 2.0 | 21.4 | 0.85 |
| **Detached** | 117 | 3.2 | 4.2 | 22.5 | 0.76 |
| **1-bed CLT** | 50 | 12.8 | 12.4 | 165 | 1.03 |
| **2-bed CLT** | 70.1 | 18.0 | 17.3 | 165 | 1.04 |

*Table 3-4. Displacement factors calculated for four housing archetypes representing extreme cases in* **(Spear et al., 2019)**. *Mass of wood and displacement factor derived from the data in the source.*

Details in an article on a case study LCA of a semi-detached timber frame house with larch cladding by **Monahan & Powell (2011)** suggest a timber content of up to 153 kg/m$^2$: insufficient information is provided to calculate a displacement factor, but appears to be more than 0.85, possibly boosted by the inclusion of the timber cladding, as also shown by **Spear et al. (2019)**.

### *Housing construction scenarios*

In order to forecast the quantities of timber required for new housing in the coming decades, estimates of the total number of homes built per year are needed, together with the average size of those homes and the proportion that are designed around timber (e.g. timber frame, or CLT).

Going through a similar process, **Spear et al. (2019)**, suggested two options for the rate of construction of dwellings. These are, firstly, low growth in activity (1% p.a.), leading to an increase from around 200,000 new dwellings per year to around 260,000 per year by 2050. The second option is a high activity scenario – enabling the UK Government's 300,000 homes per year target to be met by 2044 – in which annual growth initially proceeds at 4% before falling back to 1%. It is worth noting that even the low activity scenario may be optimistic, as even this low growth rate would result in housing construction at rates not seen since the 1970s, despite many Government commitments in the intervening decades.

In terms of penetration of the house construction market by timber, **Spear et al., (2019)** identified four scenarios. Firstly, timber construction fixed at current levels, resulting in a falling market share. Secondly, timber construction maintaining its current market share.

[31] Frame alone.

Thirdly, a moderate growth rate, whereby timber frame takes 35% of the market by 2025 and 40% by 2050, with engineered timber climbing from the current negligible levels to just 5% by 2050. And finally, a high growth rate, with timber frame and engineered timber reaching 80% and 20% respectively (i.e. combining to cover the market completely) by 2050.

The Structural timber Association **(STA, 2017)** has stated that in Scotland, timber frame had already reached 83% of the market for new dwellings by 2016, which suggests that the high growth scenario may be possible for the UK as a whole (for which the overall rate was 28%). At the time, the STA was forecasting a 67% increase in timber home-building by 2021 (to 88,000 dwellings per year).

Based on the above information and data, it is reasonable to assume that a pessimistic scenario for timber usage housing construction in the UK is that, despite fluctuations as the strength of the housebuilding industry and the wider economy varies, on average there will be no increase in the rate of construction, with the average annual figure sticking at around the Pre-Covid-19/Brexit level of around 200,000 homes of 70 $m^2$ on average. Furthermore, there is no guarantee of continued growth in market share for timber frame and engineered timber.

A more optimistic scenario is that construction rates rise rapidly to meet the Government's target of 300,000 homes per year, albeit a few years after the target date of 'the mid 2020s'. A scenario might be considered in which a 5% annual increase is maintained until the target level is reached. This very high market growth times high timber usage growth scenario might also be based on all domestic construction being in timber by 2050, with a linear progress towards that target in the preceding decades: with timber frame assumed for houses and engineered timber for flats. Note that apartment buildings currently account for approximately 25% of the current 200,000 dwellings per year market **(NHBC, 2019)**.

The information presented above allows an envelope for 'high timber' use in new housing construction to be explored. For instance, an upper limit – after a suitable transition period – might reasonably be based on 300,000 dwellings constructed per year, with a quarter of these (i.e. all apartments) constructed from CLT, and all others from timber frame, and with significant usage of timber on facades. A less extreme vision would be for a CLT/glulam system becoming the standard option for apartments by 2050 (25% of 200,000 dwellings per year), with any shortfall in CLT/glulam use being compensated by increased timber frame in the other 75% of dwellings. By this measure, an increase of 280 kt of timber per year can be expected over and above the 0.555 $hm^3$ per year (approximately 220 kt) used currently. This is consistent with 3% annual growth for 30 years.

The rate of non-domestic building construction is not subject to government targets, and the monitoring and reporting that goes with it. Furthermore, there is a great deal of variety in the form and function of non-domestic buildings (theatres, stadia, hospitals and shopping malls, for instance). Evaluation of the potential for the use of timber in each situation and in aggregate would require substantial primary research. A possible approach is to apply a multiplier to the material used in new house building to cover that used in new non-domestic construction and also in refurbishment. This might be based on construction activity statistics. For instance, Figure 3.1 shows a disaggregated time series for the annual construction value in the UK, to 2020 (with long-term growth interrupted from 2008 by the financial crash and then again from 2020 by the Covid-19 pandemic). Relative to the other categories, new housing construction has taken an increasing share of the market, reaching 33% in 2019 (non-domestic new construction also 33%, housing refurbishment at 20% and non-domestic refurbishment at 13%).

As stated above, in 2014, 0.555 $hm^3$ of sawn wood was used in new home construction out of 5.95 $hm^3$ overall in construction. This amounts to 18.8 $m^3$ per million pounds of construction value. By this metric, the rest of construction industry uses 65.5 $m^3$ of sawn wood per million pounds. These multipliers can be used as a basis for projections. Therefore, it can be conservatively estimated that new non-housing requires as much timber as new housing, meaning that between them, the two categories require a minimum of 1.11 $hm^3$ of sawn wood, which would have a mass of approximately 500 kt.

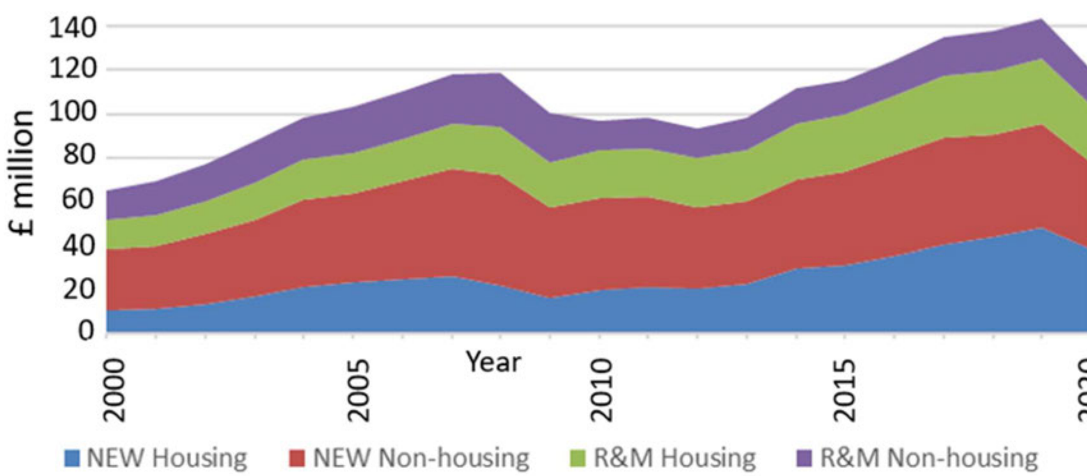


*Figure 3.1. Annual value of UK construction and refurbishment based on current prices* (**ONS, 2021**). *R&M is refurbishment and maintenance, and the values are in GBP £ millions.*

## 3.3 The Landfill Harvested Wood Product Pool

Whilst landfill sites are potentially long-term stores of biogenic carbon, they also permit – to an extent – the microbial, anaerobic decay of organic matter; and indeed it is possible to manage a landfill as a 'bioreactor' to accelerate the decay and the associated production, capture and utilisation of methane. In this work, the strategy of designing and managing landfills to limit biodegradation is of more interest, as the protection of the landfill carbon pool is more important than the substitution benefits associated with the landfill gas, so long-term storage in systems that minimise or eliminate degradation and landfill gas production is preferable. Even at landfill sites with landfill gas (LFG) management facilities, a significant proportion of LFG escapes into the atmosphere **(DEFRA & Golder Associates, 2014)**, which offsets the substitution benefits associated with LFG utilisation.

The quantity of carbon in the landfill HWP pool is governed by rate of addition to the landfill and the rate of degradation within the landfill. The latter can be difficult to calculate given the variation of contexts in which landfills exist (e.g. climates and management practices), and the dearth of studies looking at the progress of degradation in-situ over periods of decades **(Barlaz, 2006; Krause, 2018)**. **Wang et al. (2013)** is one of a small number of cases reporting on the decomposition of forest products in landfill, but only over a period of less than three years. Excavations of closed landfill sites by **Ximenes et al. (2008)** suggest that decomposition factors used to estimate LFG production, which are based on laboratory studies, significantly overestimate the decomposition of wood in landfill, at least in Australia, and together with the work of **Micales & Skog (1997)** suggests the possibility of negligible levels of degradation. Referring back to some of the aforementioned studies, the latest documentation from the **US Environmental Protection Agency (2019)** suggests that 88% of the carbon in 'dimensional lumber' is permanently stored when landfilled (thus a degradable organic carbon fraction – DOCf – of 0.12): this is approximately in line with the suggestion by **(Heyer et al., 2018)** to assume a DOCf of 0.10 instead of the IPCC default of 0.5. A wide range of rate constants can also be found in or derived from the literature, with values ranging from 0.01 to 0.12 $yr^{-1}$ **(DEFRA & Golder Associates, 2014; Heyer et al., 2018; US EPA, 2019)**.

## 3.4 Carbon Capture and Storage Pool

Another potential route to long-term storage of end-of-life HWP carbon is through bioenergy carbon capture and storage (BECCS) when wood waste is recovered and burned[32]. CCS and BECCS are usually given prominence in decarbonisation strategies through to 2050 and beyond, and yet there are still many unknowns about the technical and economic viability of CCS in general, and real concern about the land-use implications of BECCS in particular **(Fajardy et al., 2019)**. BECCS – to its proponents – is used as justification for scaling up the biofuels industry, rather than a method for mitigating the emissions associated with the inevitable disposal of bio-based products. That said, if and when CCS technology is implemented at scale, it can offer a means of securing biogenic carbon from waste wood almost indefinitely, and it is included in the modelling in this work.

The IEA Sustainable Development Scenario **(IEA, 2020b)** for reaching net zero $CO_2$ emissions from energy systems by 2070 imply CCS (actually CCUS) capacity of 800 $MtCO_2e$/yr by 2030, rising to 5800 in 2050 and 10400 Mt/yr in 2070 The numbers for the IEA's Net Zero by 2050 scenario suggest a more aggressive roll-out, achieving 1670 $MtCO_2$/yr by 2030 and 7600 $MtCO_2$/yr by 2050 **(IEA, 2021)**[33]. Results to date offer some hope, but equal grounds for caution as early progress in the industry did not transfer to accelerated project development. Indeed, the combined capacity of projects in operation, construction or supposed to be in advanced development was less in 2020 (~78 $MtCO_2e$/yr) [34] than it was in 2011 **(Global CCS Institute, 2020)**, although year-on-year progress appears to have returned. The capacity of plant in operation increased from ~20 to 38 $MtCO_2e$/yr in the same period, to a number equivalent to the emissions from at least 10 GW of CCGT generating plant, but this represents only 0.1% of the 33 Gt mentioned above.

The CCC suggests that for the UK to achieve its 2050 net zero ambition, 173 TWh of the 200 TWh of bioenergy resource available in the UK[35] in 2050 will need to be used with CCS **(CCC, 2019)**. From this, an expectation can be inferred that around 86% of wood combustion will be supported by CCS.

In addition to the market penetration of CCS, the effectiveness and efficiency of the process are also key parameters, as not all of the $CO_2$ in a stream is necessarily captured and stored, and the capture and storage process incurs an energy cost. Processes with

[32] Note that BECCS is a sub-set of CCS, which is itself a subset of CCUS (carbon capture utilisation and storage). Potentially capture and utilisation can lead to long-term storage when the utilisation is in the form of stable mineral products rather than for use in food and agriculture for instance. But for simplicity, the discussion here focusses on CCS or BECCS, as appropriate.
[33] Approximately 5% (2030) and 7.5% (2050) to be achieved from bioenergy power generation.
[34] Numbers estimated from graphs.
[35] The report assumes 83% of this supply will be indigenous.

higher capture rates, typically require more energy input: for instance processes using post-combustion oxy-fuel system, which avoid the need to separate $CO_2$ from the much greater volume of nitrogen in the flue gas of conventional combustion systems. In their assessment of the carbon intensity of electricity generation, **Pehl et al. (2017)**, assume CCS is 90% effective. This is at the centre of the 85-95% range suggested by the IPCC **(Metz et al., 2005)**, who have also suggested that capture and compression add some 10-40% to the overall energy cost.

### 3.4.1 Other Long-term Storage Options

As well as reuse and recycling, landfill, and CCS, at least one alternative method of achieving long-term storage of HWP carbon exists: namely the production of biochar from end-of-life timber. This can be thought of as a hybrid of carbon storage and energy recovery. The principle is that biomass (for instance, pelletised or shredded wood) is pyrolised, producing biochar alongside a blend of liquid and gaseous fuels (gases include CO, $H_2$, $CH_4$ and $C_2H_6$ and others). The fuels are burned with energy recovery, while the biochar is returned to the soil, or potentially used as a manufacturing raw material. **Crombie & Mašek (2015)** tested various samples in different conditions, and found that pyrolising wood pellet at relatively high temperature (650°C) yielded good results in that the yield of fuel is quite high, and whilst the total biochar yield is lower than when pyrolised at 350°C, the yield of stable biochar (capable of storing carbon for at least 100 years) is higher. Overall, about 28% of the carbon in the wood is retained in stable biochar, and the calorific value of the fuels is about 30% of that of the feedstock. Whilst at first sight this scenario might appear to be a poor substitute either to complete energy recovery or long-term storage of in-tact wood, there are additional benefits to soil and fertility, making the topic ripe for detailed LCA.

## 3.5 The Forest Carbon Pool

Inflows to and outflows from the forest carbon pool are considered in this section. Modelling scenarios in subsequent chapters will consider an enhanced role for UK commercial timber production in the provisioning of its construction industry, alongside the possibility of continuing reliance on trade. Accordingly, this section considers UK production and production in countries which supply to the UK.

Outflows from the forest carbon pool are also inflows to the HWP pool, which have already been discussed in relation to construction timber (section 3.2.3), but the topic is here addressed more broadly, considering the range of co-products and also the location of the forest.

### 3.5.1 Key parameters and conversion factors

The sources of data and information referred to in this section sometimes use different conventions for reporting timber quantities, including volume, mass, dry mass, mass of carbon, and whether – when applied to logs or standing timber – this is with or without bark (overbark – ob, or underbark – ub). Unless the conversion to the form of data needed for this work is provided by the data source, then the following assumptions apply.

- Carbon content. Wood is taken to be 50% carbon by dry mass **(BSI, 2014a)**.
- Moisture content (MC) is presented consistently on a dry basis, i.e. it is the mass of water in the wood divided by the oven-dry mass of the wood.
- Timber construction products are taken to be at 12% MC unless otherwise stated.
- The density of softwood varies, and for the purpose of modelling in this thesis the mean value is of most interest, in contrast to some structural design work where the extremes can be more important considerations. An EPD for kiln-dried sawn softwood (mix of species) from UK forests gives the density as 479 kg/m$^3$ at 15% MC **(Wood for Good, 2017)**. Wood for Good's LCA for CLT used (but not grown) in the UK quotes a density of 488 kg/m$^3$ at 12% MC **(Wood for Good & PE International, 2013)**: the higher density potentially being attributable to the mass of the binder as well as to the different origin of the wood. The Forest Research statistics suggest 550 kg/m$^3$ for sawn softwood at 25% MC **(Forest Research, 2020)**. The Gradewood project's assessment of 6000 samples from various European sources indicates average density for spruce of 423 kg/m$^3$ (at 12% MC, measured near the fracture location during testing) **(Ranta-Maunus et al., 2011)**.  The Trada database quotes figures in close alignment with this, namely 450 kg/m$^3$ for 'British spruce' (comprising a species mix of Norway spruce and Sitka spruce, and well representative of sawn wood used in the UK), at 15% MC **(Trada, n.d.).**
- Forest Research standard conversion factors predict that 1 m$^3$ of standing timber will produce 0.9 m$^3$ of felled timber overbark, equating to 0.818 green tonnes (so the density is 0.909 t/m$^3$) and 0.804 m$^3$ underbark **(Forest Research, 2020)**[36].
- Shrinkage: as wood is dried below the fibre saturation point (approximately 30% MC), it starts to shrink, progressively losing its chemically bound water following the evaporation of the free water. The loss of volume when green softwood such as sitka spruce, underbark, is dried to 12% MC is approximately 8% **(Ray, 2014)**.

[36] Section 11.2.11

- Wood raw material equivalent (WRME) is the ratio of the quantity of wood underbark required to produce a quantity of product. In the case of sawn softwood, Forest Research gives the WRME as 2.00, implying a recovery rate of 50%, whilst for 'chips, sawdust, etc.' the WRME is 1.20 **(Forest Research, 2020)**. This is a volumetric calculation and requires careful interpretation as the loss in volume comes in two forms: loss of material, and shrinkage of the remaining material, which means that in terms of dry mass (and therefore mass of carbon) the recovery rate may be better. Furthermore, in many of the statistics that might be used to corroborate the implied recovery rate of 50% clarity is lacking on a number of relevant factors such as the moisture content of the sawn wood and whether or not it has been planed. Depending on the assumptions made, it is possible to conclude that the 50% recovery rate might be either an overestimate or an underestimate.[37] It has been noted **(UNECE and FAO, 2010)** that different countries and regions adopt differing approaches, giving the example of freshly cut sawn softwood with a recovery rate of 0.64 (with the balance being sawdust and chips; if the recovery rate is measured after drying, then the recovery rate is 0.57; but for planed and trimmed timber ready for shipping to the customer, the recovery rate is 0.48. To calculate the recovery rate for planed and trimmed timber in terms of oven dry tonnes, shrinkage needs to be allowed for, resulting – in this example – in a recovery rate of 0.50. The same report also notes the relationship between recovery rate and stem diameter: thicker stems result in a lower loss rate around the circumference.
- Root-to-shoot ratio for living biomass (below ground biomass: above ground biomass) = 0.36 **(Forest Research, 2020)**. It follows that 73.5% of biomass is above ground.

### 3.5.2 UK Production & Trade

In terms of production, between 2010 and 2019, the UK has produced between 3.05 and 3.72 $hm^3$ of softwood sawn wood each year (average 3.469 $hm^3$); this is primarily from the 9.2 to 11.5 million green tonnes p.a. from 2010 to 2019 (equivalent to 11.2 to 14.2 $hm^3$ ob standing) **(Forest Research, 2020)**. The proportion of this destined for the construction market has ranged between 27% and 33% from 2015-2019 based on data from the larger sawmills (defined as those producing more than 25,000 $m^3$ p.a.) which cover 85% of the market **(Forest Research, 2020)**. If this data is also representative of smaller sawmills, then the total quantity of domestically produced sawn wood supplied to the construction

[37] A recovery rate of 56% can be derived from Chapter 2 of the Forest Research statistics, but this is before the wood is planed and trimmed.

sector has been 0.9 to 1.2 $hm^3$. The contrast between this range and the construction industry demand figure quoted above (~6 $hm^3$) clearly illustrates the construction sector's continuing reliance on imports, and the challenge in overturning stereotypes about UK-grown softwood being unsuitable for construction **(Ridley-Ellis, 2020)**. The sawn softwood imported to the UK in 2019 amounted to 6.4 $hm^3$ **(FAO, 2020)**.

Extrapolating from the data for larger sawmills, in addition to the 3.41 $hm^3$ of soft sawn wood produced in the UK in 2019, a total of 2.94 Mt of sawmill co-products were also produced, which is either used internally for energy (8% in 2019), or sold on as chips or sawdust to wood processing industries (53%), sold as bioenergy including to pellet manufacturers (20%), or sold to other users (19%, approximately half of which is bark). See Figure 3.2.

Data on UK panel production 2010-2019 **(Forest Research, 2020)** show that inputs of UK roundwood, UK sawmill co-products, and recycled fibre have been stable over the period (following a sharp rise in the latter category in the previous decade). Total inputs range from 3730 to 4150 thousand tonnes[38] p.a., with the average split being 33% UK roundwood, 43% sawmill products, 23% recycled fibre, and 1% imports. Negligible quantities of hardwood are used. Panel production has been in the range 3.00 to 3.38 $hm^3$over the same period: average 3.147 $hm^3$, 76% being particleboard (including OSB), with the remainder being fibreboard (including MDF). Additionally, in 2019, a similar quantity (3.7 $hm^3$) of panels were imported. There is insufficient information available to determine the proportion of this material destined for the construction sector **(Forest Research, 2020)**.

Other trends related to UK timber production include significant declines in inputs to integrated pulp and paper mills from the early 1990s as mills closed, but this has levelled off since around 2006. These inputs amounted to 525,000 green tonnes in 2019, which is 0.516 $hm^3$, 88% of this being from domestically produced roundwood with the remainder from sawmill products. This material, however, only accounts for a small fraction of UK paper and paperboard production (3.85 Mt in 2019), which is primarily from recycled fibres and from imported pulp **(Forest Research, 2020)**. However, more of the paper and paperboard used in the UK is manufactured overseas (5.02 Mt in 2019) than domestically **(FAO, 2020)**.

[38] Based on a given assumption that the inputs have 25% moisture content: clearly a simplification, as roundwood and sawn wood co-products may have more moisture than the recycled fibre, which has previously been dried (although there is no guarantee that it remains dry).

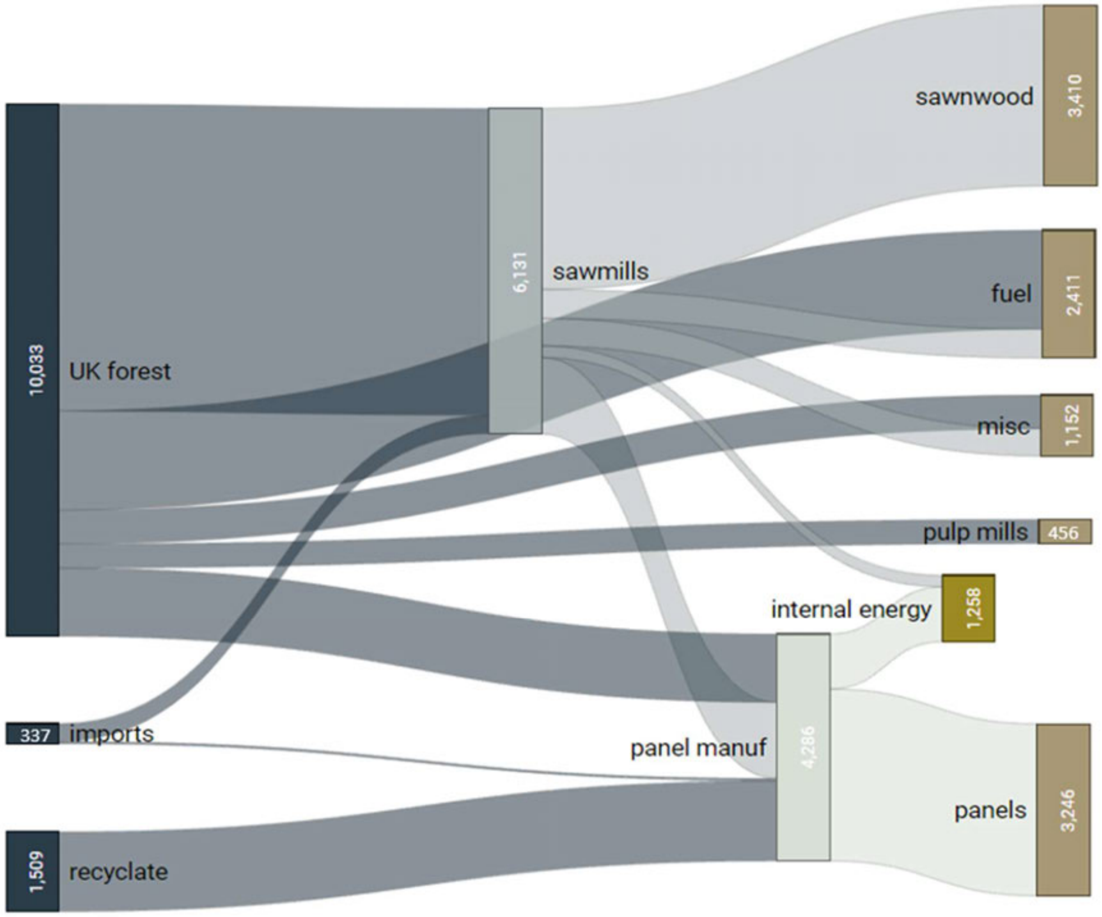


*Figure 3.2. Production of coniferous sawn wood and wood-based panels in the UK: a snapshot of material flows in 2019 based on Forestry Commission statistics. Units are thousand cubic meters under bark throughout. The imports shown are limited to those supplying sawmills and panel manufacturers: the much greater quantities of final product (e.g. sawn wood and paper) and raw material for pulp mills are excluded. The miscellaneous category ('misc') includes exports, round fence posts, and bark, amongst other things. Panels include all wood-based panels made predominantly from shreds, chips, and fibres, including OSB, particleboard and MDF, but not laminated timber products. The quantity used internally for energy is assumed to be the loss of material at the relevant processing site.*

Trends in other uses for UK softwood roundwood since 2010 have been dominated by the more than 100% increase in woodfuel production, which in 2019 accounted for 1.9 million green tonnes[39]. The use of post-consumer recovered wood for fuel has been rising sharply, from 0.55 Mt in 2010 (25% MC) to 2.39 Mt in 2019. Wood pellet production in 2019 was 298 kt, from 570 kt of delivered material (including moisture): over the five years to 2019, 57% of the deliveries to pellet manufacturers have been of roundwood, and the remainder has been sawmill residues **(Forest Research, 2020)**. UK wood pellet production is a small

[39] This is out of a total of 2.35 million green tonnes of softwood in the 'other' category. Additionally 0.7 M green tonnes of hardwood woodfuel.

fraction of trade, with imports amounting to 9.08 Mt in 2020 with an average annual growth rate of 6.7% maintained over five years, and more in earlier years **(FAO, 2020)**.

### *UK Softwood availability*

Overall production and consumption of wood, measured by wood raw material equivalent, underbark (WRME) have both been relatively stable for the ten years to 2019. The mean and standard deviation for wood production in that period are 10.6 and 0.48 $hm^3$ respectively. The apparent consumption of wood products (imports + production – exports) estimated through stock changes, has averaged 51.7 $hm^3$ with a standard deviation of 5.1 **(Forest Research, 2020)**.

The 10.6 $hm^3$/yr WRME produced in the UK on average requires forest stands with a stock of 13.2 $hm^3$ ob standing. Softwood availability forecasts illustrated in Figure 3.3 suggest sufficient slack in the system to at least satisfy current and recent levels of demand well into the 2030s, but a subsequent decline sees availability fall below current production levels in the early 2040s. This implies that there will be problems in meeting growth in demand from domestic forests in the medium term, suggesting a reliance on imports to support such growth. This conclusion is also supported by separate projections in **(Forest Research & UKCEH, 2020).**

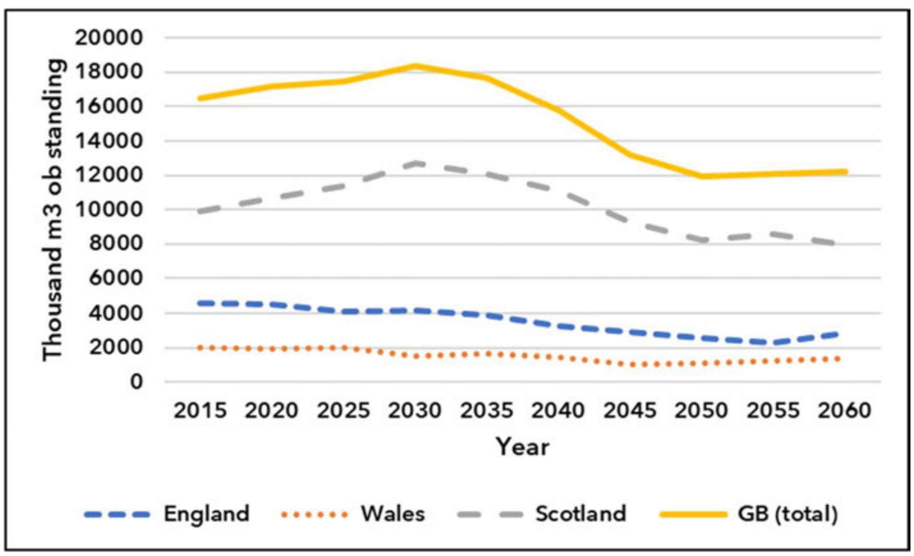


*Figure 3.3. Softwood availability forecast* **(Forestry Commission, 2014)** .

### *UK Land use, land use change, and forestry (LULUCF)*

#### LULUCF Carbon

Forest carbon stocks in the UK are currently on a long-term rising trend. Since 1990, forest carbon stocks (including soil) have increased by an annual average of 28 $MtCO_2e$, with 9.9 $MtCO_2e$ p.a. being above ground. Over this 30-year period, the above-ground stock has increased by 79%, whilst total forest area has increased by 17% **(Forest Research, 2020)**. These data imply a significant increase in the age profile and maturity of the national forest whilst supporting the conclusion that timber extraction is consistently more than offset by forest productivity.

NAEI projections **(Forest Research & UKCEH, 2020)** show the size of the forestry sink declining to a level of around 10 $MtCO_2e/yr$ by 2040 (all scenarios), with significant recovery thereafter in the enhanced scenarios (the 'stretch scenario' which requires 'an ambitious climate change mitigation programme exceeding current policy aspirations and existing public funding', including – importantly – more than 30 kha of afforestation per year by 2022). Projections for UK HWP show the existing annual sink evaporating in the 2050s. However, under the stretch scenario, an increase in timber production and the associated HWP sink is supported thereafter.

LULUCF – methane and nitrous oxide

| Transition | Gross change in area (kha) | $N_2O$ emissions (kt) | $CH_4$ emissions (kt) | Emission factor $tCO_2e/ha$ |
|---|---|---|---|---|
| **Cropland to forest land** | 1.1 | 0.0263 | - | 7.1 |
| **Grassland to forest land** | 10.8 | 0.2101 | - | 5.8 |
| **Settlement to forest land** | 1.6 | 0.0309 | - | 5.8 |
| **(Forest land total area 24439kha)** | - | 2.166 | 4.8 | 0.03 $tCO_2e\ ha^{-1}yr^{-1}$ |

*Table 3-5. Land use change, forestry, and associated non-$CO_2$ emissions in 2019* **(Ricardo Energy and Environment, 2021)**. *The land-use change emissions are direct $N_2O$ emissions from N mineralization / immobilization. The last line combines emissions related to drainage, rewetting and other aspects of forest soils management, plus emissions arising from fire.*

Data on non-$CO_2$ emissions related to UK forest land is shown in Table 3-5 **(Ricardo Energy and Environment, 2021)**. The first three rows show $N_2O$ related to land-use change, which are one-time emissions associated with conversion of land from some other use to forest land. The final row shows $CH_4$ and $N_2O$ emissions from the whole of the UK forest estate. Note that the emission factor is – in effect – a weighted average emissions factor for all forest types in the UK: the emission factor for forest on drained organic soils would be a much higher figure of 1.08 $tCO_2e\ ha^{-1}yr^{-1}$ (and 9.91 including the loss of soil carbon) according to **Evans et al. (2017)**. Compared to previous versions of the NAEI report for earlier years, the land-use change emissions per hectare are reduced, but the increase in land-use emissions per hectare is enough to result in significantly higher overall emissions (the total of the emissions in the table amounts to 0.85 $MtCO_2e$).

### Afforestation Targets in the UK

Afforestation in the UK is central to the increased construction timber scenario modelled and reported in Chapter 6. If a positive case exists for use of more construction timber, in terms of climate change, then afforestation to support this is required. This afforestation has two key purposes in this context: firstly, to promote an accelerated replacement of forest carbon removed at harvest, compared to just replanting the harvested areas; and secondly, to support the increased supply of timber required in future (from ~2060s onwards in any case: demand occurring before then would have to be met from existing trees). It is also necessary for this afforestation to be specifically linked to the strategy for harvesting more timber, and that the timber demanded is not taken from forests which exist for other reasons.

The England Trees Action Plan **(UK Government, 2021)** refers to the UK Government's commitment to increasing uptake of timber in construction. It also notes the wide disparity between the penetration of the house-building market by timber frame in England and Scotland, suggesting significant growth potential in the larger English market. Stated benefits of growing the market include 'locking away carbon long term and driving investment into tree planting'. In terms of tree planting, the target is 30,000 ha per year by the end of the current UK Parliament (2024): a step change from current rates. The potential gains from maintaining this rate of afforestation for 30 years has been modelled by **Forster et al. (2021)** who conclude that terrestrial carbon gains through commercial forestry and harvest would amount to around 450 MtC after 100 years (in 2120), with about 50 MtC of this achieved by 2050; in most scenarios, conservation forestry achieves less (from approximately 150 MtC in 2120). It is worth noting, however, that timber supply is only one of many motivations for afforestation, and that supporting biodiversity, landscape value, social benefits, urban greening, and water quality in rivers and streams are all aspects of the Action Plan. Not all of these values can be aligned easily with timber supply. Furthermore, the Action Plan also acknowledges that there is pressure to deforest some areas in the name of peatland restoration.

### 3.5.3 UK sawn wood imports

The UK is heavily reliant on imports for its supply of sawn softwood for the construction industry. For the years 2015-19, the average annual consumption of sawn softwood was 10.4 million $m^3$ (all uses), of which 7.0 million $m^3$ was imported (after a sharp increase in 2017).

The main sources for imported sawn wood softwood were as shown in Table 3-6, and – if UK forestry is unable to meet an increased demand from the construction industry – the implications of asking for more timber from these countries need consideration. However, the dynamic nature of the situation is illustrated by the fact that by 2019, Canada, for instance had dropped out of the list of top suppliers in 2013-17, and Russia was down to 6% **(Forest Research, 2020)**.

As well as sawn softwood, the following products were overwhelmingly imported from EU countries in 2019, particleboard, fibreboard, and paper and paperboard. On the other hand, the vast majority of UK imports of plywood and wood pellets in 2019 came from countries outside the EU **(Forest Research, 2020)**.

| Country | Volume 2013-17 ave [$m^3$] | Market share 2013-17 | Market share 2019 |
|---|---|---|---|
| **Sweden** | 1626800 | 27% | 42% |
| **Russian Federation** | 882000 | 15% | 5% |
| **Canada** | 851200 | 14% | 1% |
| **Latvia** | 749200 | 12% | 18% |
| **Finland** | 642200 | 11% | 14% |
| **Germany** | 445800 | 7% | 7% |
| **Ireland** | 213291 | 4% | 6% |
| **Others** | 626944 | 10% | 7% |
| **TOTAL** | 6037435 | 100% | 100% |

*Table 3-6. Main suppliers of coniferous sawn wood to the UK: market share 5-year averages 2013-17* **(FAO, 2020)***, and the market share in 2019* ***(Forest Research, 2020)***.[40] .

In order to understand the carbon implications of growing the UK's construction timber in the countries indicated in Table 3-6, an understanding of the context in each country is needed. This is approached here through a limited selection of metrics coupled with qualitative review. This understanding is needed in order to evaluate and model the potential impacts associated with a sharp increase in demand for timber in the UK, which may require increasing supply from the same countries. Taking relevant hypothetical scenarios to illustrate the point, vastly different GHG impacts would result from procuring increased supplies from forest industries with the following characteristics.

- Country 1. Extensive forests and sustainable management result in reliable annual increases in living biomass. A high conversion rate from harvested wood to sawn

[40] 2017 is the final year in the FAO trade flow series. The FAO figure for Canada is at odds with the figure from Forest Research (and for earlier years). The Timber Trade Federation agrees with Forest Research that Canada is not a major supplier of sawn soft wood to the UK **(Timber Trade Federation, 2020)**. Canada is, however, a significant supplier of plywood and wood pellets to the UK.

wood maximises the carbon storage potential of harvested timber. In this case, increases in demand do not automatically lead to reductions in living biomass: such demand can incentivise better management and further afforestation or reforestation, and – in any case – unmanaged forests eventually head towards an equilibrium in which carbon gains are offset by losses as trees die (whether 'naturally' or through more anthropogenic influences such as fire, pests and diseases, or climate change).

- Country 2. Increasing demand for timber from a variety of sources has already pushed the country's forests to the point – or even beyond it – where removals outweigh new growth. Whilst every additional hectare of forest harvested may be reforested with rapid turnaround, it takes decades to replace the lost carbon. In this case, the marginal cost of the sawn wood obtained from harvest of that additional hectare is (in somewhat simplified terms) the total carbon stored in the trees on that land. In the worst case, this loss of carbon from the land will be compounded by a poor conversion rate to sawn wood and durable products, with a high proportion of the crop marketed as chips, fuel and pulp.

As a warning that this is a genuine concern, a study looking at Finland noted that an increased harvest scenario would lead to a loss of carbon from the system as a whole (the forest and product pool combined), which would not be justified by any likely substitution benefits **(Seppälä et al., 2019)**. Another recent study suggests that any official metrics implying countries are safely in the sustainable category may be behind the curve, and instilling a false sense of security **(Ceccherini et al., 2020)**. Although the methods and conclusions are hotly disputed **(Palahí et al., 2021; Wernick et al., 2021)**, the Ceccherini analysis of satellite imagery suggests that the long-term increase in forest carbon in Scandinavia and the Baltics has already gone into reverse. Across 26 European countries, they found an average annual harvest increase of 69% in the years 2016-18 compared to 2011-15, with the prominent role of Finland and Sweden highlighted in particular, with the Baltic states only just behind. In response to the debate, some of the analysis was revisited using an adjusted methodology **(Ceccherini et al., 2021)**, with the scale of increased harvest revised down for Finland and Sweden, but this has not put an end to the debate **(Breidenbach et al., 2022)**.

