FOREST PRODUCTS:

CONTRIBUTION TO CARBON STORAGE AND CLIMATE CHANGE MITIGATION

Giuseppe CARDELLINI

Supervisors:
Prof. Bart Muys (KU Leuven)
Prof. Liesbet Vranken (KU Leuven)
Prof. Wouter Achten (ULB)

Members of the Examination Committee:
Prof. Erik Smolders, chairman (KU Leuven)
Prof. Karen Allacker (KU Leuven)
Prof. Marie-Françoise Godart (ULB)
Prof. Karel Van Acker (KU Leuven)
Prof. Jos Van Orshoven (KU Leuven)
Prof. Gabriele Weber-Blaschke (TUM)

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Dai diamanti non nasce niente,
dal letame nascono i fiori.
(Nothing grows out of diamonds,
flowers grow out of dung.)
Fabrizio De André

A Clara,
per avermi spinto quando la strada era in salita,
e a Flora,
per avermi donato una bellissima discesa.

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Table of Contents

Summary

Climate change is one of the biggest threats for our earth. Mitigation of climate change is thus an urgent challenge our society needs to take up. Many benefits are provided by forests, and one is their potential to mitigate climate change. This mitigating effect can be achieved in many ways, for example increasing the stock of carbon in managed forests or replacing more emission-intensive goods with wood-based products. To maximize the climate mitigation potential of forest and wood products use it is important to correctly quantify their climate mitigating role. A tool to do so is life cycle assessment (LCA), which estimates the environmental burdens of services and goods over their entire life cycle. While this method has been widely used in the past in the forest sector, its application still poses many challenges. Here, we worked to improve the capability of LCA to be used as a tool to assess the climate mitigation potential of forests and wood products. This general context of the thesis is presented in chapter 1.

In the first part of the thesis the challenge was addressed at a more generic LCA level.

Chapter 2 focused on the collection and analysis of data on the current state of forest management practices in Europe. Based on the collected information the free and open EFO-LCI (European Forestry Operations Life Cycle Inventory) database was built. The collected data showed that European forests are quite diverse in many aspects like rotation length, amount and assortments of wood products harvested and machinery used in the interventions. This diversity in the management is also translated into different life cycle impacts. The variability of the input data proved to be an important factor in determining the variability of the Global Warming impact of raw wood production, with the estimated anthropogenic impacts ranging from 0.4 to 73.1 kg CO_2eq/m^3 in EFO-LCI and the biogenic impacts from 1.6 to 451.9 kg CO_2eq/m^3. The release of our regionalized inventory can serve to improve the accuracy of life cycle studies aiming at assessing the relative environmental role of wood production.

Chapter 3 tackled the issue from a more general methodological viewpoint. The lack of temporal resolution in LCA, and of a methodology to solve the Life Cycle Inventory (LCI) dynamically, was addressed due to the relevance of the issue for the forestry sector. Network analysis and convolution were used in combination with the traditional matrix-based structure of life cycle inventory to both solve the LCI dynamically and consider time also in the impact assessment. Following the open source philosophy, the developed approach was also translated into a free and open software named Temporalis. The functioning of the method and the advantages of using a dynamic approach were illustrated with a real-case example. The dynamic life cycle of glulam was performed to show how considering its temporal information can offer new insights into the environmental role played by wood products. If was found that the temporal parameters (i.e. rotation length and product lifetime) used to model the dynamic of biogenic carbon fluxes can greatly influence the results which, for the same system, could range from -71 kg CO_2eq 443 kg CO_2eq when considering a temporal horizon of 20, from -901 kg CO_2eq to 667 kg CO_2eq when considering a temporal horizon of 100 years and from -546 kg CO_2eq to -120 kg CO_2eq when considering a temporal horizon of 500 years.

In the second part, a more applied approach was followed.

Chapter 4 combined the work of the previous two with other data and modelling approaches to assess how European forests would be affected by a change in the management strategies in terms of carbon fluxes, timber harvesting and climate change impact. It was found that timber production is a relatively efficient production chain, with an estimated GWP impact ranging from -1986 kg CO_2eq/m^3 harvested wood to -2989 kg CO_2eq/m^3 harvested wood depending on the year and the scenario. Looking at the overall performance of the system, changing management increases the climate change impact of the system at most of 11% by 2050, with this effect mostly driven by the increased emissions of soil carbon. In the study also the future wood demand was considered and this economic consideration proved to be a decisive factor in shaping the future evolution of European forests. In fact, the realizable changes in forest management were buffered by the constraint posed by the relative demand for timber.

In chapter 5 a dynamic and consequential life cycle-based assessment framework to estimate the climate mitigation potential of actions and policies in the forest-wood sector was proposed and illustrated with an example. In the analyzed case-study it has been shown that the estimated net climate change impact of the systems could range from - 274 to -111 tonnes of CO_2eq/ha/yr by the year 2030 in function of the methodological approach followed. The used accounting procedure influenced the estimated substitution effect which, eventually, was secondary in comparison to the benefits yield by the reduced climate change impact of the system. The results suggested that increasing the climate efficiency of the whole chain should be prioritized over the maximization of the substitution benefits.

This work contributed to improving the quality and availability of the inventory data in the European forestry sector and provided a solution for the issue of temporal consideration in LCA, which allows dealing better with the long production cycles of the forestry-wood sector. It was also learnt that the theoretical mitigation potential of forest management might be constrained by the economy and that reducing the climate change impact of the wood sector rather than maximizing the substitution benefits might be the best climate strategy.

Future research should, among others, focus on the better understating dynamic of wood in the wood sector, for which data are still way to scarce and very little is known about how the resource wood is effectively used along the chain.

Samenvatting

Klimaatverandering is een van de grootste bedreigingen voor onze planeet. Bijgevolg is de mitigatie ervan één van de meest dringende uitdagingen voor onze samenleving. Bossen leveren vele voordelen, waaronder het potentieel om deze klimaatsverandering tegen te gaan. Deze mitigatie kan op verschillende manieren worden ingevuld, bijvoorbeeld door de koolstofvoorraad in beheerde bossen te vergroten of door meer emissie-intensieve goederen te vervangen door houtproducten. Om het mitigatiepotentieel van bos- en houtproducten maximaal te benutten, is het van belang hun mitigerende rol correct te kwantificeren. Een mogelijk instrument hiervoor is levenscyclusanalyse (LCA) die een schatting maakt van de milieubelasting van diensten en goederen gedurende hun gehele levenscyclus. Hoewel deze methode in het verleden veel werd gebruikt in de bosbouwsector stelt de toepassing ervan nog steeds vele uitdagingen. In dit doctoraat werd gewerkt aan het verbeteren van het vermogen van LCA als een instrument om het mitigatiepotentieel van bossen en houtproducten te beoordelen. Deze algemene context werd aangebracht in hoofdstuk 1.

In het eerste deel van dit proefschrift werd de uitdaging aangepakt op een meer generiek LCA-niveau.

Hoofdstuk 2 was gericht op het verzamelen en analyseren van gegevens over de huidige stand van zaken omtrent bosbeheer in Europa. Op basis van de verzamelde informatie werd de, gratis en open, EFO-LCI-databank (European Forestry Operations Life Cycle Inventory) opgezet. De verzamelde gegevens tonen aan dat Europese bossen behoorlijk divers zijn, en dat in meerdere opzichten zoals bedrijfstijd, hoeveelheid en assortiment van houtproducten die worden geoogst en machines die bij de interventies worden gebruikt. Deze diversiteit in het bosbeheer vertaalt zich in verschillende *life cycle impacts.* De variabiliteit binnen de invoergegevens voor LCA blijkt een belangrijke factor bij het bepalen van de uiteindelijke variabiliteit in opwarmingseffecten bij de productie van ruw hout, met de geschatte antropogene effecten van 0,4 tot 73,1 kg CO_2eq / m^3 in EFO-LCI en de biogene effecten van 1,6 tot 451,9 kg CO_2eq / m^3. Het openstellen van onze regionale inventaris kan helpen om de nauwkeurigheid van LCA studies te verbeteren bij het beoordelen van de ecologische rol van houtproductie.

Hoofdstuk 3 behandelde de LCA vanuit een meer methodologisch standpunt. Het gebrek aan temporele resolutie in LCA alsook het ontbreken van een geschikte methodologie om de Life Cycle Inventory (LCI) dynamisch op te lossen werden aangepakt met oog op de hoge relevantie voor de bosbouwsector. Netwerkanalyse en convolutie werden gebruikt in combinatie met de traditionele matrix-gebaseerde structuur van levenscyclusinventarisatie om de LCI op dynamische wijze op te lossen enom tijd als factor op te nemen in de effectbeoordeling. Met de ‘*open source* filosofie’ in het achterhoofd werd de ontwikkelde methode vertaald in een gratis en open software, met name Temporalis. De werking alsook de voordelen van de dynamische benadering werden geïllustreerd aan de hand van een voorbeeld uit de praktijk: de dynamische levenscyclus van glulam laat zien hoe het incorporeren van temporele informatie nieuwe inzichten kan bieden in de ecologische rol van houtproducten. We vonden dat de temporele parameters bedrijfstijd en levensduur van producten de resultaten sterk kunnen beïnvloeden, met voor éénzelfde systeem een waaier van resultaten tussen -71 kg CO_2eq en 443 kg CO_2eq bij een tijdshorizont van 20 jaar, tussen -901 kg CO_2eq en 667 kg CO_2eq bij een tijdshorizont van 100 jaar, en tussen -546 kg CO_2eq en -120 kg CO_2eq bij een tijdshorizont van 500 jaar.

In het tweede deel werd een meer toegepaste benadering gevolgd.

<u>In hoofdstuk 4</u> werden de vorige twee hoofdstukken gecombineerd met nieuwe gegevens en modellen om zo te beoordelen hoe Europese bossen zouden worden beïnvloed door een verandering in de beheerstrategieën in betrekking tot koolstoffluxen, houtkap en klimaateffecten. Er werd aangetoond dat houtproductie een relatief efficiënte productieketen is met een geschatte GWP-impact variërend van -1986 kg CO_2eq / m^3 geoogst hout tot -2989 kg CO_2eq / m^3 geoogst hout, afhankelijk van het jaar en het scenario. Wanneer er wordt gekeken naar de totale prestaties van het systeem blijkt dat een verandering in beheer de klimaatimpact van het systeem met hooguit 11% zal verhogen tegen 2050. Dit effect zou voornamelijk een gevolg zijn van de verhoogde uitstoot van koolstof uit de bodem. In deze studie werd ook de toekomstige vraag naar hout in rekening gebracht. Deze economische overweging bleek een beslissende factor te zijn voor de toekomstige evolutie van Europese bossen. De realiseerbare veranderingen in bosbeheer worden voornamelijk gebufferd door de beperking die de relatieve vraag naar hout oplegt.

In <u>hoofdstuk 5</u> werd een dynamisch levenscyclus-gebaseerd kader voorgesteld om het mitigatiepotentieel van beleid en acties in de bos- en houtsector te beoordelen. Dit werd vervolgens geïllustreerd met een *case study* die aantoonde dat, in functie van de gevolgde methodologische aanpak, de geschatte netto klimaatimpact van de systemen kan variëren van -274 tot -111 tonnes CO_2eq/ha/yr tegen het jaar 2030. De gebruikte accountingsmethode beïnvloedde het geschatte substitutie-effect dat uiteindelijk ondergeschikt is in vergelijking met de voordelen door de verminderde klimaatimpact van het systeem. De resultaten suggereren dat het verhogen van de klimaatefficiëntie van de hele keten prioriteit moet zijn boven het maximaliseren van de substitutievoordelen.

Dit werk draagt bij aan de verbetering van de kwaliteit en beschikbaarheid van de inventarisgegevens in de Europese bosbouwsector en biedt een oplossing voor het gebrek aan temporele resolutie bij LCA. Als gevolg is LCA beter in staat de lange productiecycli van de bosbouwsector te evalueren. Bovendien toont dit werk aan dat het theoretische mitigatiepotentieel van bosbeheer kan worden beperkt door de economie en dat het verminderen van de klimaatimpact van de houtsector een betere strategie is dan het maximaliseren van de substitutievoordelen.

Toekomstig onderzoek zou zich, onder meer, moeten richten op de huidige stand van zaken binnen de houtsector, waarvoor gegevens nu nog steeds erg beperkt zijn, en hoe hout als grondstof effectief wordt ingezet over de hele keten.

List of abbreviations

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CF characterization factors

Cfun characterization functions

CH_4 Methane

CO_2 Carbon dioxide

CO_2eq Carbon dioxide equivalent

dCF dynamic characterization factors

DF Displacement factor

dLCA dynamic Life Cycle Assessment

DQD Data Quality Distance

DQG Data Quality Goals

DQI Data Quality Indicator

EFO-LCI European Life Cycle Inventory of Forestry Operations

ESPA enhanced structure path assessment

EU European Union

FoU Forest Unit

FU Functional Unit

GHG greenhouse gas

GPP Get Primary Productivity

GWP Global Warming Potential

IA Impact Assessment

ISO International Organization for Standardization

LCA Life Cycle Assessment

LCI Life Cycle Inventory

LCIA Life Cycle Impact Assessment

LULUCF land use, land use change and forestry

N_2O Nitrous oxide

NFI National Forest Inventory

NPP Net Primary Productivity

PEM partial equilibrium model

QC Quality Class

RCP Representative Concentration Pathways

RF Radiative Forcing

RI Replacement intensity

SF_6 Sulfur hexafluoride

SS silvicultural system

tDQI total Data Quality Indicator

TH time horizon

TSG tree species group

UNFCCC United Nations Framework Convention on Climate Change

List of units

Mt	Megatonnes
kg	kilograms
MJ	Megajoules
t	tonnes
ha	hectares

List of tables

List of figures

Chapter 1

Introduction

1. Introduction

Climate change is happening and it is mostly driven by human activities (IPCC 2013). The order of magnitude of the future changes in the global climate is still uncertainty, nevertheless, it might strongly affect our daily lives. It has been estimated that an increase of the global temperature of 3 °C by the end of this century could lead to a reduction of up to 25% of the global GDP (Burke et al. 2018). Failing to mitigate climate change might have tremendous costs for the society. It is necessary to act now and with all possible instruments to reduce the risk of unintended consequences. The forest sector can contribute to achieving this goal (Canadell and Raupach 2008), and the goal of this dissertation is to contribute to a better assessment of the potentiality of the sector as climate mitigator in an EU context.

1.1 Climate change

The warming effects of human activities on the climate system are clear and this dynamic will continue until the emission of greenhouse gases (GHGs) will be reduced (IPCC 2013). The warming effect is mainly due to the burning of fossil fuels and to land cover change, which together increases the concentration of atmospheric GHGs. These GHGs modify the radiative budget of the earth system, leading to a net increase in the atmospheric temperature. The main responsible and biggest contribution to this alteration of the radiative forcing is CO_2 (IPCC 2013) which changed from the 280 ppm of the pre-industrial era to the current concentration of 409 ppm (NOAA 2018). By the end of the 21st century, the global surface temperature is expected to increase between 1 and 4.3 °C in comparison to the average levels from year 1850 to 1900 (IPCC 2013). Other impacts like sea level changes, increase in the disparity between wet and dry seasons and regions, the shrinking of the Arctic sea ice cover and the increased acidification of the oceans, are also anticipated. To limit the harmful effect of climate change substantial and sustained reductions of GHGs are needed and mitigation strategies are being put in place to reduce these emissions.

A landmark toward a transition to a low-carbon society has been the implementation of the Kyoto protocol. During the 90s the United Nations initiated a negotiation that led to the foundation of the United Nations Framework Convention on Climate Change (UNFCCC) in 1994. Eleven years after the signing of the UNFCCC treaty the Kyoto protocol came into effect in 2005. With the signing of the protocol 37 countries and the EU-15 committed themselves to reduce their GHGs emissions based on the defined binding targets for the initial commitment period (2008-2012). The follow-up to the Kyoto protocol was represented by the Paris Agreement, which was signed in 2016 and ratified by 175 of the 197 signatory parties by April

2018. The central goal of the new agreement is to limit global warming below 2 °C above pre-industrial levels for this century.

1.2 Forests and climate

Already in Kyoto, the international community acknowledged the role played by forest ecosystems in combatting climate change. The actual role of forests in meeting the identified reduction targets was defined in the Accord of Marrakech in 2001. But it was only in the Paris agreement that the potentiality of the forest-based-sector to provide climate benefits hit the headlines. Here most countries expect the LULUCF sector to make a substantial contribution towards meeting their emissions reduction targets. The global LULUCF is in fact estimated to contribute for some 25% of the total Intended Nationally Determined Contributions (INDCs) of the parties (Grassi et al. 2017).

Forests can influence the global carbon cycle by capturing the atmospheric carbon through photosynthesis and by storing it in the forest ecosystem (Mackey et al. 2013). But the role of forest in the climate arena is not restricted to its sink effect. Broadly speaking forests and their products can contribute to climate change mitigation in 3 different ways:

- Capturing the carbon from the atmosphere through photosynthesis and sequestering it in the biomass and in the soil before it is released back into the atmosphere after a certain period.

- Storing carbon in wood products for an additional time, short or long depending on the lifetime of the product and the final use of it (i.e. if recycled, disposed in landfill or burned), before it is fully or partially reemitted into the atmosphere.

- Substituting fossil fuel, directly by using biomass for energy instead of fossil sources (named fossil fuel substitution), and indirectly by substituting other materials or products with equivalent function but made of more fossil energy-intensive materials (product substitution)

All these processes can influence the carbon cycle, the GHGs concentration in the atmosphere and their consequential global warming impact. Despite the scientific community agrees about these mitigating effects, it is not clear yet what has to be the optimal way of using the resources to maximize the benefits provided by the sector and the debate within the scientific community is still active (De Wever 2017). This controversy is also due to the inherent methodological difficulties of quantifying the relative contribution of the different stages along the chain and the importance of the three aforementioned effects. The accounting of the mitigation effect of the forest sector is related to questions about the system boundaries definition, GHGs to include, the definition of the baseline against which the effects has to be measured, the spatio-temporal pattern of emissions and removals and their climate change impact (Holtsmark 2013, Helin et al. 2013).

Box 1: The role of forests in the carbon cycle

Circa 44% of the emissions from human activities are accumulated in the atmosphere, 26% in the ocean and the remaining 30% on land. With a global area of some 40 thousands km2 forests cover about a third of the total land area (Keenan et al. 2015) and the net global forest carbon sink from 1990 to 2007 was estimated to be equal to 2.4 ± 0.4 Pg C year-1 without taking into account the land use changes (Pan et al. 2011). 9% of the global annual GHGs emissions are offset by the forest sink while, at European level, forest sinks are almost 10% of the anthropogenic emissions (Nabuurs et al. 2015). The global forest carbon stock is around 300 Pg C (estimated range 240–500 Pg C) and two-third of world forests are managed. In EU-27 the stock of carbon in wood products has been estimated to equal 814 Mt C in 1990, roughly 6 % of the carbon stored in the whole forestry sector (living trees + soil) (Karjalainen et al. 2003).

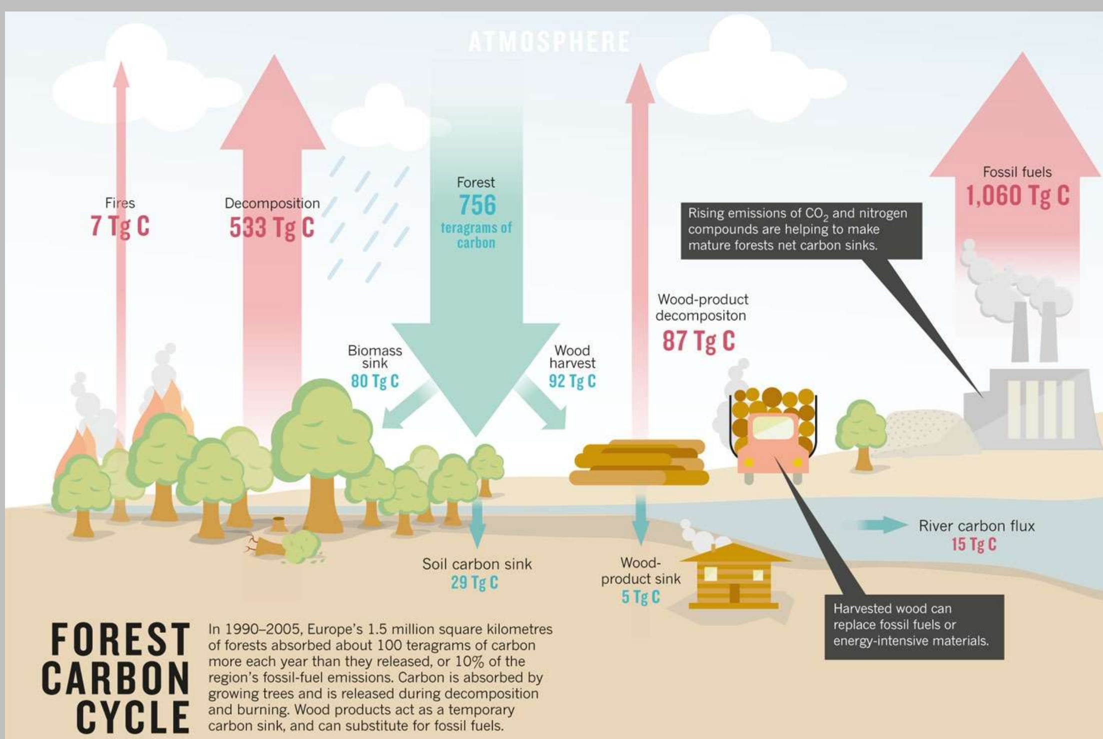


Figure 1.1 Representation of the forest carbon cycle of Europe. Fluxes are in Tg C yr-1 over a forest area of 1.46 x 106 km2. From Bellassen and Luyssaert (2014).

Forest and wood products play also a central role in the ambitious EU bioeconomy strategy to reduce its dependency on fossil fuels (Van Renssen 2014). A future increase in the demand for wood-based products is expected, and in Europe some extra 160 million m^3 yr^{-1} of roundwood are estimated to be required by 2030 (Mantau 2015). This increased demand for wood rise some concerns about the environmental consequences of this future intensified biomass extraction and use (Schulze et al. 2012). To ensure a climate-smart and sustained sourcing of wood it is important to correctly quantify the climate change impact of the sector and of its future changes.

1.3 Life Cycle Assessment (LCA)

Life cycle assessment (LCA) is one of the techniques used to assess the environmental sustainability of products and actions. According to the International Organization for Standards (ISO), LCA is a method used to quantify and help in the understanding of the potential impacts associated with products manufacturing, use and disposal aiming to (i) spot opportunities for improvement of environmental performance; (ii) give information to the industrial and policy decision-makers on the design or re-design of manufacturing processes and on the product development; (iii) choose and quantify indicators of environmental performances (ISO 2006a). The holistic focus on products' lives and/or related processes and functions and the consideration of upstream and downstream activities is the central characteristic of LCA. Life Cycle Assessment covers all the stages in the life cycle of a product, material, service or activity in order to include all environmental impacts potentially arising from it. These stages generally hold the acquiring of raw material, the processing phase, the distribution phase, the use phase and the disposal phase. The methodological LCA framework consists of four distinct though interdependent phases Figure 1.2: a) goal and scope definition, b) inventory analysis, c) impact assessment and d) interpretation (ISO 2006a, 2006b). The phase in which the goal and scope of the LCA are defined is determining for the whole study. In this phase not only the general aim and specific objectives of the LCA are defined, but also methodological choices need to be made on the functional unit, the system boundaries, the impact categories that will be assessed and on the allocation procedure that will be used, in case a process delivers multiple useful outputs.

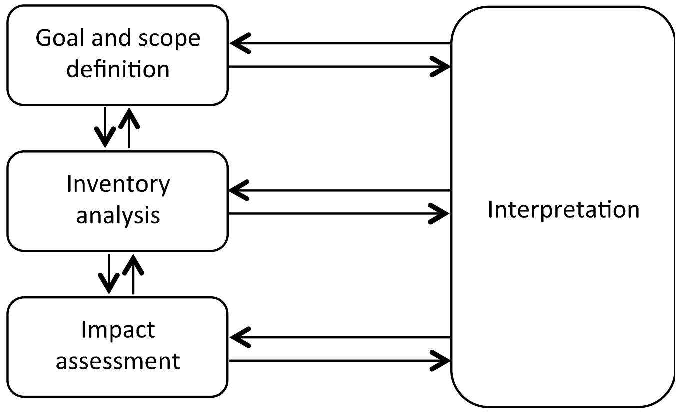


Figure 1.2 LCA stages, based on Rebitzer et al. (2004)

In case of a comparative LCA also a reference system should be defined to which the products or services under research can be compared. The other phases are executive steps to unfold the methodology defined in the goal and scope phase. In the inventory analysis all input and output data, relevant to the system boundaries, are gathered and linked to the related substances emitted to and extracted from the environment (soil, water, air). In the impact assessment phase, these substances are classified and characterized among different environmental impacts in order to quantify the magnitude of these potential environmental impacts. After the classification, where each inventory parameter is assigned to the choosen impact categories, the different environmental impacts are quantified based on the amount of substances emitted to the environment and their characterization factor.

1.4 Attributional vs consequential LCA

There are essentially two main types of LCA: attributional (ALCA) and consequential (CLCA). These two approaches aim to answer different questions and consequently have distinct applications and approaches (Table 1.1). ALCA seeks to provide information about the burden associated with the production, use and disposal of a specific product or process (the functional unit). It is essentially an accounting procedure that looks at the direct effect of the system and studies its average attributes without looking at the effects arising from a change in the systems. On the contrary, CLCA attempts to provide information about the direct and indirect consequences of a proposed change in the system under study under the form of a change in the functional unit. The different scope of the two approaches is reflected also in the different way the systems are modelled. For example, while ALCA is based on average data, CLCA use marginal ones.

Table 1.1 Overview of the key differences between Attributional (ALCA) and consequential (CLCA) LCA.

Characteristic	Type of LCA	
	ALCA	CLCA
Research question	How things are?	What if?
Purpose	Accounting	Decision making
Perspective	Retrospective	Prospective
System boundary	Completeness	Affected processes
(Co-)products accounting	Allocation/system expansion	System expansion (substitution)
Uncertainty of results	Lower	Higher
Indirect effects	Not included	Included
Data	Average-historical	Marginal-future

1.5 LCA and the forest sector

Despite being widely used in the past (Klein et al. 2015), the life cycle assessment of the forestry-wood sector has always posed many challenges (Schweinle 2007, Sandin et al. 2016). The relative complexity and variability of the forestry-wood system, together with a general shortage of data available, make the use of life cycle techniques in this context arduous. A good example of the methodological difficulties encountered when applying LCA in this sector is represented by the treatment of allocation. Another important aspect is the relatively long time-frame of the life cycle of wood-based products when compared to other products studied with LCA. The temporal aspects of the studied system are typically not captured in the traditional LCA approach. The lack of consideration of the temporal and spatial variation of flows and emissions, was in fact considered one of the major shortcomings of LCA practice until recent times (Reap et al. 2008). While for the spatial limitations methodologies have been proposed (Mutel and Hellweg 2009) and operationalized (Rodriguez et al. 2014), only a few approaches to deal with the temporal aspects have been proposed (Beloin-Saint-Pierre et al. 2014, 2017, Tiruta-Barna et al. 2016) but are not yet operationalized. This inadequate consideration of the temporal dynamics of the sector still represents an important limitation to the correct application of LCA in the forest-based sector (De Rosa et al. 2017a).

1.6 The issue of biogenic carbon accounting

Biogenic carbon fluxes are those originating from biological sources like trees and soil. In contrast, anthropogenic fluxes are those associated with human activities (e.g. land use change and fossil fuel burning). The accounting of biogenic carbon flows and the translation of these flows into their warming impact is particularly challenging for the forest sector (Sandin et al. 2016). Historically, LCA has always treated biogenic carbon as climate neutral (Johnson 2009, Haberl et al. 2012), with its accounting that was neglected (Pawelzik et al. 2013). This assumption was based on the rationale that the biogenic carbon emitted via the decay and/or combustion is sequestered back in the growing biomass when the condition of sustainable forest management is met. The validity of the carbon neutrality assumption is very case-specific and depend, among others, on the spatial scope of the study (e.g. stand vs landscape-level study) (Cintas et al. 2017) and the baseline against which the biogenic carbon dynamic in the forest is accounted for (constant vs changing carbon stock levels) (Holtsmark 2013). The limitation of the neutrality approach has been acknowledged from the LCA community and several technical standards dealing with LCA and carbon footprint of wood-based products require now the explicit accounting of biogenic carbon (Tellnes et al. 2017). Despite this step forward, no consensus exists between standards on how to include biogenic carbon modelling in LCA (Table 1.2). To complicate things further, the temporal explicit accounting of biogenic carbon fluxes does not suffice to estimate their warming impact. This is due to the temporal shift between the emission and sequestration of biogenic carbon from forests that contribute to a transient change in the radiative balance in the atmosphere that is not proportional to the carbon fluxes (Helin et al. 2013). Even in those cases when it might be valid to assume the carbon neutrality of forests, this carbon neutrality is not automatically translated into the climate neutrality. To properly estimate the warming impact of this biogenic carbon it is thus necessary to know the temporal profile of the biogenic carbon emission and sequestration and convert them into their climate change impact using the so-called GWPbio (Cherubini et al. 2011a, 2012). This metric allows accounting for the peculiar behaviour of biogenic carbon in comparison to the anthropogenic one.

Table 1.2 Overview on how technical standards for carbon footprint of products and EPDs methodologically deal with biogenic carbon accounting. Modified from Tellnes et al. (2017).

Criteria	**EN-15804 (2012)**	**ISO-21930 (2015)**	**EN-15804 (2012) +A1:2013**	**CEN/TR - 16970 (2016)**	**EN-16485 (2014)**	**PEF PilotGuide v2.2**	**ISO/TS - 14067 (2013)**	**PAS-2050 (2011)**
Instant oxidation allowed	Not specified	Not specified	Not specified	Not specified	Not specified	Yes	Compulsory for emissions less than 10 years	For food
Considers biogenic carbon in by-product allocation	Yes	Yes	Yes	Not specified	Yes	Not specified	Yes	Not specified
Consider biogenic carbon flows on GWP	Not directly, but by reference to ILCD method	Yes	Not specified	Yes	Yes	Yes	Yes	Yes
Modular approach to emissions required	Yes	Yes	Yes	Yes	Yes	No	Yes	Not specified
Criteria for separate biogenic carbon flows in inventory	Not specified	Yes	Not specified	Yes	Yes	Yes	Yes	Not specified
Considers sustainable harvest of biomass	Not specified	Yes	Not specified	Yes	Yes	-	Yes, but with land use change	Not specified

Criteria	EN-15804 (2012)	ISO-21930 (2015)	EN-15804 (2012) +A1:2013	CEN/TR - 16970 (2016)	EN-16485 (2014)	PEF PilotGuide v2.2	ISO/TS - 14067 (2013)	PAS-2050 (2011)
Possible to include effect of delayed emissions on GWP	Not specified	No	Not specified	Not specified	No	No	No	No
Possible to include effect of delayed emissions separately	Not specified	Yes	Not specified	Not specified	Yes	Yes	Yes	Yes
Final storage	Not directly, but sets a limit for 100 years	Yes	Not directly, but sets a limit for 100 years	Yes, for landfill	No	Yes	No	Yes
Requires additional information relevant to biogenic carbon	Not specified	Biogenic carbon in materials leaving the product system as technical scenario information	Not specified	Not specified	Apparent density and moisture content of wood, amount of biogenic carbon stored	-	Not specified	Use phase removals and emissions included shall be recorded, carbon storage, land use change

1.7 The role of wood products substitution

Forests can mitigate climate change by sequestering atmospheric CO_2 in biomass and in the soil. But, also after biomass is harvested and extracted from the forest, this biomass can continue to contribute to the climate change mitigation potential of forests. By using the biomass in wood products, for example, it is possible to increase the stock of carbon in wood products or in landfills and as such contribute to mitigate climate change (Brunet-Navarro et al. 2016). But, the use itself and the function such forest products deliver can also trigger the substitution effect, additionally contributing to CO_2 mitigate the changes in climate (Sathre and O'Connor 2010). Some wood products, in fact, can be used as a replacement of other more GHG intensive products that provide equivalent services or functions. This replacement can trigger a substitution effect and that results in climate change mitigation through reduced GHG emissions obtained by reducing fossil-fuel based production (Lippke et al. 2011). The substitution effect exerted from a product depends on the reduced warming impact obtained by replacing (and as such avoiding the production of) a conventionally used product with the same function (Gustavsson and Sathre 2011).

Calculating substitution effects is not straightforward. Quantification of this effect faces three main problems: (i) which product or technology is substituted? (ii) how much of this product or technology is effectively substituted? (ii) when (if) are these positive benefits exerted?

It can be argued that a new product will likely substitute the least preferred products or technologies (e.g. based on costs, financial or societal) delivering the same function. However, the reality shows that a product can substitute several existing products. E.g. bioenergy can substitute for oil, coal, gas, but also for an already renewable energy source (Hu and Cheng 2017).

A new product is often assumed to fully replaces or substitutes the substituted product. However, in reality this will be rarely the case. By putting an extra product on the market direct and indirect effects are triggered with a real replacement rate that might be different than expected. For instance, York (2012) showed that over the last half-century less than one-quarter of a unit of fossil-fuel energy was displaced by energy use from non-fossil-fuel sources and each unit of electricity produced by non-fossil-fuel sources displaced less than one-tenth of the one coming from fossil-fuel. In consequential-based life cycle analysis (Earles and Halog 2011) these aspects are normally tried to be included and taken into account. Attributional LCA aims at attributing the environmental impacts associated with a specific product system and, as such, is essentially an accounting procedure. Consequential LCA, on the contrary, has the main goal of estimating the environmental consequences of changes in the system and, as such, tries to consider also indirect environmental effects.

1.8 The issue of transparency in LCA and in the forest sector

One of the fundamental issues faced by the LCA community is the lack of reproducibility (Vafi and Brandt 2014), which is in turn mostly driven by a general lack of transparency in the sector (Frischknecht 2004, Pauliuk et al. 2015).

This issue of data accessibility and transparency is quite widespread around the scientific community nowadays. A 2010 survey across scientists coming from different fields, for example, found that the lack of access to research data was considered to hinders the progress in science from most of the respondents, while nearly half of the participants agreed about the fact that the ability to answer scientific questions is limited by this non transparent behaviour (Tenopir et al. 2011).

The widespread use of commercial and proprietary software and databases, together with the reluctance of data owners to publicly disclose information, severely contribute to this general reduction of the quality and confidence in LCA. Also for the LCA studies in the forestry and wood sector reproducible and comparable results are hardly published (Klein et al. 2015). While some initiative has already been put in place to make the sector more transparent (Pauliuk et al. 2015, De Rosa et al. 2017b, Hertwich et al. 2018) still a lot of work needs to be done.

1.9 Aim and objectives

The general objective of the thesis is to contribute to the improvement of the assessment of the climate change mitigation role of the forest sector, with a focus on the European context.

It was first deemed necessary to address two of the main aspects which makes the application of LCA in this context difficult and uncertain, namely the scarcity of regionalized data and the methodological limitation owing to the lack to temporal consideration in the LCA analysis.

It was thus hypothesized that to advance in the field, it was necessary to:

- Improve the availability of regionalized data about forest management and operations in Europe
- Tackle the methodological limitation linked to the lack of the explicit consideration of temporal aspects in LCA

These hypotheses led to the following research questions:

1. To what extent the forest management practices differ in Europe?
2. Are these differences (if existing) affecting the climate change impact of wood production?
3. Can the traditional static LCA methodology be modified to consider time in both the LCI and LCIA?

4. Does the previous methodological advance provide added value for the life cycle accounting of the climate change impact of wood products?

Once solved these more general LCA issues, the added value that such improvements could bring to the assessment of the climate mitigation potential of the forest sector could be investigated. The hypotheses were that:

- The data collected and the dynamic LCA method developed could be integrated with other modelling approaches to assess the impact of European forest management
- The temporal explicit accounting of the substitution benefits of wood products could help to better understand the climate mitigation effect of such products

Based on these two hypotheses the following questions were answered:

5. What can the use of an integrated approach in which dynamic LCA and partial equilibrium model are integrated with forest model tell us about the effect of different forest management practices in Europe?
6. What is the effect of considering temporal aspects in the assessment of the substitution effects of wood products?

1.10 Outline of the thesis

The lack of regionalized data, linked to the research questions 1 and 2, was addressed in chapter one. Here a new open source LCI database of forestry operations for the European region was developed and released under *Creative Commons AttributionNonCommercial-ShareAlike (CC BY-NC-SA)* license (Creative Commons 2016). In chapter 3 the temporal limitations of the current LCA practice were tackled and the research questions 3 and 4 were answered. In this chapter, a dynamic LCA methodology was developed and operationalized as an open source extension of the Brightway2 LCA software (Mutel 2017b). The methodology and the software allow resolving the LCA dynamically both at the level of the LCI and the LCIA and considering the temporal evolution of the fluxes and impacts.

The two works made in chapters 2 and 3 laid the foundations for the work performed in chapter 4 which aimed at answering the research question 5. The EFO-LCI data were combined with the simulations coming from the FORMIT-M simulator (Härkönen et al. 2018) and the partial equilibrium model EFI-GTM (Kallio et al. 2004) and analysed in Temporalis to perform a dynamic life cycle-based analysis of the climate change impact of the European forest management. Finally, in chapter 5 the research question 6 was addressed and the importance of temporal aspects in the assessment of substitution effects of wood products use was assessed and a dynamic and consequential life cycle-based approach to account for these effects has been proposed.

In Figure 1.3 an overview of the structure and research presented in this thesis is provided.

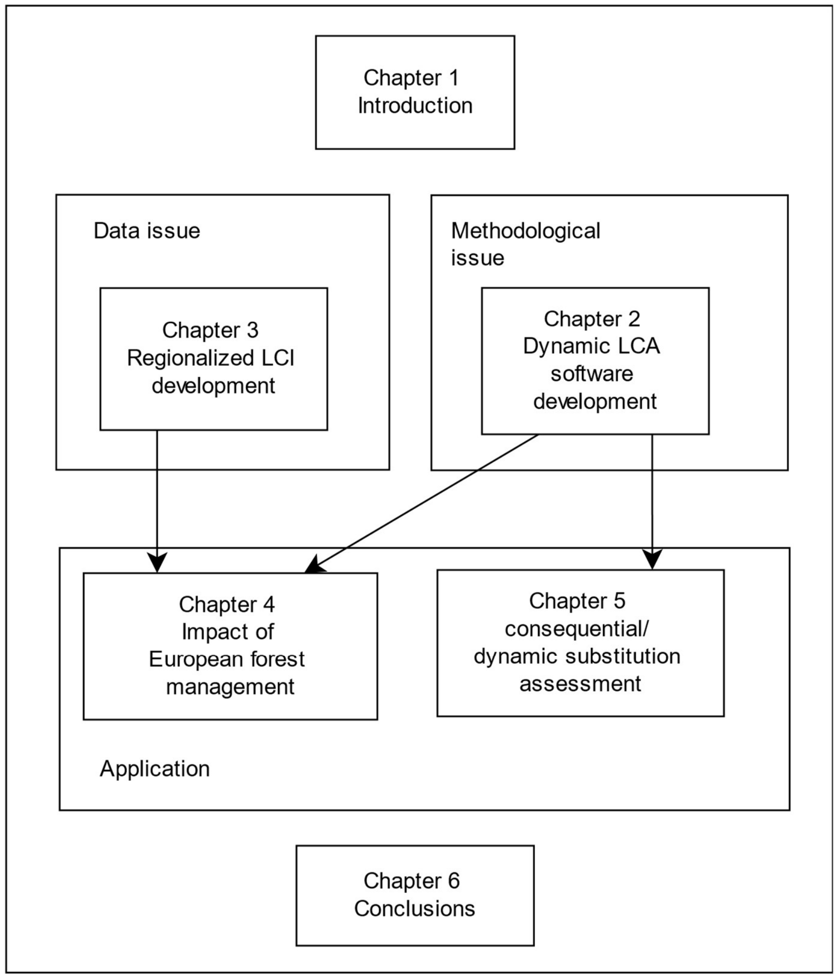


Figure 1.3 Outline of the PhD thesis structure

Chapter 2

EFO-LCI: a new Life Cycle Inventory database of Forestry Operations in Europe

Adapted from: Cardellini, Giuseppe, et al. "EFO-LCI: A New Life Cycle Inventory Database of Forestry Operations in Europe". Environmental management 61.6 (2018): 1031-1047

1. Abstract

Life Cycle Assessment (LCA) has become a common methodology to analyze the environmental impacts of forestry systems. Although LCA has been widely applied to forestry since the 90s, the LCAs are still often based on generic Life Cycle Inventory (LCI). With the purpose of improving LCA practices in the forestry sector, we developed a European Life Cycle Inventory of Forestry Operations (EFO-LCI) and analyzed the available information to check if within the European forestry sector national differences really exist. We classified the European forests on the basis of "Forest Units" (combinations of tree species and silvicultural practices). For each Forest Unit, we constructed the LCI of their forest management practices on the basis of a questionnaire filled out by national silvicultural experts. We analyzed the data reported to evaluate how they vary over Europe and how they affect LCA results and made freely available the inventory data collected for future use. The study shows important variability in rotation length, type of regeneration, amount and assortments of wood products harvested and machinery used due to the differences in management practices. The existing variability of these activities sensibly affects LCA results of forestry practices and raw wood production. Although it is practically unfeasible to collect site-specific data for all the LCAs involving forest-based products, the use of less generic LCI data of forestry practice is desirable to improve the reliability of the studies. With the release of EFO-LCI we made a step towards the construction of regionalized LCI for the European forestry sector.

2. Introduction

About 44% of European land area (including the Russian Federation) is covered by forests and other wooded land, representing approximately 25% of our global forest resources (Forest Europe et al. 2015, Keenan et al. 2015). Forest ecosystems contribute greatly to the provision of several marketed and non-marketed goods and services like construction wood, fuelwood, recreation, fresh water supply, soil protection and climate regulation. Within the context of this last ecosystem service, forests play an important environmental role as they are estimated to annually remove 719 million tons of CO_2 from the atmosphere, about one-tenth of the European greenhouse gas emissions (Forest Europe et al. 2015). Among the many functions provided, forests produce wood, the main renewable resource used to produce both wood products and energy. With its annual production of 429 million m^3, EU-28 is the largest manufacturer of roundwood within the G20 (Eurostat 2014). Furthermore, wood and other solid biomass are the largest contributors to the mix of European renewable energy sources (Eurostat 2014). According to Eurostat statistics, half of Europe's renewable energy came from wood and wood waste in 2010 (Eurostat 2015). The importance of forest bioenergy in Europe is very likely to increase in the near future due to the momentum created by the European climate and energy policies (EU 2009). The consequent expected increase in demand for wood-based products poses serious concerns about the environmental consequences of this intensified biomass extraction and use (Schulze et al. 2012). The correct quantification of the eco-environmental impacts from forestry practices is thus crucial to keep management sustainable. While the concept of sustainability gained general acceptance only after the publication of the Brundtland report (WCDE 1987), the principle was already well established and operationally implemented in the forestry sector long before (Wiersum 1995). Life cycle assessment (LCA) is one of the main techniques used to assess environmental sustainability which consists of three main phases: (i) goal and scope definition, inventory analysis and (iii) impact assessment. In the goal and scope the functional unit is defined, i.e. the function fulfilled by the studied system, which provides a reference against which inputs and outputs are related. In this step also system boundaries and the allocation method are defined. The inventory analysis is the phase involving the compilation and the quantification of inputs and outputs between the nature and the system studied (elementary flows) throughout its life cycle. The Life Cycle Inventory (LCI) model for the studied system is built in this phase and typically consists of two components: the foreground model, which is the modelled part whose production can be affected from the commissioner and for which primary data are typically used, and the background model, in which data normally come from generic LCI databases. In the impact assessment, the elementary flows are translated into their potential environmental impacts. This is obtained by multiplying the amount of each elementary flow with their characterization factors for the impact categories chosen. The overall environmental load of the system is then obtained summing up the indicator scores calculated for all the flows.

The first LCAs of the European forestry and timber industry already appeared during the early 1990s (Frühwald and Wegener 1993, Solberg and Frühwald 1995). Despite this early development of life cycle thinking in the sector, a harmonized, consistent and accepted methodology has yet to be found (Klein et al. 2015).

2.1 Main methodological issues in forestry LCA

Due to the relative complexity of the forestry-wood sector (Schweinle 2007, Sandin et al. 2016), many challenges are posed to the correct implementation of LCA analysis in this context (Bosner et al. 2012, Sandin et al. 2016), namely:

- ✓ the long time frame of wood production and use compared to other products studied;
- ✓ the spatial variability of forest stands and the site specificity of growth and management;
- ✓ the broad variety of joint products arising from forest production and the variety in their usage pathways e.g. the production of goods coming from thinning activities (roundwood, sawnwood, pulp, chips, etc.);
- ✓ the multifunctional nature of forests that leads to the production of an array of services and functions other than timber (e.g. water, non-wood forest products, recreation, carbon sequestration);
- ✓ the difficulties in accounting and estimating the effects on growth and biogenic fluxes of natural disturbances and climate change;
- ✓ the uncertainty of the end-of-life accounting of long-lived products due to the unpredictable future technological changes.

It has already been shown that the absence of a commonly agreed upon methodological approach, together with the lack of a standardized definition of both the functional unit and the system boundaries, makes the different forest-wood based LCA studies hardly comparable (Sathre and O'Connor 2008, Klein et al. 2015). Recently, the scientific discussion on the forest LCA methodology has been reinvigorated since the proper accounting of biogenic carbon gained a lot of attention due to the so-called carbon neutrality issue (Johnson 2009, Zanchi et al. 2011, Schulze et al. 2012). The alleged carbon neutrality of forests, namely the assumption that all removed biomass coming from sustainably managed forests will be entirely sequestered in the future, and hence can be neglected in LCA, has been questioned and disproved (Johnson 2009, Cherubini et al. 2011a, Zanchi et al. 2011, Schulze et al. 2012, Wiloso et al. 2016). This change in the traditional paradigm called for a more robust and consistent accounting of biogenic carbon which affects the definition of system boundaries, GHGs to include in the analysis, baselines identification and the spatio-temporal pattern of CO_2 fluxes. The methodological difficulties briefly presented so far have been extensively discussed in the literature, with several solutions already proposed (Jungmeier et al. 2002a, 2002b, Werner et al. 2006, Bright et al. 2012, Helin et al. 2013, Downie et al. 2014, Klein et al. 2015, De Rosa et al. 2016, Sandin et al. 2016). Despite the ongoing research on the methodological issues, it is crucial for any representative LCA of a forest-based production system to carefully consider what happens in the forest by means of a proper identification and description of its characteristics and management. In this work, we addressed this need by developing a regionalized LCI of the forest management practices in the European region.