### *Supplier Metrics*

A detailed review of the forest and timber industries in each of the UK's main supplier countries is beyond the scope of this work, but it may be instructive to present some metrics which illustrate some aspects of the industry, as shown in Table 3-7 presents roundwood conversion metrics together with one sustainability metric (fellings as a proportion of growth).

The sawlog production metric shows the conversion from coniferous roundwood to sawlogs and veneer logs: in each case, the remaining roundwood is mainly allocated to pulpwood, but with some also classified as wood fuel (especially Germany), or as 'other roundwood' (for sleepers, poles, piling etc.). The categorisation system used in FAO data is shown in Figure 3.4.

| | Canada | Finland | Germany | Ireland | Latvia | Russia | Sweden | UK | Mean |
|---|---|---|---|---|---|---|---|---|---|
| **Sawlog / veneer log production** | 95% | 46% | 66% | 59% | 68% | 70% | 54% | 62% | 65% |
| **Roundwood conversion** | 40% | 25% | 41% | 30% | 43% | 26% | 26% | 34% | 33% |
| **Fellings/NAI (2015)** | - | 80.4% ↑ | 76.5% ↓ | 64.5% ↑ | - | - | 93.9% ↑ | 62.9% ↑ | |

*Table 3-7. Sawn wood supplier metrics. In order, the rows represent the proportion of the harvest that is classified as sawlogs; the proportion of all roundwood that survives into sawn wood (industrial roundwood is net of trade); and the fellings to net annual increment ratio together with the trend (increasing ↑ or decreasing ↓ in comparison to previous period).Traffic light warning colours added for Fellings/NAI. Data from (FAO, 2020) and (Forest Europe, 2020)*

The roundwood conversion efficiency shows the percentage of the coniferous roundwood that is ultimately converted to sawn wood and veneers (although only a small fraction of the latter): this metric combines the performance of sawmills with the degree to which the industry is oriented to supplying the sawmills. By this metric, the UK's main suppliers fall into two clear groups, with Germany and Latvia for instance substantially higher than the 33% average, with Sweden, Finland and Russia substantially lower. Of these, Sweden and Finland allocate a high proportion (> 40%) of their harvest to pulpwood, while Russia's metric is impacted by its low conversion rate from sawlog to sawn wood.

Amongst these countries, there is no relationship between the roundwood conversion metric and the production of wood pellets in the country. Except in Latvia, where wood pellet production is equivalent to 20% of the total coniferous roundwood harvest, domestic production of wood pellets is a relatively minor concern, amounting to just 1%-4% of roundwood harvest. The fact that in Latvia the relatively high wood pellet production has no obvious impact on the conversion of roundwood to durable products may be linked to one of the following factors:

- Within limits, wood pellet production does not have to compete with sawn wood production, as sawdust – a by-product of sawmills – is a feedstock for wood pellets.
- Latvia also harvests significant quantities of hard woods, which can also be feedstock for the pellet industry.
- Countries with advanced pellet manufacturing facilities may import raw materials for processing from their neighbours.

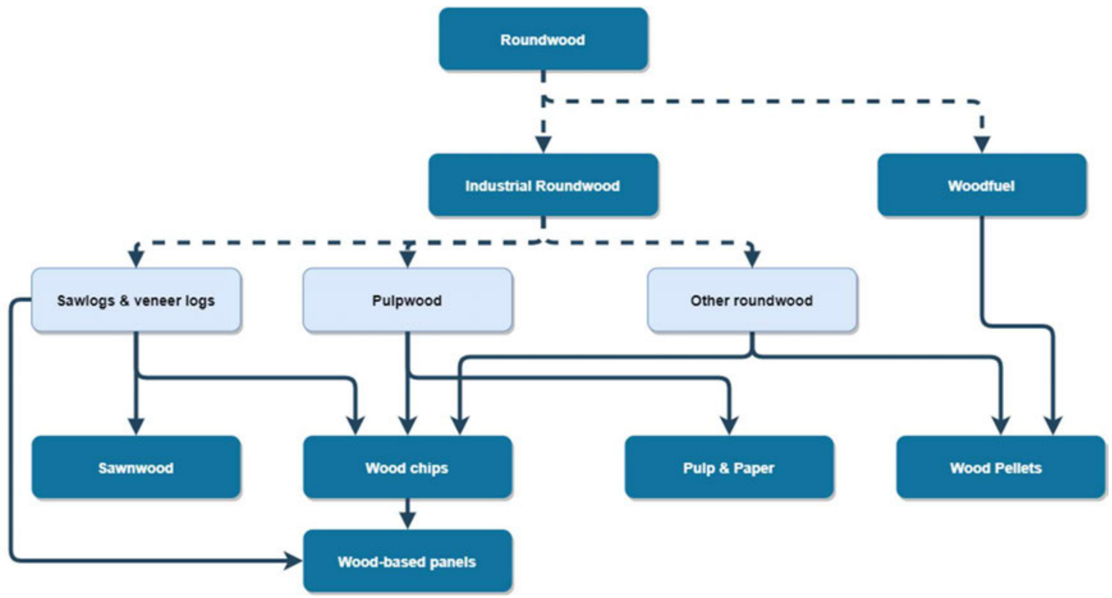


*Figure 3.4. Simplified Illustration of the FAO system* **(FAO, 1994)** *for categorisation of wood in production and trade, with focus on the categories most relevant to this work. Solid line connectors indicate physical transformations, whilst dotted lines indicate sub-categorisation. Dark text on light background (e.g. sawlogs) indicates that the category appears in production statistics only, the remaining categories appear in both production and trade. Pulp, paper, and pellets are reported in tonnes, whilst all quantities of all other materials listed here are reported in $m^3$ of timber without bark.*[41]

Fellings/NAI shows the fellings as a percentage of net annual increment (NAI), which is the net increase in standing biomass (over bark) after natural losses. The denominator is the quantity (again, over bark) felled, including any residues left in situ. Although published in the State of Europe's Forests 2020 report **(Forest Europe, 2020)** it is important to note that the data relates to the year 2015, since when the situation has altered, as noted by **Ceccherini et al. (2020)**. And as the indicator is trending upwards in four out of five countries reporting, this presents grounds for concern. The European Environment

[41] Each category has an extended definition. Some observations: wood fuel includes wood that will be used for fuel directly, or will be processed into fuel, but excludes those processed fuels (such as pellets); wood chips excludes wood chips made directly in the forest (which are included in the wood fuel category); coniferous and non-coniferous wood can be disaggregated at some of the higher levels (industrial roundwood, wood fuel, sawlogs and sawn wood), but not the lower levels (wood chips, pulp, pellets and panels); units are in volumes ($m^3$) in most cases, but tonnes for wood pellets, pulp and paper.

Agency has recommended a fellings/NAI ratio of 'approximately 70%' **(EEA, 2017)**. Arguably, any country with fellings/NAI of more than 70% and trending upwards, should be on amber alert; and any greater than 90%, or 80% and trending upwards should be on red alert, as illustrated in Table 3-7. The risk is that these countries reach, or have reached, a point when an increase in harvest results in a real-terms decrease in forest carbon (not just a reduction of the potential growth foregone when trees are harvested sustainably).

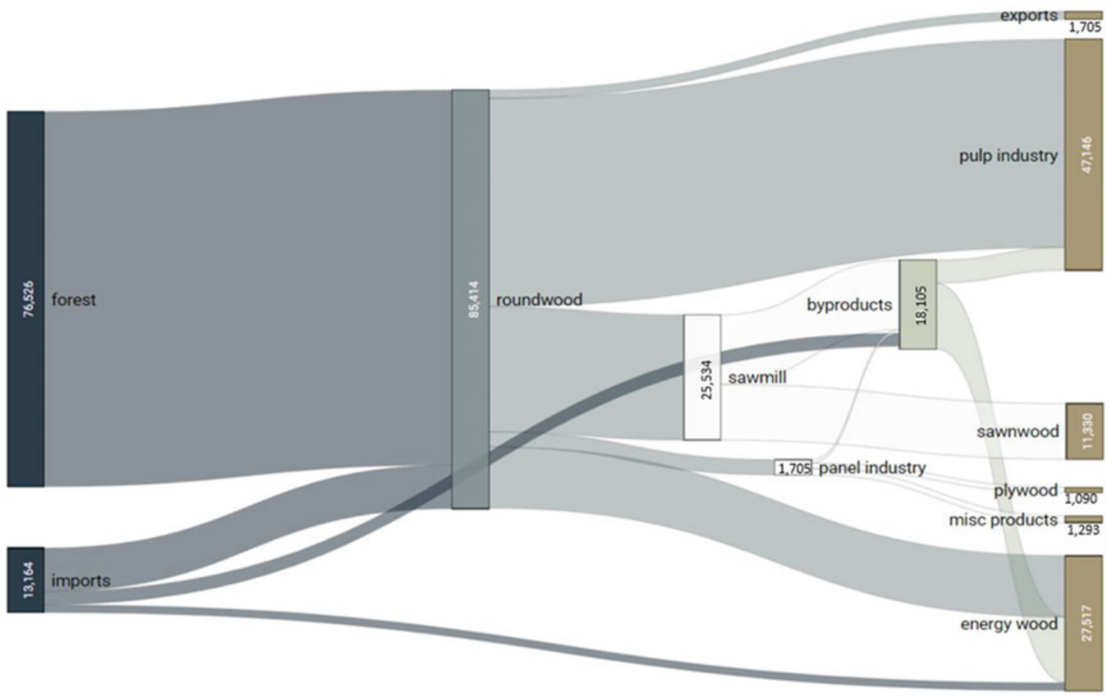


*Figure 3.5. Finland forest industry material flows, based on 2019 data from* **(LUKE Natural Resources Institute Finland, 2020)**. *Units are thousand cubic meters. In some cases net flows are shown, e.g. the net production of byproduct by the panel industry, and the net consumption of it by the pulp industry. Note that 26% of the roundwood is hardwood (mainly birch), but the quantity of this material processed at sawmills is almost negligible.*

As an example of one of the UK's supplier sawn wood suppliers, a material flow visualisation for Finland[42] is presented in Figure 3.5, which provides a marked contrast with the UK's own industry (Figure 3.2). For instance, because of the relative significance of the pulp industry, a relatively small proportion of the total harvest is embodied in sawn wood. But even so, Finland produces more than three times as much sawn wood as the UK (and about forty times as much on a *per capita* basis), which is why Finland supplies countries such as the UK alongside meeting its own needs.

[42] Analysis of FAO data for Sweden implies that its forest industry has similar characteristics.

## 3.6 The Energy and Material Substitution Pools

### 3.6.1 Approach

As discussed in Chapter 2, many researchers tracking carbon exchanges between the atmosphere, the forest and the anthroposphere conflate physical and virtual carbon pools, and routinely overstate the latter. These pools consist of the theoretically measurable and quantifiable carbon stored in the forest and wood product pools, and the carbon 'stored' in the virtual pool of GHG emission savings associated with the use of wood in preference to non-biogenic construction materials. This can be a convenient method of demonstrating that if the gains in the physical pools are small or non-existent, the ever-increasing substitution pool guarantees that the system overall has significant value in terms of climate mitigation.

One option for this thesis would be to argue that as substitution benefits are routinely overstated, and theoretically dubious, it would be better to focus solely on physical carbon. Such a choice, however, would miss the opportunity to influence how others continue to approach the topic, and furthermore the work would – in a sense – lack completeness as it would ignore those substitution benefits that very probably do exist. A better approach is to bring substitution benefits into the analysis, but with a sense of reality to their quantification and projection, and provide clarity about when physical or virtual results are being reported separately and together.

The research follows this sequence:

1. Define a baseline displacement factor ($D_f$) range for this study, based on published LCA of timber and non-timber construction options.
2. Make adjustments to the application of $D_f$. These adjustments are to compensate for effects such as wood products sometimes substituting for themselves (where a wood product is the preferred or default product for a given situation it can hardly be claimed to be substituting another product), and the future decarbonization of the industries and economies in which products are manufactured.
3. Then contrast with results from applying the crude approach of projecting substitution benefits based on an 'off-the-shelf' $D_f$ and an assumption that this will apply indefinitely to all future use of wood in construction. Note that this is emphatically not to endorse this approach, but to illustrate the significance of the adjusted approach.

### 3.6.2 The Displacement Factor

Displacement factors are discussed in Chapter 2, where an indicative range of $D_f$ values is presented, obtained from LCA studies. Corrections are applied to this range, as detailed below.

#### *Change in carbon footprints of products over time*

Progress in a number of aspects of industry and the economy can be expected to reduce the carbon footprints of both biogenic and non-biogenic construction products. Improvements may include innovation in manufacturing systems, material switching (e.g. cement substitution, or new types of concrete, plasterboard, or alloys), and a decoupling of energy systems from GHG emissions. This subject is discussed in detail in section 3.6.3.

#### *Phantom substitution and self-substitution*

As stated in section 3.2.3, in 2014, timber frame had 25% of the UK residential construction market (and the figure for Scotland is over 80%, which hints at the potential for further growth in market share across the UK). If a large sample of randomly chosen new UK residential building projects are required to use timber systems instead of masonry, concrete or steel, then we can be confident that about 25% of these projects would have used timber in any case, and that no substitution occurs in that set: if this substitution is counted (as it often is), then it is 'phantom substitution'. The substitution benefit per project should be the combined benefit of any additional substitution that occurs in the remaining 75% projects averaged across the full sample. Furthermore, as long as timber usage is seen as a valid climate mitigation tool, government policy and market drivers are likely to continue developing, shifting preferences and expectations with them. As such, the BAU expectation must be recalibrated at every shift in policy. If the dial shifts on preferences to such an extent that timber use can be expected by default, then no substitution takes place when timber is used, although any non-use of timber would then erode the substitution pool. This effect is modelled by introducing a preference change factor (an annual percentage) that further adjusts future $D_f$ as preferences change. It is worth adding, here, that it is already the case that the global community should be making all reasonable efforts to eliminate GHG emissions, and arguably no credit can be claimed for going about our business in a way that is somewhat less polluting (per unit of output) than the path we might otherwise have proceeded along. In this view, the substitution pool is not only a 'virtual' pool; it is an entirely fictitious one that encourages a delusional view of the benefits of timber usage.

One other consideration not generally accounted for is the double-counting that might occur when one timber building is removed and replaced by another (here referred to as self-substitution). This only needs to be considered when the expected lifetime of a timber building is less than that of its non-timber counterparts; in that case, some complex questions arise, as even the concept of one building replacing another is highly loaded (does the building have to be on the same footprint, site, or in the same region as the building removed, or belong to the same entity? Must it have the same dimensions and function?). For the purpose of this work, it is assumed that new timber buildings have the same life expectations as non-timber counterparts.

The approach taken to excluding self-substitution effects is illustrated here by example. In year zero, assume a baseline displacement factor ($Df_0$) for timber housing systems compared to conventional systems, and that the fraction of new housing produced using timber systems is $x_0$. The carbon intensity of the economy is $E_o$

In year n, under BAU, the fraction of new housing produced using timber systems may have changed to $x_n$ without intervention through existing market drivers, but a new set of wood market development policies increases this to $x'_n$. The carbon intensity of the economy has reduced to $E_n$.

The effective displacement factor ($Df_{eff,\,n}$), if applied to all timber housing construction in year n, is:

$$Df_{eff,n} = Df_0 \times (E_n/E_0) \times (x'_n - x_n)/x'_n \qquad \text{Eq 3—3}$$

If, for the purpose of illustration, the baseline $D_f$ = 1.0, the economy decarbonises by 20% between year zero and year 8, and the proportion of timber housing increases from 25% to 35% in the same period, as opposed to an increase to 30% anticipated under BAU:

$$Df_{eff,8} = 1 \times 0.8 \times (0.35 - 0.3)/0.35 = 0.11$$

It is interesting to note that in this scenario (which is well within the range of possibilities over the next decade) the $Df_{eff}$ is a tenth of the baseline $D_f$, which is itself around half the level of the 'off-the-shelf' displacement factors generally used in the literature, discussed in section 2.9.

Despite this, $Df_{eff}$ may still over-state the power of substitution. For instance, Eq 3—3 does not take account of 'carbon leakage', as discussed by **Harmon (2019)**. Carbon leakage, in essence, is the fraction of the substitution benefit that will not be realised, because market forces will push another actor into burning some of the extra fossil fuel freed up by a decision by the initial actor to use less. In an ideal policy environment, carbon leakage would be zero, because targeted GHG emission caps would prevent the transfer of emission reduction budgets between actors. Under a cap-and-trade system however (or, of course, in an unregulated market limited only by the accessibility of fossil fuel reserves), carbon leakage could in theory approach 100%, as emission savings in one sector are bought by another, giving it a licence to increase GHG emissions. **Harmon (2019)** tested scenarios involving leakage rates varying between zero and 12% per year, but without arguing where on this spectrum the true value might lie as in practice, carbon leakage is very difficult even to estimate.

An alternative to forecasting $x_n$ and $x'_n$ for Eq 3—3 is to replace these terms with a discount factor *Pc* ($0 \leq Pc \leq 1$) to capture preference change *and* other issues such as carbon leakage, and the sensitivity of results to the value of *Pc* tested in scenario analysis (Eq 3—4).

$$Df_{eff,n} = Df_0 \times (E_n/E_0) \times (1-Pc)^n \qquad \text{Eq 3—4}$$

Therefore the modelling in this thesis takes three fundamentally different standpoints.

1. The heavily moderated view of substitution discussed above.
2. The more cautious view that the substitution pool should not be counted at all because it provides a false and unsustainable justification to continue along a path of increasing consumption.
3. The outrageous position taken by some that the substitution pool carries on growing, with constant $D_f$, without limits and with no negative feedback. This position is taken for illustration only.

Neglecting substitution altogether may be the best option in some cases, but a strong case can be made for including it in others. For instance, at the project level when a construction project is destined to go ahead come what may, there is likely to be a climate advantage for opting for timber rather than alternatives with higher embodied carbon. However, there is no justification for aggregating this advantage for all such projects now and in the future and claiming it as a great and permanent ocean of avoided GHG emissions.

### 3.6.3 Industry Decarbonisation Scenarios

In order to evaluate the future substitution benefits associated with increased use of timber, future energy supply and demand scenarios must be considered.

When end-of-life wood products are used for energy supply (whether directly through incineration, or indirectly through landfill gas), the GHG emissions of the displaced supply must be considered. This is likely to include electricity and potentially heat (in the case of incineration), and the displacement of fossil fuel methane (as long as it remains a significant part of the energy mix) in the case of landfill gas.

In the case of combustion with electricity generation, the grid mix at the time of combustion is the main consideration; a choice also has to be made about whether to assume CCS is used in generating processes involving the combustion of hydrocarbons (whether fossil or biogenic), including the combustion of the wood. By contrast, in the case of landfill gas combustion, the assumption is that the gas simply displaces fossil methane in whatever gas-burning infrastructure is in use: as such, the presence or absence of CCS does not need to be considered, as conditions are identical for the displacing and displaced fuel.

The evaluation of future material substitution benefits also involves the consideration of the rate of decarbonisation of the economy or – ideally – the rates of decarbonisation of different sectors of the economy (e.g. forest products, steel, concrete). Information, research, and projections related to these topics are discussed below.

### *Electricity*

Dramatic strides have already been with the decarbonisation of the UK power network, with – for instance – the carbon intensity of electricity consumed reducing from 0.49 to 0.24 kg$CO_2$e/kWh in the ten years to 2019 **(DBEIS, 2021)**. This process is set to continue, with the projections for a reduction from ~170 g$CO_2$e/kWh to ~40 g$CO_2$e/kWh in 2035 **(DBEIS, 2019)**. This is as a result of an increase in low carbon generating capacity and supporting infrastructure: the reference scenario for 2035 includes 12 GW of nuclear capacity[43], 42 GW of renewables, 15 GW of interconnectors and 8 GW of storage.

Projections beyond 2035 of course come with even greater uncertainty. In its presentation on reaching net zero in 2050 **(CCC, 2019)**, the Committee on Climate Change notes what firm low carbon generating capacity will have to displace fossil fuel generation and – at the same time – double the overall generating capacity to meet increased demand from heating and transport. Only about 5% of the capacity will be gas, and that will be decarbonised in some way (e.g. CCS). CCS is a 'necessity not an option' (reaching 75-175 Mt$CO_2$/yr: mostly

[43] This now looks unlikely. All bar one (Sizewell B: ~1.2GWe) of the UK's existing fleet of nuclear power stations are slated for closure by 2030, and currently a maximum of two new large stations are likely to come on stream by 2035. These are Hinkley Point C, which is under construction, and Sizewell C, which has not yet achieved planning consent. Both developments comprise a pair of European Pressurized Reactors rated at ~1.6 GWe per reactor.

from BECCS). Any remaining $CO_2$ emissions will be limited to minor losses from CCS projects (~5g$CO_2$/kWh overall, under the 'further ambition' scenario, and ~10g$CO_2$e/kWh under the core scenario).

### *Industry low carbon transitions – steel and cement industries*

Cement and steel production both illustrate the challenges of decarbonisation beyond the power sector. Both currently dependent on high temperature processes with relatively small electrical contributions, with the exception of electric arc furnaces used for recycling scrap steel.

By its very nature, the rate of innovation and its resulting impacts are hard to predict. Because of this, and the highly incomplete information available, predicting the relative rates of change between the carbon footprints of different construction materials over time is likely to be biased by factors such as differing levels of industry hype, interest and economic value in them. As such, a reasonable approach would be to assume and apply an across-the-board decrease in the carbon intensity of relevant industry sectors over a defined time period. Application of this approach will lead to $D_f$ that also decreases over time.

For the purpose of forecasting the change in displacement factor associated with the use of timber instead of other construction materials, the forecast decarbonisation rates of the European steel and cement industries are used as a proxy, and an average is taken. **Habert et al. (2020)** found that a halving of the carbon intensity of concrete by 2050 can be achieved by implementing marginal improvements throughout the value chain, with more speculative technologies and investments (including CCS) required to go much beyond that point. The European Climate Foundation's report on pathways to a net zero by 2050 **(Pestiaux et al., 2018)**, and associated online tool[44] offers scenarios for reaching net zero by 2050, along with the situation at decadal intervals until then. The 'shared effort' scenario (combining demand reduction and technological solutions) in that report is used in this work, along with the EUREF-16 scenario to represent BAU. It is worth stating that 'net zero' in this context does not imply leaving fossil fuels entirely unextracted, nor does it imply that emissions from large sectors of the economy are net zero. In reality, the European net zero scenarios still involve considerable emissions from most sectors of the economy, especially industry, agriculture and transport. But this is offset by increased negative emissions from LULUCF, with the shared effort scenario requiring the release of land from agriculture for instance (through efficiency and demand reduction), with 76% of

[44] ***https://stakeholder.netzero2050.eu/***

this land afforested and 20% for grassland. And with an assumed reduction in forest harvesting intensity of 25%[45], either through set-aside or across-the-board intensity reduction, this justifies the claim that the LULUCF sink might offset emissions from the rest of the economy. There are grounds for questioning this. For instance, climate change may reduce the resilience of future forests, and the ongoing sequestration and storage of carbon by them. And the material and fuel switching implied by the decarbonising industry scenarios might directly or indirectly lead to increased extraction from the biosphere, thereby undermining the requirement to *reduce* harvesting intensity. Leaving these concerns to one side, pathways for the production of European steel cement are shown in Table 3-8.

| | 2010 | 2020 | 2030 | 2040 | 2050 |
|---|---|---|---|---|---|
| **Steel – EUREF16** | 1.18 | 1.21 | 1.19 | 1.03 | 0.89 |
| **Steel – shared effort** | 1.18 | 1.21 | 1.17 | 0.88 | 0.63 |
| **Cement – EUREF16** | 0.76 | 0.75 | 0.71 | 0.48 | 0.33 |
| **Cement – shared effort** | 0.76 | 0.75 | 0.65 | 0.26 | 0.12 |
| **Combined change (from 2020) – EUREF16** | - | 0 | -3% | -25% | -41% |
| **Combined change (from 2020) – shared effort** | - | 0 | -8% | -46% | -66% |

*Table 3-8. GHG emission intensity from European production of steel and cement: kg$CO_2$e/kg of material produced. The combined change is the average reduction in emissions intensity. From* ***(Pestiaux et al., 2018).***

For steel, the shared effort scenario requires a substantial (67%, 2020-50)[46] reduction in production and demand, as steel is exchanged for carbon fibre for instance, coupled with a shift towards electrification, increased use of recyclate, and the gradual introduction of hydrogen and biomass with CCS into new plant. Likewise, the shared effort scenario for cement requires a large production reduction (70%, 2020-50), coupled with a similar array of technology and process shifts, with CCS for the emissions from calcination being particularly important, and enabling a steeper reduction in emissions intensity than for steel.

The IEA/WBCSD roadmap for the cement industry **(IEA, 2018)** sees a BAU global increase in emissions of 4% between 2014 and 2050, for a growth in demand of 12%, which represents a drop in the embodied carbon of a unit of cement of 7%. A more ambitious suite of policy and technology solutions – including CCS (especially), fuel switching, clinker percentage reductions and energy efficiency – will result in a 24% decrease in $CO_2$ emissions, representing a 32% reduction in carbon intensity over the same period (4% by 2030 and 15% by 2040). The report also hints (with little detail offered) at a

[45] 2050 compared to 2015.
[46] Note that the reductions in production are actually greater than the reductions in emissions intensity.

more ambitious possibility, even more heavily reliant on CCS, which reduces emissions by ~60% (so a 64% reduction in carbon intensity). **Millward-Hopkins et al. (2018)** have projected an absolute change in emissions related to UK concrete in the range of -40% to +15% for 2017-2050, with the largest decline benefitting from substitution effects in a 'high timber' scenario: this is against a backdrop of rising demand for cementitious materials overall, and reduced availability of pulverized fuel ash and ground-granulated blast-furnace slag. The scenarios tested eschew the use of CCS and novel cements because of lack of evidence that these will be ready to play a significant role by 2050.

The UK Government's iron and steel roadmap **(DECC & DBEIS, 2015)** shows that – in terms of the efficiency of UK steel production – the easy gains have already been taken, with the specific energy consumption per unit of steel produced declining steeply from the 1970s to the mid-1990s, but then settling at a level of around 19 GJ/t, although the practical limit is thought to be 10.4 GJ/t of crude steel with a world best (so far) of 14.8 GJ/t. Coal and coke are still the main fuels used: natural gas is an alternative reductant, but is more expensive. The roadmap identifies pathways involving advanced technologies (including CCS and rebuild), retrofit with CCS, and other somewhat less significant interventions such as stove flue gas recycling. Under BAU conditions, emissions will decline by ~14% in gross terms by 2050 from the 2014 level; the 'max tech' scenario offers the possibility of a 60% reduction (gross), and intermediate pathways suggest 28% and 46% reductions. The European Steel Association **(EUROFER, 2019)** suggests a wider range of possibilities, although the 4% reduction in carbon intensity of European steel by 2050 from a 2015 baseline assumes no progress in the energy supply chain; scenarios that take such progress into account suggest a carbon intensity reduction of 71%, or 77% if various innovations are followed through by the steel sector. These innovations might include carbon capture and utilisation, and hydrogen or electricity-based metallurgy, instead of carbon, for the iron-ore reduction stage.

### *Industry low carbon transitions – timber*

The consideration of opportunities to decarbonise the timber industry (excluding consideration of biogenic carbon at this point) depends on the availability of data on how and where energy is demanded in the supply chain. For instance, diesel in the forest and on the roads, and electricity and heating fuels in production plants. A minority of EPDs have some relevant data: for instance, one EPD for CLT **(Studiengemeinschaft Holzleimbau e.V., 2017)** indicates that 49% of A1-A3 emissions are associated with electricity. Whilst a detailed breakdown of the remaining 51% is not given, there is clear potential for the electrical emissions to be nearly eliminated by 2050, making a 50% reduction in carbon intensity within reach, possibly even representing a BAU scenario.

### *Low carbon transitions - gas and liquid fuel networks*

Eliminating greenhouse gas emissions from global use of gaseous and liquid fossil fuels is arguably a more challenging problem than doing so from electricity networks. Whilst much of the western world is learning to survive with a tiny fraction of its previous coal use (partly *because* of a shift to gas), the same cannot yet be said for natural gas and transportation fuels for instance.

There are, however, options for solving the problem. For the gas network, the simplest option is substitution of natural gas with biomethane produced, for instance, in anaerobic digestion facilities. This is an incremental approach, requiring no significant modification of the gas network, and is already being used albeit at low percentages (< 1% in 2017 in the UK **(POST, 2017)**). A more radical option is to substitute natural gas with 'green' hydrogen[47], a concept which is being tested at pilot scales. The thinking is that moderate percentages of gas (~20% by volume, but only 6-7% by calorific value) can be substituted by hydrogen without major investments; beyond that (potentially up to 100%), any ageing pipelines still remaining in the ground will need to be replaced with modern plastic systems, and gas burners (in domestic boilers, for instance), will need attention or replacement. An alternative which would avoid these investments is to synthesise methane from green hydrogen and carbon dioxide, but this adds to the energy losses incurred in the production of the hydrogen, thereby undermining overall efficiency. Industry and academic bodies have plotted speculative pathways to decarbonisation by 2050, including **Gas for Climate Consortium (2020)** which suggests moderate progress to 2030 (primarily through demand reduction and biomethane), followed by steep reductions as buildings further reduce demand and other sectors switch to hydrogen (whether from gas or liquid fuels). This is represented in the model (NZ2050 scenario) by a linear decline in the emissions factor of all fuels from 2030, down to a nominal value representing supply chain emissions.

For the transport network, electrification appears to be the best hope, although battery weight creates constraints, especially for aviation and haulage. Green hydrogen has potential, either used directly or as a precursor to synthetic fuels. Biofuels should not be seen as an option except at the margins: existing policy on biofuels for transport in the EU,

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[47] Hydrogen produced by using renewable power to electrolyse water. Such hydrogen still represents a very small percentage of the market, with only 4% of global production produced through electrolysis, and not all of that 'green' (**(POST, 2017)**). Note, the other option is 'blue' hydrogen, which requires the development of CCS to capture the carbon from the fossil fuels used in – for instance – steam reformation of methane.

US and Brazil for instance may already have caused enough environmental problems **(Oliveira et al., 2017)**.

#### *Decarbonisation scenarios used in model*

The increasing inter-connectedness of different parts of the energy challenge (power, fuels, industry, transport, hydrogen, storage, demand reduction and response for instance) makes it harder than ever to analyse any one element in isolation. As such, a relatively uncomplicated scenario is justifiable, and for the decarbonisation pathway used to calculate substitution benefits in the BioCarp model the following assumptions are made.

- The power sector continues its rapid progress, with emissions intensity falling linearly to zero, from 2020.
- Little or no progress outside of the power sector until 2030.
- Steel and cement industries decarbonise to 2050 according to the schedule outlined by the European Climate Foundation (shared effort scenario) **(Pestiaux et al., 2018)**. Emissions intensity then declines linearly to zero emissions in 2070.
- Emissions intensity from any fossil fuel consumption outside of the power sector decline linearly to zero from 2030 to 2070.
- Biomass combustion emissions to be mitigated by CCS starting from effectively zero in 2030 to 100% in 2070.

The sensitivity of the model to these assumptions is tested by running the model with no progress made on decarbonisation.

## 3.7 Summary

This chapter has considered the data and information required to develop realistic scenarios and to build and populate the model presented in Chapter 4. Issues covered include the potential increase in the usage of timber in UK construction; the residence time of biogenic carbon in construction products; waste management options for timber products and the implications for carbon storage, and potential changes in future; substitution benefits associated with using timber instead of other construction materials; and aspects of the UK forest industry and trade which have the potential to impact the forest carbon pool.

# 4 Biogenic Carbon Pools Model and Climate Impact Assessment

*A full description of BioCarp – the model developed to evaluate the carbon pools and climate impact of the chosen scenarios, including equations and ranges for the variables used in the model.*

## 4.1 Introduction

The purpose of the modelling described in this chapter is to explore how changes in how end-of-life timber is managed and how changes in construction timber demand impact carbon storage in HWP and on land, and – ultimately – on atmospheric GHG concentrations and radiative forcing effect.

The first part of the analysis – carbon storage – is achieved with a model (The Biogenic Carbon Pools Model - BioCarp) developed for this thesis. This tracks carbon through the various storage 'pools' identified below. Throughout, each pool relates to the quantity of carbon stored in a given future year (year t) that can be associated with a change from BAU. Thus, all pools are set to zero when t=0. The model employs a stochastic system to allow for uncertainty associated with the input parameters.

The second part of the analysis (determining how HWP use drives GHG concentrations and radiative forcing) is much more than a case of changing the sign in front of the net terrestrial carbon storage effect, for the following three reasons. Firstly, a focus entirely on carbon storage neglects the influence of non-$CO_2$ GHGs, with $CH_4$ and $N_2O$ being of particular interest in this work. Secondly, increasing use of HWP can arguably drive down global GHG emissions through timber's role in material and energy substitution. And thirdly, consideration of temporal aspects of carbon storage and emission are complicated by the fact that each GHG has its own finite residence time in the atmosphere. As a consequence of this, for example, a pair of scenarios reaching the same point in terms of carbon storage may have different cumulative radiative forcing effects if they have reached that point by different pathways.

Quantification of non-$CO_2$ GHGs and material and energy substitution are included in BioCarp. Temporal effects are analysed by importing results from BioCarp into the Temporal Climate Impacts model developed at the University of Bath **(Cooper, 2020)**. This translates an inventory of annual net fluxes of each GHG to the atmosphere into the additional (or reduced) quantity of each gas in the atmosphere in every year from the start of the model. This takes account of the natural decay of atmospheric GHGs as they are chemically altered or taken up by receptors on land (e.g. oceans and forests). This then allows calculation of the radiative forcing effect in each year, and the ultimate impact on average global temperature in each year.

The most established tool for analysing carbon stocks in the forest value chain is CARBINE **(Forest Research, n.d.-b)**, which has significant differences compared to BioCarp. CARBINE is populated with growth data for a number of representative tree species, yield classes and management regimes, and is therefore able to make estimates of forest carbon stocks at the forest stand level and assess wider areas by combining stand-level assessments. This option is not available in BioCarp, which is specifically developed for a national assessment, allowing only a high-level approach to modelling forest carbon, with a growth function intended to represent the average of all commercial coniferous forestry. Similarly, CARBINE offers different assorting options which allocate harvested timber to different pathways according to tree species, based on expert opinion. Again, BioCarp opts for a single approach to represent the national situation.

Key areas where BioCarp has additional or distinctive capability are:

- The inclusion of end-of-life modelling beyond the instantaneous oxidation assumption offered by CARBINE. BioCarp includes the carbon dynamics of landfill, and carbon capture and storage (with efficiency and energy cost accounted for).
- BioCarp includes a dynamic element, in which variables can change over time. These dynamic variables include emission factors, displacement factors, and the split between end-of-life options. Potentially this approach could be extended to further variables such as HWP decay rates. The dynamic approach does not yet extend to forestry, as is also the case with CARBINE, which assumes future growth will be the same as historic growth rates (all else being equal).
- With respect to uncertainty, BioCarp assigns simple probability density functions to variables in the model which are randomly and repetitively sampled, meaning results illustrate a realistic range rather than single values.
- Both BioCarp and CARBINE model displacement effects, but in BioCarp this modelling is based on more up-to-date analysis, and the inclusion of a dynamic element is an important part of its capability.
- BioCarp takes a demand-driven rather than forest-centred perspective, whereby additional demand for HWP is modelled, and the downstream effects (HWP and end-of-life carbon) and upstream effects (impact on forest carbon and need for further forestry) follow from that.

- Finally, BioCarp has been configured to produce results in a format that can be directly entered into a dynamic climate change impact calculator to determine the radiative forcing and temperature change effects.

## 4.2 BioCarp

The biogenic carbon pools model (BioCarp) calculates the annual flow into and out of each pool in every year, through an iterative process starting in year t=1, with flows in year t+1 driven (in most cases) by the size of the pool in year t.

Figure 4.1 gives a general overview of BioCarp, with specific elements illustrated in more detail in subsequent figures.