2.2 Main inventory issues in forestry LCA

As acknowledged by (Werner et al. 2007), the Life Cycle Inventory of wood is strongly affected by tree species and forest management. But a detailed description of forestry systems and the inventory of the related material and substance flows is often not easy, as forests are very diverse in terms of species composition, management regime, site productivity, machinery used and socio-economic conditions (Barbati et al. 2006). Seventy-eight European Forest Types are identified and classified in Europe according to structural, compositional and functional key-factors (Barbati et al. 2014). If one also considers the different harvesting systems within forest types, one gets an even more complex myriad of different forest production systems. González-García et al. (2014b) compared the life cycle impact of forest operations in Europe and showed a large variation in terms of biomass productivity and associated environmental impact, even if only seven stands were studied. Despite the described complexity, LCA practitioners commonly base their analysis on Life Cycle Assessment software and their integrated LCI databases (Graedel and Allenby 2010). Also the LCAs of wood-based products typically relies on the generic forest LCI models included in databases like Ecoinvent (Wernet et al. 2016) and GaBi (P.E. International 2016). Forest LCI models included in the standard LCA software packages are based on a limited number of studies in which only a few forest types and management practices are considered, typically temperate plantations or natural forests in optimal conditions and intensively managed (Newell and Vos 2012). Although these databases represent an important source of information, they are not always able to correctly represent the conditions existing in other European forests (Lewandowska et al. 2008). This lack of sufficiently specific data is mirrored in Ecoinvent, which offers LCI data on hardwood and softwood production for nine systems in total, comprising six species (beech, birch, oak, pine, spruce and mixed) and only three countries (Germany, Sweden and Switzerland) for the European region (Wernet et al. 2016). Also GaBi refers to a model developed for forest plantation of central Europe in the wood products models documentation (Frühwald et al. 1997). All these limitations can lead to differences between existing LCI databases and the reality if these data are not suitable for the local conditions studied (Lewandowska et al. 2008). Much progress has been made to overcome the methodological limitations of LCA when applied to forestry sector (Jungmeier et al. 2002a, 2002b, Werner et al. 2006, Bright et al. 2012, Helin et al. 2013, Downie et al. 2014, De Rosa et al. 2016, Sandin et al. 2016, Daystar et al. 2017). The same cannot be said for data quality and availability, for which considerable efforts are still needed (Lippke et al. 2011). With this work we want to contribute to this end and, as such, to a better and more accurate profiling of the LCI of forest operations. We accomplished this through a large survey about the forest management practices in Europe. In this paper, the main findings of this data collection will be presented and the main differences in silvicultural practices found in the European forests will be highlighted to demonstrate the importance of more regional specific modelling of the management regimes in LCA. The need for better inventories of the sector will not only be claimed by presenting and discussing the results of the survey and their impact on LCA, but has also been operationalized by disclosing and making all the data freely available and usable under Creative Commons Attribution-NonCommercial-ShareAlike (CC BY-NC-SA) license (Creative Commons 2016). With this disclosure, we want to contribute to a more accurate life cycle study of the European forest production systems, and also further stimulate a major level of transparency and reproducibility in LCA studies, for which the importance has been already stressed in several previously published studies (Frischknecht 2004, Finnveden et al. 2009, Pauliuk et al. 2015).

3. Methods

3.1 System boundaries

The geographical boundary of this analysis consists of 28 countries within the European subcontinent, as shown in Figure 2.1, subdivided into the five main eco-regions defined in the State of Europe's Forest report (Forest Europe et al. 2015). The process-based system boundary of the study is from cradle-to-forest road and includes the four mandatory systems (i.e. from PG 1 to PG 4) as defined by Klein et al. (2015). The data collected thus characterize all the activities carried out from site preparation to harvested wood delivery at forest road including all relevant primary (namely Site preparation: PG 2; Site tending: PG 3; Silvicultural operations: PG 4 and Secondary processes: PG 1) of the entire forest production chain. The "whole rotation approach" (Klein et al. 2015) of the current management of European forests has been followed for the temporal system boundaries identification. Therefore, also from a temporal perspective, the entire forestry system is taken into account by considering the forest over the whole rotation period, and including all age classes and processes over the development of the stand. In this study, rotation is defined as the time from regeneration to final exploitation for silvicultural systems 3 and 4, between consequent cuttings (i.e. the cutting cycle) for silvicultural systems 5, 6 and 7 and between consequent selection cuttings for silvicultural system 2 (see Table 2.1).

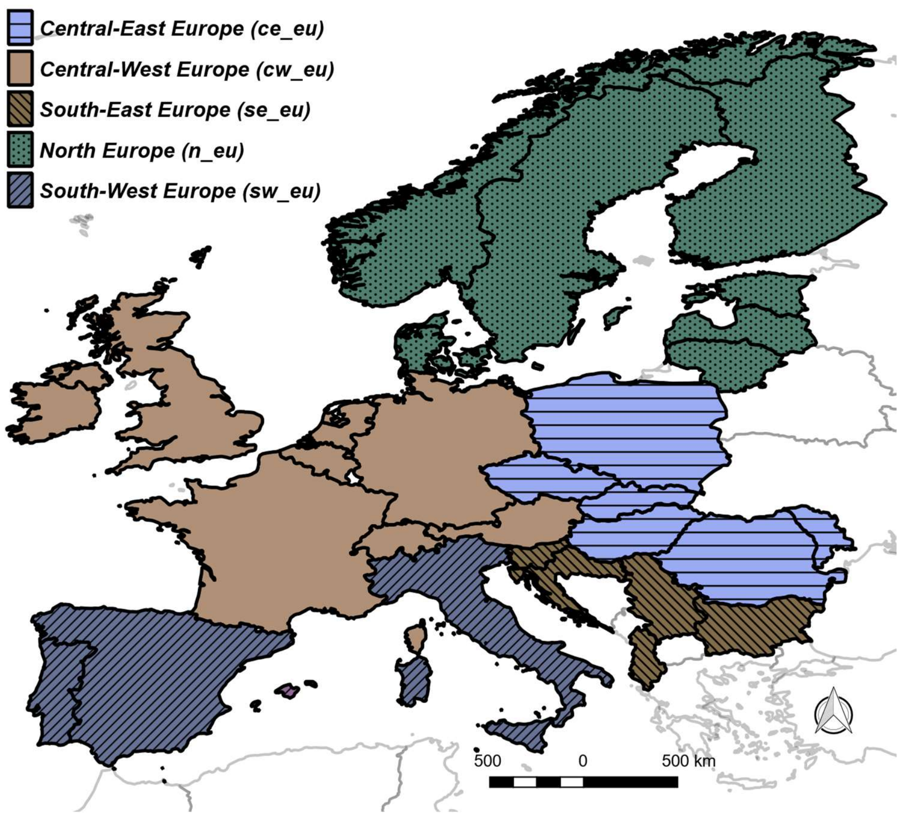


Figure 2.1 Surveyed countries grouped by ecoregion

3.2 Forest classification: Forest Units (FoUs)

To capture and describe all the possible forest types present in Europe, they had to be first qualitatively clustered and classified in a meaningful way to reduce the number of possible combinations without losing relevant information. We followed the classification adopted in the FORMIT (2016) project, where the forests have been classified on the basis of species composition and silvicultural systems, the two aspects that affect inventory results the most, as will also be shown in the following sections. The silvicultural systems were combined considering regeneration methods, forest structure and how the tree canopy is removed (Table 2.1). Forest tree species have been grouped on the basis of their ecological characteristics and their growth strategy, and all the species included in the ICP Forest tree species list (Seidling and Michel 2016) have been considered (Table 2.2).

Table 2.1 Silvicultural systems

Code	System	**Definition**
1	Unmanaged forests	No management
2	Continuous cover forest management	Continuous cover forest management • Selection cuttings based on target diameter
3	Even-aged forest management with shelterwood	Even-aged forest management Regeneration: natural Thinnings Shelterwood cut after a certain mean diameter (or age) has been reached
4	Even-aged forest management: uniform clearcutting system	Uniform forest management Regeneration: planting or natural Thinnings Clear-cut after certain target diameter (or age) has been reached
5	Coppice	Woodland which has been regenerated from shoots formed at the stumps of the previous crop trees, root suckers, or both, i.e., by vegetative means
6	Coppice with standards	Coppice system under low-density uneven-aged high forest
7	Short rotation	Plantation forestry including exotic species

Table 2.2 Tree species groups

Code	**Species group**	**Species**
A	Light-demanding conifers	Pinus sylvestris, Larix spp., Pinus nigra, Pinus cembra, Pinus heldreichii, Pinus leucodermis, Pinus radiata, Pinus uncinata, Pinus mugo, Pinus contorta, Pinus strobus, Cedrus spp., Juniperus spp.
B	Shade-tolerant conifers	Picea abies, Abies spp., Pseudotsuga menziesii, Thuja spp., Taxus baccata, Tsuga spp., Chamaecyparis spp.
C	Mediterranean conifers	Pinus pinaster, Pinus halepensis, Pinus pinea, Pinus canariensis, Cupressus spp., Pinus brutia
D	Fast-growing deciduous	Betula spp., Populus spp., Alnus spp., Salix spp., Robinia pseudoacacia, Eucalyptus spp.
E	Slow-growing light-demanding deciduous	Quercus robur, Q. petraea, Q. cerris, Q. pubescens, Q. faginea, Q. frainetto, Q. macrolepis, Q. pyrenaica, Q. rubra, Q. trojana, Q. hartwissiana, Q. vulcanica, Q. macranthera, Q. libani, Q. brantii, Q. ithaburensis, Q. pontica, Fraxinus spp., Castanea sativa, Rosaceae (Malus, Pyrus,Prunus, Sorbus, Crataegus, etc.), Juglans spp., Cercis siliquastrum
F	Slow-growing, shade-tolerant deciduous	Fagus spp., Carpinus spp., Tilia spp., Ulmus spp., Buxus sempervirens, Acer spp. Ilex aquifolium
G	Mediterranean evergreen trees	Quercus suber, Quercus ilex, Q. coccifera, Q. lusitanica, Q. rotundifolia, Q. infectoria, Q. aucheri, Tamarix spp. Arbutus spp., Olea europea, Ceratonia siliqua, Erica spp. Laurus spp., Myrtus communis, Phillyrea spp. Pistacia spp. Rhamnus spp. (R. oleoides, R. alaternus), Ilex canariensis, Myrica faya

The combination of a species group and a silvicultural system results in what has been called a Forest Unit (henceforth FoU). Within these 49 FoUs (all possible combinations of species groups and silvicultural systems) it is possible to capture and describe the most typical, currently applied, forest management strategies in Europe. By further combining this information with the geographical location, it is possible to take into account also for their geographical diversity.

3.3 Data collection

The data has been collected through a questionnaire developed in MS Access and consisting of three front end forms each of which linked to a back-end relational database (see Appendix I for the link to download it). The questionnaire was organized into four main sections and consisted of a mix of closed-ended drop-down choice and open-ended text box questions. Details on the average silvicultural practices of the forest management were described as follows:

1) FoUs in the Country: a table where the major FoUs present in the country out of the 49 possible had to be indicated.
2) General description: a form where information on the general characteristics and management regime of each FoU were asked for:
   a) composition (main and secondary species);
   b) moisture content of green wood and wood density;
   c) regeneration information of the system, including length of rotation period, type of regeneration, and, if plantation, type of seedlings and density of plantation;
   d) distance traveled (one way) from forest for harvesting equipment and staff;
   e) data source.
3) Interventions: a form where all the activities carried out in the FoUs with their relative timing was described in detail for:
   a) type of intervention;
   b) year of intervention;
   c) species concerned by intervention;
   d) equipment used (main and additional equipment, type, mass, total hours of use during the whole life, productivity, and consumption);
   e) inputs (type and amount);
   f) amount and assortment of harvested wood per hectare.

The questionnaire was submitted to all the 12 project partners of the FORMIT (2016) project. They filled it out for their own country and submitted it to other national forestry experts of neighbouring countries. Each respondent's name has been recorded together with the data source used. An introductory session to the survey was organized with all participants before the launch. A protocol containing all the practical information on how to proceed step-by-step accompanied all the questionnaires and, during the execution, a help-desk was provided to clarify all the doubt arising during the completion of the survey. The respondents were asked, first, to point out the FoUs that they considered the most important in their countries and then, to describe their current, most representative, average silvicultural management applied. The choice of FoUs was done on the basis of their extension as well as on their ecological and economic importance in the country. All the questionnaires were filled out between October 2014 and May 2015.

3.4 LCA impact calculation

To evaluate how the variability of the data influence the LCA impact, the climate change impact for the production of one m^3 of overbark (m^3 ob) raw wood (i.e. harvested wood including bark) was calculated. The analysis considered the GHG emissions of all inputs and management operations occurring along the aforementioned process-based system boundary (see section 2.1). All the foreground data used to model the emissions were taken from EFO-LCI. Specifically, from the database were used the data about: (i) distances travelled from staff; (ii) forest road density; (iii) regeneration type and, in case of artificial regeneration; (iv) number of seedling per ha; (v) type of intervention; (vi) if wood was harvested during the intervention, its amount (in m^3/ha); (vii) type and (viii) operational productivities (in m^3/h or h/ha) of the equipment used; (ix) type and (x) amount of inputs used (e.g. fertilizer) plus (xi) rotation length. Ecoinvent 3.3 and Agribalyse 1.3 (Colomb et al. 2015) were used for the background LCI data about the GHG emission per hour of use of the machinery, unitary seedling production and material used for road maintenance and fencing (see Table A.1). All management interventions reported in EFO-LCI were distinct into generic and specific as reported in Table A.1. In the former are included the activities that are not related to the amount of timber harvested, being dependent on the area of forest regenerated (e.g. ripping), and for which the emissions were calculated on a per ha basis. For the latter (e.g. primary felling) the emissions were directly estimated from the m^3 ob of wood harvested in that intervention for one ha. Based on the operational productivity of the machinery, their total demand per ha over one rotation (in h of use or km for manual activities) was calculated for all generic and specific interventions occurring over a single rotation in each FoU as shown in equation 2.1.

$$D = \begin{cases} C * \dfrac{P_{h/ha}}{8}, & if\, manual\, in\, generic \\ C * \dfrac{W}{\dfrac{P_{m^3/h}}{8}}, & if\, manual\, in\, specific \\ \dfrac{P_{h/ha}}{W}, & if\, generic \\ \dfrac{W}{P_{m^3/h}}, & if\, specific \end{cases} \qquad \text{equation 2.1}$$

with:

D=total use of machinery (in h/ha or km/ha) for the intervention under consideration over one rotation

$P_{h/ha}$= operational productivity (in h/ha) of the machinery for the intervention under consideration

$P_{m3/h}$= operational productivity (in m^3/h) of the machinery for the intervention under consideration

W= m^3ob/ha of wood harvested over one rotation for the intervention under consideration

C=commuting distance for staff (km)

Continuing with the example above, when in a FoU the intervention 'ripping' is reported and the machinery used is 'ripper', the operation productivity in h/ha is used. The total hours of use of the ripper are derived from the third case in equation 2.1. In manual activities only the transport of worker is considered, assuming a working day of eight hours and that the task is performed from two workers. Knowing the daily distance travelled by staff and the operational productivity of the intervention (both from EFO-LCI) the total distance travelled is calculated using first and second cases in equation 2.1. Forest road maintenance was modelled based on the skidding trail and road densities reported in EFO-LCI and assuming that overhauling measures take place every 15 years. Impacts from both generic and specific interventions over a single rotation were then scaled to one m^3 based on the ob volume of wood harvested over one rotation. The Appendix I contains also the Jupyter notebook (Shen 2014) with the LCA analysis.

Climate was chosen as impact category due to the relevant role played by biogenic carbon and its importance in the carbon neutrality issue. For the anthropogenic emissions due to forestry operations, the Global Warming Potential over 100 years (GWP-100) is calculated (IPCC 2013) while for biogenic carbon the impact is assessed using the GWPbio (Cherubini et al. 2011a) over a time horizon of 100 years (GWPbio-100). In chapter Dynamic climate IA methodology in Temporalis of Appendix II is reported the methodology used to calculate both GWP and GWP-bio. The climate change impact has been calculated for all FoUs and compared with the climate change impact of the nine European unit processes available in Ecoinvent 3.3 (Table 2.3) for raw wood production.

Table 2.3 The nine ecoinvent 3.3. unit processes compared with EFO-LCI and their modelling assumptions.

		hardwood forestry, beech, sustainable forest manageme nt, DE	***hardwood forestry, oak, sustainable forest manageme nt, DE***	***softwood forestry, spruce, sustainable forest manageme nt, DE***	***softwood forestry, pine, sustainable forest manageme nt, DE***	***hardwood forestry, birch, sustainable forest manageme nt, SE***	***softwood forestry, spruce, sustainable forest manageme nt, SE***	***softwood forestry, pine, sustainable forest manageme nt, SE***	***hardwood forestry, mixed species, sustainable forest manageme nt, CH***	***softwood forestry, mixed species, sustainable forest manageme nt, CH***
Interventi on	**Unit**									
mechanical site preparation,with forwarder,tractor or excavator										
	PMH/ha					1.1	1.1	1.1		
stand establishment, with tractor										
	seedlings/PMH	500	500	500	500		7	7		
	seedlings/ha	8000	10000	3000	8000		500	500		
tending, with brush cutter										
	interventions/ha	1	1	1	1	1	1	1		
	PMH/ha/interventi on	14	14	14	14	14	14	14		
young growth tending, with brush cutter										
	interventions/ha	2	2	0	1	2	0	1		
	PMH/ha/interventi on	15	15	15	15	15	15	15		
selective cleaning, with power saw										
	interventions/ha	1	1	1	1	1	1	1		
	PMH/ha/interventi on	15	15	15	15	15	15	15		
systematic cleaning, with mulcher or tractor										
	interventions/ha	1	1	1	1	1	1	1		

		hardwood forestry, beech, sustainable forest management, DE	***hardwood forestry, oak, sustainable forest management, DE***	***softwood forestry, spruce, sustainable forest management, DE***	***softwood forestry, pine, sustainable forest management, DE***	***hardwood forestry, birch, sustainable forest management, SE***	***softwood forestry, spruce, sustainable forest management, SE***	***softwood forestry, pine, sustainable forest management, SE***	***hardwood forestry, mixed species, sustainable forest management, CH***	***softwood forestry, mixed species, sustainable forest management, CH***
Intervention	**Unit**									
	PMH/ha/intervention	15	15	15	15	15	15	15		
liming, with helicopter										
	PMH/ha/intervention	3	3	3	3					
lime	kg/ha/intervention	4500	4500	4500	4500					
Harvesting and thinning (average over one rotation)										
power saw	PMH/m3	0.383	0.303	0.445	0.406	*0.00138*	*0.00138*	*0.00138*	0.38578	0.62636
tractor	PMH/m'	0.0852	0.0673	0.0741	0.0677	*0.00134*	*0.00134*	*0.00134*		
harvester	PMH/m3	0.0156	0.0263	0.0166	0.0195	*0.139*	*0.139*	*0.139*	0.0156	0.00804
forwarder	PMH/m'	0.0195	0.0328	0.0222	0.0260	*0.107*	*0.107*	*0.107*	0.0195	0.0107
Total harvest										
	m3/ha over one rotation	822	815	977	768	486	542	426	718	874
Rotation										
	years	140	140	100	120	60	80	80	130	130

4. Results

4.1 Forest Units reported

For the 28 European countries surveyed, a total of 235 Forest Units have been reported, with Central-West Europe as the region with the highest amount (95 FoUs). In total 62 Forest Units unmanaged (i.e. silvicultural system 1) were reported, especially in Central-East and Central-West Europe (13 and 24 FoUs respectively). In Central-East, West and North European regions, even-aged forest management with shelterwood (respectively 13, 7 and 11 FoUs) and clearcutting system (11, 18 and 16 FoUs), together with the continuous cover forest (2, 12 and 7 FoUs) are the silvicultural systems with the highest amount of Forest Units described. Unmanaged and shelterwood FoUs are the ones described for the highest number of species groups (all the seven for both) while coppice with standards only for two. Overall, there is a rather even distribution of species groups reported between ecoregions, except for the North, where conifers are predominant (38 FoUs of species groups A and B vs. 10 of species groups D, E and F), and the two southern regions with the geographical specificity of the two Mediterranean species groups (Figure 2.2).

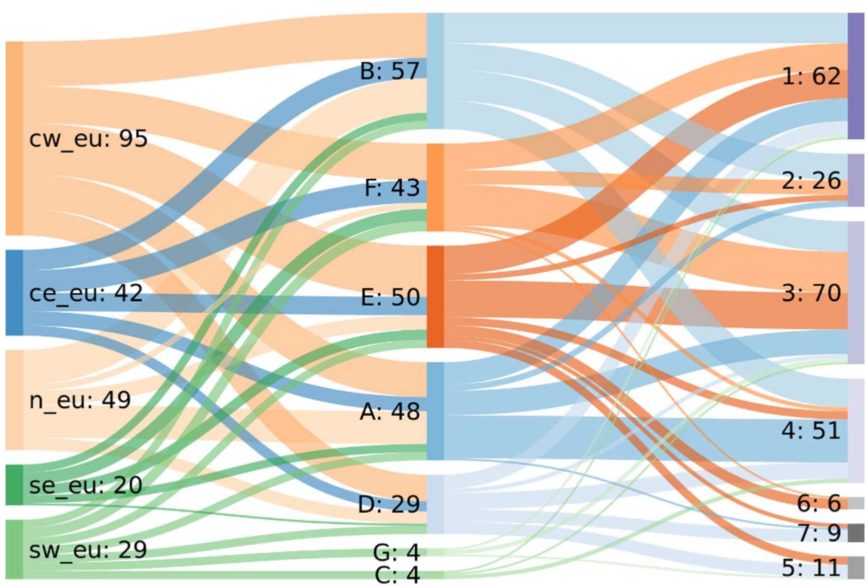


Figure 2.2 Alluvial diagram with the distribution of Forest Units between ecoregions, species groups and silvicultural systems (respectively first, second and third column). The width of the node represents the flow quantity (i.e. number of FoUs, also indicated), letters stand for

species groups (Table 2.2) and numbers for silvicultural systems (Table 2.1), for ecoregion codes see Figure 2.1.

4.2 Data available in the database

All collected data were grouped in two tables, one containing the FoUs reported and their general description, and the other one with the detailed description of all their interventions (see section 3.3). All collected data, their variable name in the spreadsheets and units are summarized in Table 2.4 and Table 2.5 and the link to download them is reported in Appendix I.

Table 2.4 Data available in EFO-LCI on FoUs description

Type of information	Field Name	Description	Unit	Comments
General information of FoU	Country	Country name		
	Ec_name	Ecoregion name		
	Eco_code	Ecoregion code		
	Man_syst	Silvicultural system		
	Sp_group	Species group		
	Man_syst_code	Silvicultural system code		
	Sp_gr_code	Species group code		
	FoU	Forest Unit code		
	Itinerary	Short narrative description of the stand and its management		
	Rotation	Rotation length	years	
	Goal_diam	Target diameter (for continuous cover forest management)	cm	
Data quality assessment	Rel	Reliability	0≤Rel≤1	
	Compl	Completeness	0≤Compl≤1	
	T_cor	Temporal Correlation	0≤T_cor≤1	
	G_cor	Geographical Correlation	0≤G_cor≤1	
	FT_cor	Further technological correlation	0≤FT_cor≤1	
	DQD	Data Quality Distance	0≤DQD≤5	
	QA	Overall Quality Assessment for FoU	A, B, C, D, E	
Composition of the stand	Main_sp	Main species in the Forest Unit group		
	Mois_field	Moisture of green wood for main species	%	
	Den_fresh	Density of green wood at for main species	t/m^3	
	Den_dried	Density of dried wood for main species	t/m^3	
	OMS	Other main species		

Type of information	Field Name	Description	Unit	Comments	
	OMS-Mois	Moisture of green wood for other main species	%		
	OMS-Den_fre	Density of dried wood for other main species	t/m^3		
	OMS-Den___dri	Density of dried wood for other main species	t/m^3		
	Sec_sp	Secondary species			
	SS-Mois	Moisture of green wood for secondary species	%		
	SS-Den_fresh	Density of green wood for secondary species	t/m^3		
Regeneration	Reg	Type of regeneration			
	Type_seed	Type of seedling			
	Age_seed	Age of seedling			
	Orig_seed	Origin of seedling			
	Dens	Density of plantation	plants/ha		
Forest transport	Equip	Distance (one way) to forest for silvicultural equipment	km		
	Staff	Distance (one way) to forest for silvicultural staff	km		
	Harv_equip	Distance (one way) to forest for harvesting equipment	km		
	Harv_staff	Distance (one way) to forest for harvesting staff	km		
	Road	Forest road density	m/ha		
	Trail	Skidding trail density	m/ha		
Interventions rules. Description of the general rules on which thinnings and regeneration fellings are based (Regeneration fellings are clearcuts, selection fellings in continuous cover and preparatory, seedling, secondary and final fellings in shelterwood)	Rul___thn	Rule on which the thinning is based (e.g. achievement of a certain standing stock, basal area, age, density etc.)		This is reported for 4 thinnings and the value of n indicates the chronological order (e.g. Rul_th1,Rul_th2,Val_th1 etc.)	Clearcutting, coppice and short rotation
	Val_thn	Value (of standing stock, basal area, age, density etc.) when the thinning is applied			
	Typ_thn	Type of thinning			

Type of information	Field Name	Description	Unit	Comments	
	Targ_thn	Target value (of standing stock, basal area, age, density etc.) after the thinning is applied			
	Un_thn	Unit of target value (of standing stock, basal area, age, density etc.)			
	Com_thn	comments			
	Rul__cle	Rule on which the felling is based (e.g. achievement of a certain standing stock, basal area, age, density etc.)			
	Val_cle	Value (of standing stock, basal area, age, density etc.) when the felling is applied			
	Com_cle	comments			
	Rul__sel	Rule on which the selection felling is based (e.g. achievement of a certain standing stock, basal area, age, density etc.)			Continuous cover forest
	Val_sel	felling intervals			
	Targ_sel	Target value (of standing stock, basal area, age, density etc.) after the selection felling is applied			
	Un_sel	Unit of target value (of standing stock, basal area, age, density etc.)			
	Com_sel	comments			
	Rul__x	Rule on which the type of felling is based (e.g. achievement of a certain standing stock, basal area, age, density etc.)		This is reported for preparation (pre), seedling (sed), secondary (sec) and final (fin) felling. The value in parenthesis is used in place of x to indicate which one (e.g. Rul_pre,Rul_sed,Val_pre etc.)	Shelterwood

Type of information	Field Name	Description	Unit	Comments
	Val_x	Value (of standing stock, basal area, age, density etc.) when the type of felling is applied		
	Age_x	Target value (of standing stock, basal area, age, density etc.) after the type of felling is applied		
	Targ_x	Unit of target value (of standing stock, basal area, age, density etc.)		
	Com_x	comments		
	Rul___grad	Rule on which the wood grading is based (e.g. achievement of a diameter, volume, quality standard etc.)		
	Val_grad	Value need		
	Com_grad	comments/additional info and brief description on how the wood is graded		
Data source	DS_rot/goal	Data source for rotation length/target diameter		
	DS_comp	Data source for composition of the stands		
	DS_reg	Data source for regeneration		
	DS_transp	Data source for forest transport		
	DS_int/rul	Data source for the description of rules applied in the interventions		
	DS_equip	Data source for the equipment used in the inverventions		
	DS_inpu	Data source for the inputs used in the inverventions		
	DS_harv/ha	Data source for harvesting volumes of the interventions		

Table 2.5 Data available in EFO-LCI on FoUs interventions

Type of information	Field name	Description	Unit	Comments
General information of Forest Unit	Country	Country name		
	Ec_name	Ecoregion name		
	Eco_code	Ecoregion code		
	Man_syst	Silvicultural system		
	Sp_group	Species group		
	Man_syst_code	Silvicultural system code		
	Sp_gr_code	Species group code		
	FoU	Forest Unit code		
General information on interventions	Interv_num	Cronological number of intervention		
	Timing_of_interv	Timing of intervention (relative to regeneration or previous selection felling)	years	
	Type_of_interv	Type of intervention		
	Sp_interv	Species concerned by the interventions		
	Pre_int_stock	Stock of the stand before the intervention		
	Pre_int_BA	Basal area of the stand before the intervention		
Equipment used	Equip_n	Equipment used in the intervention		This is reported for three machines (1st, 2nd, 3rd) and the value of n indicates them (e.g., Equip_1, Equip_1, Power_1, etc.)
	Power_n	Power of the main equipment used (if motorized)	CV	
	Mass_n	Mass of the main equipment used	tons	
	Consum_n	Consumption of motorized machine	l/h	
	Hrs_use_life_n	Hours of use during whole life for equipment	h	
	h/ha_n	Average productivity of the equipment used during the intervention	h/ha	
	m3/h_n		m^3/h	
	fresh_t/h_n		tons/h	

Type of information	Field name	Description	Unit	Comments
Input used	input_n	Type of input used (fertilizer, herbicide, etc.)		This is reported for two inputs and the value of n indicates which one (e.g., input_1, input_2, active_pr1, etc.)
	active_prn	Active principle		
	Amount_n	Amount applied	kg/ha	
Wood removal	Stem Stem_and_res Stumps	Tick boxes indicating the type of wood removed (if)		
	m3_ob_x	Amount of wood removed during the intervention	m^3/ha (overbark)	This is reported for three assortments (Logs, firewood, and pulp) and the value of x indicates it (e.g., m3_over_bark_Logs, etc.)
	m3_ub_x		m^3/ha (under bark)	
	dry_t_x		tons/ha	
	St_m3_Firewood	Firewood removed during intervention	m^3/ha (stacked)	
	m3_chips	Chips removed during intervention	m3/ha	
	Loose_m3_x	Amount of chips and stumps removed during the intervention	m^3/ha (loose)	This is reported for chips and stumps the value of n indicates which one (e.g dry_t_chips, etc.)
	dry_t_x		tons/ha	

4.3 Data quality

The data quality was assessed for each Forest Unit following the pedigree matrix approach proposed by Weidema and Wesnaes (1996). While Weidema et al. (2013) was followed for the definition of the indicator scores, the methodology based on the Data Quality Distance (DQD) approach proposed by Lewandowska et al. (2004) was used to assign the overall Quality Class (QC) to each FoU. The lower the DQD, i.e. the differences between the quality requirements (Data Quality Goals, DQG) and the actual quality of the inventoried data, the higher the data quality of the indicator considered and consequently its Data Quality Indicator (DQI). The sum of the DQI of all the indicators (tDQI) determines the overall quality of each FoU. All the tDQI were grouped into five QC with decreasing quality from A to E (tDQI ranges: 0 to 0.5; 0.5 to 1;

1 to 1.5; 1.5 to 2; 2 to 2.5). In the analysis of the following sections, only the results of the FoUs with a quality class from A to B will be presented. This decision was taken to remove the bias introduced by the low data quality, in order to guarantee that the observed variability is, in fact, due to real differences in forest management. Of the 235 reported Forest Units, the 133 that are actively managed (i.e. silvicultural systems 2 to 7) and that meet the aforementioned data quality requirements, will be presented and discussed in the next sections.

4.4 Rotation length

The average rotation length of the reported FoUs is 86-90 years and varies between 6-10 years for the FoU 7-D in Switzerland and Austria, and 196-200 years for the Belgian FoU 4-E (Figure 2.3).

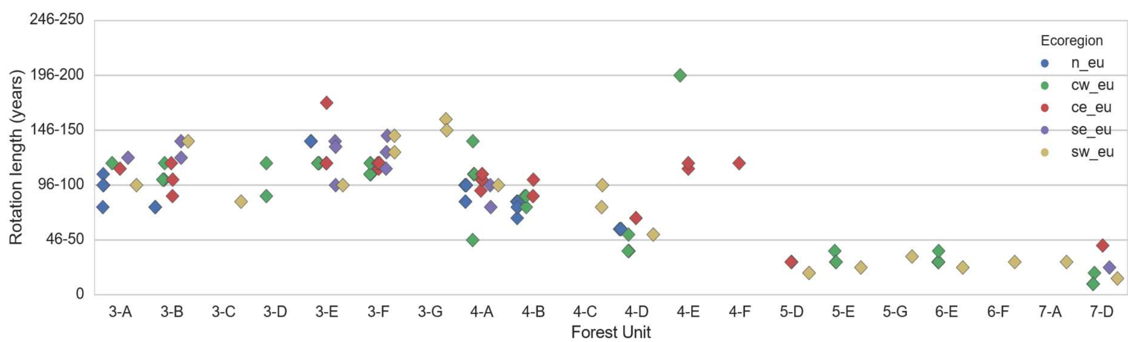


Figure 2.3 Rotation length reported as a function of Forest Units and ecoregions. Letters stand for species groups (Table 2.2) and numbers for silvicultural systems (Table 2.1), for ecoregion codes see Figure 2.1.

When FoUs are grouped per silvicultural system, the following average ranges in rotation lengths are observed: short rotation (16-20 years), coppice (26-30 years), coppice with standards (26-30 years), even-aged forest with shelterwood (86-90 years) and even-aged forest: uniform clearcutting system (116-120 years). A clear trend can also be found if the FoUs are grouped by species group, where the following average ranges in rotation lengths are observed: fast-growing deciduous (41-45 years), Mediterranean conifers (86-90 years), light-demanding conifers (91-95 years), shade-tolerant conifers (96-100 years), slow-growing, light-demanding deciduous (96-100 years), slow-growing, shade-tolerant deciduous (111-115 years) and Mediterranean evergreen trees (111-115 years).

4.5 Regeneration of the stands

Data were collected on how FoUs are established and regenerated (Figure 2.4), as these variables can influence the life cycle impact of forest management (Michelsen et al. 2008). As expected, the type of regeneration is mainly driven by the management but, remarkably, some differences between ecoregions and species groups are found. Excluding the short rotation, where artificial regeneration is always applied in all reported countries, it can be seen that, broadly speaking, there is a tendency toward natural regeneration in both southern ecoregions (15 FoUs out of 22 in the west and 10 out of 14 in the east). Notable is the widespread adoption of mixed regeneration in the shelterwood systems of the three ecoregions of central and north Europe. This can be explained by the enrichment planting typically used to increase the density of the desired tree species in such a silvicultural system.

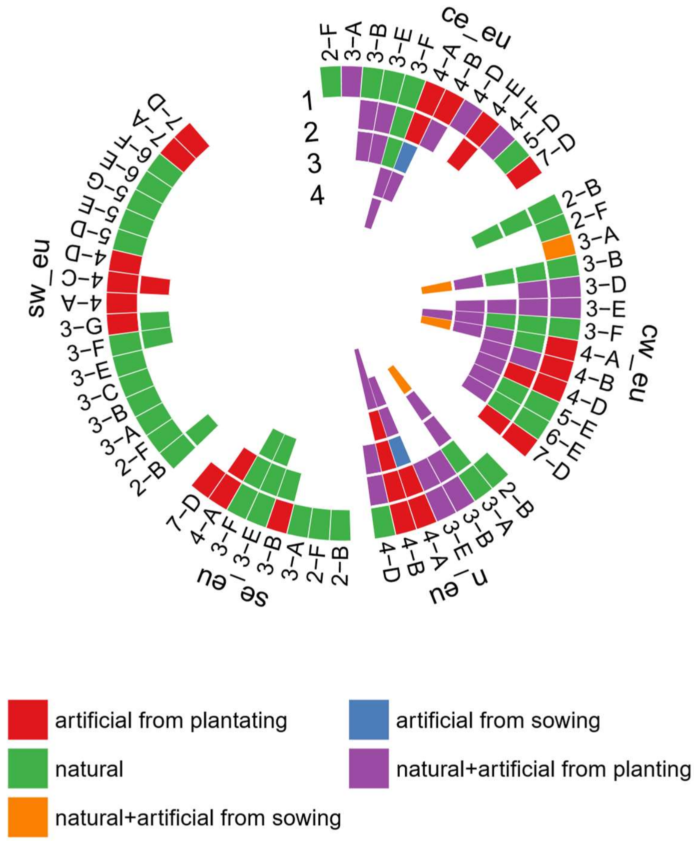


Figure 2.4 Type of regeneration reported by FoU and grouped by ecoregion. Letters stand for species groups (Table 2.2) and numbers for silvicultural systems (Table 2.1), for ecoregion codes see Figure 2.1.

4.6 Moisture content of wood

The moisture content of wood is a very important variable in all forest LCA studies, given its strong effect on both heating value and wood density. Despite this, Klein et al. (2015) found that moisture content is not reported in half of their reviewed studies. In most of the FoUs, the moisture content of wood at forest road (dry basis) was reported to be between 35% and 55%, with the highest value (u=94%) reported in the even-aged forest with shelterwood, light-demanding conifers FoU of Estonia (Figure 2.5).

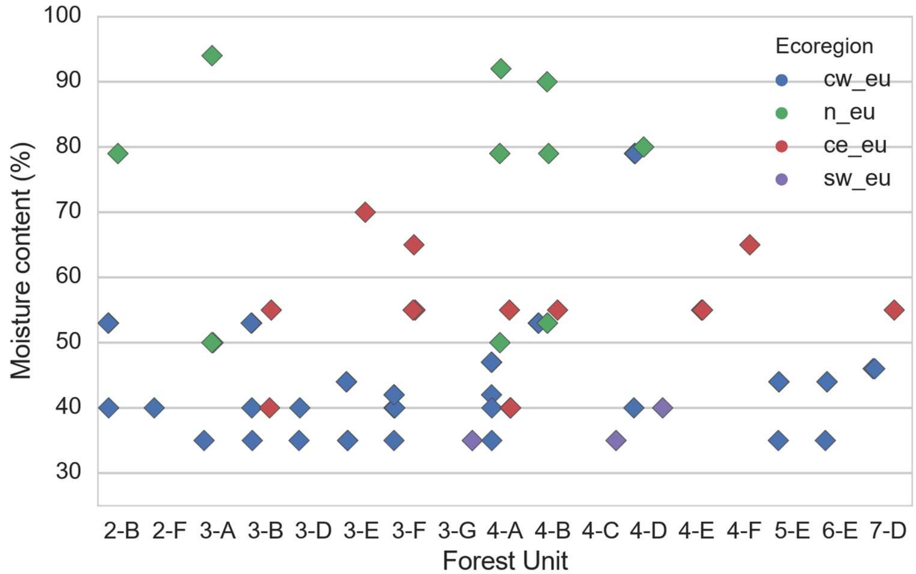


Figure 2.5 Moisture content of green wood. Letters stand for species groups (Table 2.2) and numbers for silvicultural systems (Table 2.1), for ecoregion codes see Figure 2.1.

4.7 Machinery used

The overall impact of the management is influenced by the type of machinery used (Berg 1997). Figure 2.6 shows the data reported in the questionnaire about the machinery used for tree felling (including both thinnings and regeneration fellings). It is clearly visible that heavy forestry machinery is the common choice in almost all types of forests in Central-West and North Europe. This is in contrast with the remaining ecoregions where, with the exception of the short rotation system, the chainsaw is normally used. In short rotation systems, the productivity of heavy forestry machinery sensibly increased due to better accessibility and plantation design together with the higher density of the stems.

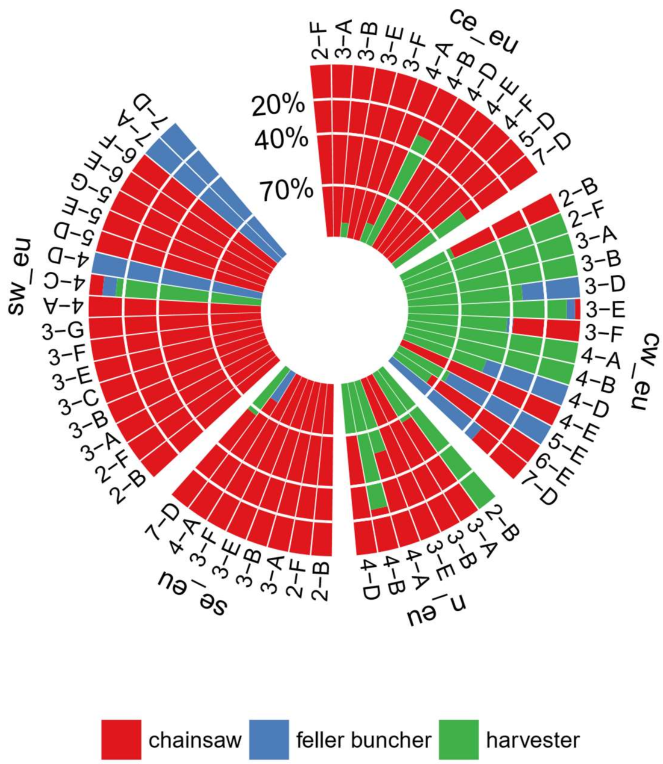


Figure 2.6 Type of machinery used in the thinning and felling reported in percentage per FoU and grouped by ecoregion. Letters stand for species groups (Table 2.2) and numbers for silvicultural systems (Table 2.1), for ecoregion codes see Figure 2.1.

The economic benefits of this high productivity outweigh the burden of the higher cost per hour in comparison to manual harvesting operations making them more economically viable, especially in industrialized countries with higher labour costs (Vanbeveren et al. 2015). The overall impact of felling is not only influenced by the type of machinery but also the productivity of the equipment used during the operation, which in turn depends on many factors like accessibility and structural characteristics of the stand. Figure 2.7 shows the distribution of the average harvesting productivities of different machinery reported in each Forest Unit for thinning and regeneration felling. It is interesting to note that, while between machinery there are clear differences in terms of productivity, for the same machinery the differences between the type of intervention are minor. For chainsaw, a median productivity of one cubic meter harvested per hour (m^3/h) for both thinning and regeneration felling is observed, for feller buncher 5 and 7 m^3/h and for harvester 6 and 12 m^3/h respectively.

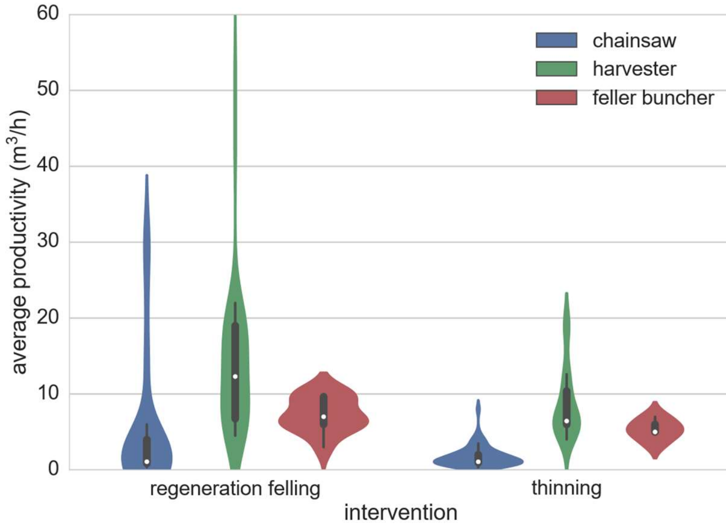


Figure 2.7 Violin plot showing the distribution of the average harvesting productivities of different machinery reported in each Forest Unit for regeneration felling and thinning.

4.8 Harvesting volumes and assortments

While ISO 14040 and 14044 on life cycle assessment recommend avoiding allocation whenever possible, the partitioning of burdens for multi-outputs processes is often necessary in the forestry context due to the high number of co-, by- and recycled products produced and used along the chain. Despite this widespread application of allocation in LCA of forestry systems, the procedure followed in many cases is not described. Following the suggestions of Klein et al. (2015) concerning allocation that should be based on mass or volume, we asked to report the volume of wood extracted during felling and thinning operations specifying the assortment (Figure 2.8). This distinction is important for the allocation of the impacts of forestry operations to the different products produced. The first thing to note is that, despite what is considered to be good management practice in silvicultural guidelines, thinnings are often not applied at all, and this is almost the standard practice in both southern ecoregions. This is because in these two ecoregions productive forests are typically in mountainous areas with high mobilization cost. When thinnings are applied, the sum of all wood removals by thinnings over the rotation period consists of an amount of wood that is often sensibly lower than the amount removed by regeneration fellings. Clearly visible is the higher wood production along the rotation for the clear-cut system compared to the other systems. With regard to the grading, it is interesting to note the biggest share of firewood produced in deciduous forests (species groups D, F and G). Overall, the assortments of products from harvesting exhibit a high variation also between silvicultural systems and countries. This variability is due to local market conditions and the relative demand for each assortment.

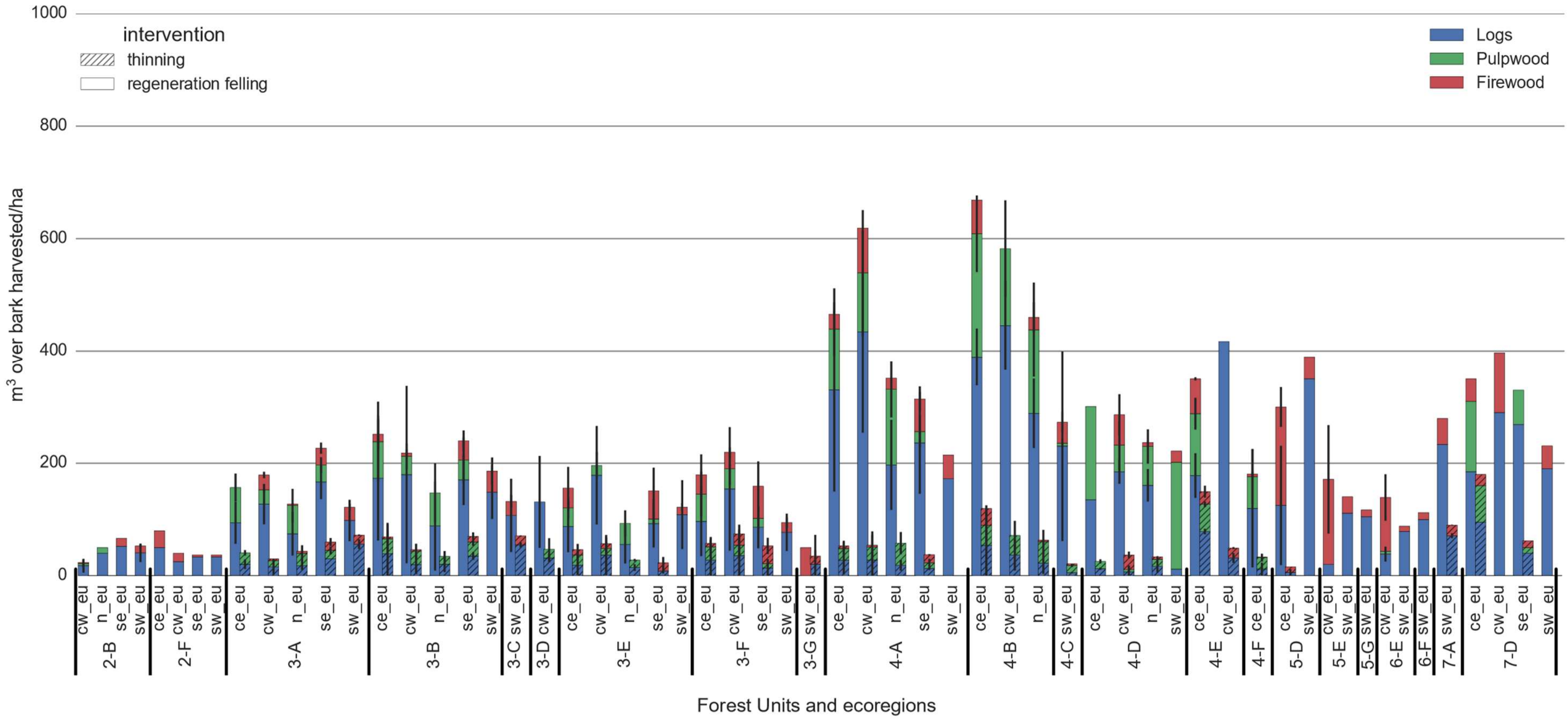


Figure 2.8 Clustered stacked bar chart showing average harvesting volume for thinning (hatched bars) and regeneration felling (solid bars) and their standard deviation (error bars) for each ecoregion grouped by FoU. Letters in main x-axis category stand for species (Table 2.2) and numbers for silvicultural systems (Table 2.1), for ecoregion codes in secondary x-axis category see Figure 2.1. In the case of thinning values are calculated summing all the thinning interventions. For shelterwood, regeneration felling values are calculated summing preparation, seedling, secondary and final fellings.