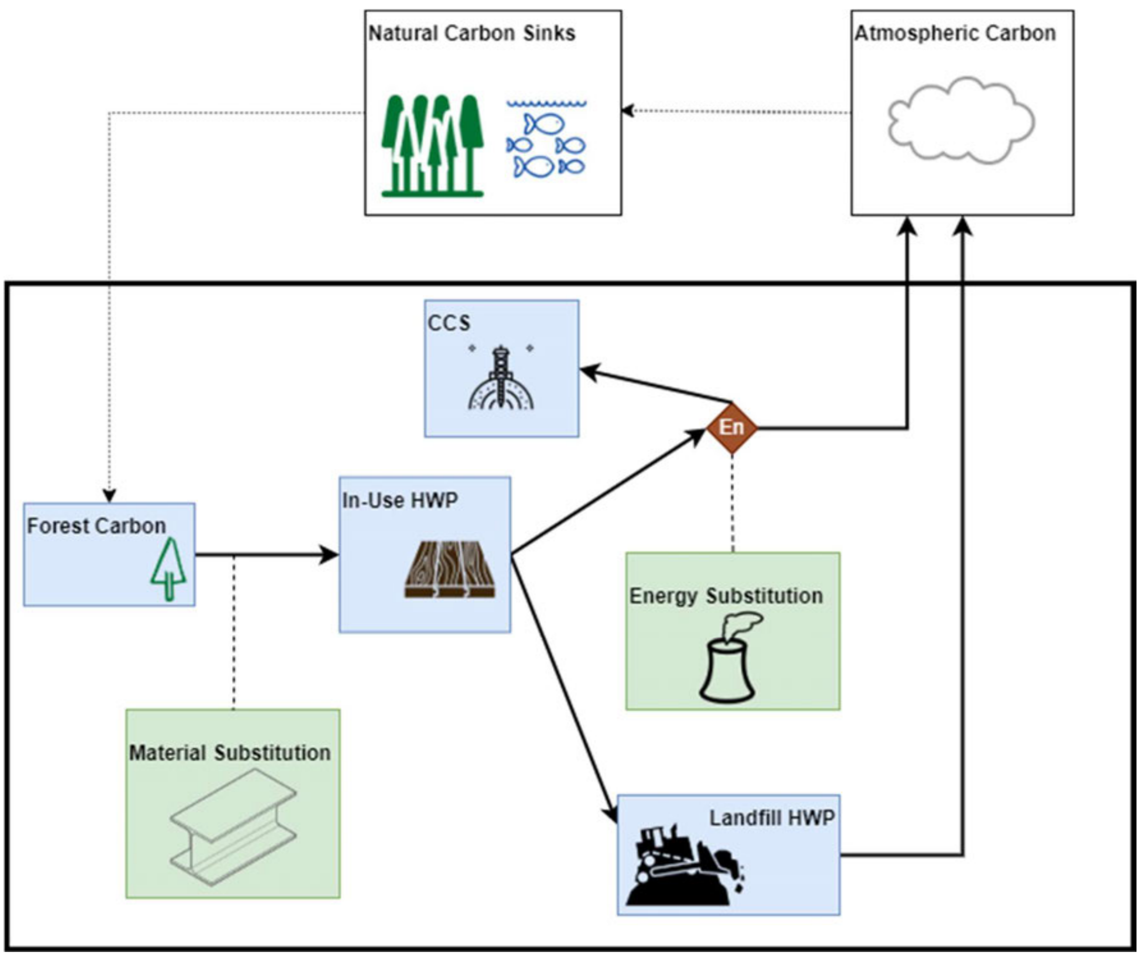



*Figure 4.1. High level overview of the main features of BioCarp. The heavy border indicates the BioCarp system boundary, with the atmospheric carbon pool evaluated separately.*

The model quantifies all physical flows of carbon within the boundaries of the system and quantifies the carbon in each physical pool (shaded blue) at each time-step. Carbon flows are indicated by solid arrows: heavy black arrows indicate one-way fluxes, and the dotted arrows (from outside the system) indicate the expected direction of net flows. Additionally, BioCarp allows for an estimation of the carbon emissions benefits associated with material and energy substitution (dashed lines connecting to green boxes representing 'virtual' carbon flows and pools respectively). The illustrated flows of carbon within the system, and into and out of the system are all quantified in BioCarp and – in the case of 'physical' carbon – subdivided into $CO_2$ carbon and $CH_4$ carbon where necessary. The flows entirely outside the boundary are analysed with a dynamic climate impacts model. The chain from forest carbon to atmospheric carbon plots the physical progress of identifiable 'packets' of physical carbon atoms from one pool to the next. By contrast, the chain from atmosphere back to the subset of forest carbon entering the system exploits the general pool of atmospheric carbon (not necessarily the carbon atoms previously embodied in HWP).

### 4.2.1 Harvested Wood Products (HWP)

The HWP modules are illustrated in more detail in the view of BioCarp in Figure 4.2.

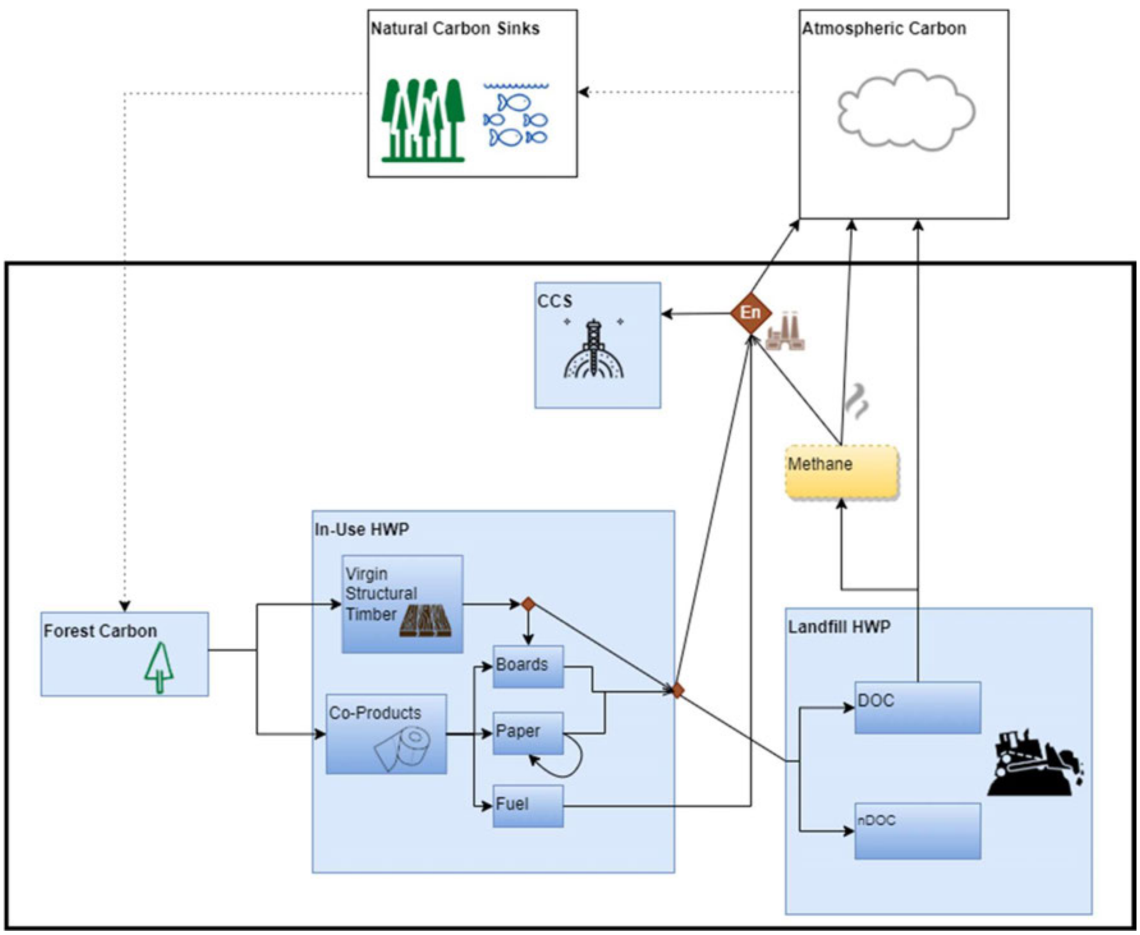



*Figure 4.2. BioCarp model, with focus on HWP (and substitution pools omitted), illustrating transfer of carbon between pools. The heavy border indicates the BioCarp system boundary, and the dotted lines indicate the likely net direction of two-way flows.*

For the ‘more timber demand’ assessments, the starting point of the analysis is always demand for virgin structural timber (VST), with the above-baseline demand for timber in each year expressed as new demand $nd_t$ in tonnes of virgin structural timber in year t. In this context, the term virgin structural timber refers to the virgin timber part of timber-based products used in construction in any context where there is a realistic expectation that the product may last in situ for the lifetime of the building, regardless of whether it actually has a structural function. Typical products that can satisfy this categorisation include dried sawnwood for timber frames, window frames, and (less commonly) cladding; oriented strandboard (OSB) for structural panels and I-joists; and engineered timber – such as glulam and CLT – for structural frames. Such material is allocated a decay constant of k1 (elaborated in section 4.2.2).

Note that in this list of structural materials OSB is an outlier, as it is manufactured from strands of wood rather than sawn wood.  When a sawmill processes a log into sawn wood, sawn wood is regarded here as the primary product, with co-products consisting of bark, chips and dust, which can used in other product categories (including chipboard, paper and fuel, which are all explicitly included in BioCarp). These categories are unlikely to include OSB, as the long strands required are themselves primary products from log processing, not co-products of sawing. Therefore a given log or consignment of logs is likely to be processed into either sawn wood plus co-products or strands plus co-products, but not both. However, a given forest stand may produce different consignments of harvest, with thinnings more likely to be used for OSB and final harvest more likely to be used for sawn wood. This means that most wood harvested is potentially suitable for one of these primary product categories. As discussed in Chapter 3, sawmills can convert more than half of a given log into sawn wood, but UK sawn softwood production accounts for only ~34% of the material harvested. The discrepancy is likely to be because much material is either unsuitable or not required for sawn wood, and one of the alternative uses would be OSB. According to FAO estimates, UK production of OSB has been increasing in recent years, reaching around 0.6 $hm^3$ in 2020, whilst UK production of coniferous sawn wood has been in slight decline, to 3.34 $hm^3$ **(FAO, 2020)**, which supports the idea that for the most part, OSB and sawn wood are alternatives for the same log.

Demand (*d*) for HWP is initially in tonnes of virgin structural timber per year, and then converted to tC. BioCarp can accommodate any chosen demand function, but offers the following as a starting point based on the discussion in Chapter 3. Demand is assumed to grow at an initial growth rate (Hgr) for time t1 years (assumed to be a high growth rate associated with a timber boosting strategy), and a subsequent growth rate (Lgr), for when the timber boosting strategy comes to an end, for instance when wood has effectively saturated the market. Hgr and Lgr are that part of the growth rates that are in excess of BAU. The simplest case, of course, is to assume that there are no existing legislative and market drivers accelerating the use of timber in construction: i.e. BAU charts a static future.

If t ≤ t1, the demand in year t, in tonnes of VST per year is:

$$d_t = d_0 \times (1 + \mathrm{Hgr})^t \qquad \textit{Eq 4—1}$$

If t > t1, the demand is

$$d_t = d_0 \times (1 + \mathrm{Hgr})^{t1} \times (1 + \mathrm{Lgr})^{t-t1} \qquad \textit{Eq 4—2}$$

The new demand is:

$$\mathrm{nd}_t = \mathrm{d}_t - \mathrm{d}_0 \qquad \textit{Eq 4—3}$$

Particleboard and MDF ('boards' in Figure 4.2, which do not include CLT, as this is virgin structural timber) are assumed to be used in contexts where the replacement cycle is faster than the replacement cycle of the building (e.g. flooring and kitchen fit-out), and is assumed to be manufactured either from recycled virgin wood or from co-products, and is allocated a decay constant of k2.

Paper is used as shorthand for recyclable products with – in the main – a short lifespan (decay constant k3). Examples are packaging boards, newsprint and printing paper. The decay constant used in the model is based on the productive life of the wood fibres in the paper, not on the lifetime of any one of the various paper products in the life cycle of the fibres before they reach the point of final disposal, as discussed in Chapter 3.

The word 'fuel' in the figure is used as shorthand for material with a very fast turnaround (decay constant k4) that will not be recycled. Of course this includes fuelwood and material sent for wood pellet manufacture, but it may also include sanitary papers.

An illustrative example of the results produced from a single forest stand, harvested, reforested, and then reharvested, are shown in Figure 4.3. This illustrates the transfer for carbon from forest to structural wood and co-products, which in turn transfer carbon to recycled products, landfill, and out of the carbon storage system altogether (i.e. to the atmosphere).

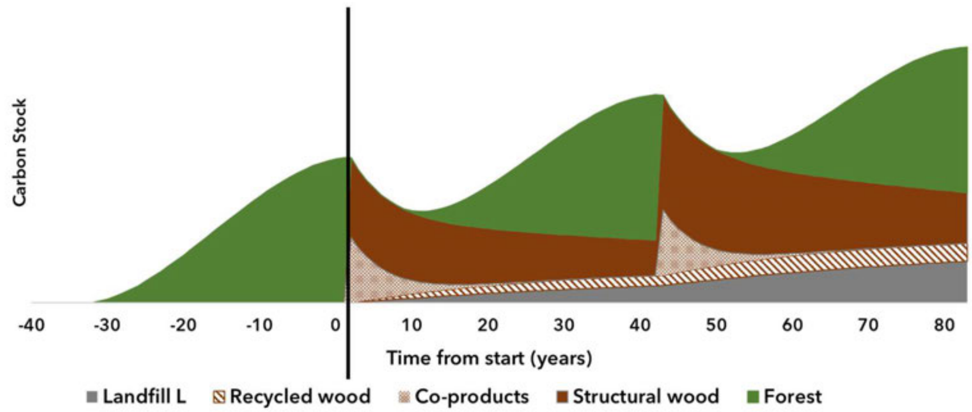



*Figure 4.3. Illustrative carbon pool development through two harvests: first harvest in year zero, forestry on a 40-year rotation.*

### 4.2.2 HWP Pool - in-use module

#### *Description*

This module tracks the wood carbon removed from the forest for use in the technosphere: in addition to logs, this includes byproduct such as stumps and brash when these are indeed removed rather than returned to the soil. All wood carbon removed from the forest is incorporated into one of two broad categories of HWP defined here: virgin structural timber or co-products. Whilst it might be argued that any distinction between primary product and co-product is artificial, in this case it is justified as virgin structural timber is the product category of interest.

As the model is driven by increasing demand for virgin structural timber, it is assumed that this increased demand in turn drives additional harvest to satisfy that demand[48]. Accordingly, the additional harvest and processing will be optimised to secure a high proportion of material suitable for structural timber.

Logs removed from the forest are taken to the mill where they are debarked, and sawn (and dried, as needed), for use in sawn wood or glulam/CLT; or, for OSB for instance, shredded and processed. The product is then transported to a construction site (potentially via further manufacturing facilities) where it is installed in a building structure. At each step of the timber journey (sawmill, factory, construction), co-product is produced such as bark, sawdust, and offcuts. This co-product has a range of potential uses, including heating fuel

[48] This is a reasonable assumption where sufficient timber and manufacturing capacity is available. Where this is not the case, then increased demand for virgin structural timber would push other timber users out of the market, with no resultant effect on forest carbon, and a beneficial effect on HWP carbon on account of a constrained supply being pushed into longer-term usage.

for the sawmill or factory; horticultural mulches; and raw material for paper, cardboard, fibreboard, particleboard, and wood pellets (another energy vector).

Virgin structural timber removed from buildings over time is routed to one of three end-of-life options, which are recycling, energy recovery, and landfill. Arguably reuse should be treated as a separate option, but percentages are likely to remain very small for the foreseeable future, so any material that might be reused is treated as recycled. Recycled material enters a separate pool from which the final exits are energy recovery or landfill, potentially following closed-loop recycling in the case of particle board: in future the same may apply to MDF, as a recycling process for this has been technically proven and independently validated **(Elias, 2011).**

### *Parameters*

Model parameters relevant to this section, used but not defined in Table 4-1 are:

- Hwf: the proportion of the harvest ultimately embodied in installed structural timber. The proportion allocated to co-products is therefore 1-Hwf
- k1: the decay or hazard constant for installed structural timber (i.e. the annual fraction of the pool removed from buildings and routed to recycling, energy recovery or landfill).
- k2: the average decay or hazard constant for wood fibres in boards made from recycled and co-products.
- k3: the average decay or hazard constant for wood fibres in paper products (potentially via multiple recycling loops).
- k4: the average consumption rate of fuel.
- Wr: the fraction of end-of-life material routed to recycling.
- Wen: the fraction of end-of-life material routed to energy recovery
- Wlf: the fraction of end-of-life material routed to landfill. Note that Wr + Wen + Wlf = 1. In situations where recycling is not an option, Wr is re-set to zero and Wen and Wlf rescaled, e.g. Wen’ = Wen/(Wen + Wlf).
- cf: the carbon fraction of wood products (taken as a fixed 0.446, assuming wood at 12% moisture content, and 50% of the dry matter being carbon).
- nd: new timber demand.

### *Equations*

The carbon flux of material into the virgin structural timber and the co-products sub-pools are functions of demand. The outgoing flux from these pools in a given year is primarily a function of the pool size and the decay rate.

The general equation for the calculation of the stock of carbon in a given pool is given by the first order decay function detailed in IPCC guidance **(Rüter et al., 2019)**, adapted as follows.

$$P_t = P_{t-1} + F_{in,t} - F_{out,t} \qquad \textit{Eq 4—4}$$

The outflux, which is itself a function of pool size, is given by

$$F_{out,t} = F_{in,t}\{1 - (1 - e^{-k})/k\} + P_{t-1}(1 - e^{-k}) \qquad \textit{Eq 4—5}$$

Thus,

$$P_t = e^{-k}P_{t-1} + \{(1 - e^{-k})/k\}.F_{in,t} \qquad \textit{Eq 4—6}$$

where

$P_t$ is the mass of the given carbon pool at the end of year t
$F_{in,t}$ is the influx of carbon in year t
$F_{out,t}$ is the ouflux (or efflux) of carbon in year t
k is the decay constant (units of $yr^{-1}$).

The decay constant is related to half-life ($t_{½}$ years) as follows:

$$k = Ln(2)/t_{½} \qquad \textit{Eq 4—7}$$

Because of the number of variations in pools and decay rates, the terminology used in the following tables of equations deviates slightly from that used above, but the following conventions are consistently applied.

- Pools are signified by a term beginning with upper-case 'P'
- Fluxes are signified by a term beginning with upper-case 'F' and – where there is potential ambiguity – conclude with an indication of flux direction with respect to the referenced pool (in/out)
- Land area is signified by terms beginning with upper-case 'A'
- For substitution fluxes and pools, terms include an upper-case 'S'
- All pool sizes are set to zero when t=0.

| Item | Equation | | Units | Ref |
| --- | --- | --- | --- | --- |
| Virgin Structural Timber Flux In | $Fvst\text{-}in_t$ | $= cf \times nd_t$ *Carbon fraction of timber demand* | tC/yr | *Eq 4—8* |
| Virgin Structural Timber Flux Out | $Fvst\text{-}out_t$ | $= Fvst\text{-}in_t \cdot \{1 - (1 - e^{-k1})/k1\} + Pvst_{t-1}(1 - e^{-k1})$ | tC/yr | *Eq 4—9* |
| Virgin Structural Timber Pool | $Pvst_t$ | $= Pvst_{t-1} + Fvst\text{-}in_t - Fvst\text{-}out_t$ *Previous year pool + influx – efflux* | tC | *Eq 4—10* |
| Boards – flux in | $Fboa\text{-}in_t$ | $= \{p \times Fvst\text{-}in_t \times (1 - Hwf)/Hwf\} + \{Wr \times Fvst\text{-}out_t\}$ *Virgin co-product + EoL structural timber* | tC/yr | *Eq 4—11* |
| Paper – flux in | $Fpap\text{-}in_t$ | $= q \times Fvst\text{-}in_t \times (1 - Hwf)/Hwf$ | tC/yr | *Eq 4—12* |
| Fuel – flux in | $Ffue\text{-}in_t$ | $= r \times Fvst\text{-}in_t \times (1 - Hwf)/Hwf$ *where p, q, r >= 0, and p + q + r = 1* | tC/yr | *Eq 4—13* |
| Secondary products flux in | $Fsec\text{-}in_t$ | $= Fboa\text{-}in_t + Fpap\text{-}in_t + Ffue\text{-}in_t$ | tC/yr | *Eq 4—14* |
| Boards - flux out | $Fboa\text{-}out_t$ | $= Fboa\text{-}in_t \cdot \{1 - (1 - e^{-k2})/k2\} + Pboa_{t-1}(1 - e^{-k2})$ | tC/yr | *Eq 4—15* |
| Paper – flux out | $Fpap\text{-}out_t$ | $= Fpap\text{-}in_t \cdot \{1 - (1 - e^{-k3})/k3\} + Ppap_{t-1}(1 - e^{-k3})$ | tC/yr | *Eq 4—16* |
| Fuel – flux out | $Ffue\text{-}out_t$ | $= Ffue\text{-}in_t \cdot \{1 - (1 - e^{-k4})/k4\} + Pfue_{t-1}(1 - e^{-k4})$ | tC/yr | *Eq 4—17* |
| Secondary products flux out | $Fsec\text{-}out_t$ | $= Fboa\text{-}out_t + Fpap\text{-}out_t + Ffue\text{-}out_t$ | tC/yr | *Eq 4—18* |
| Secondary product Pool | $Psec_t$ | $= Psec_{t-1} + Fsec\text{-}in_t - Fsec\text{-}out_t$ | tC | *Eq 4—19* |
| In-use HWP pool total | $Phwp\text{-}use_t$ | $= Pvst_t + Psec_t$ | tC | *Eq 4—20* |

*Table 4-1. Equations underpinning the in-use HWP module. Notes in italics are intended as very brief reminders of what selected equations are about, but for details refer to the text and the list of parameters and abbreviations.*

### 4.2.3 HWP Pool – Landfill Module

It can be argued that a forward-looking analysis does not even need to consider landfill. The days when landfill was the default option for disposal of wastes of all types has passed, thanks to Directives and taxes targeting the 'problem' of landfill. A high proportion of wood waste is now segregated at source, and when this is done, other treatment options (recycling or energy recovery) are much more cost-effective.

On the other hand, unsegregated construction waste in the UK contains a proportion of timber, and landfill is still a likely destination for this material. A more important consideration, however, is that landfilling waste wood may currently be the most straightforward and lowest cost (albeit barely acknowledged) form of carbon capture and storage.

The point about low cost is clearly true if the most significant cost element – the landfill tax – is waived for carbon storage schemes. UK landfill tax is £94.15 per tonne, which amounts to ~£58/$tCO_2$ initially stored[49]: this is within the wide range of potential costs for CCS with gas-fired power plant in the range of 10 – 146 $/$tCO_2$ (2015 US$) identified in a meta-analysis **(Budinis et al., 2018)**. The European Commission's Zero Energy Platform identified a cost range of 72-92 €/$tCO_2$ for CCS with gas-fired power **(ZEP, 2011)**. Accordingly, one objective of this work is to explore the potential role of landfill (or, to give it an alternative framing, 'long-term secure wood storage') in mitigating carbon emissions.

Taking such considerations into account, the term 'landfill' can be interpreted broadly here, to include as-yet unspecified types of storage facility, for carbon-rich products, that might be developed as lower-cost alternatives to CCS. It might also include the case where material is returned to the ground after (for instance) being left in situ when its functional life is over. Modelling the former case as landfill may be conservative (as there is potential for slowing down wood decay), but this is less likely to be the case for uncontrolled disposal of timber in the ground.

BioCarp quantifies landfill HWP carbon using the following logic. Every year, a fraction of the previous year's in-use HWP pool reaches end of life, and a fraction of that material is removed to landfill, adding to the pool of carbon stored in landfill. The carbon in landfilled wood is divided into two fractions. Firstly, there is the degradable organic carbon fraction (DOCf) – which is the carbon that can eventually be released from the cellular matrix of the wood by microbial action if conditions are right, as illustrated in Figure 4.4. Secondly, there is the non-degradable carbon fraction (NDOC), which is considered to be permanently stored. NDOC consists of the lignin in the wood, and that part of the cellulose and hemicellulose (or – collectively – holocellulose) that is rendered inaccessible by the surrounding lignin **(X. Wang et al., 2013)**. Additionally, wood also includes a percentage (up to around 10%) of organic carbon that is not part of the cell wall structure and is thought to be largely resistant to decomposition **(Kim & Singh, 2000)**. This material includes a wide range of organic compounds, collectively known as 'extractives', some of which are resistant to decomposition 'by design', in that they have a role in

[49] Wood at ~12% MC stores carbon at 1.63 $kgCO_2e$/kg of wood.

defending the living tree from microbial action; but other extractives (with roles in tree metabolism for instance) may actually contribute a share of the degradable carbon that is attributed to HOC in Figure 4.4.[50] The carbon that degrades is emitted mainly as carbon dioxide and methane: although the ratio of $CO_2$:$CH_4$ and the fate of the methane is important, it does not affect the physical carbon balance and is discussed in section 4.2.5.

Additional model parameters used but not defined in Table 4-2 are:

- DOCf: degradable organic carbon, which is the fraction of the total organic carbon landfilled that can degrade.
- k5: decay constant for DOCf.

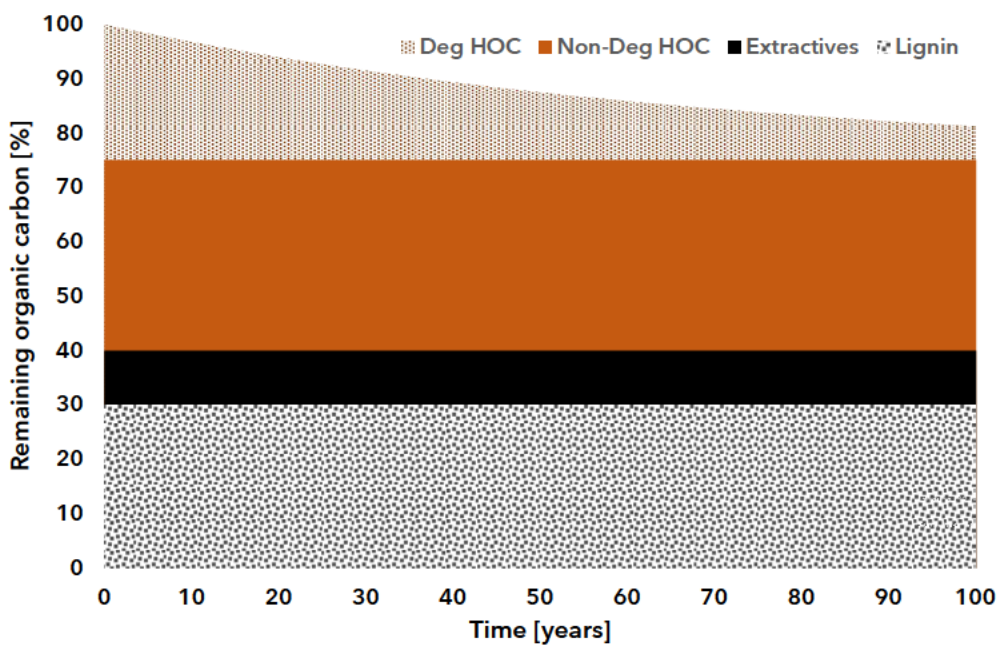



*Figure 4.4. Illustrative graph showing the decline in carbon content of a single addition of wood to landfill at year zero. Hololcellulose (HOC) is divided into degradable and non-degradable groups. Values represented are indicative: In this example, the degradable HOC has a half-life of 50 years.*

[50] In theory this should be a concern, as all of the different organic molecules and polymers present in wood are likely to have their own decay characteristics, but the understanding of how materials decay in landfill is not sufficiently advanced to account for this in any detail.

| Item | Equation | | Units | Ref |
|---|---|---|---|---|
| Landfill Flux in | $Flf_t$ | $= (Fvst\text{-}out_t \times Wlf) + (Fboa\text{-}out_t + Fpap\text{-}out_t) \cdot Wlf/(Wen + Wlf)$ *Sum of the influx to landfill from each of the 3 HWP in-use sub-pools* | tC/yr | *Eq 4—21* |
| Of which DOC | $Fdc\text{-}in_t$ | $= DOCf \times Flf_t$ *Degradable organic carbon in the influx* | tC/yr | *Eq 4—22* |
| And NDOC | $Fndc_t$ | $= (1 - DOCf) \times Flf_t$ *Non-degradable organic carbon* | tC/yr | *Eq 4—23* |
| DOC Flux out | $Fdc\text{-}out_t$ | $= Fdc\text{-}in_t \cdot \{1 - (1 - e^{-k5})/k5\} + Pdt_{t-1}(1 - e^{-k5})$ *Organic carbon degradation* | tC/yr | *Eq 4—24* |
| DOC Pool | $Pdc_t$ | $= Pdc_{t-1}(1 - Ld) + Fdc\text{-}in_t - Fdc\text{-}out_t$ *Previous year DOC pool + influx – efflux* | tC | *Eq 4—25* |
| NDOC Pool | $Pndc_t$ | $= Pndc_{t-1} + Fndc_t$ *Previous year NDOC pool + influx (no efflux)* | tC | *Eq 4—26* |
| Total Landfill Pool | $Plf_t$ | $= Pndc_t + Pdc_t$ | tC | *Eq 4—27* |

*Table 4-2. Equations underpinning the quantification of landfill carbon.*

### 4.2.4 Bioenergy with Carbon Capture and Storage (BECCS) Pool

In the same way that landfill currently offers a medium-to-long-term carbon storage option for end-of-life wood products, in the near future BECCS may be able to provide a similar service. This may prove to be more effective than landfill if $CO_2$ storage proves secure, although a higher financial cost can be expected.

Additional model parameters used but not defined in Table 4-3 are:

- $beccs\text{-}pen_t$ is the penetration of the CCS penetration of the wood energy industry in year t.
- beccs-cef is the efficacy of the CCS capture process – i.e. the fraction of the carbon stream that is successfully captured and stored. Storage is assumed to be permanent (zero flux out of the pool).

| Item | Equation | | Units | Ref |
|---|---|---|---|---|
| Energy Flux in | $Fen_t$ | $= (Fvst\text{-}out_t \times Wen) + (Fboa\text{-}out_t + Fpap\text{-}out_t) \cdot Wen/(Wen + Wlf)\} + Ffue\text{-}out_t$ | tC/yr | *Eq 4—28* |
| BECCS Flux in | $Fbeccs_t$ | $= beccs\text{-}pen_t \times beccs\text{-}cef \times Fen_t$ *The fraction of the HWP to energy flux captured for CCS.* | tC/yr | *Eq 4—29* |
| BECCS Pool | $Pbeccs_t$ | $= Pbeccs_{t-1} + Fbeccs_t$ | tC | *Eq 4—30* |

*Table 4-3. Equations underpinning the quantification of carbon stored via CCS.*

The CCS capture rate for landfill gas is set at zero by default, on the basis that landfill gas is most likely to be used in smaller on-site facilities such as gas engines, or will be cleaned up and injected into the gas network for use in buildings.

### 4.2.5 Substitution Pools

In contrast to the HWP pool, and the forest carbon pool, the substitution pools are 'virtual' carbon accounting pools, which record an estimate of the life cycle GHG emissions avoided through the choice of HWP instead of non-timber options such as steel, concrete and masonry. As discussed in section 2.9, whether and how to include these pools is the subject of debate. The method chosen here embraces the following features and qualities:

- For transparency, to present results in a way which always allows the reader to separate out the contribution of the substitution pools to the combined pools totals.
- A check is applied to the future materials substitution pool influx through the use of a discount factor, which may be construed as a method to take account of changing preferences.
- And furthermore, the displacement factor ($D_f$) itself reduces over time, in line with the decarbonisation of energy networks and industry generally, as discussed in section 3.6.3.
- Carbon leakage potentially provides exits from both the materials and energy substitution pools, whereby fossil fuel use avoided through this strategy is instead taken up in other sectors, and this is considered in sensitivity analysis.

The relationships between the substitution pools and the HWP pools are shown in Figure 4.5.

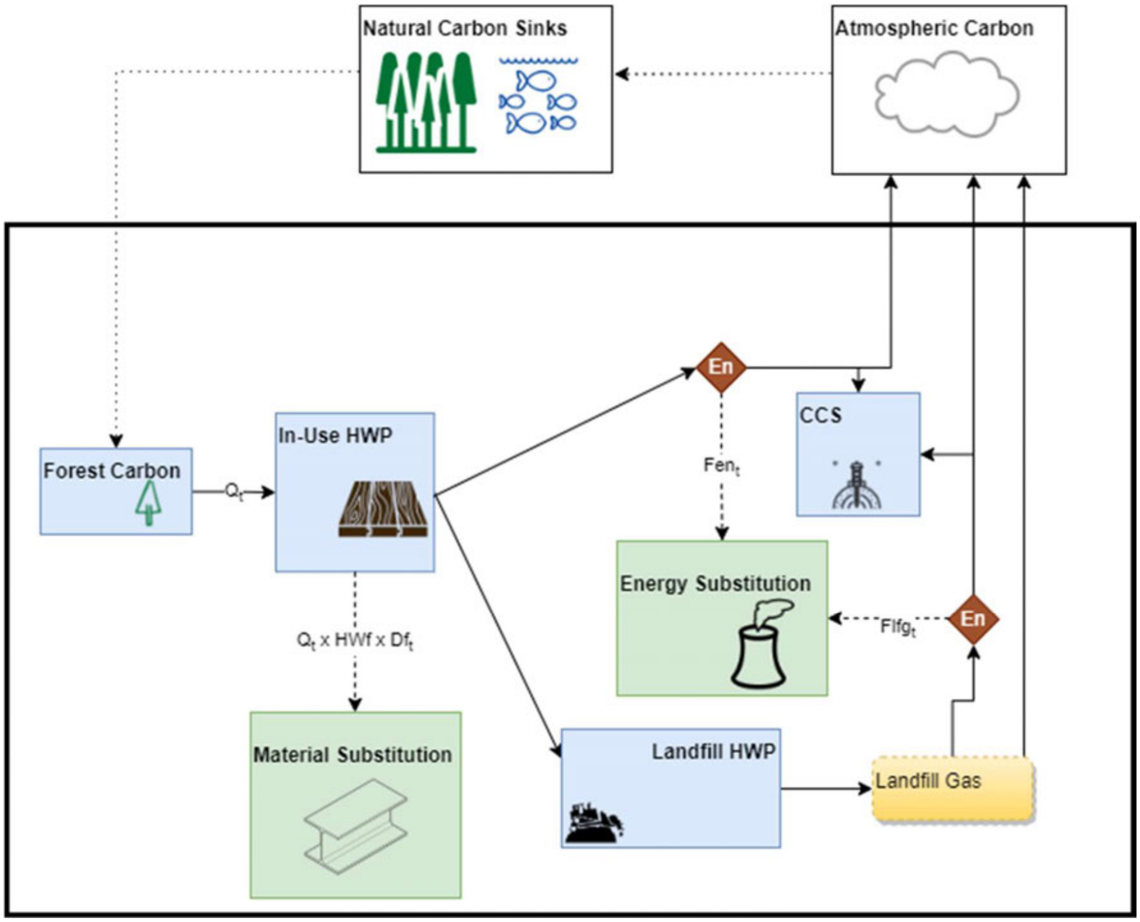



*Figure 4.5. BioCarp model with focus on the relationship between the substitution pools and HWP demand and disposal. The heavy border indicates the BioCarp system boundary, solid lines show flow of carbon within the system, dashed lines show 'virtual' carbon flows, and the dotted lines indicate the likely net direction of two-way flows.*

### *Material Substitution Pool*

The material substitution pool relates only to the flux of VST into the HWP pool, not the subsequent recycled wood. There are two main reasons for excluding the recycled wood, as follows:

- Recycled wood is most likely to be in the form of particleboard, which is a mainstream product with a wide range of uses both within and outside the construction industry. It is, for instance, routinely used for low-cost, non-durable furniture, and it is difficult to argue that it displaces another material. Although the recycled wood in the particleboard displaces virgin round wood, the latter is itself a relatively low-carbon product, so there is little potential for any displacement benefit. Thus there is great uncertainty about the displacement factor to be used, although it is likely to be low.

- Any displacement associated with use of the particleboard at significant levels only occurs a decade or so into the simulation, by which time the displacement factor is heavily discounted. As such, the impact of ignoring this displacement is negligible.

Additional model parameters used but not defined in Table 4-4 are:

- $Df_t$: the displacement factor.
- $cx_t$: correction factor in year t.
- $Ci_t$: carbon intensity of industry index figure ($Ci_0 = 1$).
- α: the displacement factor discount rate.
- β: the carbon leakage rate.

The annual material substitution benefit (tC/yr) is:

| Item | Equation | Units | Ref |
|---|---|---|---|
| Displacement factor | $Df_t = Df_0 \times cx_t$ | tC/tC | *Eq 4—31* |
| Correction factor | $cx_t = Ci_t \times (1-\alpha)^t$ | - | *Eq 4—32* |
| Material substitution flux in | $FSm\text{-}in_t = Fvst_t \times Df_t$ | tC/yr | *Eq 4—33* |
| Material substitution flux out (carbon leakage) | $FSm\text{-}out_t = \beta \times PSm_{t-1}$ | tC/yr | *Eq 4—34* |
| Material substitution pool | $PSm_t = PSm_{t-1} + FSm_t$ | tC | *Eq 4—35* |

*Table 4-4. Equations underpinning material substitution*

### *Energy Substitution Pool and related emissions*

The underlying basis of the energy substitution pool is that both – firstly – landfill gas (LFG) utilisation and – secondly - wood fuel and end-of-life timber routed towards energy recovery, facilitate the avoidance of fossil fuel use for the provision of the energy services now met by the use of this timber.

The methane component of landfill gas is assumed to be a like-for-like replacement for natural gas. This is a necessary simplification, as (1) it neglects the cost of cleaning up the landfill gas, (2) similarly (and on the opposite side of the equation) it neglects the upstream costs associated with the counterfactual natural gas, (3) neither natural gas nor biomethane are pure methane, and (4) the usage of LFG methane does not necessarily reflect the economy-average usage of natural gas. The calculation itself is straightforward, as the substitution effect equates to the carbon content of the methane component of the LFG.

For the HWP energy recovery, the calorific value of the HWP is determined, and the efficiency with which it can be used in electrical generating plant permits the estimation of electricity generated per tonne of carbon ($MWh_e/tC$), and this is compared to the emission factor of electricity in the given year. The net efficiency used for this calculation accounts for the energy required to run the CCS process, based on the following logic.

- The CCS capture efficacy – beccs-cef – falls within the range 0.85 – 0.95 **(Metz et al., 2005).**
- The energy cost of CCS capture – the additional energy required to run the CCS process - (a fraction of 0.1 to 0.4) is related to the capture efficacy, with a higher efficacy bringing a higher energy cost **(Metz et al., 2005)**. A linear relationship is assumed between efficacy and energy cost within these ranges.