4.9 LCA impact

The climate change impact of all FoUs compared to the nine European unit processes available in Ecoinvent 3.3 are presented in Figure 2.9. The variability of results in EFO-LCI is higher than when using Ecoinvent. The anthropogenic impact ranges from 11.1 to 16.4 kg CO_2eq/m^3 in Ecoinvent and from 0.4 (FoU 5-D in ce_eu) to 73.1 (FoU 3-F in sw_eu) kg CO_2eq/m^3 in EFO-LCI while the biogenic impact ranges from 73.1 to 143.6 kg CO_2eq/m^3 in Ecoinvent and from 1.6 (FoU 7-D in cw_eu) to 451.9 (FoU 4-E in cw_eu) kg CO_2eq/m^3 in EFO-LCI. In both cases, being the GWPbio calculated assuming that the stands are carbon neutral, the differences are exclusively due to the variability in the rotation lengths. The total climate change impact (GWPbio-100 + GWP-100) in Ecoinvent and EFO-LCI is respectively in the range from 158.7 to 421.8 kg CO_2eq/m^3 and from 7.6 (FoU 7-D in cw_eu) to 471.0 (FoU in 3-F sw_eu) kg CO_2eq/m^3. In EFO-LCI a higher number of ecoregions are covered in comparison to Ecoinvent, which has data only for two of them. Furthermore, when the results are disaggregated by silvicultural systems and species groups, their intra- and extra-group variability can be quite important. Ecoinvent processes are averages of different management styles and the silvicultural system applied is not explicitly mentioned but, from the description, it seems to be an average between silvicultural systems 3 and 4. While for these two systems the order of magnitude of the biogenic impacts is similar, for all the other systems impacts are systematically lower in EFO-LCI, especially in silvicultural systems 2 and 7 due to their shorter rotation period. Also when looking at species group, biogenic impacts in EFO-LCI tend to be lower than in Ecoinvent. For both species groups and silvicultural systems, the median anthropogenic impacts of the FoUs are spread around the impacts found in Ecoinvent processes.

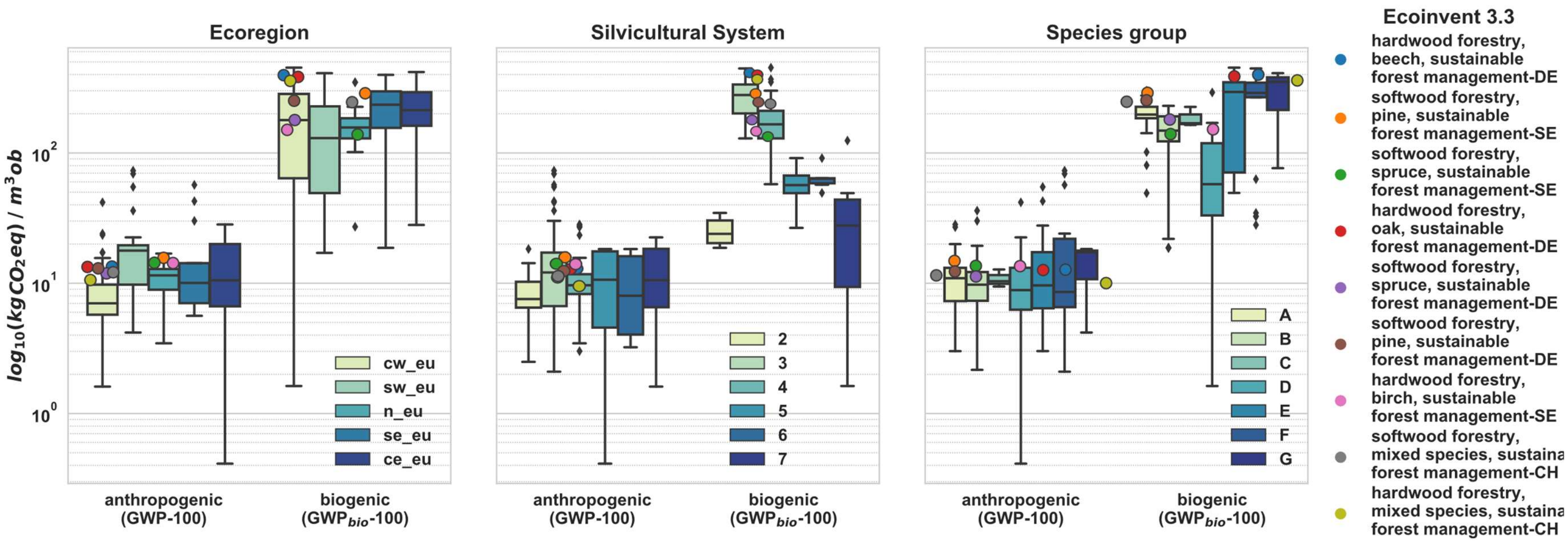


Figure 2.9 Boxplots of the climate change impact (log10 scale) per m³ overbark of raw wood harvested in the Forest Units measured as GWP-100 for anthropogenic emissions (left) and GWPbio-100 for biogenic carbon fluxes (right) and compared with Ecoinvent 3.3 (dots). EFO-LCI results are grouped by ecoregion (see Figure 2.1 for codes), silvicultural system (see Table 2.1 for numbers) and species group (see Table 2.2 for letters). Ecoinvent results are plotted against the relative ecoregion and species group while for the silvicultural system they are placed at the centre being an average between silvicultural systems 2 and 3.

5. Discussion and Conclusion

The present study shows that there is a remarkable variation in rotation length, type of regeneration, amount and assortments of wood products harvested and machinery used in interventions, depending on tree species and management practices applied, as well as on the specific country where the production takes place. All these differences are important and significantly affect the life cycle impact of raw wood production. Although we have shown how climate change impact assessment of wood production would benefits from the use of regionalized inventory, how these improvements are translated into better LCA of wood-based products is difficult to estimate. The relative role played by raw wood production obviously depends on the product studied, but also on the type of environmental impact considered and the methodological choices made. For example in González-García et al. (2014b) the abiotic depletion potential of hardboard manufacturing is solely due to raw wood production but, on the opposite, its relative contribution to human toxicity and photo-oxidant formation is only about two% of the overall impacts. In Martínez-Alonso and Berdasco (2015) the relative contribution of raw wood production to the carbon impact of sawn timber manufacturing ranges from 8 to 34% depending on whether the wood is kiln or air dried. Those are just two examples of how the choice of impact assessment method and technological assumptions can change the relative importance of forestry when put in a broader perspective. Also other methodological aspects like the system boundary of the analysis (e.g. cradle-to-gate vs cradle-to-grave) and the allocation used can be of crucial importance. With regard to the system boundary, as we have also shown, the relative contribution of biogenic carbon to the total climate change impact is rather important and, in our case, relatively higher than the anthropogenic ones. Considering that many of the wood-based products LCA's in the literature assumes the climate neutrality of wood, the relative role played by forestry, at least for what the climate change impact is concerned, can be higher than expected. It is realistically unfeasible to collect site-specific data for all the LCAs involving forest-based products. Despite this, a higher level of disaggregation and regionalization of inventories would improve the accuracy and credibility of life cycle studies of forest production systems. The difficulty of acquiring site-specific data is one of the main challenges faced when trying to build regionalized inventories (Hellweg and Canals 2014). We tried to address this problem by collecting and disclosing FoU specific data on the management practices of European forests with the final goal of reducing the uncertainty of forestry LCA. In collecting this information, we tried to follow as much as possible the methodological guidelines proposed by Klein et al. (2015), with the goal of continuing the process of harmonization proposed by the authors.

While EFO-LCI includes only part of the information necessary to develop forestry LCI (namely on management and harvesting practice), the missing information on forest growth and carbon sequestration can be taken from the study of Neumann et al. (2016b), where the same FoU classification is used.

A limitation of EFO-LCI is that it presents a sub-optimal organization of the data since it has not been normalized. Due to the relatively small and manageable size of the database, this is not expected to influence its usability. Nevertheless, the organization of the EFO-LCI data in a more efficient manner to reduce data redundancy and ensure their integrity is advisable in the future.

The choice of disclosing all the data collected under the CC BY-NC-SA 4.0 license has been taken to further stimulate transparency and reproducibility in LCA and to fight the so-called "reproducibility crisis" (Baker 2016). The use of open science is seen as one of the main instruments to increase scientific integrity and minimize the two aforementioned problems (Wicherts et al. 2012, Ram 2013, Nosek et al. 2015, McNutt et al. 2016), and we hope that in the near future this will become the standard practice in the LCA world, as it is already for other sectors with high scientific standards (Anonymous 2016). Our final hope is that the database developed, being conceived as a collaborative one, will not only be used but also integrated and improved by other researchers and practitioners.

Chapter 3

Temporalis, a generic method and tool for dynamic Life Cycle Assessment

Adapted from: Cardellini, Giuseppe, et al. "Temporalis, a generic method and tool for dynamic Life Cycle Assessment". Sci. Total Environ., vol. 645, 15 Dec. 2018, pp. 585-95, doi:10.1016/j.scitotenv.2018.07.044

1. Abstract

The limitations of the static nature of Life Cycle Assessment (LCA) are well known. To overcome the loss of temporal information due to the aggregation of flows in the Life Cycle Inventory (LCI), several dynamic LCA methodologies have been proposed. In this work, we present a new generic and operational methodology for dynamic LCA that allows for the introduction of temporal information in both in the inventory and the Life Cycle Impact Assessment (LCIA) phases. The method makes use of graph traversal and convolution to calculate the temporally differentiated inventory and makes it possible to use several types of dynamic Impact Assessment. We describe our method and apply it to a cradle-to-grave dynamic LCA of a glued laminated timber (glulam) product. We also test the sensitivity of the global warming results to temporal explicit LCI data. There is a considerable difference in outcome between the static and dynamic approaches. We have implemented our framework in the free and open source software Temporalis that is fully operational and can be used with existing LCA databases.

2. Introduction

Life cycle assessment (LCA) is a well-established method to estimate the potential environmental impacts of services and products throughout their entire life cycle. One of the shortcomings of LCA practice is the lack of consideration of the temporal and spatial variation of flows and emissions (Huijbregts 1998). Already in the early days of LCA Finnveden and Nielsen (1999) stressed the importance of considering the long-term emissions from landfills. The lack of temporal considerations is still considered an unresolved problem and an important limitation for the accuracy and representativeness of LCA (Reap et al. 2008, McManus and Taylor 2015). Methodologies to include time and space in LCA have been proposed (Mutel and Hellweg 2009, Beloin-Saint-Pierre et al. 2014, 2017, Tiruta-Barna et al. 2016, Yang and Heijungs 2016), but it is still difficult to easily perform dynamic and spatialized LCA for practitioners. This is also due to the lack of accessible and transparent (possibly open source) software. Since "LCA is primarily a steady-state tool" (Udo de Haes 2006) the conventional approach sums all the emissions for a given pollutant into a single value in the Life Cycle Inventory (LCI), regardless of its time of occurrence. Subsequently, the impacts of the aggregated environmental interventions are characterized during the Life Cycle Impact Assessment (LCIA), irrespective of their timing.

Time can be explicitly considered at the level of: (i) the Functional Unit (FU), by giving it a temporal dimension (e.g. one year of energy use); (ii) the LCI, by explicitly considering the temporal relationship between flows; (iii) the LCIA, by using dynamic characterization factors (dCF) or characterization functions (CFun) in place of characterization factors (CF) and (iv) the weighting of impacts, for example by discounting them (Hellweg et al. 2003, Collet et al. 2014). Regardless the level of complexity considered, to take time into account in LCA, the LCI must be dynamic, which means that emissions and resource consumptions are explicitly distributed over time. In their seminal book on the computational aspects of LCA, Heijungs and Suh (2002) already discussed a theoretical extension of the matrix-based method to include both spatial and temporal differentiation of the inventory. But already at that time the authors warned the reader that, despite the solid theoretical base, the method's operationalization posed problems. This is due to the huge amount of temporal data required and its high computational demand. In the first studies talking about dynamic LCA (dLCA) (Pehnt 2006, Kendall et al. 2009, Zhai and Williams 2010) time was not explicitly considered. In these works the temporal changes in the processes studied were implicitly considered and eventually both emissions and impacts were still aggregated following the traditional LCA approach.

To be dynamic a LCI must be able to locate and differentiate activities and flows in time. This ability to consider and compute temporal characteristics in LCIs, to the best of our knowledge, has been presented in three methodological proposals. In Collinge et al. (2013) the traditional approach based on matrix inversion is used and improved with the inclusion of temporal information. Although it is possible with this method to consider time for each dataset in the LCI, it shows the important operational limitations already recognized from Heijungs and Suh (2002). Beloin-Saint-Pierre et al. (2014) developed the enhanced structure path assessment (ESPA), which extends on structural path analysis, a widely known technique in input-output analysis. It makes use of power series expansion to solve the dynamic inventory, and the matrix inversion is replaced with a product of convolution of the discrete distribution functions. The ESPA has recently been further integrated with the possibility to consider time also at the level of LCIA by applying time-dependent characterization factors (Beloin-Saint-Pierre et al. 2017). The major drawback of this approach is that it is still insufficiently documented and, to date, it has not been made operational and thus not available for the LCA community. A final approach consists in the direct traversal of the supply chain graph, as done by Tiruta-Barna et al. (2016). They recently introduced a very promising method for dynamic LCI that has been developed as a prototype web application. It is based on a process flow network structure and makes use of a graph search algorithm to build the temporal model. Despite the promises of this methodology, it is still a proof of concept that needs to face the implementation challenges of a desktop application. For example, the need for a reduced utilization of memory and computational resources in comparison to a server application. Moreover, it is not coupled with a LCIA framework and it is not clear if the method can deal with datasets without temporal information, raising doubts over its integration potential with existing LCA databases. Regarding the treatment of the LCI as a graph, it is worth mentioning that this approach poses a key methodological challenge due to the cyclic nature of the supply chain graphs. Loops can be encountered, and a cutoff function must be applied to halt potentially infinite loops in supply chain traversal.

Available temporal information can be absolute (e.g. May 25, 1978) and relative (e.g. two weeks ago) in time. While for most impact assessment methods it is necessary to know the absolute calendar date of the emissions (Beloin-Saint-Pierre et al. 2014), both relative and absolute distributions can be encountered in the inventory. This is essentially dependent on how the data are collected during the LCI construction and there are no specific indications to use one or the other. The work of Collinge et al. (2013) is based on absolute temporal data while Beloin-Saint-Pierre et al. (2014) and Tiruta-Barna et al. (2016) use relative temporal information. Ideally, both types of temporal information can be handled by a dynamic LCA framework.

The timing of emission is also relevant in impact assessment (IA). In conventional LCIA methods, emissions are integrated over the life cycle, hence they are treated as a pulse rather than a temporally distributed flux. But the moment when the emissions occur can affect the impact. An example are those impact categories influenced by the background concentrations of the pollutants, like aquatic eutrophication (Udo de Haes et al. 2002) and acidification (Potting et al. 1998). Photochemical smog production (Shah and Ries 2009) and water scarcity (Kounina et al. 2013) are other examples of time-dependent environmental responses. The timing of emissions is also relevant when the impact assessment is performed on a finite time horizon (TH). The typical example of a time horizon-dependent CF is the Global Warming Potential (GWP). This metric, in fact, is very sensitive to the time horizon considered, and the impacts are directly related to its length (IPCC 2013). In the non-dynamic approach, it is implicitly assumed that all the life cycle emissions occur at year 0 and remain in the environment for the entire TH. Levasseur et al. (2010) applied time-dependent CFs to temporally differentiated LCI, overcoming the inconsistencies due to the application of a static approach in the IA.

Numerous authors have demonstrated how neglecting time consideration in LCIA can lead to misestimation of impacts (Levasseur et al. 2010, 2012, 2013, Kendall 2012, Lebailly et al. 2014, Pinsonnault et al. 2014, Almeida et al. 2015). The limits of the non-dynamic approach are further amplified when biogenic carbon and long life cycles are studied (Jørgensen and Hauschild 2013). To address the issue of emissions timing in LCA Kendall (2012) also proposed the use of the Time Adjusted Warming Potential (TAWP), a static, time-corrected GWP metric that weights the global warming impact on the basis of the timing of the emissions.

While the systematic introduction of temporal dynamics would increase the representativeness of the LCA results, the process needs to be confronted with the increase in complexity of the LCA modelling and the lack of temporal parameters in LCI databases. In addition, the collection of temporally differentiated data can be a long and costly task, and it should be undertaken only for those datasets that are more sensitive to time. Pinsonnault et al. (2014) demonstrated that temporally differentiated information, on first approximation, is not needed for every process, and their use can be restricted to the ones more sensitive to time. Collet et al. (2014) also introduced a method to identify the specific flows requiring such a temporal differentiation. The method uses a stepwise approach to assess the sensibility of the results to the temporal variability of environmental and product flows. Despite the limitations due to the upfront choice of the LCIA method, this method can represent an important instrument to help in understanding where temporal explicit data are needed and further efforts are necessary during data collection. The possibility to deal also with datasets without temporal parameters is a necessary feature of a dLCA framework.

In short, despite the substantial work done on developing dynamic LCA in the past ten years, no methods have been defined and implemented to provide (i) efficient resolution of temporally differentiated life cycle inventories (LCI); (ii) handling of both absolute and relative temporal distributions, as well as exchanges with databases that have no temporal information; (iii) dynamic characterization of emissions, including both distribution over time and characterization as a function of time; (iv) correct temporal accounting of biogenic carbon (i.e. no carbon neutrality assumption); (v) implementation in accessible and open source computer code. In this paper, we present a novel numerical computational approach to dynamic LCA which meets all our criteria. We implemented our approach in the open source Temporalis software library, built on top of the Brightway2 LCA framework (Mutel 2017b). In this work, we will present the methodology, validate it with a virtual example, and introduce its software implementation. We will use Temporalis to calculate the cradle-to-grave climate change impact of 1 m^3 of glued laminated timber (glulam) and show how, by explicitly considering the temporal information, the LCA results diverge from the conventional steady-state approach.

3. Methods

We first introduce the computational framework and the way temporal information is stored. We then explain the functioning of the implemented best-first graph traversal used to solve the dynamic inventory problem, validating it with a virtual example. Finally, the dynamic impact assessment and software implementation are explained. To ensure the necessary transparency and reproducibility of the study requested by several scholars (Frischknecht 2004, Pauliuk et al. 2015), all the analysis have been performed using Jupyter notebook (Shen 2014) and have been uploaded on a public GitHub repository (Ram 2013) with its web-link reported in Appendix II.

3.1 Framework

The solution to the dynamic inventory problem in our method is rooted in the traditional matrix-based approach for LCI computation proposed by Heijungs and Suh (2002):

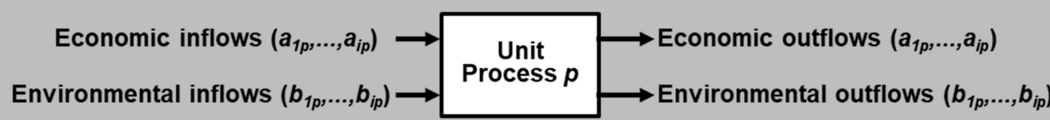
$$\vec{s} = A^{-1}\vec{f};\ G = B\hat{\vec{s}} \qquad \text{equation 3.1}$$

where $\rightarrow$ is used for vector notation and $\frown$ denotes diagonalization.

In the technosphere matrix A, each element $a_{i,p}$ represents the flows from the products i to the processes p; in the biosphere matrix B, each element $b_{j,p}$ represents the biosphere flow j due to the processes p and $\vec{f}$ is the demand vector (i.e. the Functional Unit FU). Here A and B are time invariant (i.e. do not change over time) and have the implicit assumption that the system is assessed over a temporal interval of adequate duration to account for all the relevant flows. The scaling vector $\vec{s}$ and the inventory matrix G represent, respectively, the amount of each process p needed to satisfy the FU demand and resulting environmental interventions j due to the process p. But while in the case of a static LCI, for each process p, we are interested in all its j environmental interventions $G_{j,p}$, in the case of a dynamic analysis we also need to know their time t. The solution to the dynamic inventory problem is thus to find all the environmental interventions $G_{j,p}(t)$ for the FU assessed. Technosphere and biosphere matrices are also adjacency matrices of weighted directed graph (Valiente 2002), where the nodes are processes and edges are exchanges.

Box 2: The matrix-based approach for LCI representation

In the matrix-based approach, the LCI is represented as an algebraic system of linear equations.



Considering the hypothetical unit process p depicted above, the following rules are applied:

- a_{ip} is used as symbol for the j^{th} technosphere flow of the unit process p;
- b_{ip} is used as symbol for the i^{th} biosphere flow of process p;
- negative values of a and b represent input flows;
- positive values of a and b represent output flows;
- value 0 of a and b represents flows that are not involved in the process.

All unit process vectors p together defines the unit process matrix P:

$$\boldsymbol{P} = \begin{pmatrix} \mathbf{A}_{\mathbf{i},\boldsymbol{p}} \\ \hline \mathbf{B}_{\mathbf{j},\boldsymbol{p}} \end{pmatrix} = \begin{pmatrix} \mathbf{a_{1,1}} & \mathbf{a_{1,2}} & \mathbf{a_{1,3}} \\ \mathbf{a_{2,1}} & \mathbf{a_{2,2}} & \mathbf{a_{2,3}} \\ \mathbf{a_{3,1}} & \mathbf{a_{3,2}} & \mathbf{a_{3,3}} \\ \hline \mathbf{b_{1,1}} & \mathbf{b_{1,2}} & \mathbf{b_{1,3}} \\ \mathbf{b_{2,1}} & \mathbf{b_{2,2}} & \mathbf{b_{2,3}} \end{pmatrix}$$

where the columns are the unit processes (unitless, since it is the process of producing something) and the rows both economic (a) and elementary flows (with physical units, like mass or energy). The matrix P is partitioned into the technosphere matrix A (flows within the economic systems) and the biosphere matrix B (environmental flow of unit processes).