The energy cost is therefore given by

$$\begin{aligned} \text{enco} &= 0.1 + 0.3 \text{ x } (\text{beccs-cef} - 0.85)/(0.95 - 0.85) \\ &= (3 \text{ x beccs-cef}) - 2.45 \end{aligned} \qquad \textit{Eq 4—36}$$

The net electrical efficiency is given by

$$\eta_{net} = \eta_{gr} / (1 + \text{enco}) = \eta_{gr}/(3 \text{ x beccs-cef} - 1.45) \qquad \textit{Eq 4—37}$$

A fraction of the LFG produced in a landfill is emitted to the atmosphere as methane. This comprises the methane that is (1) neither captured nor oxidised in the landfill cap, thereby escaping directly from the landfill, (2) methane slippage – methane that leaks in the utilisation process. Although this emission makes no contribution to energy substitution, it is included in Table 4-5 for convenience. Additional model parameters used but not defined in Table 4-5 are:

- CH4C is the relative molecular mass of methane to carbon = 16/12
- LFGm is the fraction of LFG carbon released as methane, with (1 – LFGm) released as carbon dioxide).
- LFGc is the fraction of LFG captured.
- LFGu is the fraction of the captured LFG that is utilised (rather than flared).
- LFGs is the 'slippage' fraction, which is the methane that leaks to the atmosphere during utilisation processes
- LFGox is the fraction of released methane oxidised as it migrates through the landfill cap.
- CO2C is the relative molecular mass of carbon dioxide to carbon = 44/12.

- WCE is the wood carbon energy density (i.e. the calorific value of dry wood in terms of MWh/tC).

| Item | Equation | | Units | Ref |
|---|---|---|---|---|
| Landfill Gas Production | $Flfg_t$ | $= Fdc\text{-}out_t$ *Outgoing flux of degradable organic carbon* | tC/yr | *Eq 4—38* |
| Landfill Gas substitution influx | $FSlfg\text{-}in_t$ | $= F\mathit{lfg}_t \times LFGm \times LFGc \times LFGu \times (1 - LFGs)$ | tC/yr | *Eq 4—39* |
| Landfill Gas substitution outflux | $FSlfg\text{-}out_t$ | $= \beta \times \mathit{PSlfg}_{t-1}$ | tC/yr | *Eq 4—40* |
| Landfill gas substitution pool | $PSlfg_t$ | $= PSlfg_{t-1} + FSlfg\text{-}in_t - FSlfg\text{-}out_t$ | tC | *Eq 4—41* |
| HWP to energy flux | $Fen_t$ | $= (Fvst\text{-}out_t \times Wen) + (Fboa\text{-}out_t + Fpap\text{-}out_t)\cdot Wen/(Wen + Wlf)\} + Ffue\text{-}in_t$ | tC/yr | *Eq 4—42* |
| Electrical generation from HWP used for substitution | $GENel_t$ | $= Fen_t \times WCE \times \eta_{net}$ *Carbon flux* x *calorific value* x *net efficiency* | $MWh_e$ | *Eq 4—43* |
| HWP energy substitution flux | $FSen\text{-}in_t$ | $= GENel_t \times EFgrid_t$ *Electricity generated* x *grid emission factor* | tC/yr | *Eq 4—44* |
| HWP energy substation outflux (carbon leakage) | $FSen\text{-}out_t$ | $= \beta \times PSen_{t-1}$ | tC/yr | *Eq 4—45* |
| HWP energy substitution pool | $PSen_t$ | $= PSen_{t-1} + FSen\text{-}in_t - Fsen\text{-}out_t$ | tC | *Eq 4—46* |
| Total Energy Subs Pool | $PSall\text{-}en_t$ | $= PSlfg_t + PSen_t$ | tC | *Eq 4—47* |
| **Non-$CO_2$ LFG Emission** | | | | |
| Methane Emitted to atmosphere. | $Fmet_t$ | $= Flfg_t \cdot LFGm \times \{(1 - LFGc) \cdot (1 - LFGox) + LFGc \cdot LFGs\}$ | tC/yr | *Eq 4—48* |

*Table 4-5. Equations underpinning energy substitution.*

### 4.2.6 Forest Carbon

When accounting for HWP carbon, the accuracy of the model is governed by the degree to which the model parameters represent reality. If, as seems possible, the 'true' values of all parameters are within the ranges assumed in the model which all draw on real-world evidence, then the true sizes of the HWP pools will be well within the ranges shown in the model outputs. As discussed above, the same cannot be said of the substitution pools, unless an extreme range of displacement factors are considered in the model (including negative values) which would render the output meaningless. Although substitution benefits can be precisely calculated, model uncertainties around issues such as carbon leakage and changing preferences mean we cannot even be certain that the correct calculation is being performed. However, such results can provide a useful basis for discussion when presented with full disclosure.

Quantification of how forest carbon responds to a long-running change in demand for structural timber is beset by related issues. Even where details are known of forest management practices and tree growth rates (often the case) and the effect of these on soil carbon (less often), there cannot be certainty about how landowners, industry and governments will respond to the change in demand. The list of unknowns includes:

- To what extent (if any), will the extra demand for virgin structural timber displace existing demand for the timber crop?
- And what will be the indirect impact in terms of displaced users opting for other materials?
- In cases where the growth of commercial forest resources seems to comfortably outstrip local or regional demand, can it be assumed that trees will be harvested later (resulting in a higher carbon stock per hectare), or will new markets and exports pick up the slack?
- How will government policy and incentives, and societal and investor expectations, combine to influence the balance between optimising forestry assets to meet different objectives, such as timber production, profit more generally, carbon storage, and biodiversity?
- To what extent (if at all) can afforestation rates respond to predictions of demand at the end of the rotation?
- In what locations will afforestation for commercial production take place – bearing in mind climate and soil carbon?

To add to these uncertainties about socio-economic factors, there are questions to face about the resilience or response of forestry to climate change and globalisation, with issues such as more extreme weather, higher atmospheric $CO_2$ concentrations, and faster spread of pests and diseases all potentially having a role to play.

Whilst all of the above areas of uncertainty might be modelled, doing so would, in all likelihood, produce results with ranges so wide as to be unhelpful. Therefore the approach taken here is to identify specific scenarios and assumptions, and caveat them as necessary, explaining the uses and limitations of each set of results. The logic behind the method is shown in Figure 4.6 and Figure 4.7, with the details presented in Table 4-6.

At the start of the simulation, all land and its associated carbon stock is outside the system. Each year in the short and medium-term (up to one rotation length of, for instance, 45 years in UK commercial forestry), forest land stocked with mature trees ready to harvest can be brought into the system to satisfy immediate demand. Also, grassland for afforestation can be imported to the system to satisfy all anticipated long-term demand (one rotation into the future) that cannot be met by the forest land already imported.

The carbon stock in each parcel of land (whether forest or grass) imported into the system is regarded as the baseline for that parcel, so does not need to be reported: instead, the model assesses the change in carbon stock over time associated with harvesting and replanting mature forest (tree carbon loss followed by recovery), and afforestation (land-use change emissions, followed by tree carbon accumulation).

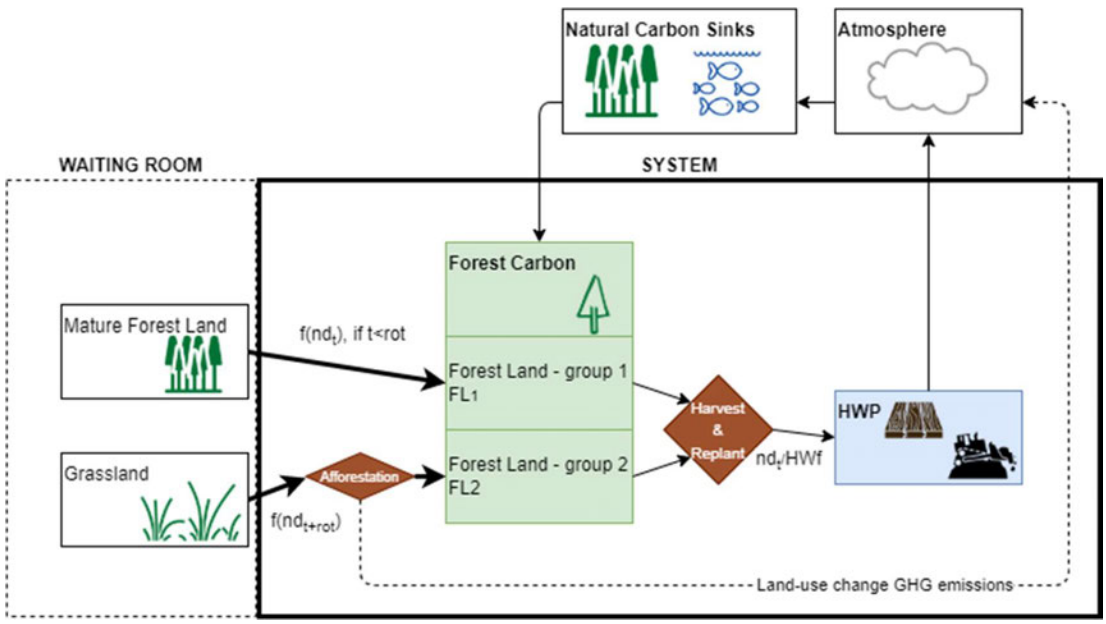



*Figure 4.6. BioCarp model with focus on forest carbon pool and land requirement. The heavy border indicates the BioCarp system boundary, with a 'waiting room' outside it, containing land available for introduction to the system when needed; heavy solid arrows indicate introduction of land area into the system; thin solid arrows indicate carbon flows; and dashed arrow shows non-$CO_2$ GHG emission (nitrous oxide).*

Any permanent increased demand for HWP by the construction industry implies an increase in extraction from existing forests for the next ~45 years (a typical rotation period for commercial conifer plantations in the UK). As such, it involves removal of carbon from the forest: some of this is transferred to durable HWP, and some is rapidly oxidised via – for instance – uses as fuel or as pulp. This transfer of carbon from the forest, partly to HWP, and partly to the atmosphere, is accounted for in the model. The counterfactual of not only leaving the forest *in situ,* but accounting for its continued sequestration of carbon is included in the model, but only for scenario analysis (i.e. not in default settings, as commercial forest – in the UK at least – has generally been planted specifically for the purpose of providing an income from harvest).

The demand for sawnwood in year 1 implies an area of mature forest to be imported into the model, harvested and the quantity of carbon to be removed from the forest in the previous year. That same area will be reforested in year 2, with subsequent growth and carbon sequestration recorded in the model. Also in year 1, an area of unforested land will be imported into the model and planted up for the first time (afforested) to meet the forecast demand in year ~46 (i.e. the rotation period plus 1).

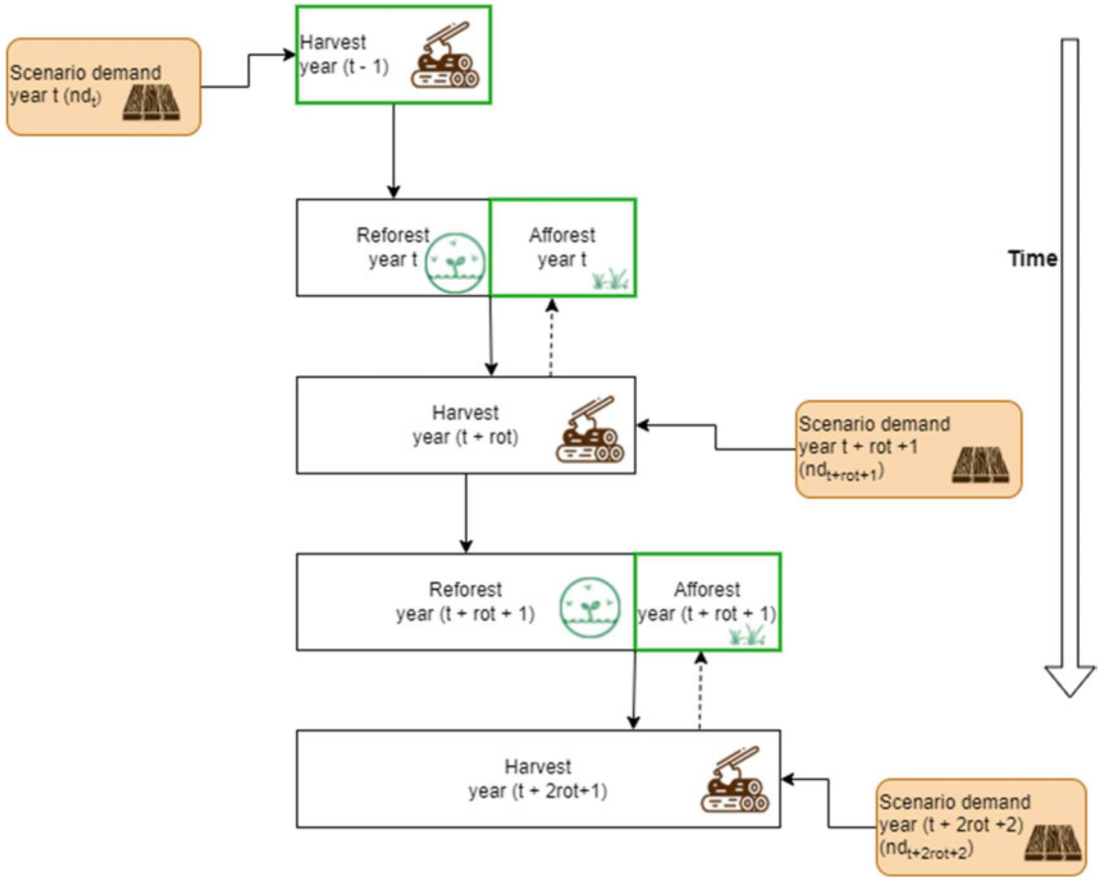



*Figure 4.7. Forest area accumulation over time, showing how a contemporaneous cohort of even-aged forest progresses through two rotations. The rectangles down the centre imply area, and the rectangles with green outlines are areas imported to the system in the year indicated. The total forest area in play can be found by summing all such diagrams for t = 1 to t = rot +1, where rot is the rotation period. The areas required are governed by the average total tree growth per hectare over the rotation period (expressed as tC/ha). The dashed arrows are included to indicate the dependence of afforestation area on the forecast demand (and hence harvest) at the end of the rotation.*

### Equations relating to forest carbon

Two rotation periods can be used in the model (although they can be set to the same number).

- rot: is the rotation period for all forest planted (reforestation and afforestation) during the simulation.
- rotb: the assumed rotation period for all forest imported into the model for immediate harvest. This is a factor in determining the area of forest initially transferred.

Other parameters introduced in this section used but not defined in Table 4-6 are:

- $fv_i$: the quantity of standing timber per area ($m^3$/ha) in a stand of age i. Derived from look-up tables showing standing volume of forest for every one-year age increment, see Figure 4.8.
- cpsv: carbon per standing volume. A conversion factor: tC/$m^3$
- $fcd_i$: the forest carbon density (tC/ha) in a stand of age i. Derived from cpsv × $fv_i$
- EFluc: Land-use change emission factor associated with $N_2O$ emissions on afforestation.

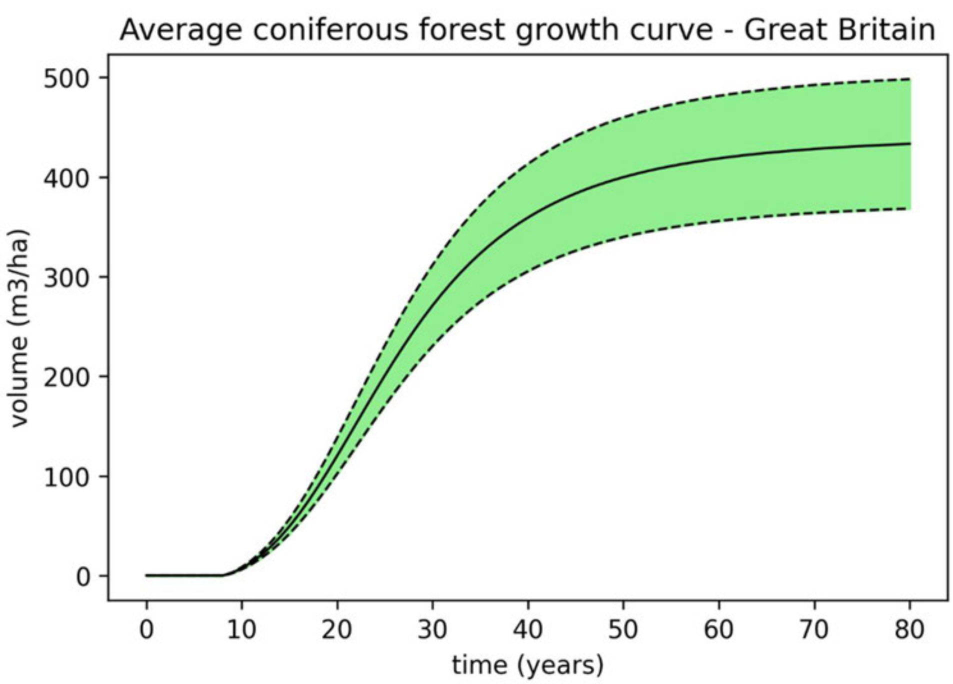



*Figure 4.8. Timber standing volume density as a function of stand age. Based on UK coniferous forest inventory. Note that the quantity of standing timber remaining at each time point is indicated (thus, the data excludes material previously removed as thinnings or by natural processes).*

| Item | Equation | | Units | Ref |
|---|---|---|---|---|
| Forest carbon transfer (imported mature forest) | $Fef_t$ | $= - Fvst_{t+1}/HWf$ (for t = 0 to rot)<br>*Negative (i.e. loss) of forest carbon in* ***existing forest*** *harvested to meet demand next year* | tC/yr | *Eq 4—49* |
| Area of forest transfer (imported mature forest) | $Aef_t$ | $= - Fef_t/fcd_{rotb}$<br>*Carbon transfer / forest carbon density in tC/ha* | ha/yr | *Eq 4—50* |
| Cumulative carbon transfer (imported mature forest) | $Pef_t$ | $= \Sigma_t Fef_t$ (for t = 0 to rot) | tC | *Eq 4—51* |
| New forest carbon pool (resulting from growth in forest planted from year 1 onward) | $Pnf_t$ | $= \Sigma_i fcd_i \times As_{i,t}$ (for i = 0 to rot)<br>*Forest carbon density of all stands of age i, and the area of those stands in year t* | tC | *Eq 4—52* |
| Total forest carbon pool (above ground) | $Pfc_t$ | $= Pef_t + Pnf_t$ | tC | *Eq 4—53* |
| Harvest area in year t | $Ah_t$ | $= Aef_t$ (for t = 0 to rot)<br>Or<br>$= Fvst\text{-}in_{t+1}/(HWf \times fcd_{rot})$ (for t > rot)<br>*Function of HWP demand the following year, and forest carbon density at end of rotation* | ha/yr | *Eq 4—54* |
| Area planted in year t (afforestation and reforestation) | $Ap_t$ | $= Ah_{rot+t}$<br>$= Fvst\text{-}in_{rot+t+1}/(HWf \times fcd_{rot})$<br>*Area needed for harvest at end of rotation, includes afforestation and reforestation* | ha/yr | *Eq 4—55* |
| Area of forest of age i in year t | $As_{i,t}$ | $= Ap_{t-i}$<br>*Area planted in year (t – i)* | ha/yr | *Eq 4—56* |
| Afforestation area in year t | $Aa_t$ | $= Ap_t – Ah_{t-1}$<br>*Area planted minus area reforested in year t (which is the area harvested the previous year).*<br>$= Ah_{rot+t} – Ah_{t-1}$ | ha/yr | *Eq 4—57* |
| Land-use change $N_2O$ emissions year t | $Fluc_t$ | $= Aa_t \times EFluc / GWP100\text{-}N_2O$<br>*Afforestation area x emission factor* | $tN_2O$ /yr | *Eq 4—58* |
| Total land tranfer in year t | $Aaf_t$ | $= Aef_t + Aa_t$ | ha/yr | *Eq 4—59* |
| Cumulative land transfer to year t | $APef_t$ | $= \Sigma_t Aaf_t$ | ha | *Eq 4—60* |

*Table 4-6. Equations underpinning forestry calculations.*

Different approaches are available for compiling the forest volume ($fv_i$) look-up table. Here, the table is derived from data related to the UK softwood forest inventory. Current forestry statistics for Great Britain **(Forest Research, 2020)** provide data from – and refer back to – the National Forest Inventory **(Forestry Commission, 2011, 2014)**. These sources provide data on the areas of coniferous forest standing in each 20-year age class across the approximately 1.3 million hectares of stocked coniferous forest in Great Britain along with the total standing volume in each age class. The inventory volumes in each age class (across all areas in GB and including both private and Forestry Commission land) are divided by the areas of coniferous forest in each age class and summed. The curve for standing volume as a function of age (Figure 4.8)[51] is obtained by linear interpolation between the values for the mid-points of each age class, then smoothing the curve by fitting a sigmoidal function (Eq 4—61), and an uncertainty band of +/- 15% added, to be used in the Monte Carlo process. The uncertainty band (or 'forest growth correction factor': range 0.85 to 1.15) can be adjusted as needed, for instance to reflect the possibility of lower productivity resulting from increased losses from storms, drought, pests and diseases, or higher productivity arising from developments in forest management and potentially productive effects of climate change, such as the impact of increased atmospheric carbon dioxide on growth rates.

$$fv_i = fgcf \text{ x } \{439.8205 + (-3.251031 - 439.8205)/(1 + (t/26.24488)\hat{}3.622534)\} \qquad \textit{Eq 4—61}$$

Where fgcf is the forest growth correction factor.

The curve in Figure 4.8 is intended to represent, as an average, how the entirety of the national coniferous stocked forest estate reached its status when the inventory was taken. In doing so, it offers a limited guide to how average new forest might develop in the future. The method therefore has its weaknesses in that it does not take account of a number of dynamic factors, such as silvicultural developments that have occurred in the decades since many of the subjects of the forest inventory were planted. Therefore, future forestry can be expected to be more productive than is indicated here, assuming that equally productive land is available. Furthermore, the inventory data is arguably too coarse to be represented accurately in this way. Nevertheless, the curve is used as the central modelling assumption in this thesis for UK forestry growth. The curve should be understood as a modelling approximation of the inventory data discussed above, which itself involves approximations associated with proxy measurements for stock volume, age and area. The curve fits approximately in the range for Yield Class (YC) of 8-10 $m^3ha^{-1}yr^{-1}$ mean annual increment as presented in **Matthews et al. (2016)**.[52] Whilst the more productive forests in GB operates at higher yield classes (potentially up to about YC24 according to **Matthews et al.**

[51] Note that if continued beyond 80 years, the curve reaches a peak in the next age band and falls away thereafter, but trees in this age band are out of scope in this thesis so this part of the curve is not shown.
[52] See figure 3.

**(2016)** for Sitka Spruce), the entirety of the national coniferous forest estate in private and public hands covers a range of species, soils, climate and management practices. A more conventional approach to defining the forest volume look-up table would be to use expert input to identify a yield class growth curve that is thought to best represent future forestry: even though individual curves for specific contexts are scientifically well-founded, there would still be an element of subjectivity in defining a curve to represent the expected average across all situations. This alternative is considered in the sensitivity analysis in Section 6.2.6.

In the management of a given area of forestry – whether optimised for timber productivity, carbon productivity or profit – detailed knowledge of the YC and growth curve is highly advantageous; but for the modelling presented in this thesis, the accuracy of the curve is relatively unimportant in BioCarp's quantification of the forest carbon pool. On the other hand, it is central to estimating the area of forest needed to meet the demand for timber.[53]

### *Nitrous Oxide*

When land is afforested for the first time, the UK National Atmospheric Emissions Inventory **(BEIS, 2019)** records a one-time emission of nitrous oxide, which is the variable EFluc, and is currently given as 5.8 $tCO_2e/ha$ (having been given as 10 $tCO_2e/ha$ in previous years). The quantity of $N_2O$ is back-calculated from the GWP100 value for nitrous oxide.

### *Soil Carbon Flux*

The quantification of soil carbon flux is highly complex, contentious, riddled with uncertainty and hugely dependent on context, such as soil type and climate **(Hart & Pomponi, 2020)**. Therefore, in its default mode this model excludes soil carbon. A justification often used for this common choice is that across a wide landscape (e.g. at regional or national level), an equilibrium is likely to have been reached, whereby losses are balanced by gains. However, this claim is especially dubious when significant land-use change is being modelled. A more honest statement would be that the science and data do not yet support the modelling of soil carbon flows to a sufficient level of accuracy to justify inclusion in BioCarp defaults.

Notwithstanding the scant data to support the analysis, the model does include a basic soil carbon module as an optional extra for scenario testing, to explore the potential importance of soil carbon in the system.

[53] Reduced productivity per hectare, for instance, will result in larger areas of forest needed to meet the specified demand for wood product, but the total carbon stored will, *ceteris paribus,* be the same.

The basic issue is that evidence exists that when afforestation takes place, there may be an immediate and ongoing loss of soil carbon (depending on the nature of the soil) that endures for the first rotation, before gradually recovering to its previous level during the second rotation. The loss of soil carbon can be significant in organic soils, particularly when drained for afforestation, potentially amounting to around 120 tC/ha. Another part of the dynamic is the accumulation of above and below ground tree carbon (in addition to the stem) during the rotation, and its subsequent oxidation after harvest.

The soil carbon model, included as an option within BioCarp, allows the average cumulative soil carbon loss per hectare after one rotation to be set. That amount is emitted to the atmosphere at a constant rate during the first rotation, and recovered at the same rate during the second, at which point it settles. Additionally, the temporary accumulation of non-stem tree carbon during each rotation is modelled as a proportion of the stem carbon.

Additional model parameters for the soil carbon module used in Table 4-7 are:

- $Psc_t$ is the soil carbon pool compared to that in year zero, and
- pcl is the peak cumulative carbon loss after one rotation.

| Item | Condition | Equation | Units | Ref |
|---|---|---|---|---|
| Soil carbon pool | t ≤ rot | $Psc_t = - pcl \times (t/rot)$ | tC | *Eq 4—62* |
| Soil carbon pool | 2rot >= t > rot | $Psc_t = - pcl(1 – (t-rot)/rot)$ | tC | *Eq 4—63* |
| Soil carbon pool | t > 2rot | $Psc_t = 0$ | tC | *Eq 4—64* |

*Table 4-7. Equations underpinning estimation of possible soil carbon losses.*

### 4.2.7 Values used in BioCarp for constraints and variables

In this section (4.2 and all subsections up to this point), the capabilities and functions of BioCarp have been presented in detail. It remains to present the values used for the model parameters that have been discussed. In the approximate order in which they have been mentioned, the default values and ranges used for model parameters are shown in Table 4-8, along with references to the relevant section of the thesis where each parameter is discussed.

| Parameter | Category | Description | Value | Ref/Evidence |
| --- | --- | --- | --- | --- |
| Hwf | Uncert | Fraction of harvested timber carbon surviving as installed structural timber | 0.34 – 0.5 – 0.6 | Section 3.5 |
| k1 | Uncert | Hazard rate for installed structural timber | 0.007 – 0.014 – 0.035 | (Brunet-Navarro et al., 2016; Penman et al., 2003) † (Section 3.2.2) |
| k2 | Uncert | Hazard rate for co-product - boards | 0.028 – 0.046 - 0.14 | (Brunet-Navarro et al., 2016; Penman et al., 2003) † |
| k3 | Uncert | Hazard rate for co-product - paper | 0.02 – 0.028 – 0.1 | (Brunet-Navarro et al., 2016; Penman et al., 2003) † |
| k4 | Uncert | Hazard rate for co-product - fuel | 0.7 to 7 | Section 3.2.2 |
| k5 | Uncert | Decay constant for DOCf in landfill | 0.010 to 0.022 to 0.12 yr-1 | (DEFRA & Golder Associates, 2014; Heyer et al., 2018; US EPA, 2016) † |
| Wr | Uncert | Fraction of end-of-life material recycled | Variable 0 to 1, constrained by Wr + Wen + Wlf = 1 | N/A |
| Wen | Uncert | Fraction of end-of-life material to energy recovery | | |
| Wlf | Uncert | Fraction of end-of-life material to landfill | | |
| cf | Quasi-constant | Carbon fraction of wood at point of use | 0.446 | (BSI, 2014a) |
| $nd_t$ | Control var | New demand for timber (in tonnes) above baseline | Modeller's pick: 3% growth from 0.5 Mt/yr | Section 3.2.3 |
| beccs-cef | Uncert | BECCS carbon capture efficiency | 0.85 to 0.9 to 0.95 | (Metz et al., 2005) |
| DOCf | Uncert | Degradable organic carbon – the fraction of wood organic carbon that can decay in anaerobic conditions | 0.01 to 0.12 to 0.5 | (Heyer et al., 2018; Micales & Skog, 1997; US EPA, 2016; Ximenes et al., 2008) |
| $Df_0$ | Uncert | Displacement factor for material substitution | 0 to 0.7 to 2.0 tC/tC | (Hart et al., 2021; B. B. Lippke et al., 2004; Sathre & O'Connor, 2010) |
| $cx_t$ | Control var | Time-dependent correction factor for displacement factor | Function. $cx_0$ = 1 | Section 3.6.2 |
| $Ci_t$ | Uncert | Carbon intensity of industry | Function. Index $Ci_0$ = 1 | Section 3.6.3 |
| α | Control var | Displacement factor discount rate | 0 to 0.05 | Section 3.6.2 |
| β | Control var | Substitution pool Leakage | | Section 3.6.2 |
| | | | | Continued... |

| ...cont | | | | |
|---|---|---|---|---|
| Parameter | Category | Description | Value | Ref/Evidence |
| CH4C | Constant | Molecular mass of methane relative to carbon | 16/12 = 1.33 | Periodic Table |
| sLFGm | Uncert | fraction of landfill gas (by volume) that is methane | 0.5 to 0.57 | **(DEFRA & Golder Associates, 2014; IPCC, n.d.)** |
| LFGc | Uncert | Fraction of LFG captured | 0.48 - 0.52 - 0.57 | **(DEFRA & Golder Associates, 2014)** |
| LFGu | Uncert | Utilisation fraction of captured LFG | 0.76 +/-10% | **(DEFRA & Golder Associates, 2014)** |
| LFGs | Uncert | LFG slippage (leakage) from combustion plant | 0.015 +/-10% | **(DEFRA & Golder Associates, 2014)** |
| LFGox | Uncert | Fraction of methane in LFG oxidised as it migrates through the landfill cap | 0 – 0.1 – 0.1 | **(DEFRA & Golder Associates, 2014; IPCC, n.d.)** |
| CO2C | Constant | Molecular mass of carbon dioxide relative to carbon | 44/12 = 3.67 | Periodic Table |
| WCE | Quasi-constant | Wood carbon energy – the lower heating value of wood | 10.6 MWh/tC | (Forest Research, n.d.-a) |
| $\eta_e$ | Uncert | Efficiency of electrical generator | 0.3 to 0.355 to 0.4 | **(Stephenson & Mackay, 2014)** |
| rot | Control var | Rotation length for all forestry planted during the simulation | 40-50 years | **(Haw, 2017; Lee & Watt, 2012; Moore et al., 2012)** † |
| rotb | Control var | Rotation length of mature forestry imported into the model | 40-50 years | |
| $fcv_i$ | Uncert | Forest carbon volume ($m^3$/ha) in a stand of age i | Growth curve function | Figure 4.8 |
| cpsv | Uncert | Carbon per standing volume | 0.16 - 0.17 – 0.18 tC/m3 | Section 3.5.1 |
| Fgcf | Uncert | Forest growth correction factor | 0.85 to 1.0 to 1.15 | Figure 4.8 |
| EFluc | Constant | Land use change emission factor: $N_2O$ | 5.8 $tCO_2e$/ha | **(BEIS, 2019)** |

†Some interpretation of the data and information presented in the sources was needed to define the ranges.

*Table 4-8. Summary of parameters used in BioCarp, including the default values and ranges used in the model, and references to the section of the thesis in which they are discussed. The term 'quasi-constant' is used to describe parameters for which variation is within a narrow or unknown range, and/or the overall impact of any variation on the carbon account will be negligible.*

## 4.3 Temporal Climate Impacts

As Figure 4.1 shows, changes in the various carbon pools correspond to changes in atmospheric carbon. In particular, combustion and degradation of HWP result in carbon being added to the atmosphere, in the form of both carbon dioxide and methane. And forest processes can be expected to draw carbon dioxide down from the atmosphere on a net basis, assuming that forest growth is not surpassed by losses to the combined forces of fire, storm, pests and diseases. Additionally, changes in the system result in changes to emissions of the GHG nitrous oxide to the atmosphere. In order to quantify the impact of these additions and subtractions of these three GHGs, the lifetimes of these gases in the atmosphere must also be factored into the analysis. To do this, the dynamic LCA method pioneered by **Levasseur et al. (2010)** is used. This they implemented as a spreadsheet tool (Dynamic Carbon Footprinter v2.0 – updated to take account of the IPCC Fifth Assessment Report, AR5 2013). A more recent implementation of the method is the Temporal Climate Impacts model previously referred to **(Cooper, 2020)**, which has added global temperature change to the outputs. A trial of the two calculators with test data (steady emissions of carbon dioxide, methane and nitrous oxide for 100 years) shows very similar results for radiative forcing, with a discrepancy of just 1% in integrated RF after 100 years. The model draws on the science and assumptions in the IPCC Physical Science Basis report of 2012 **(Myhre et al., 2013)**, as outlined here.

A single emissions pulse of any GHG leads to a raised concentration of that GHG in the atmosphere, which will decline over time either – in the long run – to the non-perturbed level, or to some intermediate level. During the period when this raised concentration of atmospheric GHG exists, there is also an increase in radiative forcing (units of $Wm^{-2}$) compared to the situation in the absence of that pulse. Integrated over time, this gives the total change in energy balance. The impact of this change on global temperature is described by an impulse response function, combining climate sensitivity (units of $K(Wm^{-2})^{-1}$) and response times for both short and long timescale effects.

### 4.3.1 Radiative forcing

Radiative forcing is the net change to the earth's energy balance resulting from some perturbation to the system, averaged over a given period. In the context of this work, the perturbation is the addition or removal of one or more GHGs from the atmosphere arising from changes to the construction and forestry management. Whilst radiative forcing can refer to the energy balance at the top of the atmosphere, the IPCC has settled on the tropopause as the interface of interest, with RF defined as *"the change in net irradiance at the tropopause after allowing for stratospheric temperatures to readjust to radiative equilibrium, while holding surface and tropospheric temperatures and state variables such as water vapour and cloud cover fixed at the unperturbed values"* **(Myhre et al., 2013)**.

The effect depends on the gas, with specific radiative forcing (units of $Wm^{-2}kg^{-1}$) and radiative efficiency ($Wm^{-2}ppb^{-1}$) being measures of the radiative forcing power of a greenhouse gas, in terms of the effect for a given mass added to the atmosphere or change in concentration change of the gas in the atmosphere. Parameters used in the model are from **Boucher & Reddy (2008)**.

### 4.3.2 Lifetime of GHGs in the atmosphere

Any dynamic account of the climate change impact of GHG emissions must include the lifetime of those gases in the atmosphere. The average lifetime of a GHG in the atmosphere can be deduced from the total atmospheric burden divided by the annual sink. This simple calculation hides many complexities, as methane is not distributed homogeneously in space and time. For instance, methane concentrations currently range from ~ 1650 ppb in the Antarctic to ~1950 to the North of the equator, with seasonal variation, and nonuniform distribution within the column **(GHGSat - Pulse, 2021)**. The many uncertainties in understanding and modelling of methane are discussed by **Turner et al. (2019)**. Putting such questions to one side however, when feedback loops are taken into account, a perturbation lifetime ($\tau$) can be derived from this average lifetime. In the case of methane, a negative feedback loop exists whereby a perturbation that increases the methane concentration consumes the hydroxyl radicals in the troposphere that are on the pathway of the main sink for methane, thereby slowing the removal and increasing the lifetime of the methane. In the case of nitrous oxide, a positive feedback loop exists involving $O_3$ and other $NO_x$, which results in the perturbation lifetime being slightly lower than the average lifetime. The perturbation lifetimes are shown in Table 4-9 and the decay in any perturbation $Q_0$ after time t is given by:

$$Q_t = Q_0 \, e^{-t/\tau} \qquad \textit{Eq 4—65}$$

For $CO_2$, the IPCC concludes that no single lifetime can be given, referring back to **Joos et al. (2013).** A fraction of any perturbation is associated with each of four nominal lifetimes $\tau_i$ (including one infinite), as detailed in Table 4-9, which combine to define a function with a rapid initial reaction to the perturbation with a very long tail, with a significant fraction surviving indefinitely.

$$\mathrm{IRF(t)} = \sum_{i=1}^{4} a_i \cdot \exp\left(-\frac{t}{\tau_i}\right) \qquad \textit{Eq 4—66}$$

Inspection of the values in Table 4-9 shows that methane and nitrous oxide are both relatively strong radiative forcers, compared to carbon dioxide, but nitrous oxide has a much longer lifetime than methane. This explains why, over a 100-year timescale, the accepted value for the GWP of $N_2O$ is an order of magnitude higher than that for $CH_4$, which is more than an order of magnitude higher than that for $CO_2$. Integrated over a shorter timescale of twenty years for instance, the GWP20 of methane is significantly increased, reducing the gap between it and $N_2O$; whilst integrated over 500 years the GWP500 of methane is much reduced.

| Gas | Specific radiative forcing $Wm^{-2}kg^{-1}$ | i | Fraction $a_i$ | Perturbation Lifetime $\tau_i$ (years) |
|---|---|---|---|---|
| $CO_2$ | 1.75E-15 | 1 | 0.2173 | ∞ |
| | | 2 | 0.2240 | 394.4 |
| | | 3 | 0.2824 | 36.54 |
| | | 4 | 0.2763 | 4.304 |
| $CH_4$ | 2.11E-13 | 1 | 1 | 12.4 |
| $N_2O$ | 3.57E-13 | 1 | 1 | 121 |

*Table 4-9. Specific radiative forcing and perturbation lifetime figures used in the model* **(Myhre et al., 2013)**. *Note that the forcing values for methane and nitrous oxide include adjustment factors for effects that these gases have on other GHGs (for instance $N_2O$).*

## 4.4 Handling of uncertainty in BioCarp

For parameters with elements of uncertainty surrounding their true value, a straightforward linear probability density function (PDF) is defined by a maximum of three values: a default value and upper and lower limit values.