The rows i and j represent the source (i.e. exchange flow from) and the columns p the destination of each edge in case of $a_{i,p}$ (i.e. exchange flow to) or the process p responsible of the exchange with the environment in the case of $b_{j,p}$. The weight is represented by the value in the cell $a_{i,p}$ and $b_{j,p}$ (i.e. the exchange amount of i and j respectively flowing to process p and to the environment) (Kuczenski 2015). These edges are dynamic, meaning that the flows occur over a time interval. In non-dynamic LCI, edges are statically represented, and flows $a_{i,p}$ and $b_{j,p}$ represent the integral over time of the flows and are represented by a single value (total flow over the operating interval). But these edges can also be represented by a temporal distribution, which explicitly represents the temporal distribution of flows over time. We introduce two further variables to represent the temporal flows of the dynamic edges, the product-process Temporal Distribution (TD_{ip} hereafter) and the biosphere-process Temporal Distribution (TD_{jp} hereafter). These two TD represent the flow (*y*-axis) per unit of time (*x*-axis) of the product *i* and the biosphere element *j* respectively, due to the process p over the operating time of the exchange (eq 2), in analogy with section 3.1 in Beloin-Saint-Pierre et al. (2014).

$$a_{i,p} = \int_{-\infty}^{+\infty} TD_{ip}(t)dt\,; \quad b_{j,p} = \int_{-\infty}^{+\infty} TD_{jp}(t)dt \qquad \text{equation 3.2}$$

Often the available temporal data in the LCI, and consequently the temporal distributions of the edges, are relative to each unit process. The advantage of using process-relative differentiation is that two relative temporal distributions can be convolved to propagate temporal information. The product of convolution (indicated with $*$) is a mathematical operation that, applied to two distributions, produce a third one which results in the integral of the product of the previous two, where one is reversed and shifted along the other. Convolution can be used in LCI networks to propagate in time the temporal information and determine the amount of each flow and when they occur (first case in equation 3.2). For the details on the application of convolution the reader is invited to consult Beloin-Saint-Pierre et al. (2014) and Maier et al. (2017) where the operator and its application to temporal distributions' propagation in life cycle analysis are explained in detail. Edges which occur at a precise point in time, such as a pulse emission, can be represented by the Dirac delta function. While such a function may seem strange upon first glance, it can be easily convolved with more normal temporal distributions (Raju 1982). Finally, edges with inputs or emissions which occur at a fixed time (i.e. with absolute temporal distribution) do not need to be convolved - these absolute temporal distributions are instead simply scaled by the amount of the edge (second case in equation 3.2). In the software implementation, these TDs are stored as discretized arrays and represented by two one-dimensional numpy arrays of the same length: *TD(i)* and *TD(t)*. The former represents the y-axis values and reports the amount of the exchanges in double precision float (numpy data-type: float64), the latter is corresponding to the x-axis value and stores the time of the exchanges in datetime (numpy data-type: datetime64). *TD(t)* can use any temporal resolution below 1 second, which is the highest resolution in current software implementation, and the software automatically converts the user-defined *TD(t)* into seconds to make all the temporal information uniform (e.g. 1 year is converted to 31556952 seconds). Both *TDip* and *TDbp* are optional, and when not reported the exchanges are automatically modelled as a one-time pulse (Dirac) with the implicit assumption that the emission happens the same year of the downstream consuming exchange and not spread over time. It is up to the user to make sure that, when this is not the case, the correct TD for the exchanges is entered in the database. TDs can be result of a function (e.g. modelling) and *TD(t)* can be also non-continuous. This approach enables the treatment of the three situations reported in the introduction and, if available, temporal distributions of different time-scales and time-steps. To solve the inventory problem another temporal information is also necessary, namely a calendar date representing the start time *t0* of the FU. This other parameter is necessary to propagate in time the flows when reported in relative time as explained in Appendix II. To solve the dynamic inventory problem the matrix-based approach is used to represent the network of flows between processes and biosphere flows and a graph traversal is used to explore all the important processes of the network and solve the inventory dynamically.

3.2 The best-first graph traversal

Graph traversal algorithms are used to explore the nodes of a network and are classified based on the order in which each node in the graph is visited. A well-known method is the breadth-first search strategy, used by ESPA. Despite its short running time, this method has high memory requirements, mainly when big databases are traversed, making its application limited for simple desktop utilization (Marvuglia et al. 2013). Another quite widespread traversal algorithm is the depth-first search strategy used also by Tiruta-Barna et al. (2016). It has lower memory requirements but a longer running time (Marvuglia et al. 2013). Here we propose to traverse the supply chain to solve the LCI dynamically based on a LCA informed best-first search strategy (Zhang and Korf 1993). Starting from the FU, the order in which each exchange $a_{i,p}$ (i.e. the node) in the technosphere matrix A is traversed is based on its relative contribution to the LCA score of the FU (see Temporalis algorithm in Appendix III for details). This means that the nodes with the highest impact relative to the impact of the FU are evaluated first. The traversal continues through the supply chain until either the impact of the traversed node is below the LCA cutoff criterion, represented by the potential relative impact of the exchange to the FU assessed (by default, 0.1%), or until the maximum number of traversal steps has been reached (by default, 10,000). Calculating the relative LCA score can be tricky for dynamic impact assessment functions, as our general-purpose methodology should allow such functions to have arbitrary complexity. The approach we have chosen to handle such functions is to evaluate them over the entire time period of interest and use a conservative worst-case strategy when solving the dynamic LCI with the traversal algorithm. An incorrect use of the CF at this stage might lead to the exclusion of important flows, but if an input is not important (in the sense of contributing to the total LCA score) applying even the highest possible characterization factors, then we can safely exclude it. Three different cases can be encountered depending on the nature of the IA used, for which the worst-case CF used to solve the dynamic LCI changes accordingly (equation 3.3).

$$worst-case\ CF = \begin{cases} CF, & if\ CF\ static \\ max(\{CF(t): t = 0, \dots, TH\}) & if\ CF\ dynamic \\ \int_0^{TH} CF(t)dt, & if\ CF\ extended \end{cases} \qquad \text{equation 3.3}$$

The simplest is when a static CF is used. In this case, the CF consists of a value that is time-independent (e.g. GWP) and the CF values are used as they are (first case in equation 3.3). In the other two situations, the impact assessment used is time-dependent. For those impacts that are subject to seasonal variations, like photochemical oxidation, the highest possible value of the dCF is used (second case in equation 3.3). By doing so we are sure that, if the impact for a certain process is below our cutoff, even with the highest possible CF, it is not prematurely excluded. The last case is when a CFun is used, namely when the impact of the flow emitted is distributed over time. When calculating the Radiative Forcing (RF), for instance, the impact is spread over time for a length that is function of the decay rate of the flow emitted. The most impacting situation is when the emissions occur at year 0. Consequently, the integral over the TH of the analysis is taken in the worst-case approach for all the environmental interventions (third case in equation 3.3). Depending on the characteristic of the IA method, the use of the worst-case strategy ensures that all potentially important flows are not prematurely excluded during the resolution of the dynamic inventory problem. The CF to use during the traversal must be decided by the user before starting the calculations based on the CF that will be used in the IA phase following the worst-case approach of equation 3.3. For example, if the goal is to estimate the climate change impact using RF, the dynamic LCI must be resolved using as worst-case CF the third case in equation 3.3. The algorithm is CF-specific, meaning that when other impact categories are required to be assessed, the dynamic LCI must be resolved against the new worst-case CF. Failing to do so can produce incorrect results since each process can have a different relative importance depending on the evaluated impact category.

A methodological problem arising from the treatment of the technosphere matrix as a graph is its cyclic nature. The presence of loops, in fact, make the traversal infinite without any stop condition. Other dynamic LCA methods that apply graph traversal use a temporal cutoff as stop condition, interrupting the iterations when exchanges occur outside a certain time window (Tiruta-Barna et al. 2016). In our case, when loops occur, they continue to be traversed until the impact of the node falls below the LCA cutoff value or the loop is repeated a certain amount of times. By default, this loop cutoff (Lco) is set to 10 iterations but can be modified according to practitioner needs (the higher the number, the higher the precision at the expense of running time). After an exchange is looped Lco times, the same approach used for static databases is applied (first case in equation 3.4). This approach avoids infinite loops; the resulting introduction of imprecision in can be reduced by increasing the Lco value.

For each node evaluated during the traversal, both process and elementary flow are calculated, temporally propagated, and all the resulting environmental interventions $g_{j,p}(t)$ are added to a timeline $T_{t,i,p}$, a three dimensional array containing all the $g_{j,p}(t)$ flows of the studied FU. In $T_{t,i,p}$ the dimension i corresponds to a specific elementary flow (e.g. kg of CO_2), the dimension t to the calendar date of that emission and the last dimension p to the process responsible of the emission, as presented in equation 3.4.

$$\mathrm{T_{t,i,p}} = \begin{array}{c} \nearrow p \\ \downarrow i \end{array} \begin{bmatrix} T_{t_0,1,1} & \cdots & T_{t_n,1,1} \\ \vdots & \ddots & \vdots \\ T_{t_0,n,1} & \cdots & T_{t_n,n,1} \end{bmatrix}_{\xrightarrow{t}} \begin{bmatrix} T_{t_0,1,2} & \cdots & T_{t_n,1,2} \\ . & . & \vdots \\ & & T_{t_n,n,2} \end{bmatrix} \begin{bmatrix} T_{t_0,1,n} & \cdots & T_{t_n,1,n} \\ . & . & \vdots \\ & & T_{t_n,n,n} \end{bmatrix} \qquad \text{equation 3.4}$$

The resulting timeline contains the time of occurrence of all environmental interventions meeting the requirements of a dynamic LCI, as given by Levasseur et al. (2010). To this timeline it is easy to apply both static and dynamic characterization factor, as well as characterization functions, as we show in the next section.

3.3 Dynamic impact assessment

Dynamic impact assessment methods that spread the impact over time, such as dynamic GWP, can be easily implemented in our proposed framework. Each characterization factor would behave the same as an edge in the supply chain graph - it would have a relative temporal distribution that could be convolved with inventory distributions. The inclusion of dynamic impact assessment functions, which produce characterization factors or temporal characterization distributions, can also be included in our method. Indeed, such functions can even take discretized temporal distributions as inputs, treating each pair of (emission amount, time) as a separate flow to be characterized.

With the timeline populated, it is possible to calculate impact for the chosen IA method both for the whole system or separately by processes and/or flows. When the whole system is assessed, the timeline $T_{t,i,p}$ is reduced by one dimension to $T'_{t,i}$ in order to reduce subsequent IA calculations time (equation 3.5).

$$T'_{t,i} = \sum_{p\ \in p} T_{t,i,p} \qquad \text{equation 3.5}$$

At this stage, it is easy to calculate the environmental impact over time of the studied FU (h_t) or by environmental interventions ($h_{t,i}$). Thanks to the nature of the timeline, which retains also information about the process responsible for each environmental intervention ($T_{t,i}$ $\forall$ $p{:}T_{t,i,p}$), it is also possible to calculate environmental impact by the process p over time simply by looping over each $T_{t,i}$. Practically, when using a static CF, the environmental interventions are multiplied by the CF and the data are grouped based on time t. We simply take our two-dimensional array $F_{t,i}$ ($F_{t,i}=T'_{t,i}$ or $T_{t,i}$ $\forall$ p:$T_{t,i,p}$) and the q_i vector with the CFs for each environmental interventions i and apply equation 3.6.

$$h_{t,i} = \sum_{i \in i} F_{t,i} q_i \; ; \; h_t = \sum_{i \in i} h_{t,i} \qquad \text{equation 3.6}$$

When a dCF or a CFun is used equation 3.7 is applied.

$$h_{t,i} = \sum_{i \in i} \sum_{t \in t} F_{t,i} \, q_{t,i} \; ; \; h_t = \sum_{i \in i} h_{t,i} \qquad \text{equation 3.7}$$

3.4 Software implementation

We have implemented our methodology in a free and open source software package written in python and called Temporalis. One of the limitations of the previous approaches for dLCA is that they are still experimental and not yet operationalized into a readily usable tool. In our case, the software has been implemented as part of the open source framework for life cycle assessment Brightway2 (Mutel 2017b). It is well known that opening up software and algorithms increases transparency, a feature that LCA still lacks as already stressed several times in recent years (Frischknecht 2004, Finnveden et al. 2009, Pauliuk et al. 2015). An increased level of openness of LCA algorithm and software development can help to get constructive feedback from other users with the final result of obtaining also better software and, broadly speaking, LCA analysis. Brightway2 is fully compatible with many existing commercial LCI databases like, among others, Ecoinvent (Wernet et al. 2016), Agrybalise (Colomb et al. 2015), the World Food LCA Database (Lansche et al. 2013) and FORWAST (Villeneuve et al. 2009). As part of the software library, we wrote a custom convolution function that does not require a fixed and continuous temporal resolution. Furthermore, dynamic IA methods for climate change impacts based on the 2013 Intergovernmental Panel on Climate Change (IPCC) methodology (IPCC 2013) are already included. They allow calculating GWP and Global Temperature Potential (GTP) dynamically and overcome the temporal inconsistency due to the use of static IA. To explicitly account for the temporal discrepancy of biogenic carbon fluxes due to their delayed re-sequestration after emission, also the methodology of Cherubini et al. (2011a) and Cherubini et al. (2012) has been implemented

3.5 Virtual example

Here we illustrate and validate the functioning of the Temporalis tool using a simple fictitious example. Figure 3.1 presents a system of six unit processes, involving a loop between process 2 and 6, and two processes (1 and 3) without temporal information (i.e. static).

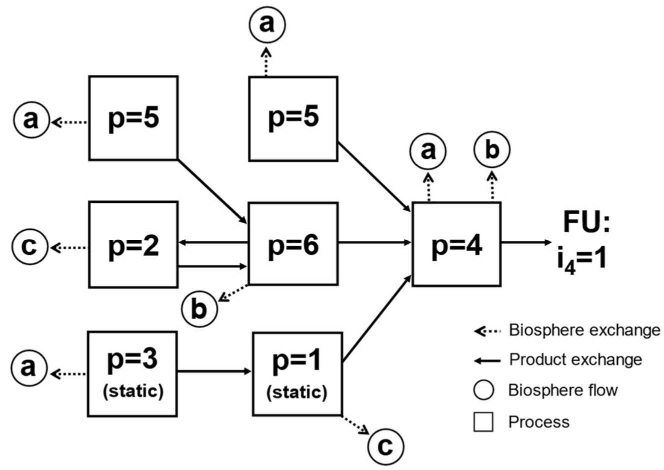


Figure 3.1 Schematic representation of the virtual example modelled. Functional Unit is equal to 1 unit of product 4. Processes 1 and 3 are static (i.e. without temporal distributions).

Three biosphere flows are considered and a fictitious CF that is equal to 1 for all the flows is used as worst-case CF. The FU for this example is one unit of the product 4 and t0 is set equal to 01.01.2017. In Table 3.1 all the exchange amounts with their relative TD used in this example are given (for the sake of clarity a one year resolution has been used both in the codes and in the figure). Figure 3.2 shows the dynamic environmental interventions for each individual process $g_{j,p}(t)$ for the analyzed FU. The results are validated by comparing the static and dynamic cumulative environmental interventions and products' supply (Table 3.1). As can be seen, the dynamic approach gives almost the same outcome as a conventional static LCA. There is a slight difference in the results due to the nature of the best-first traversal methodology. This difference is in the order of magnitude of the LCA cutoff chosen and can be reduced by simply lowering the cutoff, at the expense of computation time.

Table 3.1 Parameters needed to perform a dynamic LCA using the software Temporalis

Variable	**Description**	**Mandatory (M) or Optional (O)**
TD_{ip}	Temporal distribution (absolute in time or relative to the consuming process) of technosphere exchange of the process j	O
TD_{jp}	Temporal distribution (absolute in time or relative to the consuming process) of biosphere exchange of the process j	O
t0	Starting date of the analysis	M
Worst-case CF	Characterization factor used during the traversal	M
LCA cutoff	Cutoff below which the process nodes are excluded during the traversal	M (in the software set by default to 0.01 of FU score)
Maximum calculation number	Maximum number of iterations of the graph traversal	M (in the software set by default to 10000)
Lco	Maximum number of iterations in a loop	M (in the software set by default to 10)

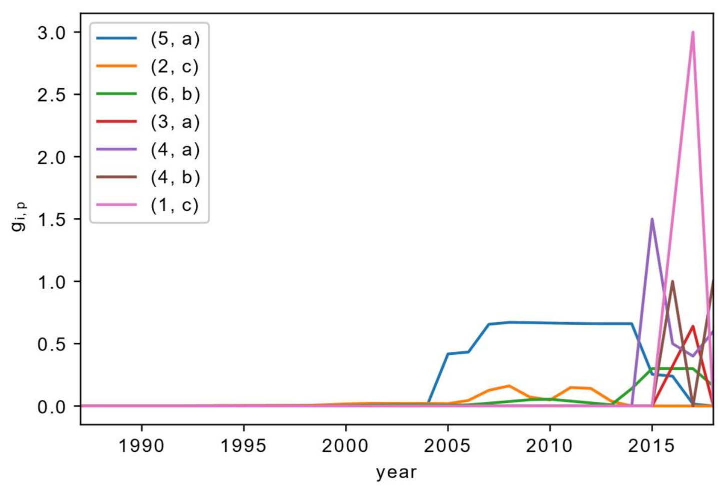


Figure 3.2 Temporally defined environmental interventions $g_{j,p}(t)$ for the virtual example i.e. environmental interventions (letter) for each individual processes (number) over time

Table 3.2 Parameters used in the virtual example and validation of the results.

Inventory exchanges			
Technosphere exchange (from-to)	**TD(t) (years, relative to the consuming process)**	**TD(i)**	$a_{i,p}$
3 to 1	static	static	0.4
6 to 2	[-3,-1]	[0.2,0.2]	0.4
1 to 4	[-1,0]	[0.2,0.4]	0.6
5 to 4	[-2,0]	[0.4,0.2]	0.6
6 to 4	[-1,0]	[0.14,0.16]	0.3
2 to 6	[-5,-4]	[0.2,0.3]	0.5
5 to 6	[-1,0,1]	[0.04,0.06,0.1]	0.2
Biosphere exchange	***TD(t)*** **(years, relative to the consuming process)**	***TD(i)***	$b_{j,p}$
c to 1	static	static	7.5
c to 2	[-5,-4,-1,0]	[1,1.5,1.7,0.8]	5
a to 3	static	static	4
a to 4	[-2-1,0,1]	[1.5,0.5,0.4,0.6]	3
b to 4	[-1,1]	[1,1]	2
a to 5	[-10,-9,-8,-7,-6,-5,-4,-3,-2,-1]	[1,1,1,1,1,1,1,1,1,1]	10
b to 6	[-2,-1,0,1]	[1,1,1,1]	4
Balance check			
Process	**Static ($\vec{s}$)**	**Dynamic**	**Difference (%)**
1	0.6000	0.6000	0.0000
2	0.1875	0.1875	-0.0053
3	0.2400	0.2400	0.0000
4	1.0000	1.0000	0.0000
5	0.6750	0.6750	-0.0030
6	0.3750	0.3750	0.0000
Flow	**Static ($\vec{g}$)**	**Dynamic**	**Difference (%)**
a	10.7100	10.7098	-0.0022
b	3.5000	3.5000	-0.0006
c	5.4375	5.4374	-0.0011

3.6 Application in a case study

The very long life cycles involved in the forestry-wood sector systems make them an exemplary field to illustrate the developed framework. An additional complication of impact assessment in this sector is due to the temporal discrepancy between the emissions of biogenic CO_2 and their capture through forest regrowth. We thus performed a cradle-to-grave dLCA for a reference flow of one m^3 of glulam. Biosphere and technosphere exchanges were modelled using own data for the foreground system and Ecoinvent 2.2 and 3.2 for the background. The choice of using both Ecoinvent databases is not casual. With this we want to show (i) that the framework can be efficiently applied to big commercial databases (Ecoinvent 3.3, consists of about 12,900 datasets) and (ii) how it is possible to temporalize also data coming from databases conceived in a static way, like Ecoinvent 2.2. In Figure 3.3 a representation of the dynamic system is given. The detailed system graph of the inventory can be found in the Github repository, together with the Jupyter notebook with the commented codes showing the step-by-step procedure followed to create the dataset.

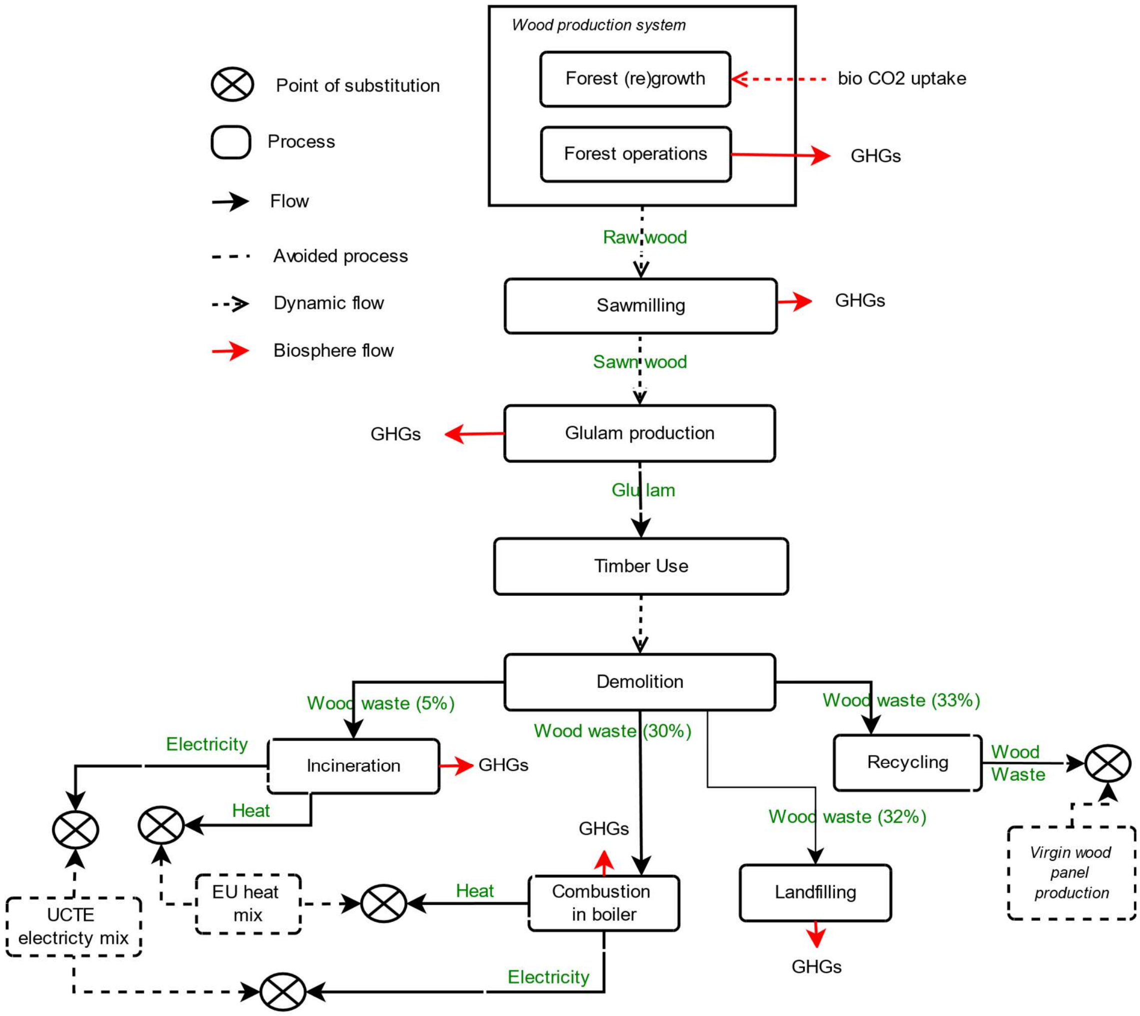


Figure 3.3 The product flow diagram of the Glulam use as modelled in the case study.