The default value is defined as the expected median of the distribution, and simple linear PDFs are defined, taking the following form, where the median value is $\mu$ at the junction of the two triangles of equal area as illustrated in Figure 4.9.

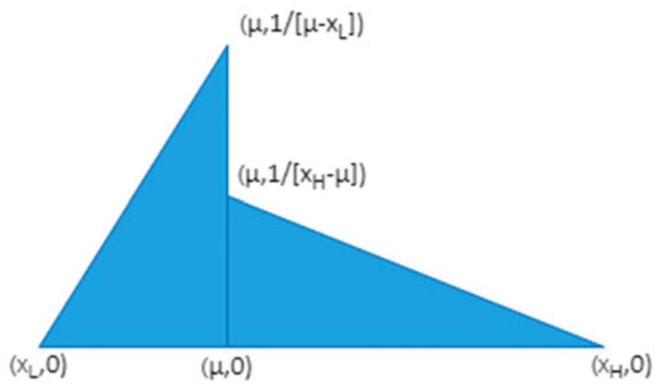



*Figure 4.9. Illustration of the probability space for a variable with a possible range spanning $x_L$ to $x_H$ and a median value of $\mu$.*

Points are selected from this space at random, with each pixel in the diagram being returned with equal probability. The x-coordinate is then returned as the parameter value.

This is implemented in the spreadsheet model through the following sequence of steps.

**Step 1.** The objective is to identify a point at random in the triangular space illustrated in Figure 4.10, with area = 1, bound by the lines, y=0, y=x, and y=2-x. All pixels within the shaded area to be equally probable.

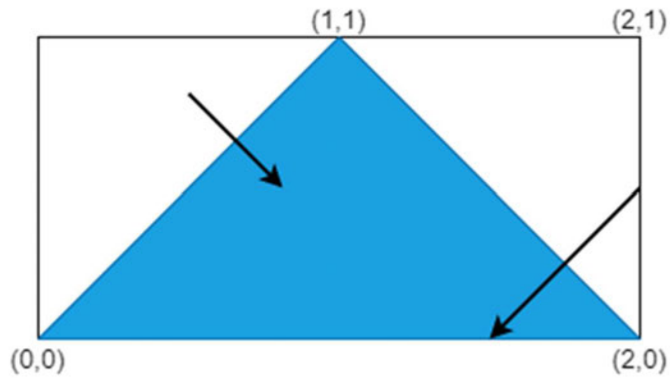



*Figure 4.10. Conversion of a random coordinate within the rectangle to a random coordinate within within the shaded triangle.*

A random number function is used to select a point (x”, y”), where x” is zero to two, and y” is zero to one (i.e. a point within the surrounding rectangle, area = 2). If (x”, y”) is within the shaded triangle (i.e. if y” ≤ x” and y” ≤ 2-x”), then the value of the parameter (x’) is given by:

$$x' = x''$$

If, however, (x”, y”) is in the upper left triangle (i.e. y” > x”), then it is reflected across the diagonal into the solid triangle. In this case:

$$x' = y''$$

This is illustrated in Figure 4.10 by an arrow, showing point (0.5, 0.8) transformed to (0.8, 0.5).

If (x”, y”) is in the upper right triangle (i.e. y” > 2-x”), then it is reflected across the other diagonal into the solid triangle. In this case:

$$x' = 2 - x''$$

This is illustrated by the arrow showing point (2, 0.4) transformed to (1.6, 0).

With repetition, an even distribution of points within the triangle can be expected.

**Step 2.** The next step is to skew the triangle on each side of the median separately, to match the actual median and limits to be used for the variable as illustrated in Figure 4.9.

Thus, if x’ ≤ 1, the value of the parameter returned is given by $x = x_L + x'(\mu - x_L)$

And if x’ > 1, $x = \mu + (x'-1)(x_H - \mu)$

**Step 3.** The same process is carried out separately for each parameter, and then repeated as many times as desired, meaning that each random coordinate is used only one time on one variable.

The results of a sample process are shown in Figure 4.11.

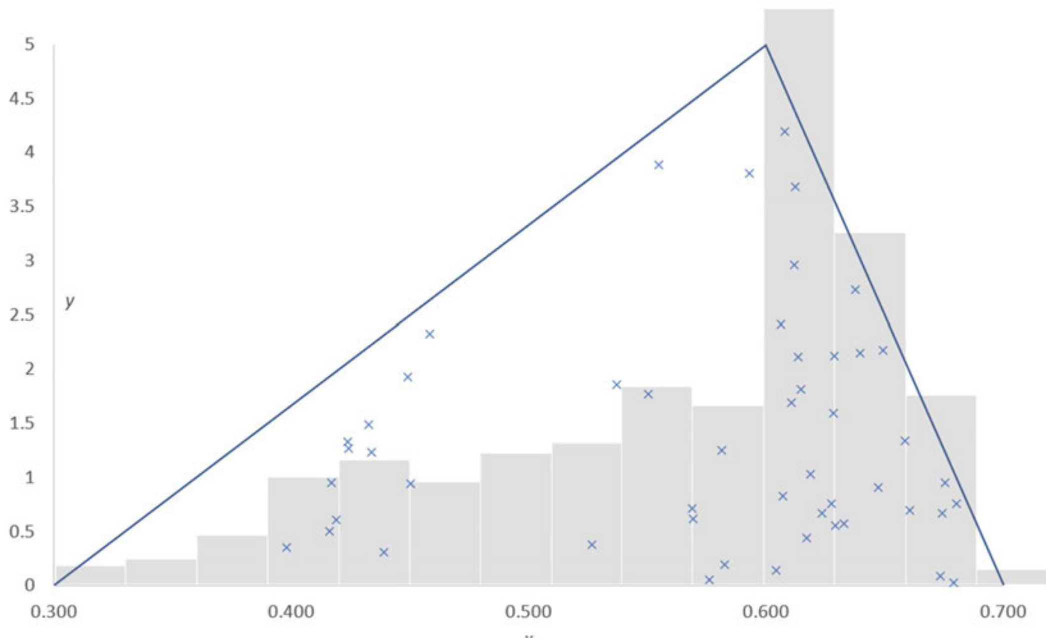



*Figure 4.11. Results illustration for a hypothetical variable with a range spanning 0.3 to 0.7 and a median set to 0.6. The points show the location in cartesian space of 50 random selections (approximately half being either side of x=0.6). The lines forming a triangle of area 1 indicate the probability space, and the histogram shows the relative distribution of x-axis values resulting from 1000 iterations.*

## 4.5 Summary

This chapter has introduced the theory behind the BioCarp model for tracking carbon flows through the forest / wood product / waste management system. Default value ranges for the variables in the system are shown in Table 4-8, cross-referenced to the relevant discussion in Chapter 3 or the applicable source. The use of BioCarp outputs for calculation of temporal impacts has also been outlined.

Chapters 5 and 6 present and discuss model outputs. In Chapter 5, the focus is on waste management, so the forest carbon and area module is not needed. Chapter 6, however, tackles the question of linking the demand for new timber products to the forest carbon issues, and so uses all of the BioCarp modules.

# 5. Results: end-of-life management

**growing the HWP pool and reducing climate impact**

*Scenarios set-up and results from BioCarp model regarding the carbon storage and climate forcing effects of UK timber waste management options.*

## 5.1 Research question and system boundary

As well as addressing questions related to an increase in demand for construction timber (Chapter 6), BioCarp enables investigation of a narrower – but still important – question relating to the role of management within existing timber consumption patterns. Given current construction timber usage and disposal patterns in the UK, what role can EoL management play in increasing the HWP pool and mitigating climate change? Specifically, *how much greenhouse gas emissions can be avoided and what associated climate benefit can be achieved by making immediate changes to timber waste management priorities in the UK?* Promoting demand for structural timber for environmental gain requires consideration of forestry and land use; but from the waste managers' perspective, this is not the case. The role of waste managers (in a broad sense – including government) is to identify the most suitable methods for treating waste and to develop the supporting infrastructure that is needed.

The purpose of this investigation is to explore the benefits of life extension strategies and recycling, the trade-offs between landfill and energy recovery, and the potential role of BECCS in the growth of the biogenic carbon pool. This also provides useful context for the following investigation of demand increase, as management of the ensuing end-of-life timber is one facet of that question.

The initial question is whether the current quantity of approximately 4.5 Mt/yr of timber reaching end of life in the UK **(Wood Recyclers Association, 2020)** could be managed differently in order to develop the HWP pool and mitigate climate change. The changes modelled are principally alterations to the market share taken by each of the primary treatment options (energy recovery, recycling, and disposal to land). The enhanced recycling scenario, however, includes a variation in which industry successfully embeds circular economy principles such as design for durability, repairability, disassembly and reuse, thereby extending product cycles and increasing HWP carbon pools. And the enhanced energy recovery scenario includes a variation with a greater role for CCS.

The question is first considered in relation to a single pulse of 4.5 Mt of end-of-life timber processed in 2021. The analysis is repeated for a hypothetical pulse of the same size in 2050, in order to consider the potential impact of developments in CCS and in the carbon intensity of the electricity network.

In any comparisons of management options made in this investigation, the quantity and quality of virgin structural timber demanded in any given year is assumed to be the same for each option. This means that forestry and forest carbon are unaffected by the choice, and are therefore outside of the system boundary[54]. For the same reason, material substitution is not considered. Energy substitution, however, is quantified as this is a potentially significant differentiator between waste management options.

The system boundary is illustrated in Figure 5.1. Carbon is imported into the system in the form of end-of-life wood product, and for the purposes of the model is regarded as 'in-use HWP' while it still has the potential to be used in any way, i.e. until it has been burned or buried. Any combined losses from the terrestrial carbon pools within the system translate to an increase in atmospheric carbon (in the form of $CO_2$ mainly, but also $CH_4$). This additional atmospheric carbon exchanges freely with receptors outside the system, such as oceans and forests, with different dynamics associated with each gas. The lifetimes of each gas in the atmosphere and associated climate change impacts are modelled with the Temporal Climate Impacts calculator **(Cooper, 2020)**.

In theory, the carbon emissions associated with transporting the wood to its next destination in the system (such as landfill site or board or pellet manufacturing facility) could be included, but such emissions are relatively low, and – in terms of the differences between the various options – likely to be negligible.

## 5.2 Baseline

For the end-of-life question, the Wood Recycling Association figures **(Wood Recyclers Association, 2020)** presented in Chapter 3 are used to represent the current situation in the UK, and therefore the baseline scenario for the subsequent analysis.

Although it can be surmised that only up to around half of the 4.5 Mt of material is likely to originate from construction and demolition **(Defra, 2012; Pöyry, 2009)**, as this question is being approached from the waste managers' perspective, it is equally instructive to consider the fate of the entire quantity. Readers who want to consider the construction industry's share can scale the results down accordingly: reducing all values in the results by about 55%. Whilst it can be inferred that the bulk of the 1.32 Mt of timber recycled as boards and bedding is 'grade A' quality timber – uncontaminated pre-consumer sawn wood

[54] This involves disregarding some potential feedbacks, for instance buildings designed for durability and reusability could in theory reduce the quantity of virgin structural timber used in future decades. However, it is also possible that such an approach simply increases the total quantity of timber materials used.

offcuts – there is more uncertainty around the constituents of the other categories (for instance the quality grade, and the mix of sawn wood and different board types).

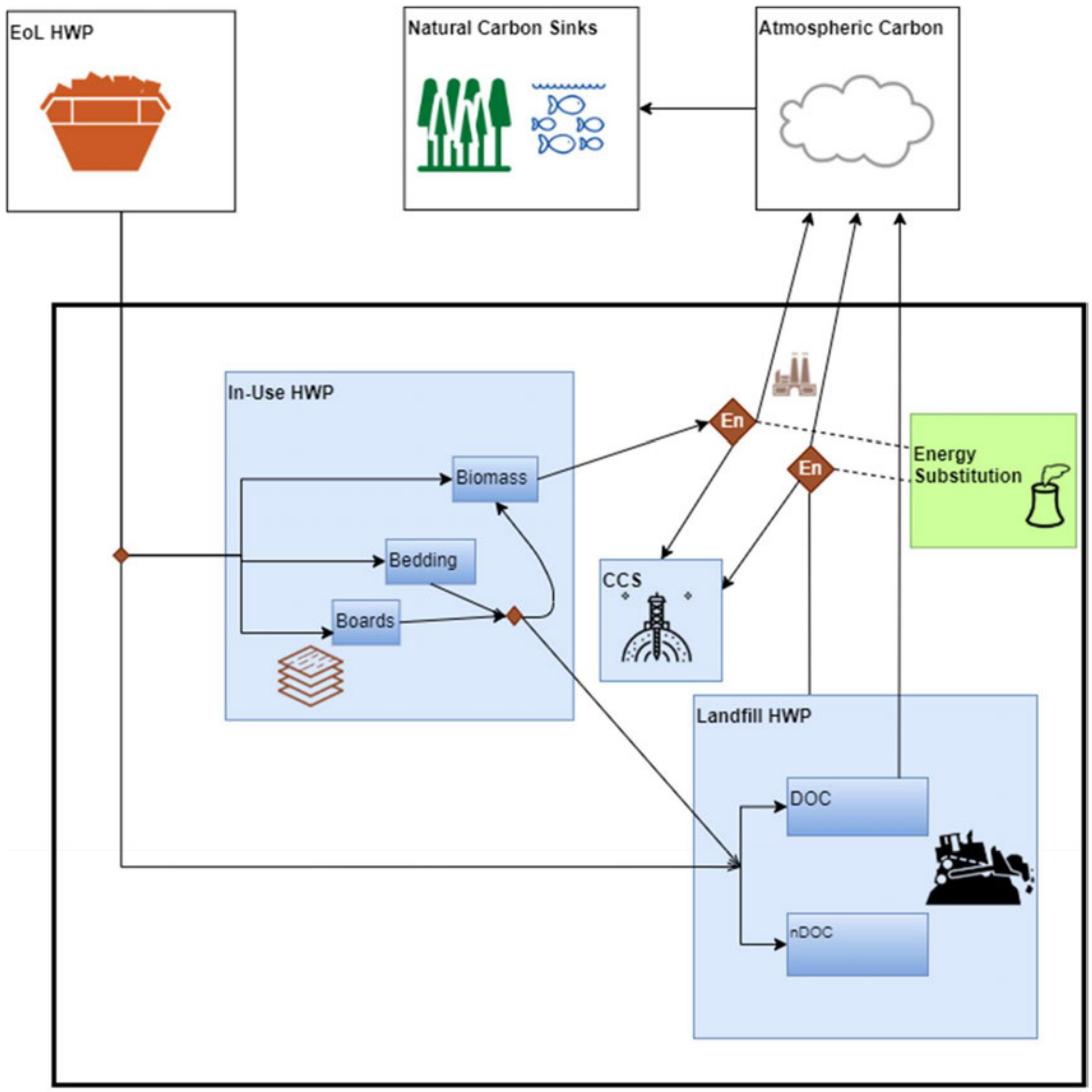
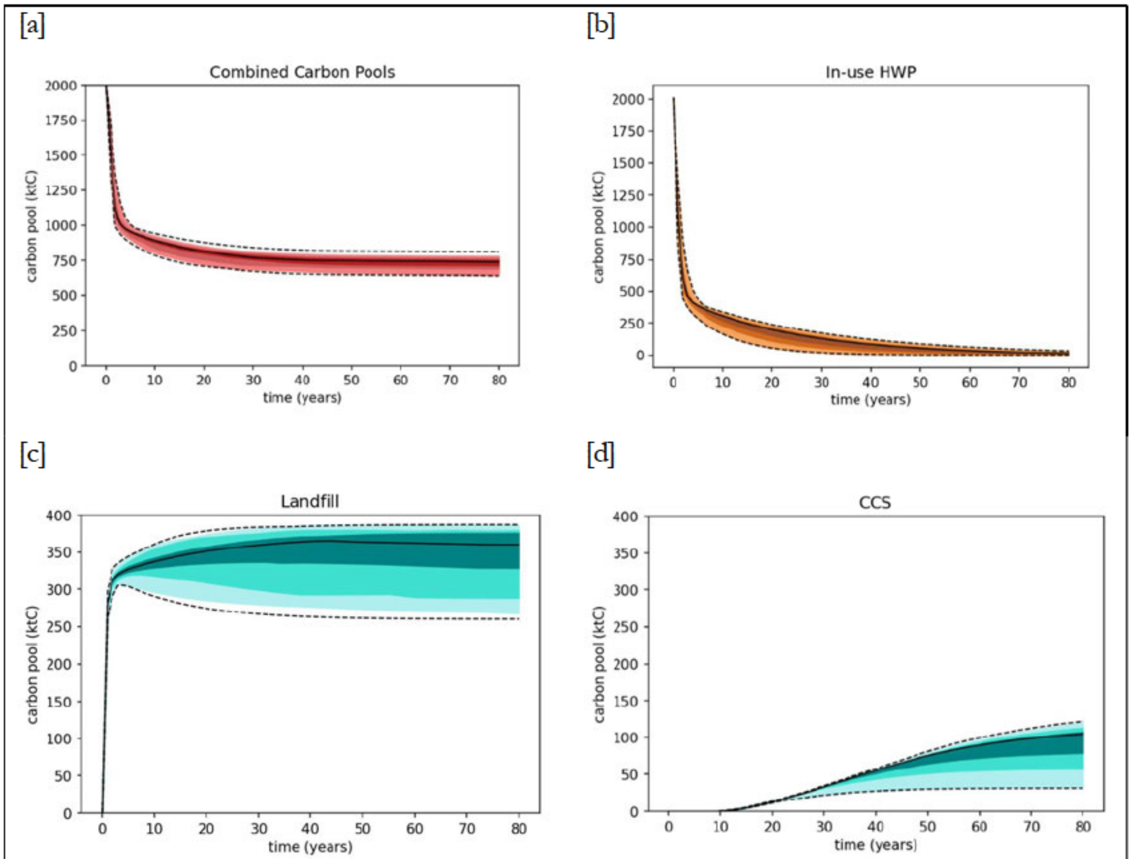



*Figure 5.1. System boundary for wood-carbon end-of-life assessment in BioCarp. The BioCarp system boundary is indicated by the heavy border; the effects and decay of atmospheric carbon (including methane as well as carbon dioxide) is calculated separately.*

The results for the development of carbon pools related to the baseline scenario are shown in Figure 5.2. In the ensuing discussion, where explicit reference is not made, it is the median curve from each figure that is under review.



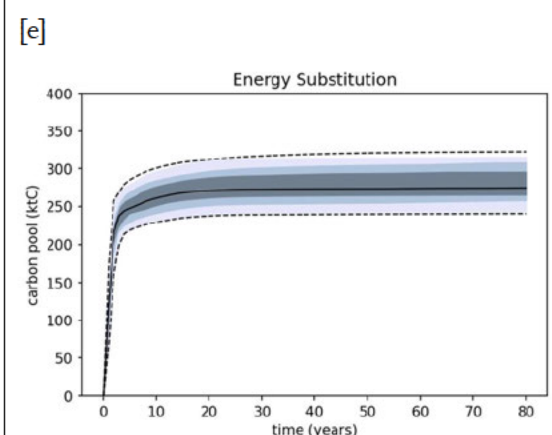




*Figure 5.2. Baseline showing the change in carbon pools over time, following the processing of 4.5 Mt of wood waste in 2021 (year 1). [a] All pools combined, [b] In-use HWP, [c] landfill, [d] CCS and [e] energy substitution. Key to shading: solid line = median; darkest colour = central 50%; +mid-colour = central 80%; +pale colour = central 95%; and the unshaded area between this and the dotted lines represents the highest and lowest 2.5%. Note that two different scales are used on the y-axis in this set of plots.*

Because of the modest baseline recycling rate, there is an immediate steep drop in carbon stored in in-use HWP (Figure 5.2[b]), from the initial level of ~2 Mt of carbon, with the fraction remaining falling below 5% after 36 years according to the median curve (range: 15-45 years, driven mainly by the board half-life). The landfill carbon pool (Figure 5.2[c]) consists of an immediate pulse to landfill in year 1, followed by a more gentle positive gradient (reaching 90% of the maximum value after 5 years, with the maximum reached after 43 years) as recycled products are retired. There is, however, a wide range of possibilities indicated: in the worst case, landfill carbon peaks after only three years, after which point additions to landfill are outrun by decay (on account of high DOCf and high decay rate values); and in the best cases, landfill carbon is still increasing at the end of the simulation (because of low values used for those two variables. As with landfill, the energy substitution pool (Figure 5.2[e]) is taken to a high level in year one by the timber used as fuel immediately: subsequently it increases further as previously recycled timber returns to the waste management system, but this slow increase is mitigated by the declining displacement factor, meaning that 95% of the pool is in place after ten years. The differences between the curves are mainly a function of electrical generation efficiency, although other factors such as the timing of emissions and LFG production rates play a part. With regard to CCS, whilst the size of the pool is small compared to the others (with the exception of in-use HWP after the first 20 years or so), the variability is wide. This is a function of the timing of biomass combustion: the low values in the distribution are associated with short half-life values picked for recycled board products, and this brings forward combustion to a time when CCS has a negligible market share.

The combined effect (Figure 5.2 [a]) is that the initial carbon stored in the waste is reduced sharply, by ~50% in four years, before settling at a level of about 37% of its initial value (and a third of this attributable to energy substitution). The balance (~75%) is emitted to the atmosphere (approximately 99% of this through combustion of biomass, with the rest arising from landfill gas emission and combustion).

## 5.3 Scenario Results & Discussion

The scenarios explored are shown in Table 5-1. These are designed to test the potential for improved outcomes if policy and market developments combine to shift the balance between the different waste treatment options. The objective is to explore the upper limit of the potential for each scenario by assuming an instant, rather than a phased, shift to the new waste treatment regime.

The baseline scenario uses current UK statistics, as discussed above **(Wood Recyclers Association, 2020)**, and the alternative scenarios are based on the hypothetical situations listed below. Results are presented in terms of the difference between each scenario and the baseline, and therefore report the potential gains (or losses) from the changes in policy and market conditions applicable to each scenario.

- Sc-Bas – the baseline scenario.
- Sc-En – the energy scenario. Stronger policy and financial support for energy recovery results in the capture of half of the share previously taken by each of the other options. Demand still exists for the recycling options, so the market is

reduced rather than eliminated; and it is assumed that disposal to land is not eliminated completely even if this is the ultimate objective. A variation of this scenario (Sc-En-CCS) is explored in the sensitivity analysis. This considers the possibility that in future, emissions from energy recovery systems will be captured and stored by default.

- Sc-Rec – recycling scenario. In this case, the hypothesis is that board manufacturing processes become able to tolerate a higher proportion of post-consumer wood (e.g. from demolition materials), for instance through innovation in the recovery and recycling process, or in the end products. The share taken by animal bedding is unchanged, as it is assumed that the need for Grade A waste wood remains, and the shares taken by biomass and landfill are reduced by the same proportion as each other. A variation of this scenario (Sc-Rec+) is explored in the sensitivity analysis. This considers the impact that circular economy initiatives might have on the median case: the analysis assumes that CE extends the half-life of board products by 10%.
- Sc-Lf – landfill scenario. In this case, reverses in support for biomass energy are coupled with a realisation that – given its carbon storage potential – the presumption against landfill by policy, taxes and incentives may have over-extended. Half of the material currently destined for biomass is instead sent to landfill or landfill-like facilities; other streams are unaffected. This runs counter to current trends, and is perhaps the most unlikely scenario to be realised, but is a useful case to study, as it is easily achievable technically.

| | **Sc-Bas** | **Sc-En & Sc-En-CCS** | **Sc-Rec & Sc-Rec+** | **Sc-Lf** |
|---|---|---|---|---|
| **Panel Board** | 23.1% | 11.6% | 46.2% | 23.1% |
| **Animal bedding etc.** | 7.4% | 3.7% | 7.4% | 7.4% |
| **Biomass** | 57.4% | 78.7% | 38.3% | 28.7% |
| **Landfill** | 12.1% | 6.1% | 8.1% | 40.8% |
| **Total** | 100% | 100% | 100% | 100% |

*Table 5-1. EoL scenarios: baseline, energy, recycling and landfill.*

### 5.3.1 Sc-En – Energy Scenario

The difference between this scenario and the baseline scenario is shown in Figure 5.3. The curves – in each case – are the result of subtracting the results for the baseline scenario from the results for the energy scenario. For each sampling of the probability density functions of the relevant variables (as described in Chapter 4), new results curves are generated both for the energy scenario and for the baseline scenario, with the latter being subtracted, and the process repeated 50 times. Values above the x-axis indicate that the scenario results in an increase in the terrestrial carbon pool, compared to the baseline, and a corresponding climate change mitigation benefit.

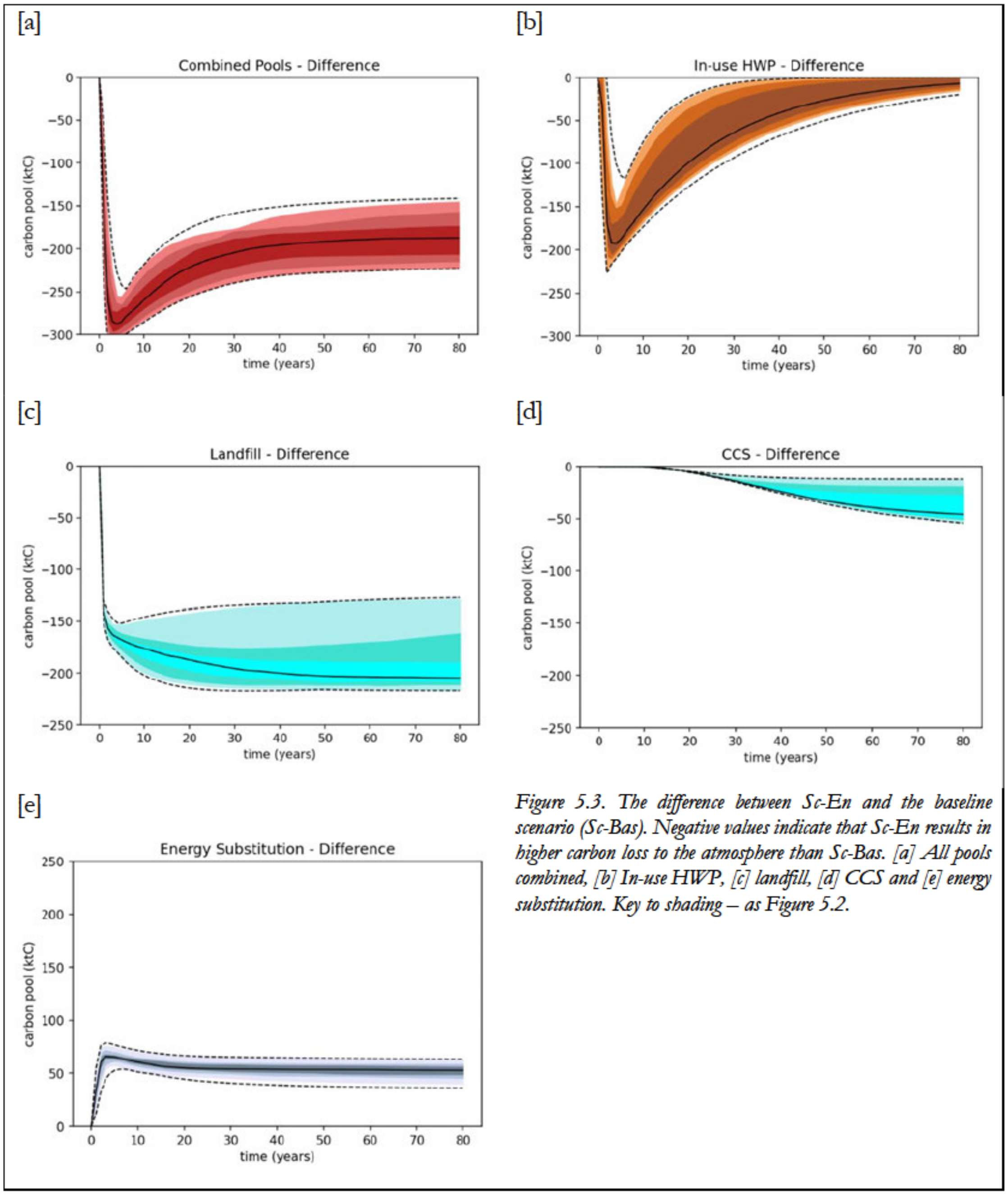



*Figure 5.3. The difference between Sc-En and the baseline scenario (Sc-Bas). Negative values indicate that Sc-En results in higher carbon loss to the atmosphere than Sc-Bas. [a] All pools combined, [b] In-use HWP, [c] landfill, [d] CCS and [e] energy substitution. Key to shading – as Figure 5.2.*

Sc-En involves reductions in the in-use HWP pool[55] because of the reduced emphasis on recycling. These reductions are not offset by the gain in energy substitution. The scenario involves a sharp rise in the quantity of wood burned in year 1, compared to the baseline

[55] The initial increase over the baseline observed in some cases for in-use HWP is for one year only, and is a result of biomass stock existing for a variable time period before combustion.

scenario, but in subsequent years there is less wood in the system to burn, so the cumulative energy substitution gain over the baseline declines slightly. The CCS pool is reduced compared to the baseline, despite the significantly greater quantity of biomass burned in Sc-En. This apparently paradoxical result is resolved by the fact that Sc-En brings forward biomass burning to the first few years of the simulation, when CCS is absent, leaving much less available material as the CCS market develops.

The curves for the combined pools in this scenario show that while there is a degree of recovery from the initial carbon loss, a substantial deficit remains at the end of the simulation, amounting to about 9% of the carbon in the original quantity of wood waste. In the long term the surviving in-use HWP pool is negligible in all scenarios, which explains – in this case – why the initial steep carbon loss associated with the extra wood burned early on is much reduced by the end.

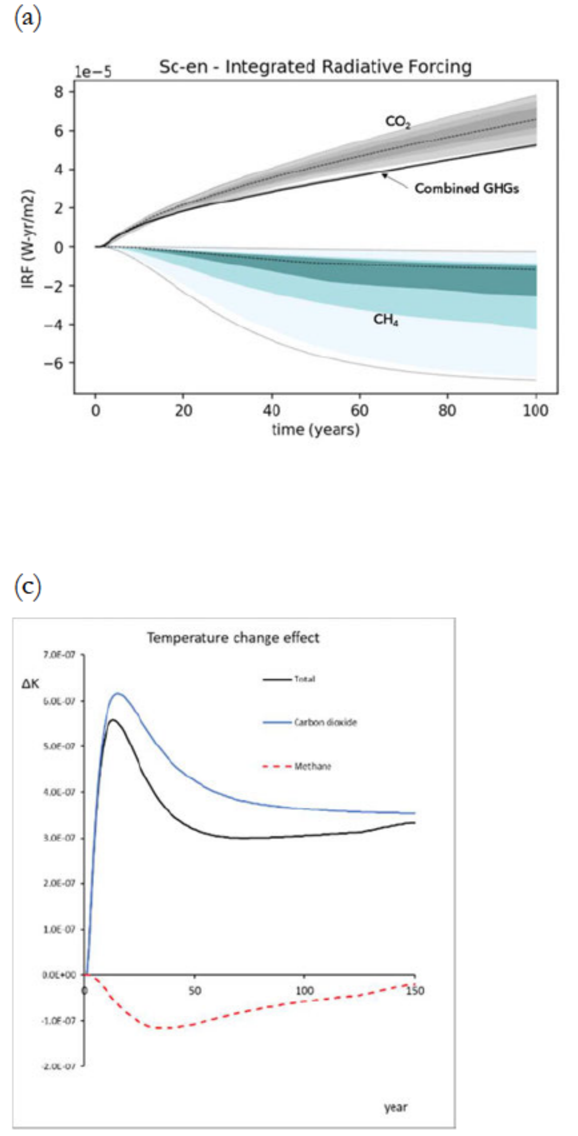
At the end of the simulation, the total is mainly determined by the balance between increased energy substitution and reduced landfill storage, with the landfill storage proving to be the more significant factor. This balance could be tipped in favour of the energy scenario if CCS is more aggressively employed, and especially if CCS proves to be a more secure carbon store than landfill.[56] This is explored through Sc-En-CCS.

Using the Temporal Climate Impacts calculator (Cooper, 2020), analysis of the effect of median curve for all pools on quantities of carbon in the atmosphere (treating $CO_2$ and $CH_4$ separately), and ultimately the impact on climate change produces the curves in Figure 5.4.

These show that whilst there is a strong recovery from the initial steep increase in atmospheric $CO_2$,[57] the effect on increased global temperature levels out at approximately 1.5E-07 K, with the reduced methane emissions arising from lower landfill utilisation mitigating the overall impact somewhat. The change to the methane content in the atmosphere peaks at -930 tonnes after 24 years: this is at least two orders of magnitude below the change to the carbon dioxide content, but because of methane's greater GWP, it is enough to have a significant impact.

[56] This seems realistic over the longer term, given optimal conditions, but landfill does have the advantage that it is possible to transfer 100% of a consignment of timber to a landfill without any carbon loss, whereas this will not be the case regarding the capture and transfer of gaseous $CO_2$.
[57] The recovery is partly because of the natural transfer to pools outside the system, but also because the sharp increase in wood burned in the early years means that there is less available in subsequent years, so the flux of carbon from in-use HWP to the atmosphere is below baseline.

*Figure 5.4. Temporal climate impacts – Sc-En (difference from baseline).*
*(a) integrated radiative forcing*
*(b) mass of $CO_2$ and $CH_4$ in atmosphere: median values*
*(c) temperature change effect: median values.*




## 5.3.2 Sc-Rec – Recycling Scenario

The recycling scenario (without the life-extension variation discussed in sensitivity analysis) results in a significant short to medium-term gain associated mainly with the increased retention of in-use HWP (Figure 5.5). Set against the gains in the in-use HWP pool are reductions in the landfill and energy substitution pools: the landfill pool curves mirror the in-use pool, but with lower absolute values; the energy substitution pool is also below the axis but never fully recovering on account of biomass burning being delayed to a period when the substitution benefits are lower. The overall effect is that, for the combined pools, it is only CCS that keeps the results above the axis after year 50.

The benefits of the additional temporary storage in the in-use pool clearly feeds through into the climate impact results shown in Table 5-2. In terms of impact, the strategy results in an initially beneficial temperature change effect (turning point of -5.0E-07 K in year twelve), which declines to about a quarter of its peak by the end of the simulation.

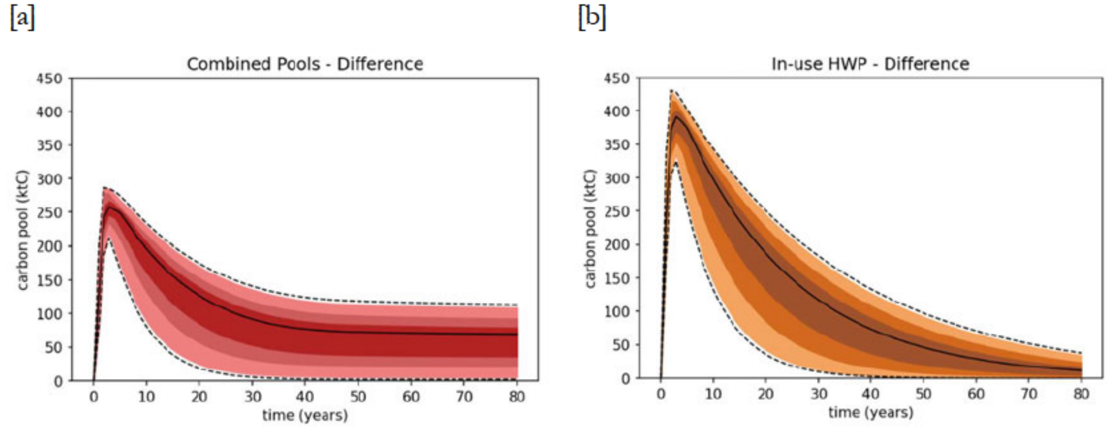



*Figure 5.5. Recycling scenario: Sc-Rec – overall improvement over the baseline scenario. [a] All pools combined, and [b] In-use HWP.*

### 5.3.3 Sc-Lf – landfill scenario

Results for the Sc-Lf scenario (Figure 5.6), to an extent, mirror or reverse those from Sc-En, in that feedstock is diverted from energy to landfill instead of from recycling and landfill to energy. The effect is that energy substitution benefits are diminished, but this loss is more than offset by the carbon stored in landfill. In this scenario, more than half of the original biogenic carbon is stored in landfill at the end of the simulation[58], approximately 310,000 tC more than in the baseline scenario.

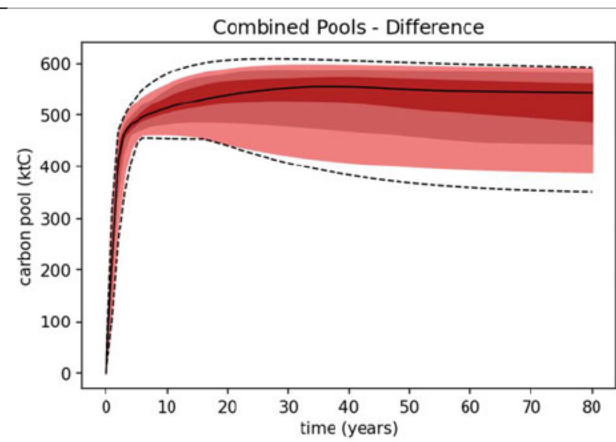



*Figure 5.6. Sc-Lf. Carbon pools in the landfill scenario: difference between that and the baseline scenario.*

[58] Although landfill only takes 40.8% of the initial waste (compared to 28.7% for energy), by the end of the simulation nearly all of the material has met one of these two fates, meaning that by this point 59% of the carbon has been placed in landfill, with 90% surviving with the remainder lost to landfill gas.

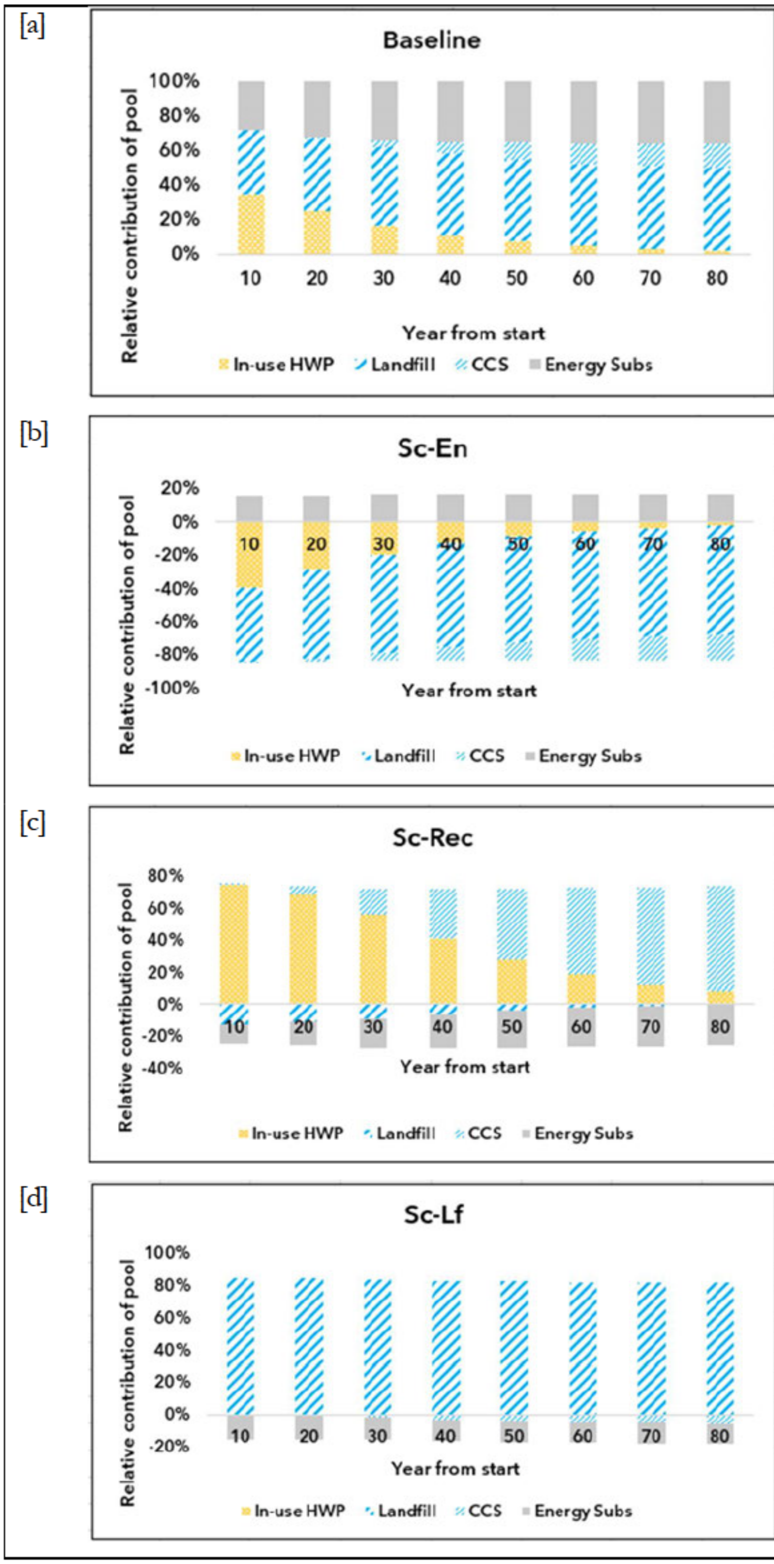



*Figure 5.7. Disaggregated results at 10-year intervals, showing the relative contributions from each pool – median values. [a] baseline, [b] Sc-En, [c] Sc-Rec, and [d] Sc-Lf.*

For each of the scenarios discussed above, the relative contributions of each pool are shown in Figure 5.7. For the baseline situation, after the recycled HWP has reached end of life, the dominant pools are landfill followed by energy substitution. For Sc-En, the benefit of increased energy substitution is more than offset by the loss in the other pools, with reductions in landfill pool becoming dominant over time. For Sc-Rec, the slight reduction in landfill and energy substitution benefits is more than compensated by the increase in the HWP pool (which plays a significant role to the end of the run) and ultimately the CCS pool that replaces it. Sc-Lf broadly mirrors Sc-En, except that in this scenario no change in recycling rates is assumed, which is reflected in the results.

Scenario results for selected years are shown in Table 5-2.

| t (yr) | | Baseline | Sc-En | Sc-Rec | Sc-Lf |
|---|---|---|---|---|---|
| 0 | All pools (tC) | 2.01E+06 | +0 | +0 | +0 |
| 10 | All pools (tC) | 8.93E+05 | -2.60E+05 | +1.96E+05 | +5.13E+05 |
| | Std. dev (tC) | 3.39+04 | 2.06E+04 | 4.25E+04 | 2.67E+04 |
| | IRF (Wyr/m$^2$) | 4.8E-05 | +1.2E-05 | -1.1E-05 | -1.8E-05 |
| | Temp gain (K) | 2.3E-06 | +5.4E-07 | -4.9E-07 | -8.4E-07 |
| 30 | All pools (tC) | 7.69E+05 | -2.04E+05 | +9.18E+04 | +5.52E+05 |
| | Std. dev (tC) | 3.71E+04 | 2.33E+04 | 3.87E+04 | 4.06E+04 |
| | IRF (Wyr/m$^2$) | 1.5E-04 | +2.5E-05 | -2.0E-05 | -4.5E-05 |
| | Temp gain (K) | 3.3E-06 | +4.1E-07 | -2.8E-07 | -8.5E-07 |
| 80 | All pools (tC) | 7.39E+05 | -1.87E+05 | +6.98E+04 | +5.43E+05 |
| | Std. dev (tC) | 3.76E+04 | 2.47E+04 | 3.19E+04 | 5.03E+04 |
| | IRF (Wyr/m$^2$) | 3.7E-04 | +4.6E-05 | -2.9E-05 | -1.1E-04 |
| | Temp gain (K) | 2.8E-06 | +3.0E-07 | -1.2E-07 | -8.6E-07 |

*Table 5-2. Scenario results for selected intervals representing short, medium and longer terms.[59] The results represent the difference between the median curve for the given scenario and the baseline. Cells shaded green show an improvement over the baseline, and buff shade show a worsening.*

### 5.3.4 Sensitivity Analysis

#### *Carbon Capture and Storage – Sc-Bas-CCS and Sc-En-CCS*

Whilst CCS does have an influence on the results presented above, this influence is limited by the (safe) assumption that the proportion of biomass burned in facilities equipped with CCS will take significant time to grow from its current level of around zero. Alongside this,

59 Longer term in the context of the simulation, as the pools near their asymptotes.

the combustion of biomass in all scenarios is skewed towards the early years, when CCS is absent. It is interesting, however, to explore the limits of the potential contribution from CCS if it could be 'switched on' for all combustion of end-of-life timber. This is modelled in the completely hypothetical scenario Sc-En-CCS. For this scenario, energy substitution benefits are ruled out of scope, on the grounds that in a world where CCS is the default, grid average electricity emissions are likely to be around zero.

In this case with respect to Sc-Bas-CCS (the baseline scenario adapted for 100% CCS application to biomass combustion), the Sc-En-CCS scenario is very little different (Table 5-3). Overall there is a loss of carbon from the combined pools, but the difference is very slight: this implies that if the carbon is as good as guaranteed to be stored underground at the end of life, then it is not critically important what happens to the wood in the meantime, and how long it is used for. On the other hand, the difference between Sc-Bas-CCS and Sc-Bas is significant, and after 80 years the difference represents more than half of the original carbon. At this point, 90% of the initial quantity of carbon is still stored: mainly CCS, but with a significant proportion in landfill and a negligible proportion remaining in in-use HWP. Thus, in a world where CCS is obligatory when wood is burned, the carbon account will be dramatically improved overall, but the balance between landfill, biomass combustion and recycling will be a matter of very minor concern.

#### *Circular Economy – Sc-Rec+*

The CE variant of this scenario, in which the half-life of the boards is extended by 20% (the uplift being applied to all three values defining the probability density function for the lifetime of the boards) shows a marked improvement in the overall carbon pool (Table 5-3), with an improvement over Sc-Rec (Table 5-2) already evident at year 10, and adding around 30% to the benefits achieved by Sc-Rec at years 30 and 80. That said, this still leaves Sc-Rec+ ranked behind Sc-Lf.

#### *Decarbonisation of the economy*

With the exception of the 'total CCS' sensitivity test, results presented above all use the NZ2050 decarbonisation scenario as a basis for the calculation. This plots a path for the decarbonisation of the grid and the phase-in of CCS. Taking the opposite perspective to total CCS, sensitivity testing has also included using a business as usual (BAU) energy scenario[60], where no further decarbonisation is assumed, and observing how this affects the energy substitution pool and, by extension, the combined pools.

Switching to BAU affects the result details, but not in such a way as to alter the discussion and overall interpretation. For example, under BAU, the improvement in the energy substitution pool (Sc-En-BAU, compared to the baseline scenario with BAU carbon intensity) is around 20% less than under NZ2050, even though the absolute values for energy substitution are higher under BAU. This is because one of the most important effects of Sc-En is to bring forward emissions that would have happened later in Sc-Bas: this is less salient in the BAU decarbonisation scenario than for NZ2050.

### *Best of all worlds*

Leaving aside 'total CCS', comfortably the most successful of the identified scenarios is Sc-Lf, followed by Sc-Rec+. These landfill and recycling/CE strategies might be combined to further improve the result. This is a scenario in which recycling rates and product lifetimes are enhanced (as in Sc-Rec+), and biomass energy is eliminated as an option in favour of landfill. Results for this scenario (Sc-Ren+Lf+) are shown in Table 5-3, with all values favourable in comparison to those in Table 5-2.

| t (yr) | All pools | Sc-En-CCS (vs Sc-Bas) | Sc-En-CCS (vs Sc-Bas-CCS) | Sc-Rec+ (vs Sc-Bas) | Sc-Rec+Lf+ (vs Sc-Bas) |
|---|---|---|---|---|---|
| 10 | carbon (tC) | +9.44E+05 | -2.95E+04 | +2.17E+05 | +1.01E+06 |
| | IRF (Wyr/m²) | -3.9E-05 | +9.5E-07 | -1.1E-05 | -3.8E-05 |
| 30 | carbon (tC) | +1.04E+06 | -1.97E+04 | +1.20E+05 | +1.09E+06 |
| | IRF (Wyr/m²) | -1.2E-04 | -2.5E-07 | -2.4E-05 | -9.7E-05 |
| 80 | carbon (tC) | +1.07E+06 | -1.00E+04 | +9.06E+04 | +1.08E+06 |
| | IRF (Wyr/m²) | -3.0E-04 | -5.1E-06 | -3.6E-05 | -2.2E-04 |

*Table 5-3. Scenario results for sensitivity test scenarios. Values represent the difference between the scenario indicated and the version of the baseline shown in parentheses. Cells shaded green show an improvement over the baseline, and buff shade show a worsening.*

Adding the baseline results back to these results shows that the final carbon pool represents 91% of the original wood carbon, or 89% if energy substitution is disregarded.[61] This scenario is very slightly more successful than the 'total CCS' option, a conclusion that

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[60] The assumption is crude, but even as the grid is decarbonised, there is a case for using a higher than average emission factor to represent the marginal generator, even though the marginal generator is very difficult to define, as it varies with circumstances (demand, weather, availability, investment, etc.).

[61] Energy substitution is relatively low in this scenario because it is limited to landfill gas alone.

still holds when methane emissions and temporal impacts are also considered. The difference is, however, marginal.

## 5.4 30-year simulation

Results presented above all relate to the journey taken by the carbon stored in wood waste from a single year. It is also instructive to repeat the analysis for wood waste produced at the same rate (in the absence of reliable forecasts about future waste production rates) for an extended period. In this section, this is done over a 30-year period up to ~2050. During this time the grid emission factor declines to zero whilst the market penetration of CCS rises linearly from zero in 2030 to 100% in 2070.

Under the baseline scenario, the combined pools are as shown in Figure 5.8. This shows the pool eventually settling at around 27 MtC, which represents a loss of 33 MtC from the wood waste.[62]

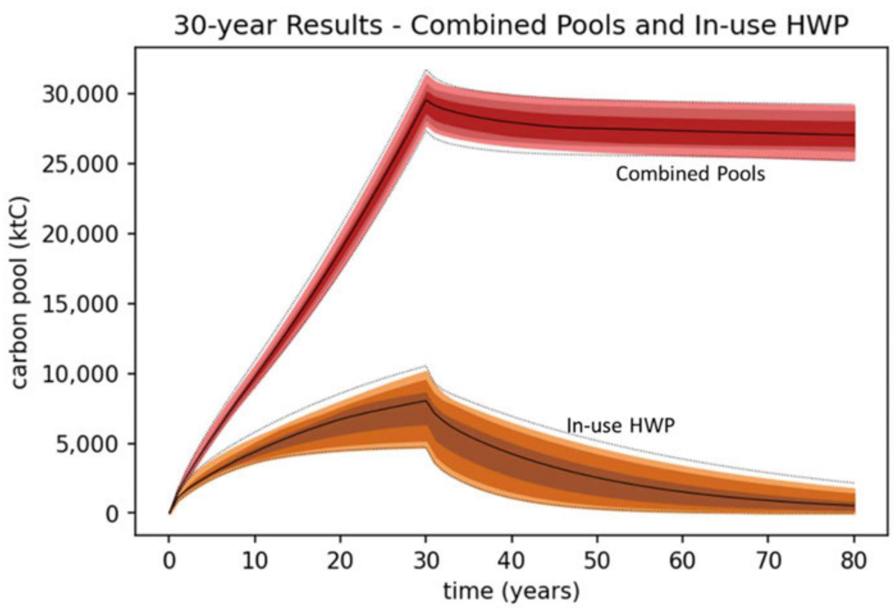



*Figure 5.8. 30-years of wood waste simulation – baseline scenario, showing the combined pools and the contribution to the total made by in-use HWP.*

As shown above, a push towards using more of the timber as fuel will make the situation poorer, so is not investigated further here.

Running the landfill scenario (Sc-Lf) – the most successful of the mainstream scenarios – for thirty years, the results are as shown in Figure 5.9, yielding a gain of 13.9 MtC relative to the baseline: or an average of 0.46 MtC for every year of waste treated in the scenario (or

[62] Depending on perspective, this is a 27 MtC gain compared to the worst possible case (burning all the wood without energy recovery) or a 33 MtC loss compared to securing the wood in a hermetically sealed bunker.

15.2 MtC excluding energy substitution). The UK's carbon budget for the middle of this period – the sixth carbon budget, 2033-37 – is equivalent to 52.6 MtC/yr:[63] this analysis shows that curtailing the use of waste wood as biomass, in line with Sc-Lf, would deliver emission reductions worth 0.9% of this budget. This would be a significant contribution: until fossil fuel extraction is banned and obsolete, climate change mitigation can only come about through the accumulation of a comprehensive set of marginal gains in diverse areas. Furthermore, this particular gain is only one of the contributions that the wood products and related industries has to offer, as will be shown in the next chapter.

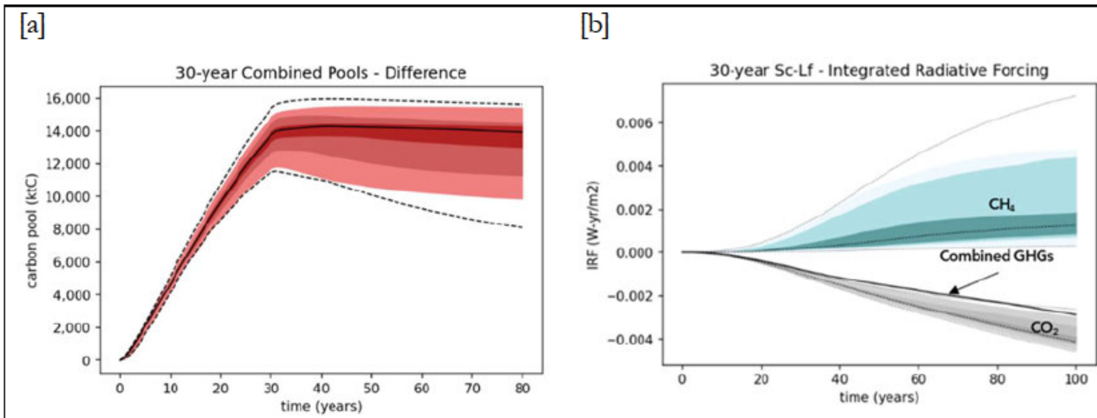



*Figure 5.9. Sc-Lf results: improvement over baseline (all pools combined) when run for 30 years. [a] carbon pools, and [b] integrated radiative forcing.*

## 5.5 Conclusions

There is a strong climate case against the current default position for the management of timber waste – energy recovery without CCS, as scenarios involving reductions in this share all reveal increases in the terrestrial carbon pool. For instance, the most technically straightforward option of reverting significant quantities of wood waste to landfill increases carbon pools by around 0.5 MtC for each one-year pulse of waste; and if this strategy is run for 30 years, the gain amounts to 14 MtC by 2050; and furthermore, if a more aggressive strategy is followed – involving the effective elimination of combustion emissions, complemented by improvements in circular economy aspects of the system – then the annual benefits exceeds 1 MtC. Although not evaluated here, the argument is also applicable to other stable high-carbon wastes that are incinerated in significant volumes: plastics, for example, whether derived from biomass or from crude oil. This information may not be what environmentalists and policy makers would wish to hear, after so many years of bad publicity (much of it well-deserved) and an adverse policy environment for

[63] The budget is 965 $MtCO_2e$ / 5 years (Climate Change Committee, 2020).

landfill. However, the results in this thesis show that society collectively should be contemplating a new class of landfill designed and managed to take specific categories of waste and to keep the carbon content of that waste in the ground for as long as is feasible.

On the other hand, the argument between landfilling wood or recovering energy with CCS is very finely balanced and – in the absence of more precise data about carbon capture and retention in CCS and degradation rates in landfill – a choice can be made on other criteria such as cost, technological readiness, and environmental impacts more generally.

If a new form of landfill is eventually considered as a carbon storage solution, then investigations into the optimal conditions to slow and limit the decay of wood need to be investigated, and appropriate standards set for design and management of such sites. The elimination of methane emissions to the atmosphere should be the primary objective, as the model shows that such emissions are a more significant factor than the loss of carbon per se. This would require focus on decelerating the decomposition of carbon-based materials, whilst also improving capture rates for the LFG that is still generated.

The next chapter investigates the carbon account for construction timber from a different perspective: increasing demand for new timber products. The management of the waste generated in future – again – is involved in the assessment, but this time the supply of the timber (i.e. from forestry to sawmill to building site) is added into the balance.

# 6. Results: demand growth and timber supply modelling carbon pools and climate change

*Scenarios set-up and results from BioCarp model regarding the carbon storage and climate forcing effects of a UK construction timber demand increase and associated forestry scenarios.*

## 6.1 Research question and system boundary

This chapter explores the change in carbon pools and the associated climate change impacts that would be associated with an increase in demand for virgin structural timber (VST) above a baseline. It answers the key research question of this thesis, i.e. *how much greenhouse gas emissions can be avoided and what associated climate benefit can be achieved over time periods of up to 100 years by coupling an ambitious but realistic increase in construction timber usage in the UK (preferably supplied from domestic forestry) with an afforestation agenda designed to meet future demand?* The case of UK new construction is taken as the context for the question and the investigation, but the model can be applied to other sectors and geographies, and with appropriate consideration of the context, the general conclusions presented here can be applied more broadly. As important as the quantification of climate benefits, the determination of whether any such benefit exists is key, and – if so – how long it takes to be realised. The system boundaries are as discussed in Chapter 4 and illustrated in figures 4.1-4.6 inclusive. The modelling is led by the above-baseline quantity of virgin structural timber demanded (shown as 'virgin wood' in the figure) in each year of the model. From this, the associated carbon moving from forest to structural timber and co-products is calculated, and so on.

As discussed in Chapter 3, data is scarce with regard to exactly where and how in the construction industry timber is used, meaning scenario development is needed. However, much can be learned from the investigation even of scenarios that will not be realised in practice, as results can be scaled as necessary. A key element of the research question is *'does the evidence support scaling up timber construction at all?'* and the mere fact of increasing timber use is likely to be more important in addressing this question than the exact quantities involved, at least until supply is constrained (e.g. if carrying out a wider regional assessment).

Section 3.2.3 concluded that a reasonable estimate for the quantity of VST used in all new UK construction is 500,000 tonnes per annum. Whilst the quantity used in repair and refurbishment of buildings is considerably higher, it is here assumed that the 500,000 tonnes is the quantity that can be influenced by changes in client and architectural preferences along with product innovation, policy environment and costs. The main scenario involves an annual increase of 3% in the demand for VST, and it is only the extra (i.e. marginal) demand (over and above the baseline 500,000 tonnes per annum) that is modelled: 15,000 tonnes in year 1, 30,450 tonnes in year 2, etc. This increase runs for 30 years and then levels off.

With respect to forest carbon, the source (in particular, the country of origin) of the additional timber must be stated and modelled. This is not so that emissions can be allocated to the relevant country, as this work is about the overall climate impact rather than the carbon accounts of a particular administration. Rather, it is because forests in different regions have their own characteristic growth rates, planting densities, management styles and other indicators of sustainability that feature in the ensuing discussion.

The base case used in this work is designed to support the notion that the UK can meet any increase in demand for construction timber from its own forest resources. Thus, the marginal demand is met from UK forestry, without displacing other users of timber. All harvested forest is promptly restocked, and an afforestation strategy is linked to the anticipated demand increase such that all demand after one rotation (40-50 years assumed) is met from forest associated with the strategy. This is referred to as the Proactive Anticipatory Demand Scenario (PADS) and uses the full scope of the modelling approach presented in Chapter 4, and the results are presented in depth in section 6.2.

Whilst PADS offers a somewhat idealised picture of how well-integrated cross-sectoral and governmental planning, policy and investment can possibly be, when presented alongside other scenarios it helps to define a realistic envelope for the results. In reality, it is unlikely that the marginal VST demand will be met entirely from one source, and the true situation may fall between PADS and the following scenarios, considered in section **6.3.3** in terms of their potential to influence the forest carbon pool.

- Scandinavian Import Scenario (SIS). In this case, VST is imported from a representative European supplier that already meets a significant part of the UK's demand, such as Finland or Sweden. In this case, there is assumed to be limited scope for afforestation in the supplier country due to lack of control over that country's policy, and thus no explicit link between forecast demand and afforestation. Furthermore, longer rotations and less dense forestry is assumed, in line with national forestry statistics.

- Reactive Unplanned Domestic Scenario (RUDS). The marginal demand is met from domestic resources, but – in contrast to PADS - no link is made between demand ramping and afforestation.

The motivations underpinning afforestation and harvest are central to the two domestic scenarios in particular, and are vital to the consideration of the impact of a general policy or societal shift towards greater timber use. For instance, with reference to PADS, a policy designed to promote VST usage without supporting supply will ultimately simply transfer carbon from forest to product and then atmosphere. Whereas if a convincing link is made to an afforestation strategy, then a case can made for significantly reducing the loss of forest carbon. This link to afforestation might be made directly – as modelled in PADS – or indirectly by having an existing afforestation strategy that references demand growth and has sufficient headroom to meet the demand growth being modelled.

## 6.2 Proactive Anticipatory Domestic Scenario Results

As stated above, this scenario is based on the increased demand for construction timber postulated here for the UK being met from domestic sources, coupled to a strong afforestation policy specifically designed to meet future demand. Aggregated results are presented first, followed by results for individual pools. Sensitivity to the parameters modelled with probability density functions (PDFs) is discussed with each set of results, as appropriate. But the implications of making alternative modelling choices, including (but not only) the alternative forest carbon scenarios (SIS and RUDS), are explored separately, in section 6.3.

### 6.2.1 Combined Pools

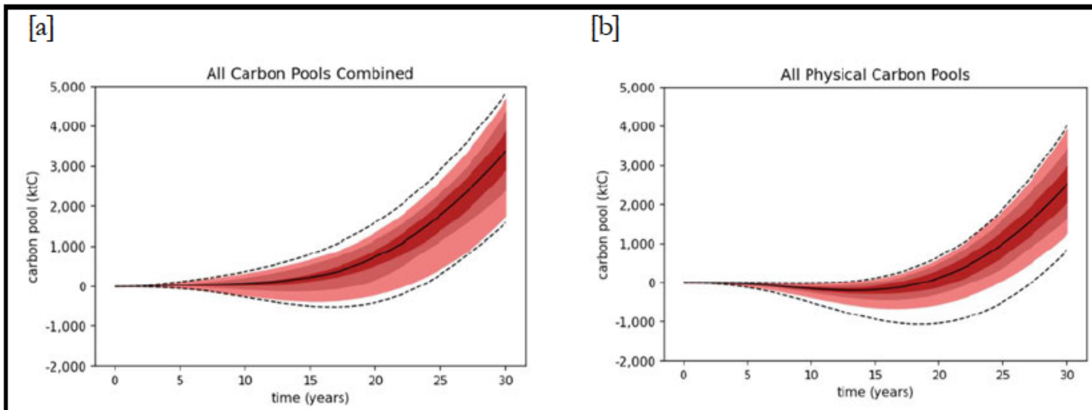
The combined results for all pools under the PADS scenario are presented in Figure 6.1, over 30-year and 100-year timescales, with and without substitution benefits.

Initial observations are as follows:

- In the long term, with even the most conservative selections from the PDFs for the variables, consistent development of the overall carbon pool is observed – climbing steadily after an initial period of around 20 years. After 100 years, the overall gain (median value) is 47 MtC. This is equivalent to approximately a third of the total net GHG emissions from the UK in a single year.
- In the short term, the situation is more nuanced, with little discernible development of the carbon pool in the first 20 years, reaching 3.4 MtC after 30 years. In the worst case, an overall carbon loss is shown over the first 15-20 years before

recovering. This is particularly pronounced if substitution is disregarded, when even the median curve is below the axis until year 18, with potential for that loss to temporarily reach 700 ktC.

Therefore, it is clear that the 'more timber' strategy can only deliver climate change mitigation in the medium to long term. In the short term, substitution effects need to be included in order for the strategy to be considered as a potential climate change mitigation measure.



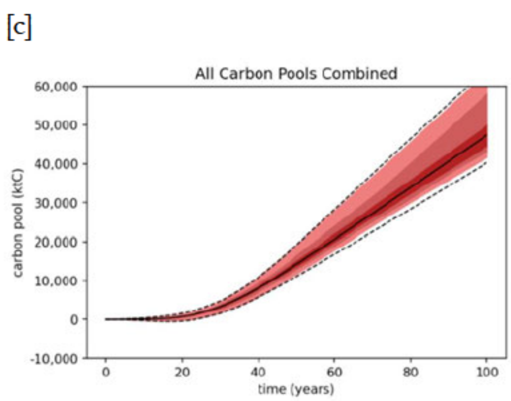




*Figure 6.1. Carbon storage in combined pools. PADS. [a] combined pools including 'virtual' substitution pool, 30 years; [b] as [a] but excluding substitution; [c] as [a] but over 100 years (note the change of scale for [c]). For these plots, and all subsequent plots with similar format, the central solid line is the median, the outer dashed lines are the limits, and the three shaded bands enclose the central 50%, 80% and 95% of values. Outlying values occupy the white space between the dashed lines and the palest colour.*

The impact of the development of the combined carbon pool, in terms of the integrated radiative forcing (IRF) associated with avoided $CO_2$ emissions is shown in Figure 6.2, along with the combined impact of methane and nitrous oxide emissions. This plot highlights a time lag between the development of the carbon pool (shown in Figure 6.1c) and the resulting climate impact, with relatively little achieved before about 40 years. The median contribution from $CH_4$ and $N_2O$ is relatively low (shifting the combined curve from the $CO_2$ curve only slightly), and – within that – the contribution from $N_2O$ is slight. However, the upper range of the emissions curves show that the potential exists for methane to have a strongly negative contribution to the climate mitigation effect from the carbon pool: this applies in cases where the value ascribed to degradable organic carbon in landfill (DOCf) is very high. A high DOCf might reflect the risk that landfills will be managed to promote or permit very high landfill gas emissions, or that the few references to high DOCf values for wood in the literature are, in fact, about right: both seem unlikely, but need monitoring.

Not perceptible at the scale shown in Figure 6.2 is the initial increase in overall IRF (above the axis) before the benefits start to accumulate. The median period before IRF turns negative is 12 years, but in the extreme case (associated with high DOCf again) this period is 45 years.

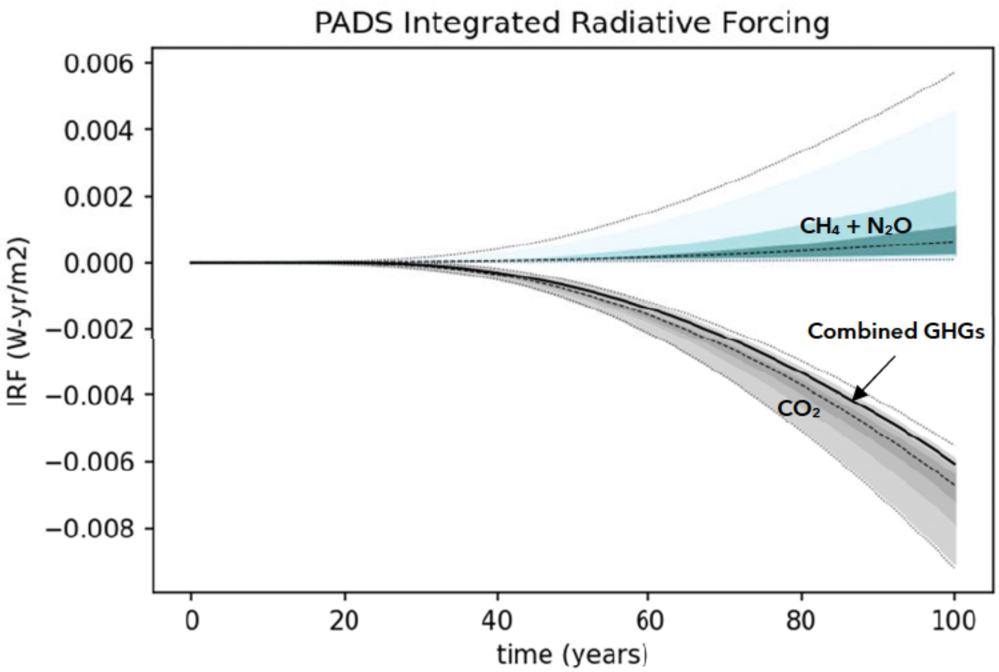



*Figure 6.2. Integrated radiative forcing (IRF) in PADS, showing the contribution of all carbon pools combined, as $CO_2$ emissions (and thus IRF) avoided (below the x-axis) and the contribution from combined non-$CO_2$ GHGs above the axis. The solid line ('combined GHGs') indicates the median line of the total for all three gases; all other lines and shaded bands report either the $CO_2$ or the non-$CO_2$ data.*

The relative contribution of each pool to the total, at intervals, is illustrated in Figure 6.3. This shows how the relative significance of the forest carbon loss is high to start with, but decreases over time. Substitution benefits are initially significant, but in the long term[64] make a near-negligible contribution relative to the other pools. Over time, the landfill and CCS pools become increasingly dominant, and it must be remembered that this outcome is dependent on 'unrecyclable' end-of-life HWP being sent down these paths, and not – as is

[64] In this chapter, the loose terminology of short, medium and long terms should be understood in the context of the 100 years of the model run. For instance, short-term being up to around 20 years, and long-term being over about 80 years.

currently the case – defaulting to combustion without CCS, which of course results in these pools being lost to the atmosphere in their entirety.

In the following subsections, the disaggregated results are dissected in more detail, and the sensitivities to key parameters are discussed.

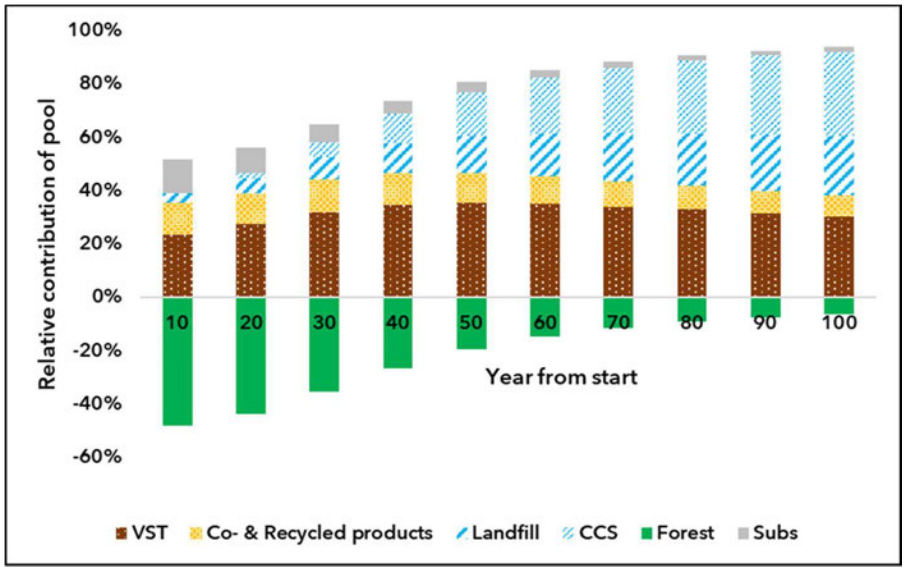



*Figure 6.3. Disaggregated results at 10-year intervals, PADS, showing the relative contributions from each pool – median values. Virgin structural timber, other in-use HWP (co- & recycled products), landfill, CCS, forest, and substitution pools.*

### 6.2.2 In-use HWP

The development of the in-use HWP pool is shown in Figure 6.4. These show that after 80 years (i.e. at the end of the century) ~14MtC in the VST survive from the original 21 MtC supplied into buildings.[65] The quantity stored in all in-use HWP (including panels, paper and fuel) is somewhat more, at ~18 MtC after 80 years, but from a much higher total input (which varies according to the forest to building sawn wood production efficiency, but the median is 42 MtC).

Interrogation of the individual curves contributing to each plot shows that the variation in the VST pool results is driven only by the decay constant (or half-life) of in-use VST. For the total in-use HWP pool, however, other variables are also influential. In particular:

- Low VST production efficiency values (represented by the variable 'Hwf' – harvested wood fraction) result in more carbon entering the system as co-product, and therefore high values for in-use HWP.[66]

[65] Remembering that VST is added all the way through the simulation, even when market penetration by timber has levelled off.

[66] Note that this parameter (VST production efficiency) affects all results involving co-products: everything but VST carbon. Accordingly, to inform the discussion about sensitivity to the various parameters, a 'shadow'

- But in at least one case, a value at the high end of the range for production efficiency is more than offset by contributions from – firstly – a high value for the fraction of co-product directed to the more durable product category (particleboard), and – secondly – a high value for the fraction of end-of-life VST recycled, and – thirdly – above average half-lives for VST and for particleboard. Set and combined in this way, these parameters slow down the circulation of carbon through the product systems, allowing the in-use HWP pool to build.

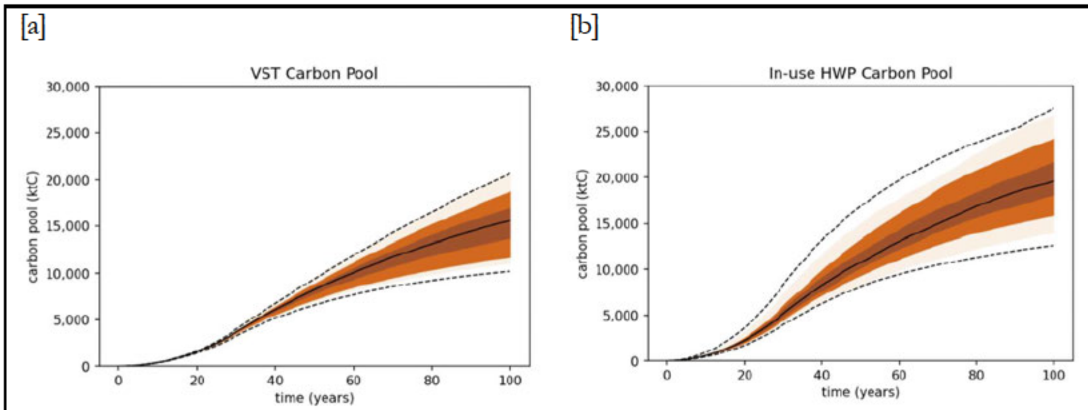



*Figure 6.4. The in-use HWP carbon pool, PADS. [a] virgin structural timber only; and [b] the entire pool of VST and co-products (boards, paper and fuel).*

### 6.2.3 Carbon stored below ground

In BioCarp, HWP carbon is stored below ground either when HWP is landfilled, or when it is burned for energy, with carbon capture and storage (CCS).

The landfill and CCS curves (**Figure 6.5**) have broadly similar characteristics, and scales. The CCS curve takes longer to escape the x-axis, because in the initial decade CCS is modelled as having zero penetration into the wood-burning energy market, before linearly rising to 100% in 2070. But in these early years, not so much of the wood carbon is reaching end of life, so – over the long term – the difference is not particularly marked.