Raw wood production in the forest has been modelled based on the Ecoinvent 3.3 unit process "softwood forestry, mixed species, sustainable forest management". This dataset represents the sustainable forest management practices related to the production of 1 m^3 of softwood under bark over a rotation length of 130 years. It includes site preparation (assuming natural regeneration) and all processes related to forest management (i.e. clearing, tending, pruning, thinning and harvesting operations). We made this unit process dynamic by adding temporal parameters to the silvicultural management practices and temporally explicit biogenic carbon fluxes due to forest regrowth based on the information reported in the unit process description from Ecoinvent. For the management practices, the original exchanges in the Ecoinvent dataset were made dynamic by equally spreading their inputs over 9 thinnings and a final harvest. It was assumed that each of these 10 interventions had the same intensity and occurred every 10 years starting from year 40. For what forest regrowth is concerned, we applied the methodology proposed by Cherubini et al. (2011a) to model its atmospheric CO_2 re-sequestration rate (see equation A5 in chapter Dynamic climate IA methodology in Temporalis of Appendix II for details). The rate of biomass re-growth (i.e. *e(t')* in equation A5) has been modelled as a normal (Gaussian) distribution with mean (μ) equal to half of the rotation length and the variance (σ) that is assumed to be half of the mean (equation 3.8).

$$g(t) = \frac{1}{\sqrt{2\pi\sigma^2}} e^{-(t-\mu)^2/2\sigma^2} \qquad \text{equation 3.8}$$

We modelled a two years gap between forest harvesting and first transformation into sawnwood, and another two years between first and second transformation to glulam. The life cycle of the glulam has been modelled following the modelling principle of the Environmental Product Declaration (EPD) standard EN 15804 (CEN 2012) and the modules A1 to A5 (Product stage and Construction process), C1 to C4 (End-of-life stage) and D (Benefits and loads beyond the system boundary) were included. While the standard has been followed for what the system boundary, co-product allocation and the end-of-life modelling is concerned, in the following sections the results are not reported following the classification into modules used in the standard and only the GWP impacts are estimated. The life cycle inventories of both first and second transformation have been modelled mostly based on Ecoinvent 2.2. In accordance with the aforementioned standard in both stages economic allocation was applied. Also steel fittings are included in the modelling of the glulam production. At the end-of-life the glulam beam was assumed to be partially recycled, partially landfilled and partially used for energy recovery according to the figures reported in Mantau et al. (2010). Following the EPD standard, system expansion is applied in this stage and substituted impacts for recycling and energy recovery are included in the calculation. It was assumed that the electricity and heat recovered substitute respectively the current European electricity and heat production grid. The part that is recycled is assumed to replace the production of wood panels from virgin wood. For the glulam, an average service life λ of 50 years has been considered and the discard rate over time (i.e. the amount of the product that reach its end of life over time) has been estimated using the gamma distribution, as already suggested by Marland et al. (2010). This distribution has been parameterized with $a = k/2$ and $b = 2$, where k is a positive integer corresponding to the year of maximum oxidation (i.e. mean lifetime of the product λ) as proposed by Cherubini et al. (2012). This parameterization of the gamma distribution is equivalent to a Chi-squared distribution with k degrees of freedom (equation 3.9).

$$\chi^2(t;k) = \frac{(1/2)^{k/2}}{\Gamma(k/2)} x^{k/2-1} e^{-t/2} \qquad \text{equation 3.9}$$

Where t=time and Γ(k/2) is the gamma function in equation 3.10.

$$\Gamma(k/2) = \int_0^{-\infty} t^{k/2-1} e^{-x} dx \qquad \text{equation 3.10}$$

We solved the LCI statically and dynamically with t0 as the year of production of glulam (01.01.2017) and calculated for both the cumulative climate change impact. As IA method we used the (static) CFs for GWP published by IPCC and implemented in Ecoinvent (Bourgault 2015) and compared them with the dynamic GWP result, which accounts also for the climate change impact of forest biogenic CO_2 emissions and removals.

4. Results

Figure 3.4 shows the cumulative climate change impact for the case study over a time horizon TH of 20, 100 and 500 years using both static and dynamic LCA. First, we compared the results obtained using static (sLCI) and dynamic LCI (dLCI) for a static GWP over 20 (Figure 3.4a), 100 (Figure 3.4b) and 500 years TH (Figure 3.4c). It can be seen that the closer *t0* to the end of TH, the greater is the discrepancy between the two results. This is due to the fact that when using a static LCI all the environmental interventions are characterized regardless the timing of their occurrence while, when using the dynamic LCI, only the environmental interventions occurring within the TH are considered. The results over the complete TH, in fact, are equivalent between the two approaches, provided that all the environmental interventions are within this time window (as in the case of Figure 3.4c). In the results, the negligible difference between dynamic and static approach (~0.01%), is explained by the approximated results yielded by the graph traversal and explained above. Next, we compared these results with the cumulative climate change impacts obtained using a fully-fledged dLCA (i.e. both LCI and LCIA dynamic) over a time horizon of 500 years (Figure 3.4d). In this case, the results revealed are quite surprising and the difference between a conventional and a fully dynamic approach with a correct accounting of forest biogenic CO_2 fluxes are substantial. The estimated impacts are lower in the static approach with a relative difference of 226 %, 406% and 42% over 20, 100 and 500 years TH respectively. Even assuming the carbon neutrality of forests (i.e. without accounting biogenic carbon) the relative difference between the two results is important (274%, 151% and 29% over 20, 100 and 500 years TH respectively). Also when comparing these dynamic results (Figure 3.4d) with those using dynamic LCI and static LCIA (Figure 3.4c) it can be seen that the temporal evolution of impacts is sensibly different and the climate change impact due to forest regrowth plays an important role bringing the system to a higher impact for the first 145 years and then lower. Notable is the fact that while a fully static approach always gives negative values (thus a positive, mitigating, climate change impact due to glulam use), a fully dynamic analysis shows positive effects only 145 years after t0.

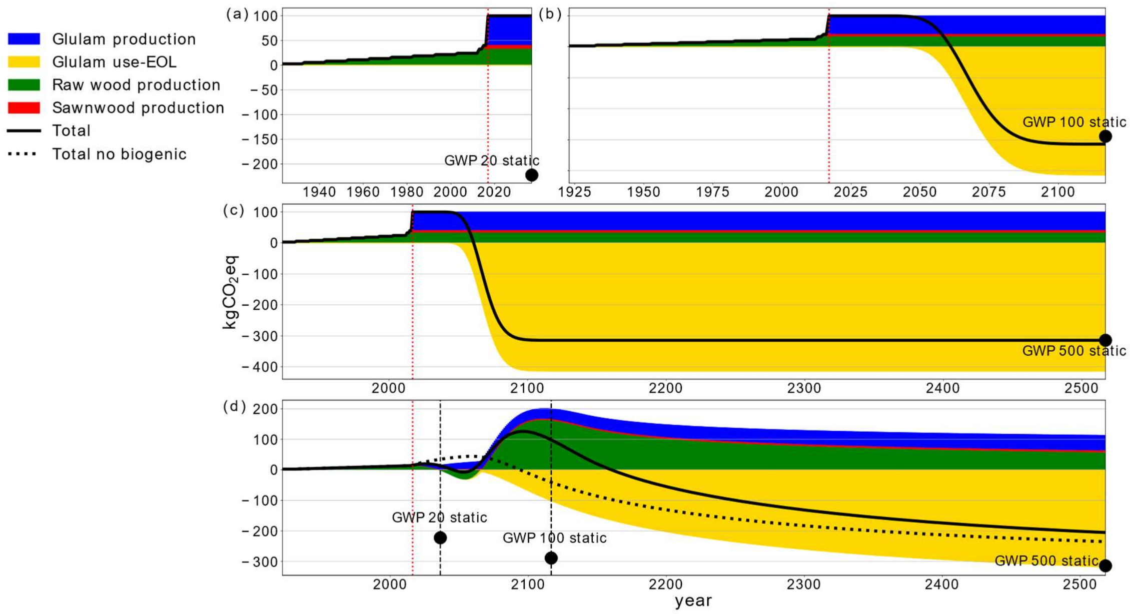


Figure 3.4 Cumulative climate change impact of the cradle-to-grave dLCA of 1 m^3 of glulam calculated over a time horizon of 20 (a), 100 (b) and 500 (c) years using static GWP and over 500 years using dynamic GWP (d). Vertical red dotted lines represents *t0* (2017). The temporal evolution of the impact is shown for each of the main four phases and for the total (black line). Black dotted line shows the total results without accounting for biogenic carbon (i.e. assuming carbon neutrality) and black dots indicate the results of the static LCA (i.e. both LCI and CF static) using different time horizons for GWP (20,100 and 500 years).

We then assessed the sensitivity of our results to the temporal parameters used evaluating the same system but with varying rotation lengths of 50, 130 and 200 years and product lifetimes of 1, 50 and 150 years (Figure 3.5). The results change quite substantially depending on these temporal parameters. For all three TH considered the shorter the rotation length the lower is the impact. This is explained by the shorter residence time of the biogenic carbon in the atmosphere. Being re-sequestered more rapidly this carbon exerts its warming effect for a shorter time period. Inversely, the longer the lifetime of glulam, the higher are the climate benefits of postponing biogenic carbon emissions. In this case the extension of the lifetime of the product let the re-growing stand sequester atmospheric carbon other than the one emitted during the previous rotation, augmenting the temporary sink effect of the stand. For the same system the GWP impact for a TH of 20 years can range from -71 kg CO_2eq (Figure 3.5b) to 443 kg CO_2eq (Figure 3.5g), from -901 kg CO_2eq (Figure 3.5c) to 667 kg CO_2eq (Figure 3.5g) for a TH of 100 years and from -546 kg CO_2eq (Figure 3.5c) to -120 kg CO_2eq (Figure 3.5h) for a TH of 500 years based on the rotation length and the lifetime of the product studied.

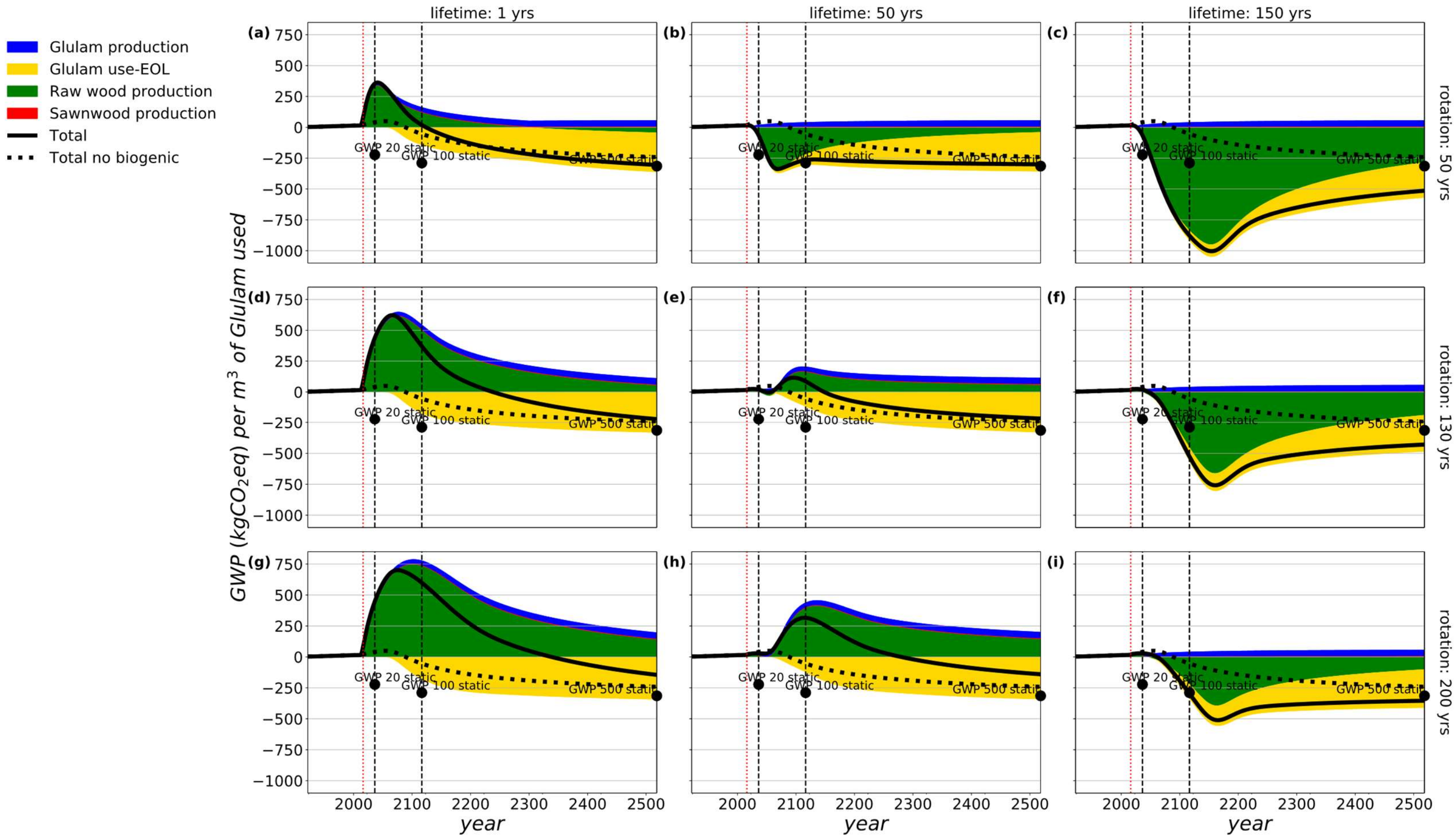


Figure 3.5 Sensitivity analysis of the cumulative climate change impact of the cradle-to-grave dLCA of 1 m^3 of glulam to rotation length and glulam lifetime over a time horizon of 500 years. Rotation length in the forest of 50 (a,b,c) 130 (d,e,f) and 200 (g,h,i) years and lifetime of glulam use of 1 (a,d,g) 50 (b,e,h) and 200 (c,f,i) years are considered. Vertical red dotted line represents *t0* (2017). The temporal evolution of the impact is shown for each of the main four phases and for the total (black line). Black dotted line shows the total results without accounting for biogenic carbon (i.e. assuming carbon neutrality) and black dots indicate the results of the static LCA (i.e. both LCI and CF static) using different time horizons for GWP (20,100 and 500 years).

5. Discussion and Conclusions

The methodology reported in this paper goes a step further compared to what has been already done in the field of dynamic LCA. It allows for the accounting of time at all the levels outlined in the introduction and is fully flexible for what the temporal information is concerned. This flexibility makes it possible to easily and efficiently use the methodology and the Temporalis software with already existing databases that traditionally lack temporal information. In our case study, for example, the dynamic LCI is solved in about 16 seconds and the dynamic LCIA in approximately 34 seconds on a regular laptop (Intel(R) Core(TM) i7-6820HQ CPU 2.70GHz, 8GB RAM), with a maximum memory usage of less than 350 MB.

Our methodology, in analogy with the ESPA and the method of Tiruta-Barna et al. (2016), makes use of graph traversal to solve the inventory, but differentiate from them from the uses of a different algorithm. With ESPA it further shares the fact that temporal information is propagated using convolution and that the technosphere and biosphere exchanges relative to each unit process are represented by means of temporal distributions. With regard to this, Temporalis has the advantage to be able to handle also exchanges occurring at a fixed time or not reported in the database, which is not the case for the other methods. Tiruta-Barna uses temporal cutoff to deal with loops, while Temporalis uses the loop cutoff (Lco) approach while, due to the type of traversal algorithm used, ESPA does not need to deal with loops.

Data availability is and will continue to be a major limitation for the application of dynamic LCA. The ability of Temporalis to combine both static and dynamic inventory data is therefore remarkable. While already operational, Temporalis and its underlying methodology can still be further refined and improved. For example, the dynamic LCIA implementation could be improved, creating a more robust framework based on an improved version of the one developed by Beloin-Saint-Pierre et al. (2017). They proposed the use of the Hadamard product between a two-dimensional matrix $G`$ representing the biosphere flow emissions (row) and the time of their emission (column) with the matrix $H$ containing specific time-dependent CF (column) for each biosphere flow (row). An improved version of this approach could be implemented using a three-dimensional matrix for the $G`$ with the inclusion of a third dimension for the process responsible of each emission to allow for a better interpretation of the results compared to Beloin-Saint-Pierre et al. (2017).

The importance of using dynamic analysis and accounting properly for biogenic carbon is confirmed by the case study results. The alleged positive climate effects due to glulam use (Sathre and O'Connor 2010), when studied dynamically, is only seen with a certain delay (from 9 to 352 years in our glulam case-study) that depends on the temporal characteristics of the system, essentially rotation length and product lifetime. This aspect is of tremendous practical importance for wood products when their sequestration and substitution effect is estimated. In fact, while in the static analysis the (potential) climate substitution effect of wood product use is always found, a temporal explicit approach reveals that this phenomenon is very much influenced by the way the analysis is performed. From our results, it can be seen that, first, the positive effects are often over-estimated, and even more importantly, that, when seen, they take place only with a delay that is depending on the temporal characteristics of the studied life cycle. Most studies, ours included, are forward-looking, assessing forest carbon regrowth (Helin et al. 2013) and thus analyzing the forest carbon dynamics from the moment of harvesting onwards. However, some authors suggest taking a backward-looking in which the past carbon fluxes due to the forest growth (and not re-growth) is considered (Sedjo 2011). While it is outside the scope of the paper to discuss which is the most correct assumption, the backward approach would reduce the time needed for the system to start exerting its substitution effect. As both approaches are discussed in the literature, the importance of an adaptable and efficient dynamic LCA tool is reinforced.

The results of this case study confirm how dynamic LCA is particularly relevant when analyzing long life cycles and, in assessing climate change impacts, when also the dynamics of biogenic carbon are accounted for (i.e. without assuming any carbon neutrality).

Further progress towards more accurate LCA analysis will certainly be obtained by coupling temporally and spatially resolved analysis. The idea is to come to a full spatio-temporal framework combinable with the computational structure of LCA software and databases in use nowadays. Being based on the traditional matrix-based approach, our dynamic methodology could be easily combined with location data, provided that the regionalized LCA methodology used fits into matrix math structure. Our next step is to work towards this spatio-temporally defined LCA approach including uncertainty, coupling our method with the matrix-based regionalized framework proposed by Mutel and Hellweg (2009) and already developed in Brightway2.

Chapter 4

The impact of European forest management under alternative silvicultural scenarios

Adapted from: Cardellini, Giuseppe, et al. "The net impacts of European forest management under alternative management scenarios". In preparation

1. Abstract

European forests provide a wide range of ecosystem services and timber production is one of the most important. While forest management should achieve multiple objectives, mitigation of climate change is becoming an increasingly important goal. Changes in forest management practices have the potential to maximize the delivery of specific functions and/or goods. However, it should be ensured that the multifunction potential of forests is also preserved, and that forest management is balanced. In this work we analyzed the effectiveness of three identified forest management strategies in Europe in terms of carbon balance, timber harvesting and climate change impact measured as Global Warming Potential, also considering the future demand of timber. We found that timber production is a climate efficient production chain, with a GWP impact ranging from -1986 kg CO_2eq/m^3 to -2989 kg CO_2eq/m^3 depending on the year and the scenario. Furthermore, we found that the realizable changes in forest management are constrained by the demand for wood and that economic considerations still play a crucial role in determining the achievable outcomes. To obtain the strategically planned goals strong incentives, e.g. in form of economic subsidies, can be necessary to modify the traditional profit-oriented forestry practices.

2. Introduction

About 33% of EU land area, corresponding to 215 million hectares (ha), is covered by forests and other wooded land, out of which around 80% is available for wood supply (Forest Europe et al. 2015). Wood is undoubtedly the most important product delivered by European forests. It is the prominent source of revenue from these ecosystems, with the wood and pulp industries contributing to one per cent of the European GDP (Forest Europe et al. 2015), and one of the most important raw materials for the rapidly evolving bio-based industries (Van Renssen 2014). The bio-economy is seen from the EU as a keystone toward the transition to a low-carbon society (EU 2011). The strong emphasis put on this transition is expected to further raise the demand for forest biomass at European level. To meet the ambitious EU climate and energy targets by 2030 an additional supply of 160 million m^3 yr^{-1} of wood would be needed (Mantau 2015). But forests are not only important for wood production and other aspects like biodiversity conservation and mitigation to climate change need to be considered when implementing future forest management strategies. Also in the new EU forest strategy (European Commission 2013), it is stressed that, despite wood is a crucial source of raw material, the role of forests is not and will not be restricted to timber production. In the document, it is highlighted that aspects related to biodiversity conservation, bioenergy production and use and climate change mitigation need to be considered. It is thus important to find the optimal management of forest resources that can ensure a sustained level of wood production without harming the other environmental services.

Among the different functions and services provided by forest, climate change mitigation is undoubtedly one of the most relevant. The role of forest management on the overall carbon sink seems to be substantially higher than expected before (Erb et al. 2013) and forest-based climate change mitigation strategies can play an important role in help fighting climate change and meet the future climate change mitigation targets. Globally land use, and particularly forests, are expected to contribute to about a quarter of the planned emission reductions of the Paris agreement (Grassi et al. 2017). But the role of forests in mitigating climate change is not restricted to C sequestration (Lippke et al. 2011) due to the mitigation role played by wood-based products. The combined use of carbon sequestration and the provision of forest-based products (e.g., timber and biomass for material and energy substitution) will augment the contribution of forestry in climate change mitigation (Canadell and Raupach 2008). Nabuurs et al. (2017) estimated that by 2050 the whole sector could mitigate an additional 441 Mt CO_2 $year^{-1}$. To maximize the climate mitigation potential of forest-based products it is crucial to ensure a climate-smart production of timber. While the renewable nature of woody biomass as energy and material is well-established, its sourcing requires also a certain amount of non-renewable inputs that inevitably affect the assumption of wood as a fully renewable material.

To maximize the climate benefits of wood use, it is important to understand the net climate change impact of its provision. So far, most of the studies on the climate mitigation potential of European forestry focus only on biomass carbon fluxes neglecting the greenhouse gas (GHG) emissions from forestry activities (Böttcher et al. 2012, Packalen et al. 2014, Pilli et al. 2016, 2017, Schelhaas et al. 2017). These studies look at the GHG fluxes occurring in the forest but neglect both the GHG emission from forestry operations and the estimation of the climate change impact of all these fluxes.

Among the many methodologies used to evaluate the impact of humankind activities Life Cycle Assessment (LCA) allows to asses environmental sustainability of the system studied from a holistic perspective (ISO 2006a). In recent years different life cycles studies have tried to assess the climate change impact of raw wood supplied from different management practices in various European countries (Klein et al. 2015). Some studies used life cycle assessment to compare the impact of forest operations on different management practices (González-García et al. 2014a, 2014c, Murphy et al. 2014, Klein et al. 2016), others to assess the direct climate change impact of biogenic carbon fluxes on forest regrowth (Cherubini et al. 2011b, 2013b). Each of them focused on different aspects and at the local scale. There is still a lack of a wider assessment of the potential climate of timber provision for the European region in consideration of the goals of the European forest management and the future demand of wood.