Landfill and energy recovery with CCS are alternative fates for a given consignment of end-of-life wood, so parameters leading to high landfill carbon values tend to correspond to curves with low CCS carbon values. Interrogating the curves at the extremes of the total underground carbon distribution shows that no single variable has overwhelming influence: the extreme values tend to result from parameter values combining to exert influence in the same direction. For instance, whilst the best results tend to be weighted towards landfill

model run was completed with this parameter fixed (at 0.5), although the figures presented all relate to the model run with this parameter varying in the defined range.

(i.e. a high proportion of waste going to landfill, including from VST and from particleboard, coupled with a low degradable organic carbon component: DOCf), low values can also arise from landfill scenarios (with high DOCf) as well as energy from waste scenarios (with low CCS efficiency).

High decay constants for VST and for particleboard consistently bring forward the influx of material to landfill and CCS, therefore having a clear and significant influence on the size of the pool in a given year.

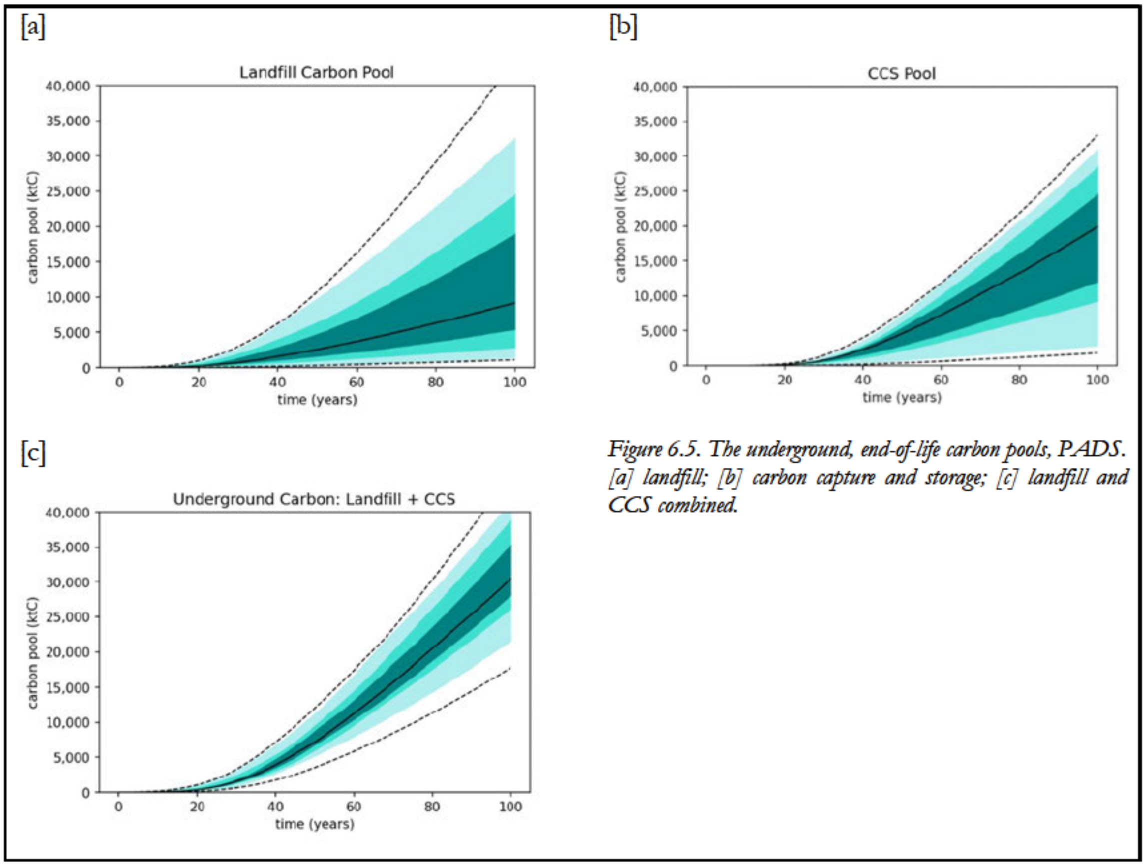



*Figure 6.5. The underground, end-of-life carbon pools, PADS. [a] landfill; [b] carbon capture and storage; [c] landfill and CCS combined.*

### 6.2.4 Carbon stored in the technosphere

Combining the in-use HWP carbon with the buried HWP carbon gives the total carbon stored in the technosphere, which is here deemed to exclude the forest (Figure 6.6). Over the longer term, the range of values in a given year is mostly connected to the production efficiency variable as – once CCS becomes established – most of the carbon that leaves the in-use pool is successfully migrated to one of the buried pools.

Over the medium term however, the range is more significant relative to the absolute value. For instance, at year 30 the carbon stored in the highest case is more than double that of the lowest. And even when Hwf is fixed (at 0.5), the range at year 30 is from 5.46 to 7.97

MtC (6.69 +/- 19%). This is because in the early years of the simulation, CCS is poorly established or absent altogether, so wood carbon directed to the energy recovery sector is lost – emitted to the atmosphere. Accordingly, the highest values are achieved especially in situations where considerably more waste is directed to landfill than to energy recovery; but the rate of removal of VST from buildings also plays a part (as the 'simplest' method of storing the carbon in the technosphere is to leave it in situ – suitably protected and maintained).

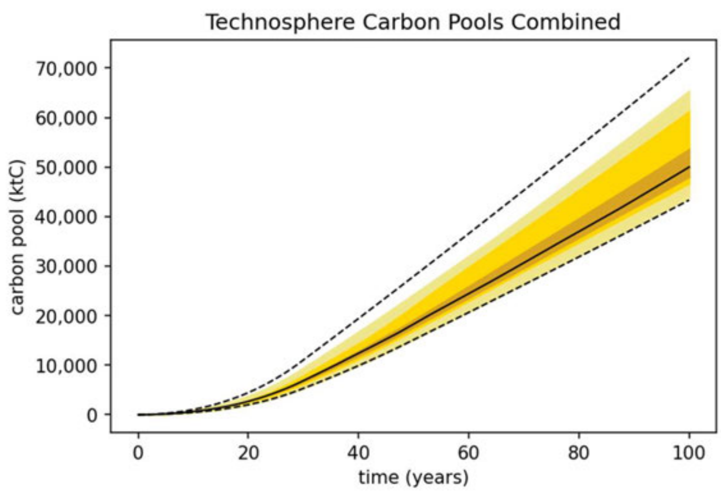



*Figure 6.6. Carbon stored in the technosphere, PADS (i.e. HWP, landfill and CCS).*

### 6.2.5 Substitution Pool

The substitution pools are the 'virtual' pools of carbon associated with wood construction products displacing non-biobased products (material substitution) and with wood fuels (including end-of-life timber) displacing fossil fuels. The development of these pools over time is illustrated in Figure 6.7. As discussed in Section 6.2.1, these pools are only large enough to exert significant influence in the short term: at the end of the simulation, the median values contribute less than 2% to the combined pools total.

#### *Material Substitution*

The wide range of values for the eventual size of the material substitution pool is entirely driven by the width of the PDF for the displacement factor. In all cases, the curves level off after 30 years, when the initial steep growth in demand is replaced by a nominal growth rate, and the effective displacement factor has – in any case – reduced to less than 10% of its starting value.

#### *Energy Substitution*

The energy substitution curves show an initial pulse which plateaus after around 20 years after the co-products have – largely – reached the end of their lives. Then, after an interregnum of another 20 years or so, some of the curves climb again as significant quantities of landfill gas start being produced.

Energy substitution in the early years is maximised when there is a high proportion of co-product used as fuel, and/or a high proportion of end-of-life HWP (including paper and panel boards) routed to energy recovery rather than landfill. The higher values in the long term are, however, achieved in cases where landfill is preferred to energy recovery and where the degradable fraction (DOCf) is above average.

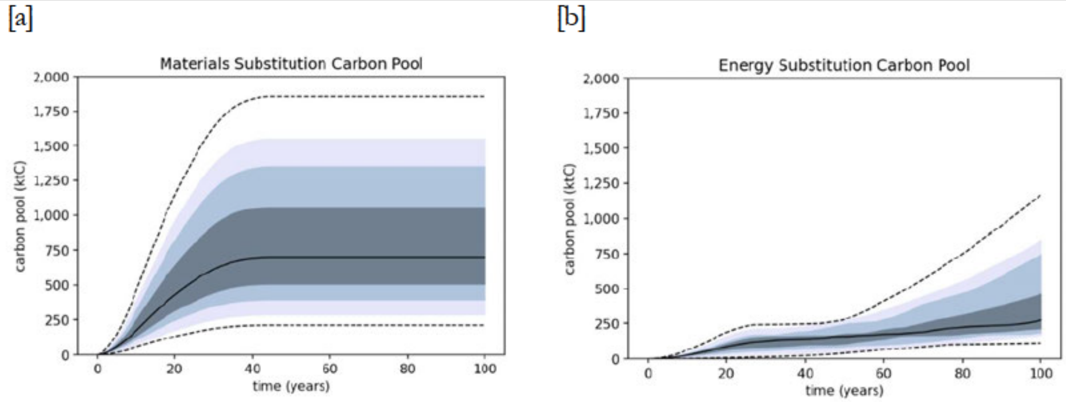



*Figure 6.7. The virtual carbon pools, PADS, [a] material substitution; [b] energy substitution.*

### 6.2.6 Forest Carbon

The development of the forest carbon pool is illustrated in Figure 6.8. In this case, despite the afforestation and the continuous restocking after harvest, an overall loss of forest carbon is recorded during the first rotation, with only a partial recovery in subsequent decades. The factors that influence the scale of the loss are the wood production efficiency (which, after demand for sawn wood is set, dictates the size of the harvest), and the rotation length, with longer rotations resulting in deeper losses. Other parameters such as the forest growth correction factor, and the rotation ascribed to forest 'imported' to the system have almost no influence[67], as changes to carbon per unit area are automatically offset by changes to the allocated area.

To avoid the initial loss of forest carbon, the first harvest would need to be delayed. In other words, the afforestation programme would have to begin twenty years before the demand ramping programme begins, so that by the time harvesting begins, there is enough new forest of sufficient maturity for its productivity to outstrip the harvest.

[67] An exception being a few cases where there is a temporary excess of forest – when BioCarp calculates that no afforestation is needed for a period, and the restocking that occurs automatically is more than immediately necessary.

The effect of rotation length can appear counterintuitive, as longer rotations result in more carbon stored per hectare both at the end of the rotation and averaged over the length of the rotation.[68] However, the mean annual increment (MAI a measure of growth rate) peaks before the forest stand itself peaks. Harvesting after the MAI peak reduces the rate at which forest stands can store carbon.

The turning point of the median curve is at 44 years. By this time, 21.8 MtC has been removed with the timber from 44 harvests, and 77% of this quantity of carbon has been recovered by the trees afforested on grassland to meet all excess demand after the first rotation, resulting in a net value of -5.02 MtC.

With regard to afforestation, the required rate decreases significantly after around two decades. The median curve, for instance, reaches a cumulative area 161 kha after 20 years, 191 kha after 30 years, and 250 kha after 100 years. In the most expansive case, 242 kha are needed in the first 20 years, which equates to 12.1 kha per year. This is well within the 30 kha per year UK Government target for afforestation, but only just within the envelope of current afforestation rates **(CCC, 2021)**, much of which is dedicated to non-productive use (in terms of harvest). This is a significant finding, as it implies that if afforestation is to be coupled with future proposed increases in demand, then every effort will be needed to secure land for the purpose – over and above the area of land that would have been afforested in any case.

In addition to land afforested after the start of the model run, the model imports mature forest land into the system every year during the first rotation period. In the median case, this reaches a cumulative 288 kha after 45 years, more than doubling the total land area used by the model, to about 2% of the UK's ~24 million hectares of land (but see below for caveats regarding sensitivity to yield curve).

At least four of the variables covered by PDFs influence the afforestation area required. Lower than average values for the following parameters all lead to less carbon converted to sawn wood per hectare, and therefore higher areas needed: carbon per standing volume of timber; forest growth correction factor; and sawn wood production efficiency. In addition to this, if the rotation period assumed for forest 'imported' to the system is lower than average, then more land is acquired by the system through this route, meaning a smaller area is required for afforestation (and vice versa).

[68] This is the case, at least, until the rotation period is extended so far that the forest is already in decline at the time of harvest, which would not be normal practice in a commercial setting.

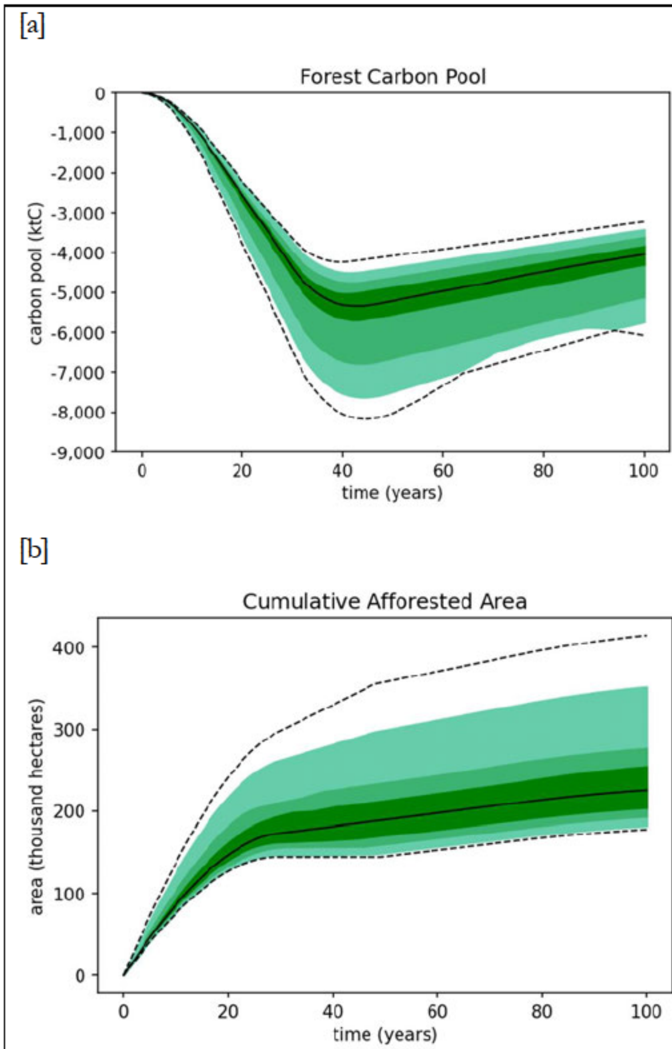



*Figure 6.8. Key BioCarp forest outputs, PADS, [a] the forest carbon pool (showing a loss compared to the situation with no increased HWP demand), and [b] the cumulative area of afforestation on grassland.*

### *Sensitivity to yield curve*

The forest yield function used to define the average growth rate of forest has an important impact on the results. The forest carbon pool is sensitive to the shape of the curve, and the afforested area is sensitive to the amplitude of the curve.

The relationship between curve amplitude (the productivity per unit area at the end of the rotation) is checked by doubling the cumulative yield at each age, i.e. instead of defining the yield by $y = f(x)$, it is defined by $y = 2f(x)$, where y is the yield and x the age. The effect is to halve the afforestation area needed, whilst leaving the forest carbon pool unchanged. The area needed is therefore highly sensitive to the locations ultimately chosen for afforestation, and to the species selection and forestry practices, meaning that the result presented in Figure 6.8b can only be useful as a first approximation. If these factors combine to produce new forest with an average of YC16, for instance, then the area required will be lower than indicated.

To test the impact of curve shape, instead of using the data from the national forest inventory to define an average growth function as used in the base assessment, a function was derived from a particular yield class (YC8) presented in **(Matthews et al., 2016)**. The difference between the curve shapes (Figure 6.9a), qualitatively, is that the NFI curve rises more steeply in the early years (approximately years 10 to 30) after which the growth rate is surpassed by the YC8 curve. The effect of skewing the curve in this way is to decrease the average carbon stored per hectare of forest even though at the point of harvest the carbon stored is about the same. Therefore the forest carbon curve (Figure 6.9b) is shifted significantly further into negative territory than in the base case (Figure 6.8a): by 45% - in the median case – after 40 years, at around the curve minimum, and by 75% after 100 years. This shows a relatively high degree of sensitivity of the model to the growth curve used. This sensitivity is potentially high enough to influence conclusions about the value of the more timber strategy over the short and medium terms, until the forest carbon pool loss is overwhelmed by increases in the other pools.

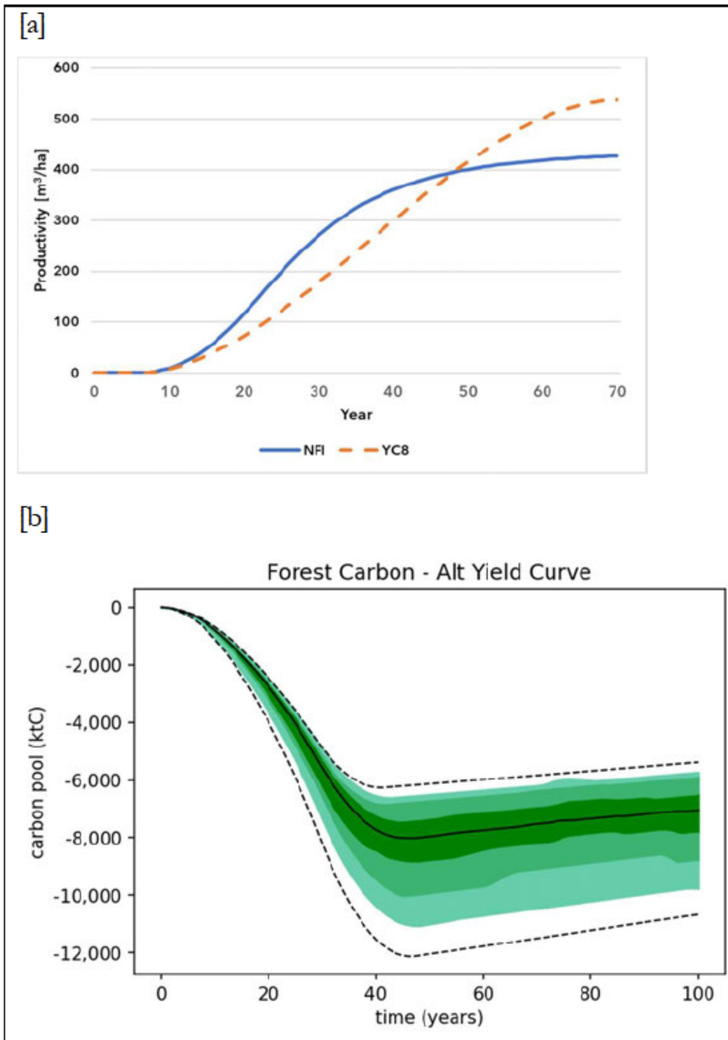



*Figure 6.9. Effect of yield curve on forest carbon. [a] alternative yield curves. NFI: curve derived from National Forest Inventory data used in base case; YC8: an example yield curve for Sitka Spruce from* (Matthews et al., 2016)*; [b] forest carbon results produced using YC8.*

## 6.3 Implication of Modelling Choices

The modelling choices used in section 6.2 have been stated and justified in previous chapters. There are, however, alternative ways of understanding and simulating some of the functions of the model, and also alternative scenarios. It is therefore important to understand the sensitivity of BioCarp to these factors, in order to develop a qualitative view of the robustness of the various results. This is explored by changing one aspect of the model at a time, and determining the effect on the results.

The intention here is not to suggest that these alternative standpoints are either better or worse than the default settings in the model, although in some cases such investigations may point to the need for further research and development.

### 6.3.1 Technosphere

#### *First order decay model – VST*

By default, BioCarp uses a first order decay model to describe the exit of materials from the in-use HWP pools (sections 3.2.2 & 4.2.2). Intuitively this seems reasonable for commoditised materials with shorter life spans, such as paper, but for construction materials – whilst following IPCC defaults – such a model seems divorced from reality. In a state of equilibrium, it is reasonable to assume that the exit from the pool would be proportional to the size of the pool itself. But when that pool is growing rapidly, the approximation begins to break down as – in the case of buildings – so much is invested in them that they are unlikely to be dismantled until they have served a useful purpose for a reasonable period of time. Such a period depends on the location and the purpose of the building, but in the UK it is likely to be decades.

The question explored here is not whether the first order decay analysis is the correct approach. Rather, the question is whether the first order decay model can produce results that are broadly consistent with those produced by what might seem to be a more realistic model. For this simple test, a building lifetime distribution is constructed based on a Weibull distribution (Figure 6.10a), which gives the building an average lifetime of 50 years. This is consistent with the default half-life of 50 years used in BioCarp, but the Weibull distribution is constructed so as to favour the removal of VST after 50 years (with 50% removed within the surrounding 25-year period).

When the in-use VST carbon pool is compared using these two approaches (Figure 6.10b), the difference is evident, but not nearly as profound as the difference between the Weibull function illustrated and the PDF implied by first order decay, which is an exponential decline from an initial value of ~0.014 (i.e. $Ln(2)/t^{½}$). Essentially the model-scenario combination has some self-correcting features: for instance, in the early years – when there is a great disparity between the functions – there is little material in the system for the functions to act upon.

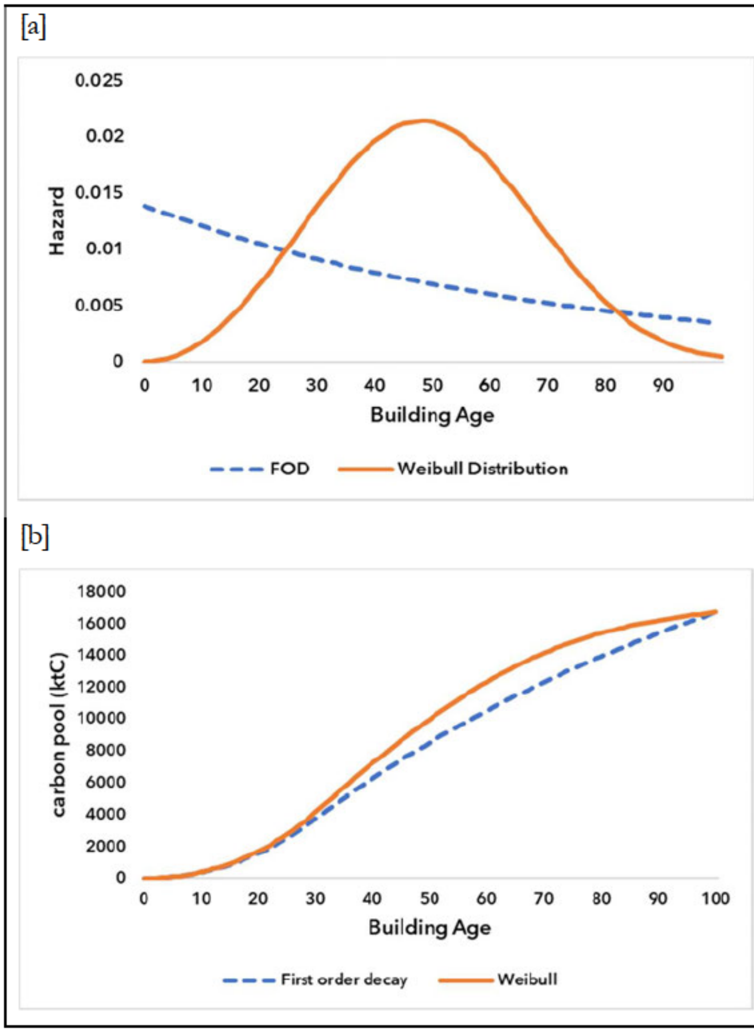



*Figure 6.10. (a) Alternative Weibull distribution for hazard function for virgin structural timber. Note that in this case the function represents the probability that a new item will eventually be removed in the given year, rather than the probability that an item of a given age will be removed in the current year (i.e. it is the fraction of the original that is removed, not the fraction of the remaining). And (b) VST carbon pool results compared using the alternative hazard functions.*

### *Landfill & CCS*

As mentioned in Section 6.2.1, the development of the landfill and CCS pools as indicated in Figure 6.3 and Figure 6.5 is dependent on (a) landfill or an equivalent form of solid wood storage, and/or[69] (b) use of CCS with biomass, being available at a large scale in the future.

There are reasonable grounds to doubt both of these propositions. With regard to solid wood storage below ground, this currently takes a small and diminishing fraction of the waste stream, and an active choice would have to be made to reverse this. This would

[69] Note that landfill and CCS can be alternative strategies here, so it would be technically possible to achieve similar levels of underground storage with either or both methods.

present challenges, as so much effort has been put into eliminating landfill as an option, both in public discourse and through government policy. With regard to CCS, it may be that the model takes an over-optimistic view of its development and application to facilities burning wood.

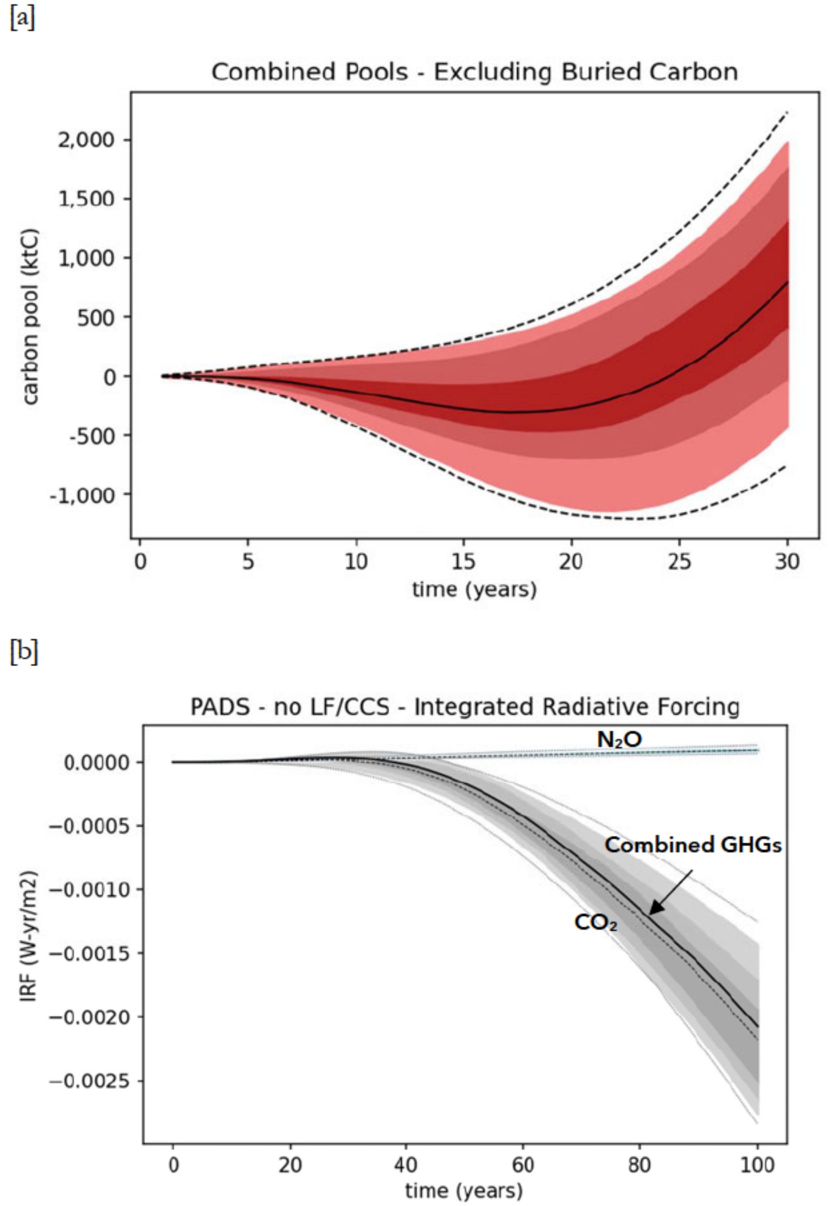



*Figure 6.11. Combined pools, without landfill and CCS. [a] All pools, physical and virtual; [b] integrated radiative forcing – $CO_2$ and $N_2O$ ($CH_4$ is zero without the landfill).*

The most straightforward variant case is to assume that neither landfill nor CCS is an option for end-of-life timber, and that all such timber deemed unrecyclable is instead burned in facilities without CCS. This has the effect of eliminating the underground carbon pools from BioCarp altogether, and with negligible compensation in other pools.[70] The effect is significant even in the first two decades, and in the long term it reduces the size of the combined pools by more than half. A reproduction of Figure 6.1(b) with this alternative data (Figure 6.11) shows an exaggerated trough and a postponement of the break-even data to ~20-30 years (median = 25, with IRF not turning negative for 40 years), and more (median = 31) if substitution pools are excluded.

A more nuanced variant to this scenario would be to postpone the roll-out of CCS by a suitable period whilst continuing to assume that landfill rates will be negligible. This has been done here by postponing CCS implementation by 15 years. For the first 30 years, the numbers are as illustrated in Figure 6.11, as CCS only starts ramping up its contribution from zero in year 26. The picture across the full 100 years is as shown in Figure 6.12 for the CCS pool, which is also the full pool of buried carbon in the absence of landfill, and so can be compared with Figure 6.5(b) and (c). Despite the delayed start, the CCS pool is larger in this scenario than in the base scenario, but not by enough to offset the absence of landfill: by year 100, the delayed CCS pool is around 15% lower (median = 26 MtC) than the buried carbon pool in the base scenario.

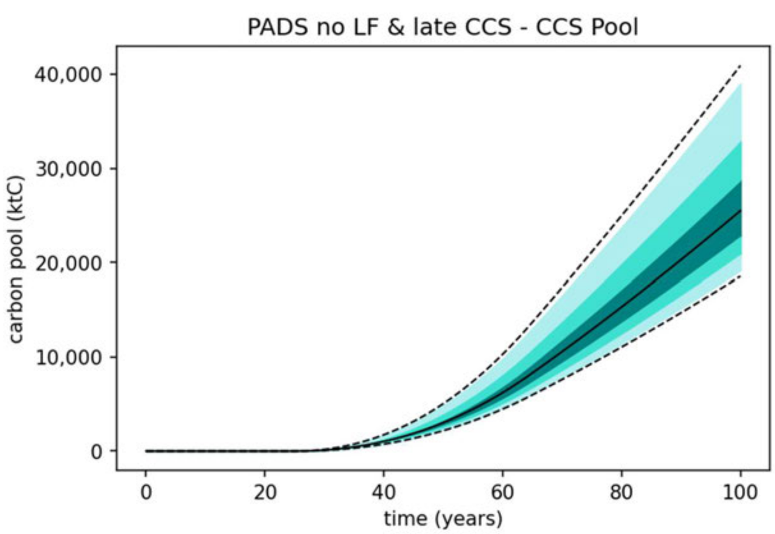



*Figure 6.12. CCS pool: Zero landfill with CCS postponed.*

[70] The only changes being to the energy substitution pool, which still constitutes a near-negligible fraction of the total.

### 6.3.2 Substitution

In this section, the impact of the modelling choices concerning energy and material substitution are considered, in particular, those concerning preferences (the discount factor), carbon leakage, and decarbonising the economy.

#### *Discount factor*

If the discount factor is eliminated from the model (i.e. set at zero, instead of at the level of 5% per annum used in PADS) because of a lack of faith in its justification, this has a significant effect on the substitution pool. After 30 years, the pool total (median values) is 1.75 MtC, and 2.4 MtC after 100 years, both being more than double the values shown in section 6.2.5. Thus the increase in the long term is insignificant relative to the total pool, but this change in assumption would significantly reduce the risk of the combined pools showing a loss of carbon in the initial two decades.

#### *Carbon leakage*

On the other hand, it might be argued that carbon leakage should be included in the model. If so, it is difficult to provide evidence for a value that should be ascribed to it, but to test the sensitivity, a value of 5% of the pool size is assumed to be lost every year. In this case, the effect increases over time, and after 100 years the pool is almost eliminated because as time passes the influxes to the pool decrease: even with no discount factor applied, the substitution pools after 100 years are only 3% of the no leakage case. In the early years the loss is less marked but still potentially significant: for instance, in years 15-20 – the years around the trough in the combined pools curve in Figure 6.1 – the substitution pool is reduced by 20-30%.

#### *Panglossian view*

As discussed previously (sections 3.6), as well as neglecting questions of changing preferences and carbon leakage, some models have also ignored the expected future decarbonisation of the economy. Instead, they assume that all future use of wood for construction or fuel earns carbon credit at the same rate per tonne as they do today.

Whilst there can be no realistic justification for taking this approach, it is at least interesting to see what the consequence would be. If, also, the default displacement factor used in much work is used (2.1 tC emissions per tC in the wood, instead of the figure of 0.7 tC/tC used in BioCarp), the substitution pool grows endlessly, without any check, at a significantly higher rate than all other pools combined. Even though CCS is taken out of this scenario, as the assumption (whether implicit or stated) in such modelling is that technology in the future will be as it is now, after 100 years, the median value for the size of the substitution pool is 63 MtC (about 90% of this being material substitution). This compares to a figure of ~1 MtC with BioCarp in its default settings. In a case of *reductio ad absurdum*, if such a model was run for long enough, then it could in theory substitute more carbon than exists on earth.

### 6.3.3 Forest

Before going on to present scenarios that show the forest carbon pool in a less favourable light than that shown in PADS, it is worth noting that it would be possible to design a more favourable variant of PADS – a 'super-PADS'. This would involve higher levels of afforestation so that the mature forest area imported into the system in the early years can be exported from the system later on: in other words, the mature forest is on temporary loan to the system until sufficient grassland has been afforested and matured to meet the system needs. This would see the initial forest carbon loss converted to a gain over the medium term. This would, however, require a significantly higher level of afforestation than that shown in Figure 6.8b, with the curve rising slightly more steeply initially and continuing at about the same rate for one rotation period (>10 kha/yr for 45 years in the median case). Whilst this is possible, it must be the case that the higher the afforestation rate 'requested' by the model, the less secure any link will be between future timber demand and afforestation.

#### *Scandinavian Import Scenario*

In this variation (SIS), a highly-forested country representing a major supplier of construction timber to the UK is chosen as a proxy for all providers of the UK's imported sawn softwood. An average forest productivity curve is derived from the EFISCEN database (EFI, n.d.)[71]. In this case, the country is Sweden, the EFISCEN data relates to spruce grown in all regions of the country (itemised separately), and the derived productivity curve reports area-weighted averages of standing volume per hectare in each age band. Longer rotations than the UK are also assumed (50-80 years rather than 40-60): if the upper limit were set higher it would make little difference to the result here because of the timescale of the model runs.

The average growth rate is slower than in the UK, which – as discussed in section 6.2.6 – is not relevant to the carbon account, but does affect the area of forest required. As a result, around 950,000 ha of pre-existing forest is imported into the system to satisfy demand.

[71] Much of this forest inventory data is of the order of 20 years old, so the area distribution of forest types and age bands is likely to be out of date. But the conclusions about forest growth rates will still be valid if (a) fundamental growing conditions and management and (b) the distribution of forest around the country have not changed significantly.

The adjustment to the carbon account in SIS comes about through the omission of any link between afforestation (in Sweden) and demand (in the UK). There are two justifications for this: one is the geographical and administrative separation makes a formal link harder to achieve. And Sweden has much higher forest coverage than the UK (75% and 13% respectively, **(Forest Europe, 2020)**), with – therefore – a lower proportion of land available for afforestation. No afforestation means less compensating growth to help the recovery of forest carbon: instead, forest carbon recovery is entirely reliant on post-harvest reforestation (which is not additional to that in PADS). Forest carbon therefore declines towards an equilibrium level that is reached at the upper end of the rotation period range (Figure 6.13).

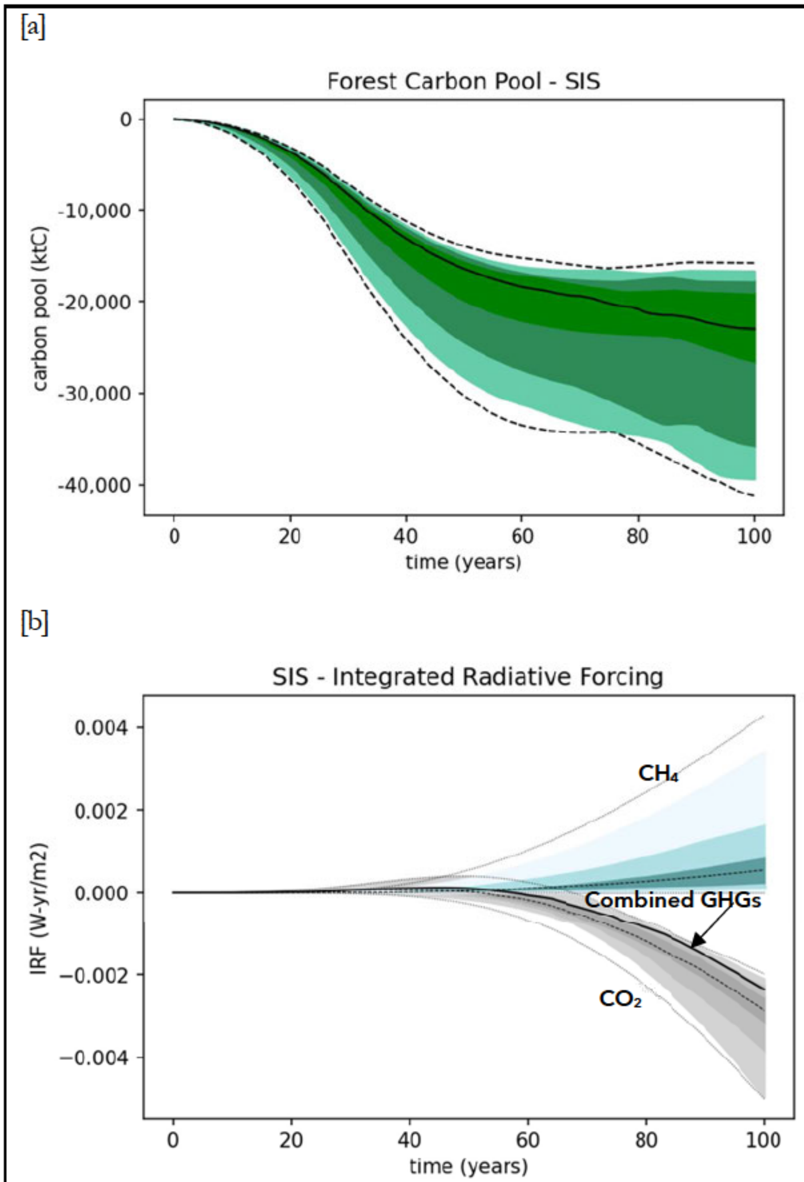



*Figure 6.13. Scandinavian Import Scenario. [a] Forest carbon pool; [b] effect of combined carbon pools and methane emissions on IRF (land-use change nitrous oxide emissions are zero).*

The effect on the combined carbon account is to delay the time until the initial carbon debt is cleared (median 43 years, or 49 years excluding substitution). Therefore it takes until the 2070s for this strategy to pay off in terms of carbon balance, and beyond 2080 for IRF to show a negative trend, but beyond then, the net benefit accelerates convincingly as the carbon accumulates in the technosphere.

The relative contribution of each pool is shown in Figure 6.14, which clearly shows the enduring influence of the forest carbon deficit, compared to PADS (Figure 6.3). The product and forest carbon pools are influential throughout the period, whilst the landfill and CCS pools increase in relative (as well as absolute) importance, and the relative contribution of substitution dwindles over time to the point of insignificance.

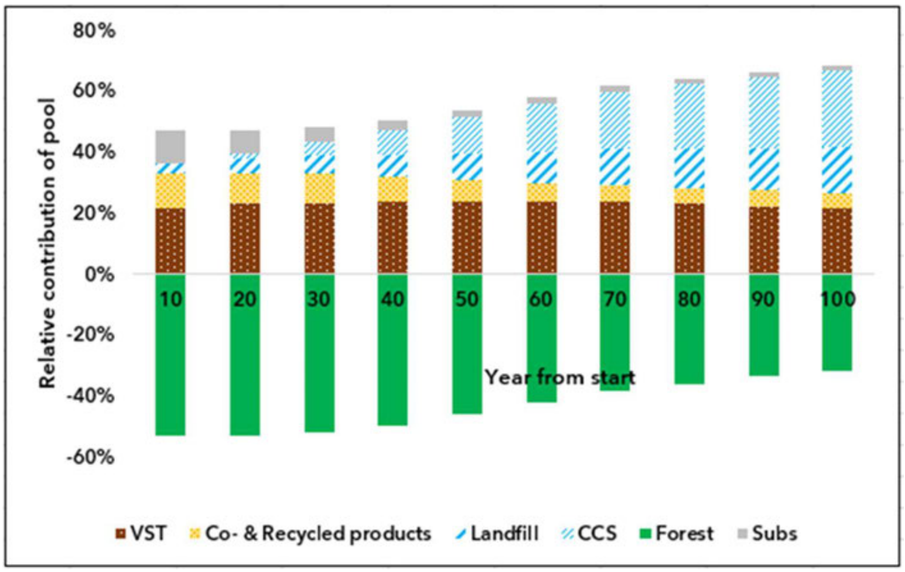



*Figure 6.14. Disaggregated results at 10-year intervals, SIS, showing the relative contributions from each pool – median values. Virgin structural timber, other in-use HWP (co- & recycled products), landfill, CCS, forest, and substitution pools.*

### *Reactive Unplanned Domestic Scenario*

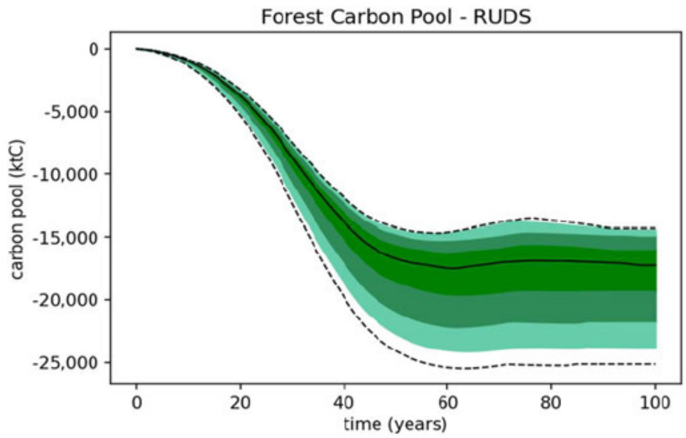
The Reactive Unplanned Domestic Scenario (RUDS) assesses what happens when future demand is met from UK forestry, but the link between future timber demand and afforestation rates is not successfully made. In this case, no link is made between demand ramping and afforestation. Therefore future demand might be met from spare capacity in existing commercial forestry or from afforestation implemented for a different purpose (thereby reducing the carbon pool and other benefits associated with that forest). In either case, tree harvest is brought forward, although in the case of commercial forest it is arguable that an earlier-than-initially-planned harvest is well within the original broad purpose of that forest, and this is factored into the model. So, as well as breaking the link with afforestation, RUDS considers the growth foregone (for an illustrative, defined time period) when a forest stand is harvested before it was otherwise due: this offsets – to some extent – the growth in the restocked stands in the years following the harvest. However, it is hypothesised that until the supply crunch potentially due in the 2040s (section 3.5.2), timber can be supplied from existing commercial forest; but after then, marginal harvest is from trees not planted specifically for timber supply (e.g. planted for environmental services, and with new products and methods available to enable a diverse range of tree types to be exploited at scale). Therefore, the growth foregone contribution to the pool is considered only from 2045 onwards.

As illustrated in Figure **6.15**a, the forest carbon loss is significantly greater than in PADS (more than tripled after 50 years, and quadrupled after 100 – compare Figure **6.8**a). The major contributor to the difference is made by the absence of the afforestation link, with the 20-year growth foregone accounting for 10 – 20% of the difference. Extending the 20-year period increases this value, but with diminishing returns as growth rate reduces[72], and with diminishing justification as the probability that the forest would – counterfactually – have survived this extra period reduces.

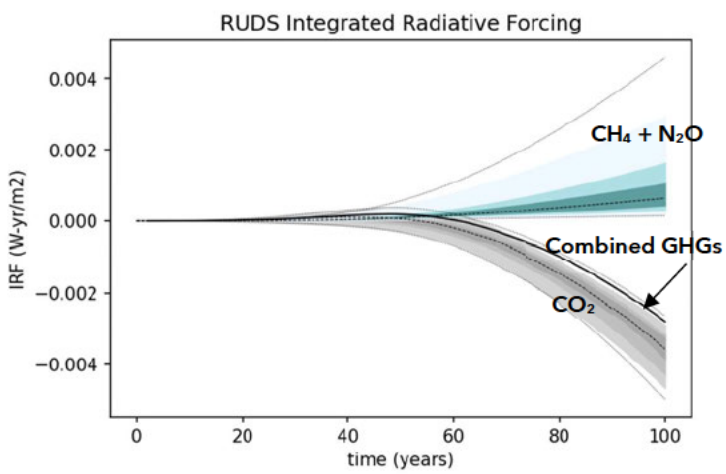
The change in the forest carbon pool resulting from the RUDS variation results in a significant delay to the combined pools showing a net increase, with approximately 45 years passing before a net positive result is achieved in the median case. Leaving substitution benefits out of the combined pools only increases this period marginally, but does increase the depth of the minimum (median = -1.9 MtC). This loss being high enough to postpone any climate benefits until around 60 years have passed, approaching the end of the century (Figure 6.15b). Overall, the results for RUDS are very similar to those for SIS. This also applies to the disaggregated figures (Figure **6.15**c), although they show a slight reduction in forest carbon loss towards the end of the period compared to SIS.

[72] Although this would not be the case for slower-growing forestry, so there are uncertainties involved.

[a]




[b]




[c]

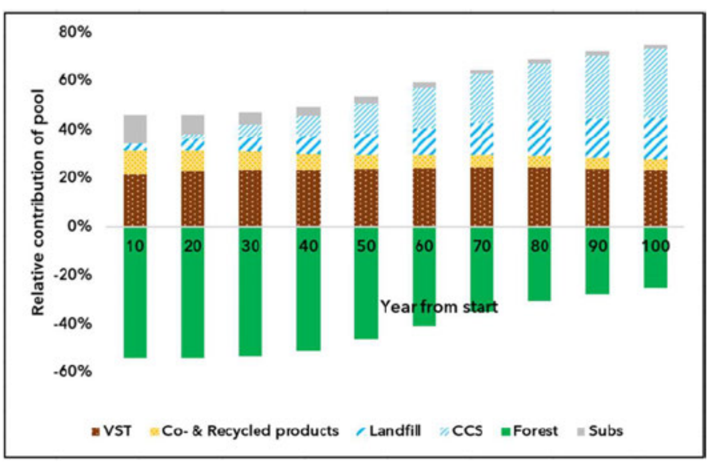



*Figure 6.15 Key outputs from the RUDS variation. [a] Forest carbon, [b] Integrated radiative forcing for all pools combined (including substitution) and the contribution from combined non-$CO_2$ GHG emissions. The solid line ('combined GHGs') indicates the median line of the combined curves ($CO_2$, $CH_4$ and $N_2O$), and [c] Disaggregated results at 10-year intervals – median values.*

### *Full forest carbon stock – including soil*

As previously stated, soil carbon is not considered in BioCarp in its default settings, and therefore in PADS. Averaged across the whole UK estate, this implies a state of equilibrium in forest soils, in that the transfer of atmospheric $CO_2$ to carbon below ground (the soil and everything in it, including stumps, roots, etc.) is in balance with the removal of carbon from soil, either directly (e.g. removal of stumps for fuel) or indirectly through oxidation of soil carbon. While this may be a reasonable assumption, it does not necessarily apply to areas subject to land-use change, which in BioCarp means afforestation.

The Read report **(Read et al., 2009)** includes a discussion of how below ground carbon varies over two rotations following the afforestation of a British moorland site, with deep ploughing at the start, and minimal disturbance after the first harvest (including leaving all above and below ground residues in situ). A soil carbon loss of around 120 tC/ha over the first 40-year rotation is recovered over the second. In addition, around 110 tC/ha of above and below ground carbon accumulates beyond the merchantable stem. This occurs in each rotation, but the gains are temporary, being lost within a few years of harvest. Whilst it in no way suggests that these dynamics represent the average soil dynamics associated with afforestation, it is at least interesting to consider the effect of including such an assumption in BioCarp.

It turns out that including this non-stem carbon in this way does not affect the conclusions to be drawn from this study, for the following reasons:

1. The reason for this is that over the first rotation (i.e. in the short and short-medium term), the soil carbon loss is approximately in balance with the growth of non-merchantable carbon stocks. Therefore the change in non-stem carbon over the first rotation makes very little difference to the total forest carbon pool.
2. At the start of the second rotation period there is significant loss from this carbon pool (lasting from years 46 to 76 on the median curve), but by this point in time the loss (reaching 3.6 MtC at 60 years on the median curve) is not significant in the context of the combined pools.
3. The very strong subsequent recovery in this carbon pool merely adds to the sums in a period where the combined pools perform very well in almost all scenarios.

### *Shorter time period*

Default PADS models carbon pools based on a 3% annual increase in demand for a 30-year period followed by a nominal 0.1% annual increase for the next 70 years. Of course there are infinite variations possible for the demand function, but there is one aspect in particular to explore here. Arguably modelling 100-years of enhanced demand is too much of a stretch, as very little policy and planning in any sphere actually deals with such timescales. In practice, the year 2050 is about the limit of much long-term policy development.[73] Therefore, an alternative approach might be needed in which:

- The only demand of interest is the marginal demand for the first thirty years. For instance, it might be the case that after 2050 the world will have moved on (again) from basing construction around the availability and use of timber.
- A destination must, however, be found for end-of-life materials that buildings will continue to issue. Therefore the end-of-life functions of the model continue to operate until the end of the model run.
- Whilst the PADS model keeps its anticipatory afforestation element in this case (as if demand were to continue after year 30 as in the base case), but no afforestation, harvesting or reforestation is modelled after year 30. After that point, productivity from forest stands planted within the model is counted until the end of the set rotation period, at which point it is assumed that the forest stand becomes part of a different accounting system.

Carbon pool results for in-use HWP and the forest are shown in Figure 6.16, together with a plot of IRF attributable to the scenario. Within the technosphere, the initial increase in the carbon pool ceases at year 30 – at a level equal to around 80% of the total carbon transferred into the pool by that point. After this, transfer of carbon from forest to system HWP stops. From then on, the primary change is the transfer (within the technosphere) from HWP to landfill and CCS, with some losses to the atmosphere. For the first 30 years, forest carbon declines as in the PADS base case, but then recovers to above zero, after importation of forest from outside of the system stops and as the trees planted within the system head towards maturity.

Overall, this variation is almost as successful as the default PADS approach after 100 years, despite the absence of HWP entering the system for the last 70 years. The reason for this is that the afforestation of the first 30 years (the same as in default PADS) is not harvested within the system, and so is effectively a gift to climate mitigation.

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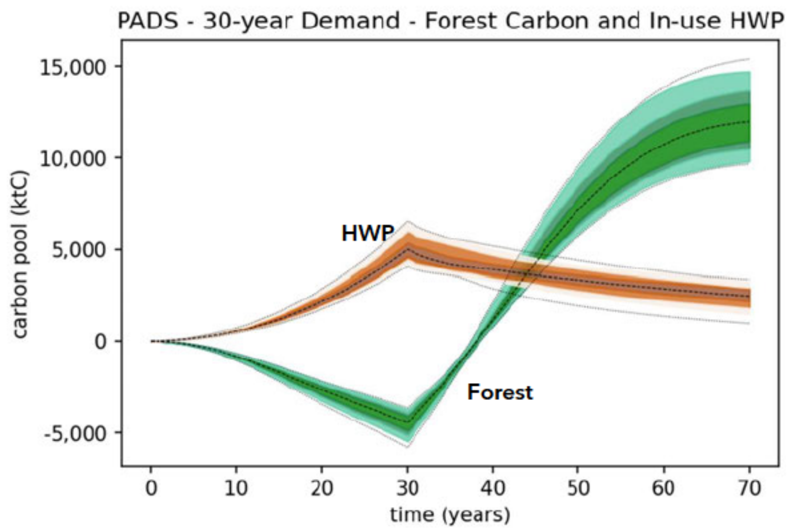
[73] Although exceptions exist, such as when countries believe that 2050 is too soon for certain objectives to be met, such as – at COP26 – India announcing its target of 2070 for reaching Net Zero emissions. And much infrastructure is built at least in the hope that it will last well beyond mid-century, even when the long life is not central to the case for investment.

[a]




[b]

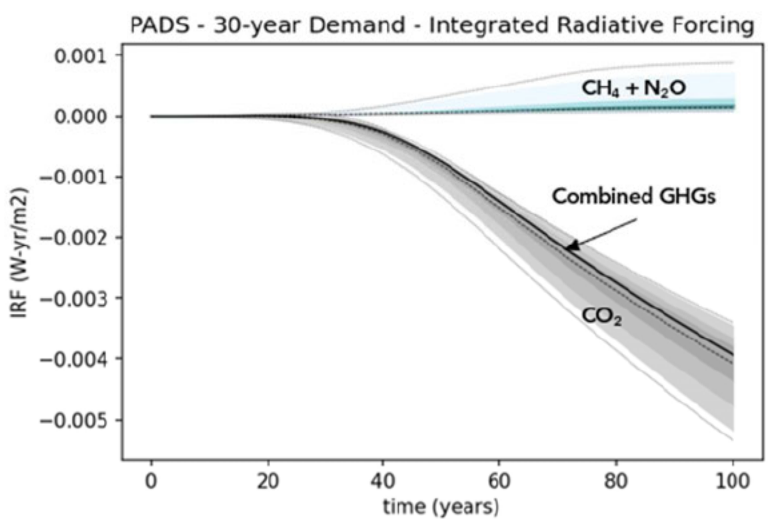



*Figure 6.16. Results for 30-year time limit on marginal demand. [a] forest carbon and in-use HWP, and [b] integrated radiative forcing.*

### 6.3.4 Scenarios Combined

As well as considering these alternative scenarios and 'framings' of the question individually, it is also worth considering their effect in combination. One particular combination seems particularly pertinent, as it may be the default path that the UK will follow without very active intervention.

Firstly, the UK may continue to rely heavily on imported construction timber, meaning that marginal demand would be met predominantly from imports. Therefore, without active intervention to change the situation, SIS may be the most realistic assumption (out of PADS, SIS and RUDS). Secondly, the existing presumption against landfill remains, meaning that landfill continues to take only a small fraction of waste wood. And thirdly, there is a possibility that waste incineration and energy recovery will be towards the bottom of the queue for investment in CCS, with facilities such as fossil fuel power stations and large industrial facilities such as cement kilns and blast furnaces cornering the available investment. As such, CCS is excluded from this worst-case scenario.

The results, presented in Figure 6.17 suggest that, under this set of assumptions, timber has no useful role to play in climate change mitigation over the next 100 years.

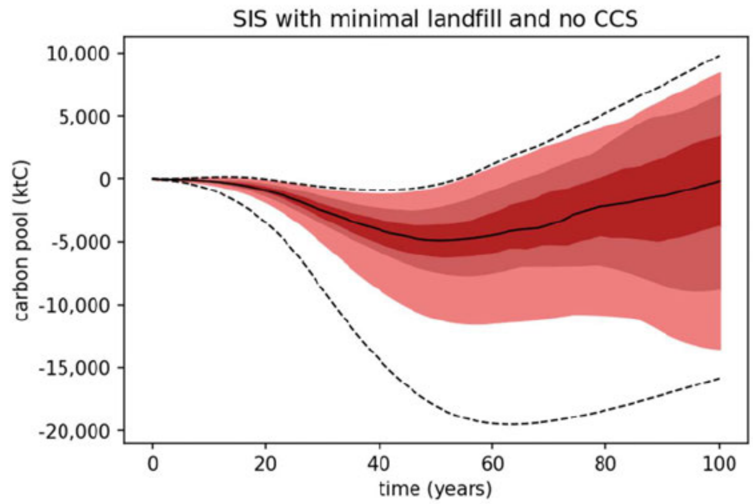
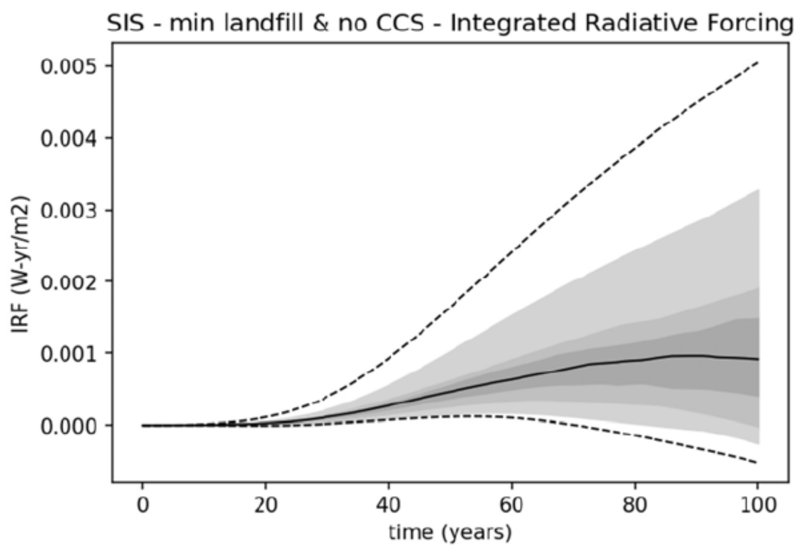



*Figure 6.17. Worst-case scenario – Scandinavian Import Scenario, with landfill only used for residual unsorted waste, and CCS never implemented for waste wood. [a] combined carbon pools, [b] integrated radiative forcing.*

[a]

[b]




The two plots in Figure 6.17 suggest that even after 100 years there is only a 50% chance that the carbon account will be in credit, and only a 10% chance of IRF being below zero, meaning a 90% chance that this scenario will increase pressure on climate change for at least 100 years.

This may be overly pessimistic in terms of the potential contribution of CCS, as the Drax biomass power station has potential for CCS, and in theory this could take the pelletised co-product included in the BioCarp, if not the end-of-life HWP (paper, boards and sawn wood products). However, it is reasonable to present this as a realistic worst case, as there is still no certainty that CCS will have as large a role as expected generally, and also about whether CCS will apply to much, if any, end-of-life wood combustion.

## 6.4 Conclusions

In this chapter, the implications of ramping up the use of construction timber in the UK have been modelled and the results presented. When this extra demand is linked to a strong afforestation strategy in the UK, then a strong climate change mitigation signal is detected over the medium to long term, with an increase in the terrestrial carbon pool of 47 MtC after 100 years and an associated reduction in integrated radiative forcing of 0.0061 $W.yr.m^{-2}$. However, the benefits achieved by 2050 are modest (the carbon pool and IRF values after 30 years being, respectively, 7% and 1% of the 100-year values), and whether any detectable benefit at all materialises within 25 years is within the model's margin of error. The alternative scenarios modelled present less encouraging pictures, with the most pessimistic assumptions around timber supply and end-of-life management resulting in no climate change mitigation within a century. For instance, if the additional construction timber demand is imported, with no link to afforestation, and waste is still burned without CCS, then the terrestrial carbon pool loss reaches 0.5 MtC before recovering (but still showing a loss of 0.18 MtC at year 100); IRF is continually positive for 100 years, peaking as late as year 91, before dropping slightly to a value of 0.0009 $W.yr.m^{-2}$ in year 100.

A fuller discussion of the conclusions and the lessons that can be drawn from this research follow in Chapter 7.

# 7. Discussion and Conclusions

*Conclusions that can be drawn from the modelling results; ideas for further research; and wider lessons for government, industry and society.*

## 7.1 Conclusions

The results and analysis presented in this thesis allow a number of clear conclusions to be drawn about the potential contribution of harvested wood products in the UK to climate change mitigation, and how HWP can be supplied, utilised and managed in order to maximise that contribution. These conclusions are presented below in a logical order (not priority order).

### *Conclusion # C1: Increasing the demand for construction takes a long time to pay off.*

All the scenarios related to increasing the use of construction timber considered in this thesis take time to develop a contribution to climate change mitigation, delivering very little in the first few decades. Despite annual increases in demand of 3%, compounded for 30 years, all scenarios deliver a very small fraction – at best – of their 100-year contribution in the first 30 years. In the less optimistic cases, increased demand will actually cause negative climate impacts over the first few decades, and potentially beyond. In the best case, however, the carbon pool will have developed by 3.4 MtC by 2050, with integrated radiative forcing being negative by this point in time. This shows that at least some contribution can be made. The 3.4 MtC is equivalent to approximately 0.17% of the UK's remaining carbon budget (assuming a linear progression from the 2019 level of 550 $MtCO_2e$/yr (150 MtC/yr) to net zero in 2050 (NZ2050). Therefore it is possible to assert that a strategy to use more timber in construction can make a contribution to meeting the UK's NZ2050 target, but – given the many factors likely to squeeze this contribution (as outlined in section 6.3) a substantial share is unlikely to be achieved. Furthermore, considerable attention must be paid to factors that can actually cause the strategy to do harm on that timescale.

### *Conclusion #C2: For a 'quick win' with respect to climate mitigation, look to waste management.*

In contrast with the time delay before the combination of timber demand and afforestation can deliver significant levels of mitigation, a rapid result could be achieved by changing the way in which timber waste is managed. Section 5.4 showed that a change to waste management priorities for existing rates of end-of-life timber production could reduce carbon emissions by a cumulative 13.9 MtC by 2050 (which is 0.7% of the UK's remaining budget). This is under the landfill scenario (Sc-lf), under which only half of the material currently burned is instead sent to landfill. A more extreme version of this strategy (i.e. combustion eliminated until CCS is used as standard), coupled with a circular economy strategy, could more than double this gain. This means that a new form of landfill (offering bulk wood storage optimised for carbon retention and potentially offering a stock of recyclate for use as recycling markets and technologies develop) should be brought into consideration at least until such time as CCS becomes standard for wood combustion.

#### *Conclusion #C3: For the increased demand strategies to succeed, unmitigated combustion of wood must cease*

Following on from #C2, it is also clear that the continued combustion of wood without CCS undermines the potential of the more timber in construction strategy to deliver climate change mitigation. The results presented in section 6.3.1 and 6.3.4 show that the continued combustion of wood without CCS incurs an initial carbon cost, and delays (at best) any benefit until the latter part of the century. And, if this effect is coupled with other parts of the system being sub-optimal (particularly regarding timber supply and afforestation), then the strategy will add to climate change pressures for at least a century.

#### *Conclusion #C4: If negative impacts are to be avoided, then domestic supply – and a strategy to secure it – is required*

Of the more timber in construction scenarios tested in this chapter, the only one that successfully avoids initial costs and delivers tangible benefits in the medium term and beyond is the Proactive Anticipatory Demand Scenario, in which any marginal demand (above baseline) is anticipated and linked to afforestation measures. Any deviation from this is likely to undermine the benefits, unless the deviation involves even greater levels of afforestation and/or a delay to the start of the demand ramping.

Therefore, any promotion of greater levels of timber use to benefit the climate must be linked to a strategy for enhanced afforestation over and above the strategies (including policy, incentives, targets, etc.) that are already in the system. Furthermore, a strategy will be required to ensure that the marginal timber under this strategy is supplied from UK forestry, rather than imported from countries operating on longer rotation periods and with limited opportunity for afforestation.

## 7.2 Limitations and Further Research

The research presented here achieves its intended purpose, as the uncertainty modelling and scenario analysis between them should capture the range of likely results. There are, however, limitations to both the model and the data behind it, and so there is potential to refine the model and also to extend it into new contexts, as discussed below. At a philosophical level, any future retrospective assessment of the work would come with some subjectivity, as – for instance – the strength of the link between the more timber in construction agenda and afforestation policies is unquantifiable.

### 7.2.1 Elements missing from the model

Much importance is attached in the conclusions of this research to the need to retain biogenic carbon in the technosphere rather than allowing it to return to the atmosphere at the end of product life. The only options considered in BioCarp (beyond recycling) are landfill and carbon capture and storage: the first is technically and economically straightforward, but politically unpalatable and potentially difficult to regulate, whilst the second is expensive and yet to achieve any significant scale globally. Therefore the model might be extended to consider other possibilities, such as biochar.

BioCarp could be extended to products from fast-growing crops, because of the simplicity of one-year growth cycles, compared to forestry. This would require the identification of realistic scenarios for the growth in demand of a given product category, a justification for any material substitution benefits, and a land allocation calculation (which would need to take account of whether the material is primary product such as hemp fibre or by-product such as wheat straw).

The stated purpose of the research is to track the flow of carbon and other greenhouse gases through the forest-product-use-recycling-disposal system in order to evaluate the climate change mitigation potential of the system. Thus it is not a fault of the model that it does not investigate other environmental impacts, apart from calculating the area of land needed by the system for afforestation. However, it would clearly be interesting to understand what non-climate environmental benefits and costs are missed by the research: work which could be started by assessing the full LCA benefits of substituting a suitable mix of construction materials with wood and scaling up.

### 7.2.2 Model Refinements

It is expected that the probability density functions used to define each variable in BioCarp are sufficiently wide to capture the realistic range of possibilities in each case. In some cases there will be scope for refining and narrowing these based on new information. For instance, the variable that governs landfill gas emissions has a PDF with a long tail (based on a single literature value), leaving open the possibility of high emissions: it would be helpful to attach more certainty to the landfill gas potential of timber in specific contexts (in this case, UK managed landfill sites).

Similarly, the lifetimes assumed for wood products are based on IPCC defaults and a limited set of literature values. Primary research to validate or revise these values would be very helpful: a sufficiently robust research programme to determine this would be a substantial undertaking, possibly involving extensive sampling of a wide range of waste flows in various time periods and locations, and designing a method for estimating the age of the wood products. This work might also involve the development of context-specific (UK in this case) building stock models. Whilst sensitivity testing showed that the first order decay model presented in IPCC guidelines for national greenhouse gas inventory reporting does not automatically lead to an inaccurate result, it would be preferable to use a model and decay rates with stronger evidence behind them.

With regard to forest carbon, there are many aspects of the model that might be adapted or delegated to a pre-existing model such as, for use in the UK, CARBINE. A better-evidenced function to represent forest growth (and surrounding range to reflect uncertainty) could enhance the credibility and quality of the results; this could even have a dynamic element whereby future forestry is treated differently to existing forestry in terms of growth functions and rotation periods, taking account of developments in silviculture. Scenarios considering importation of timber could include such functions for each significant supplier country. Another example is forest risk to extreme climate events and pests and diseases. In BioCarp this is handled with a user-defined correction factor that adjusts the productivity of the forest uniformly: the true impact of such risk is – however – likely to increase over time, and furthermore the vulnerability of a forest stand may vary during the course of a rotation (e.g. vulnerability to drought early on, and windthrow subsequently). Other developments might include offering a set of context-specific (e.g. species, soil, latitude) growth curves and rotation lengths. This would need to be reinforced with research as to where future afforestation is likely to take place, as there is no guarantee that this will look like the current average context: an illustration of this is the justifiable pressure to allow commercial forestry in Caithness to revert to peat bog.

With regard to the scenario investigated for the growth in timber demand, this is – necessarily – speculative, as there is no national target for this, but it may be worthwhile to consult on and model alternative scenarios, and apply more specific displacement factors to different parts of the scenario. This might for instance include a displacement factor for CLT/glulam frames and another for regular timber frames, with values adjusted to take account of additional transport needs if the timber is imported, and potentially with further sub-division for different building archetypes. Any method chosen to model the substitution pool would divide opinion, but a case for a sceptical stance to the continuous accumulation of such benefits is argued for and consistently implemented here, leading to the substitution pool generally being a relatively small part of the whole.

### 7.2.3 Policy Development

Finally, the results and conclusions raise the question of how policy can be developed to take advantage of the opportunities identified. For instance, in order to shift wood from energy recovery to other uses and fates, a range of interventions is likely to be needed, which might each have undesired side-effects. These will need careful evaluation. Similarly, the concept of promoting increased demand for domestically-produced production timber and linking this to levels of afforestation that will eventually meet this demand requires a great deal of cooperation between government, landowners, and the forest and construction industries in order to develop and support realistic scenarios for timber usage, and associated land-use planning.

## 7.3 Lessons

In addition to the firm conclusions drawn directly from the analysis, a number of softer conclusions, or 'lessons' can be inferred.

#### *Lesson #L1: More consideration needs to be given to the potential for fast-growing crops to sequester and store biogenic carbon*

All the scenarios presented in this thesis have one thing in common. That is, the long pause before substantial climate change mitigation can be realised. And in the worst case - the pessimistic but still plausible combined scenario (section 6.3.4) – no such mitigation can be achieved within a century. There are two reasons for the existence of this pause. One is the simple logic that it takes many years of changed policy and market conditions to accumulate significant quantities of above-baseline carbon in the technosphere: the only way this can be addressed is to try and move policy and market as fast as reasonably possible. But the real challenge to the strategy is that all the scenarios presented begin with a loss of forest carbon, and with associated losses to the atmosphere as this carbon is transferred to HWP, with a substantial time-lag between harvest and forest carbon recovery. Depending on the scenario, this initial loss eventually levels out or recovers as the trees planted in the model develop, and in some cases the loss is eventually overwhelmed by the development of the technosphere pools. However, this takes time, and no scenario shows potential to contribute substantially – or even at all in some cases – towards the UK's NZ2050 goal.

This begs the question of whether, collectively (for the most part), we are looking in the wrong place for a solution, and that a more effective way of leveraging the power of the construction industry to shift carbon from the atmosphere to the built environment is via crops that grow much faster than trees: with a rotation time of a few years (e.g. bamboo) or a single year (e.g. hemp and straw). Additionally, we might make more use of recycled wood fibres in construction, either by enhancing manufacturing processes to facilitate the use of recyclate (e.g. MDF or wood fibre insulation) in board production (building on what is already being done with particle board) or by increasing the role of currently niche applications such as light earth construction (which can use recycled wood chip or fast-growing crop residues). Where material supply is constrained (e.g. wood recyclate), then the most climate-friendly uses for it need to be identified. Whilst this is already understood by some researchers and practitioners, the message needs a wider hearing, and encouragement for more product development in this category, as natural insulating products still only have a small share of the market, and there are few structural products using fast-growing crop materials.

#### *Lesson #L2: For the construction client or practitioner – wood should not automatically be seen as the 'low carbon' option*

Whilst this work shows that a broad strategy to increase timber use in construction has the potential to be beneficial in the medium-to-long term, this is by no means assured. And this matter is out of the hands of individual actors in the construction industry. Unless a convincing strategy to link afforestation and construction timber growth strategies is implemented, advice to the industry should be as follows. When the embodied carbon of different products and building designs are compared in order to influence selection, the benefit of biogenic carbon storage in wood products should be disregarded entirely. Thus the carbon cost of producing construction materials should be included in the production stage LCAs, but the biogenic carbon content should not be subtracted, which is as required by international standards such as EN 15978:2011 (Sustainability of Construction Works ). This advice does not necessarily apply to products from fast-growing crops, but the relevant standards do not make any distinction between products from trees and from crops.

Even without this accounting advantage, timber-based options for building designs still have lower embodied carbon than non-biobased materials in many contexts, and this will continue to be the case until the economy is substantially decarbonised. But this advantage should not be taken for granted: practitioners should continue to assess the role of timber products in the whole life embodied carbon of the building, including how they impact on maintenance, refurbishment and longevity of the building, as well as energy consumption within it. Therefore this thesis makes a distinction between the widespread ramping-up of timber usage in construction and the targeted application of timber in situations where it is the best material to fulfil the required role in the building.

### *Lesson #L3: Re-evaluation needed of carbon storage options*

After years of dialogue on the topic of zero waste and the consequent elimination of landfill from our future waste management infrastructure, it is time to acknowledge that zero waste is a gravity-defying chimera and, accordingly, the role of landfill in carbon storage should be re-evaluated, and modern methods of long-term storage of end-of-life timber products and associated carbon developed as an alternative. If a realistic view is taken of the slow rate of decomposition of timber in landfill, and landfills are managed to optimise storage rather than decomposition, then landfill might be the best option for timber that is reaching end of life. That said, any re-embrace of landfill as currently practised is politically unlikely and comes with practical risks. For instance, it might be argued that the subsidy for biomass burning and the tax on landfill of wood should be exchanged, but this would create perverse incentives that would very probably be exploited by less scrupulous actors: examples might include reburying the same wood in a sequence of landfill sites, or hiding other materials in amongst the wood. A better strategy might be to eliminate biomass subsidies as fast as is practically possible, whilst developing new methods of solid timber product storage, whether above or below ground. Whilst sustainability criteria for biomass combustion generally are more stringent than when many existing contracts were entered into, new power generating contracts under Contracts for Difference still allow the combustion of end-of-life harvested wood products, albeit subject to conditions around energy conversion efficiency. A possible alternative may be found in biochar, but it is worth noting that carbon storage potential is lower.

### *Lesson #L4: Caution needed when projecting substitution benefits into the future.*

Advocates of timber construction's role in climate change mitigation often rely on substitution benefits to make the case. Whilst this can work at the project level (as discussed above in Lesson #L2), the argument breaks down when aggregated substitution benefits are projected into the future. One fundamental objection to this is that a 'climate emergency' is increasingly acknowledged by activists, academics and politicians alike, with the need for the elimination of net GHG emissions by 2050. Therefore, the impacts of construction activity over the next few decades should not be compared with something worse, they should be compared with 'no build' strategies. As such, substitution benefits at the national / global level may be a chimera and should not be included in longer term scenarios. Therefore a discounting system – as used in this work – should be applied to future substitution benefits whereby benefits accrued immediately are worth 100%, and benefits accrued after 2050, or even sooner, will be calculated as being of negligible worth. In some studies, substitution benefits are presented in a way that appears to encourage an almost reckless approach to the use of timber (the more wood used the greater the benefits) regardless of efficiency. This runs counter to the principles of sustainability and circular economy.

### *Lesson #L5: 'Scaling up' can be the problem, not the solution*

As awareness of environmental issues increases around the world, so does interest in the impacts of products, services and technologies, and the choices we make. 'Eco-friendly', 'green', and 'low-carbon' options are identified (often understood as being almost synonymous with 'natural'), pro-environmental consumers become keen to adopt them, industry identifies an opportunity for profit, and governments facilitate the transition.

However, scaling up this green transition can reveal previously hidden impacts (e.g. associated with resource extraction for batteries and turbines), and can push technologies into arenas in which the case for green-ness disappears. A prime example is the combustion of wood, justifiably seen as a green way of heating rural homes from local sources of firewood instead of coal or oil: a solution worthy of support in such – often disadvantaged – communities. But scaling up takes the idea into urban settings, with wood transported longer distances, and emitting particulate and NOx pollution into an already pollutant-loaded atmosphere, whilst also supporting the import of millions of tonnes of wood pellets to burn in power stations.

The patina of green-ness worn by wood products is potentially another example. Whereas in the (somewhat distant) past, building in wood involved a high reliance on what could be obtained in the locale, wood products are now a global commodity, imported from Europe (sometimes via processing sites in the Far East in the case of laminated flooring for instance) and from further afield. Certificates may confirm sustainable forestry practices, but – as this work makes clear – this is far from sufficient to assure low-carbon construction. Therefore, those planning to use timber in buildings should look first at whether they can optimise the quantity they need and get this from local sources. If those sources are found wanting, then they should ask themselves what they can do to support the development of those sources, including afforestation and the development of products and infrastructure that can exploit any resources that are available in abundance.

The other problem with the green image of wood (along with electric vehicles, wind turbines, fuel cells, etc.) is that it subverts the need for deeper systemic change. Instead, it provides the excuse that we need to carry on doing the things we are doing, but in a lightly adjusted variation.

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