It is widely accepted that forest management provides a good instrument to fight climate change (Bellassen and Luyssaert 2014, Ussiri and Lal 2017) and that forest management can be changed to further augment this mitigating effect (Köhl et al. 2010, Bright et al. 2014). However, silviculture choices are influenced by many factors and, among them, economic aspects often play a predominant role (McConchie 1976, Eggers et al. 2014, Sarvašová et al. 2014). The adoption and effectiveness of continental-scale forest strategies should thus be confronted with the needs and the decisions taken by local actors. As the relative demand of timber is one of the strongest drivers of decision-making for forest managers, to estimate the effectiveness of such management strategies this aspect should also be considered.

In this study, we estimated the impact of a set of management strategies with different underlying objectives on the European forests. To achieve this, we developed 3 different management scenarios, reflecting 3 different main management objectives, and modelled the responses of the forests in terms of carbon balance, timber availability and global warming impact. In the analysis, we considered the dynamic of carbon in the forest as well as the anthropogenic GHG emissions from the silvicultural operations using a life cycle-based approach. All these effects were estimated from present (2011) to 2050, taking also into account the future demand of harvested timber.

3. Material and Methods

3.1 General approach

The development of European forests was estimated from 2011 to 2050 for current management practices and two alternative management scenarios (see section Forest management scenarios). We used the FORMIT-M forest simulator to model the future response of European forests to current and alternative forest management under expected climate (see section The FORMIT-M simulator). The future harvest levels in each year were defined on the basis of the identified management rules and the future wood demand modelled by the partial equilibrium model (PEM) EFI-GTM (Kallio et al. 2004, Moiseyev et al. 2011). The simulated above and below ground biogenic forest C fluxes and wood harvesting levels were used in the life cycle analysis The anthropogenic GHG emissions due to forest management operations were estimated using the EFO-LCI database (see section GHG emissions from forestry operations). With the anthropogenic and biogenic GHG fluxes calculated we estimated the cumulative GWP of the scenarios and their relative timber production impacts (see section Global Warming Impact). Each of these steps is explained in more detail in the following sections.

3.2 Study area and forest classification

The geographical boundary of the study consists of 32 countries within the European subcontinent. Countries were subdivided into 5 main eco-regions (Figure 4.1) as defined in the State of Europe's Forest report (Forest Europe et al. 2015).

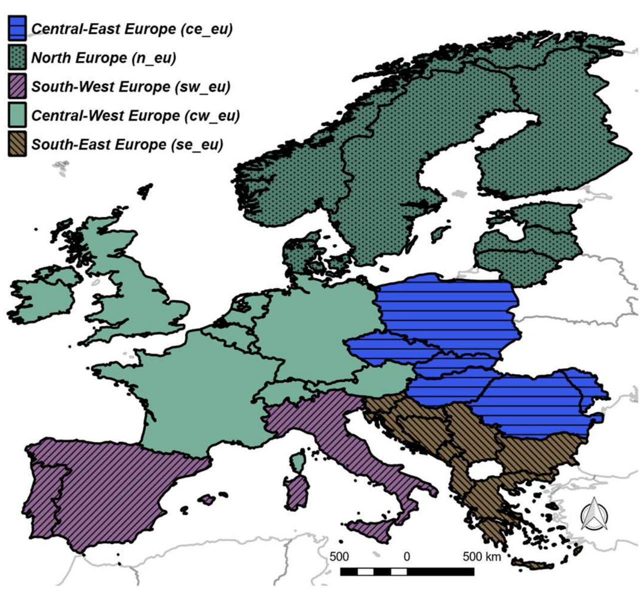


Figure 4.1 Countries included in the analysis grouped by ecoregion.

A FoU (Cardellini et al. 2018b) is defined by the combination of silvicultural system (SS) (Table 4.1) and tree species group (TSG) (Table 4.2). Combined with eco-region (ER) or country, the FoUs allow considering also the geographical differences of the forests.

The current area covered by European forests (177 million hectares) was produced using the Forest Cover Map of Kempeneers et al. (2011) as base layer. Initial TSGs distribution has been derived from Brus et al. (2012), which reports the species distribution of the 18 main European tree species. Each of these species was reclassified and aggregated according to the TSGs in Table 4.2.

The current share of silvicultural systems in each country was based on available national statistics and expert judgement collected with the EFO-LCI questionnaire (Cardellini et al. 2018b). The answers were based on existing forestry statistics whenever possible, and on expert judgment where needed. For each FoU we modelled stand dynamics and forestry operations from 2011 untill 2050 based on the identified management practices (see section Forest management scenarios). Being the goal of the study to assess the effect of forest management alternatives, land-use was assumed constant (no afforestation or deforestation) in all the scenarios. Only forest management changes were modelled and studied, namely change in the type of management within a FoU and in the relative area between FoU (see Appendix III for simulated area changes).

Table 4.1 Silvicultural systems

Code	System	Definition
1	Unmanaged forests	No management
2	Continuous cover forest management	Continuous cover forest management • Selection cuttings based on target diameter
3	Even-aged forest management with shelterwood	Even-aged forest management Regeneration: natural Thinnings Shelterwood cut after a certain mean diameter (or age) has been reached
4	Even-aged forest management: uniform clearcutting system	Uniform forest management Regeneration: planting or natural Thinnings Clear-cut after certain target diameter (or age) has been reached
5	Coppice	Woodland which has been regenerated from shoots formed at the stumps of the previous crop trees, root suckers, or both, i.e., by vegetative means
6	Coppice with standards	Coppice system under low-density uneven-aged high forest
7	Short rotation	Plantation forestry including exotic species

Table 4.2 Tree species groups

Code	Species group	Species

A	Light-demanding conifers	Pinus sylvestris, Larix spp., Pinus nigra, Pinus cembra, Pinus heldreichii, Pinus leucodermis, Pinus radiata, Pinus uncinata, Pinus mugo, Pinus contorta, Pinus strobus, Cedrus spp., Juniperus spp.
B	Shade-tolerant conifers	Picea abies, Abies spp., Pseudotsuga menziesii, Thuja spp., Taxus baccata, Tsuga spp., Chamaecyparis spp.
C	Mediterranean conifers	Pinus pinaster, Pinus halepensis, Pinus pinea, Pinus canariensis, Cupressus spp., Pinus brutia
D	Fast-growing deciduous	Betula spp., Populus spp., Alnus spp., Salix spp., Robinia pseudoacacia, Eucalyptus spp.
E	Slow-growing light-demanding deciduous	Quercus robur, Q. petraea, Q. cerris, Q. pubescens, Q. faginea, Q. frainetto, Q. macrolepis, Q. pyrenaica, Q. rubra, Q. trojana, Q. hartwissiana, Q. vulcanica, Q. macranthera, Q. libani, Q. brantii, Q. ithaburensis, Q. pontica, Fraxinus spp., Castanea sativa, Rosaceae (Malus, Pyrus,Prunus, Sorbus, Crataegus, etc.), Juglans spp., Cercis siliquastrum
F	Slow-growing, shade-tolerant deciduous	Fagus spp., Carpinus spp., Tilia spp., Ulmus spp., Buxus sempervirens, Acer spp. Ilex aquifolium
G	Mediterranean evergreen trees	Quercus suber, Quercus ilex, Q. coccifera, Q. lusitanica, Q. rotundifolia, Q. infectoria, Q. aucheri, Tamarix spp. Arbutus spp., Olea europea, Ceratonia siliqua, Erica spp. Laurus spp., Myrtus communis, Phillyrea spp. Pistacia spp. Rhamnus spp. (R. oleoides, R. alaternus), Ilex canariensis, Myrica faya

3.3 Forest management scenarios

A business as usual (BAU) and two alternative forest management scenarios were developed. In the alternative scenarios, the objective management was modified from the BAU so as to fulfil their main objective using the silvicultural advice provided by country experts. To factor out the legacy effects of past management on future forest development, we used a dynamic, forward-looking baseline approach (Böttcher et al. 2008) where the continuation of the present forest management practices (i.e. the BAU) is considered to be the reference system of the study. The overall objective of each scenario and their translation into management practices are described in Table 4.3.

EFI-GTM BAU's general economic assumptions followed the Shared Socioeconomic Pathways 2 (SSP2) from IPCC (Riahi et al. 2017), which is considered to be a "Current Trends Continue" scenario. The specific wood demand related assumption is the same of the "Crunch" scenario in Kallio et al. (2015). BioEne differs from BAU for the higher demand of wood for energy production while scenario Biodiv, demand side, does not differ from the BAU scenario.

Table 4.3 Forest management scenario description with relative management rules used in the study

Code	Scenario	Description	Management Rules
BAU*	Business as usual	Continuation of the present management practices for the entire simulation period until 2100.	• Thinning rules followed are described in Appendix III; • The forest stand is established always with the same species and silvicultural system after the final cut.
Biodiv	Biodiversity and Conservation	Forest management is focused on biodiversity and nature conservation and assumes that the contribution to climate change mitigation can be obtained with higher stocks of carbon in the forests. The area of unmanaged forest is increased by 20%, the regeneration is done with species groups representing the "Potential natural vegetation of Europe" and mixed stands are preferred wherever possible. Deadwood is retained with a share of 20% of the harvested wood except in Mediterranean regions (for the high fire risk) and the rotation length is increased by 25%.	• 20% of the plots are left unmanaged (the plots located on protected areas + randomly selected plots); • Regeneration of new trees is done based on the map describing the potential tree species for that region; • 20% of the harvested stems are left as dead wood; • Final cut is postponed (25% longer rotation time); • Harvest residues are left in the forest i.e. not used as bioenergy (as for BAU).
BioEne	Maximum Bioenergy	The goal of forest management is to sustain the production of bioenergy. In terms of forest management this was simulated by applying no thinnings as there are no requirements towards wood quality; harvesting is done when stands reach maximum mean annual increment (MAI), a high portion of the harvest residues are used for bioenergy and there is an increased regeneration with fast-growing tree species.	• 66% of all harvest residues are utilized as energy wood • Spruce stumps are used for energy wood in the North • Final cutting is done when the mean annual increment of stem + branch biomass is highest • No thinnings are done • regeneration: • North: birch/spruce fertile sites (SC 2-3) and pine for others • Central Europe: broadleaved trees -> TSG D; conifers to TSG B

*Reference scenario

3.4 The FORMIT-M simulator

The effect of future climate and forest management on forest growth and stand structure was modelled using the FORMIT-M forest growth simulator (Härkönen et al. 2018), a hybrid stand growth simulator developed by combining several process-based and empirical summary models of forest ecosystem functioning. An earlier version of the model was validated for Finland in Härkönen et al. (2011) and Härkönen et al. (2010). The simulator is climate sensitive and uses forest inventory plots as a basic source of input data. It can be used to estimate forest development in Europe taking into account both climate and forest management actions. It comprises multiple regional sub-models (Figure 4.2), which are based both on earlier studies (GPP and NPP calculation, biomass and volume equations etc.) and new models fitted to country-wise national forest inventory (NFI) data (model describing the relationship between stem volume, stem diameter and stand density). The simulations of forest development were calibrated using NFI data from 11 European countries. For countries without available NFI data, regional functions were parameterized using neighbouring country's data and their choice was based on similarity of forest management, silvicultural systems and growth conditions (see Appendix III). The model was initialised simulating the forest development in all NFI points available, and all results were aggregated at FoU level. The year of initialisation depended on the NFI data available and was always from the first year of measurement available to the year 2010. Biomass equations, GPP and NFI-based NPP estimations used by the model were taken from Neumann et al. (2016a) and Neumann et al. (2016b). Future climate conditions were assumed to follow the Representative Concentration Pathways (RCP) 4.5 scenario (IPCC 2013) and the data used were produced by the MPI-M-MPI-ESM-LR model version CCLM4-8-17 (Giorgetta et al. 2013). In the simulator, the heterotrophic respiration of soil is simulated using the soil carbon model YASSO07 (Tuomi et al. 2011) with annual litterfall data inputs modelled by the simulator. For all scenarios, the harvest level in each FoU followed a *demand-limited* approach in which the total cuttings level per country for the years of the simulations were defined by soft linking FORMIT-M with the EFI-GTM model. By doing so the simulated harvest level took into account both the forest management objective and the predicted market demand of wood. The procedure took several steps. In a first step, FORMIT-M gave the harvest supply based on the conditions of the stands and the defined management rules, while EFI-GTM calculated the market demand by country. After this first run, the cutting levels were adjusted to satisfy the PEM demand, ensuring that the harvesting levels defined by the rules (Table 4.3) were not exceeded. These results were used as input for the next EFI-GTM run and the process was iteratively repeated until the cutting levels from FORMIT-M and the EFI-GTM demand were in balance i.e. when the difference between wood demand and supply was less than 5%.

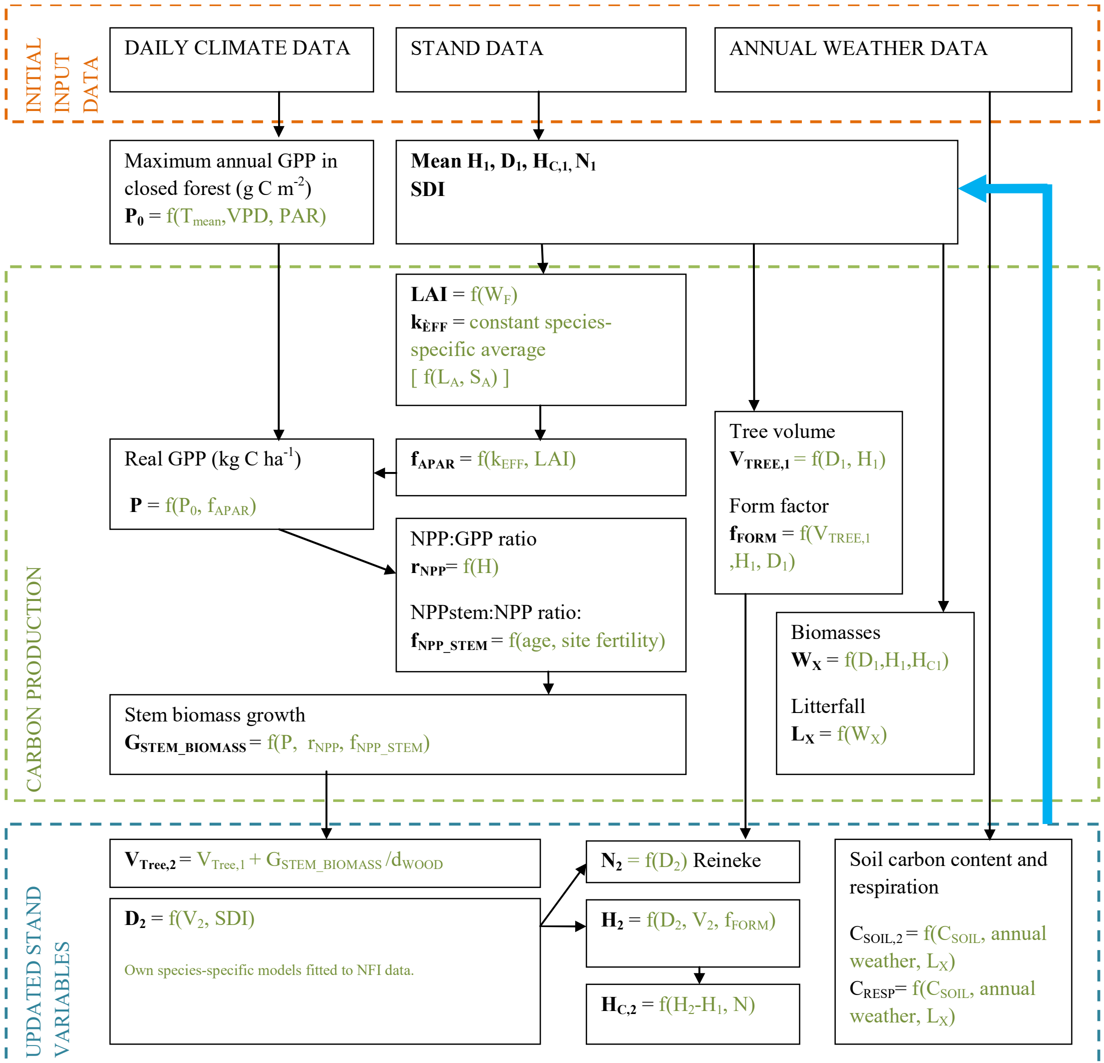


Figure 4.2 Schematic overview of the FORMIT-M forest growth simulator; maximum annual GPP is calculated on basis of climate scenario data and realized GPP, NPP, biomass allocation and growth are derived from this; harvest regimes are implemented reflecting forest management in the European regions.

The FORMIT-M simulator structure, parametrisation and validation for Europe against FAO statistics is described in Härkönen et al. (2018). In the same work are also described in detail the modelled processes, the model spin-up and initialization, the regional functions and data used in the simulations of this study.

3.5 GHG emissions from forestry operations

We modelled the GHG emission due to forestry operation following a life cycle approach. The foreground data used to model the emissions of forestry operations came from the European Life Cycle Inventory of Forestry Operations database EFO-LCI (Cardellini et al. 2018b). The database contains a detailed spatio-temporal life cycle inventory (LCI) of the forest management practices of 235 European FoUs in 28 European countries and was built based on surveys filled in by national silvicultural experts. EFO-LCI was used to obtain data on: (i) distances travelled by staff; (ii) density of forest roads; (iii) regeneration type and, in case of artificial regeneration; (iv) number of seedling per hectare; (v) type of intervention; for the equipment used type (vii) and (viii) operational productivities (in m^3/h or h/ha); (ix) type and (x) amount of inputs used (e.g., fertilizer) for all the FoUs reported. For those FoUs not described in EFO-LCI, we used proxy FoUs based on their similarity with the missing FoUs following the similarity rules reported in the section Identification of proxy FoUs of Appendix III. The background LCI data for the GHGs emissions of machinery use and production, seedling production and the material used for road maintenance and fencing were taken from Ecoinvent 3.3 (Wernet et al. 2016) and Agribalyse 1.3 (Colomb et al. 2015) (Table A.2). The forest management interventions reported in EFO-LCI were divided into generic and specific (Table A.1). With generic we refer to activities that are not related to the amount of timber harvested and depend on the area regenerated (e.g. ripping) and for which the operational productivities in EFO-LCI are reported in h/ha. Their total hours of use per FoU was calculated based on the area of forest regenerated yearly as modelled by the simulator. In contrast, specific interventions (e.g. secondary fellings) are directly related to the respective amount of timber harvested. In EFO-LCI these operational productivities are reported in m^3 overbark of wood harvested per ha (m^3 ob/ha) and their total hours of use per FoU was calculated based on the yearly amount of m^3 overbark of wood harvested as simulated by the forest model. Based on the amount of inputs used and the operational productivity of the machinery, the total use of each type of machinery (in hours of use) and input (amount) was calculated for both general and specific interventions in all FoUs as by first and second cases in equation 4.2. For manual activity, only the transport of the worker is considered. Based on the daily distance travelled by staff and the operational productivity of the activity, the total distance travelled is calculated following third and fourth cases in equation 4.2, which assumes a working day of 8 hours and that the work is performed from two workers. Forest road maintenance was modelled based on the skidding trail and road densities reported in EFO-LCI and assuming that overhauling measures take place every 15 years.

$$D_{FoU}(t) = \begin{cases} A_{reg}\ (t) * P_{h/ha}, & \textit{if generic} \\ W(t) * \dfrac{1}{P_{m^3/h}}, & \textit{if specific} \\ \dfrac{P_{h/ha} * A_{reg}(t)}{8}, & \textit{if manual in generic} \\ \dfrac{W(t) * \frac{1}{P_{m^3/h}}}{8}, & \textit{if manual in specific} \end{cases} \qquad \text{equation 4.1}$$

with:

D_{FoU} (t)=total use of machinery in hours/FoU in year *t*;

A_{reg} (t)= Area regenerated in FoU in year *t* (from FORMIT-M simulator);

$P_{h/ha}$= operational productivity of the intervention in h/ha (from EFO-LCI);

$P_{m3/h}$= operational productivity of the intervention in m^3/hours from (from EFO-LCI);

W (t)= m^3ob of wood harvested in the FoU in year *t* (from the FORMIT-M simulator).

3.6 Global Warming Impact

The results of previous steps were used to calculate the climate change impact of the system. The system boundary of this analysis included all silvicultural activities from site preparation to wood hauling to forest road. Both the emissions from forestry interventions and forest biogenic CO_2 fluxes were considered. The forestry related activities were combined into the three main process groups: site preparation (P), site tending (T) and biomass harvesting (H). Each of them embeds their secondary and logistic processes, which include respectively off-site processes and non-biomass transports entailed in the provision of raw wood (Klein et al. 2015, 2016). Biogenic C fluxes where grouped into the three main forest fluxes groups, namely net primary production (NPP), heterotrophic respiration (Rh) and harvested wood oxidation (Hw).

The climate change impact of each FoU measured as global warming potential (GWP_{FoU}) has been calculated summing the cumulative GWP contribution of each compartment over the time horizon:

$$GWP_{FoU} = GWP_{NPP} + GWP_{Rh} + GWP_{Hw} + GWP_P + GWP_T + GWP_H \qquad \text{equation 4.2}$$

The contribution due to biogenic CO_2 is represented by GWP_{NPP}, GWP_{Rh}, GWP_{Hw}, respectively the GWP impact of NPP, Rh and Hw. GWP_P, GWP_T and GWP_H are the anthropogenic GWP contribution due to site preparation, stand tending and harvesting operations. GWP has been estimated using methods and parameters reported in the 5th Assessment Report (AR5) of (IPCC 2013). First, the impulse response function (IRF) of CO_2 and the other greenhouse gases (GHG) *g* was calculated as follows:

$$IRF_g(t) = exp\left(-\frac{t}{\tau_i}\right) \quad \text{equation 4.3}$$

$$IRF_{CO_2}(t) = a_0 + \sum_{i=1}^{N} a_i\, exp\left(-\frac{t}{\tau_i}\right) \quad \text{equation 4.4}$$

The IRF describes the temporal decay of a GHG pulse emissions to the atmosphere based on its decay τ. Note that IRF for CO_2 is more elaborated since some 20% of the emitted CO_2 stays in the atmosphere for millennia due to the equilibrium response of the ocean-atmosphere system. For atmospheric CO_2 concentration, its response *f(t)* to any CO_2 flux perturbation can be computed via convolution of the IRF of CO_2 with the net CO_2 profile of the perturbation *p(t')*:

$$f(t) = \int_0^t p(t')IRF_{CO_2}(t-t')dt' \quad \text{equation 4.5}$$

While for the IRF of the GHGs emitted by forest operations equation 4.4 and equation 4.5 are used, we used equation 4.6 to calculate the IFR of biogenic carbon. With $p_{bio}(t')$ as the net biogenic CO_2 flux over time (i.e. from carbon oxidation or sequestration) in each FoU for the three components (NPP, Rh and Hw). The relative impulse response function (IRF_{bio}) is calculated as:

$$IRF_{bio}(t) = \int_0^t p_{bio}(t')IRF_{CO_2}(t-t')dt' \quad \text{equation 4.6}$$

Being the aim of the study to assess the impact of timber production, the system boundary is restricted to the forest and consequently it is assumed that the harvested wood is immediately oxidized after harvesting.

Based on the obtained impulse response functions the instantaneous radiative forcing for each GHG (RF_g) is calculated multiplying its atmospheric mass (IRF_g) by the relative radiative efficiency of the gas (RE_g):

$$RF_g(t) = RE_g IRF_g(t) \quad \text{equation 4.7}$$

$$RF_{bio}(t) = RE_{CO_2} IRF_{bio}(t) \quad \text{equation 4.8}$$

For the anthropogenic GHG emissions due to forest management operations, the absolute global warming potential (AGWP) for each GHG emitted at time *i* is calculated as:

$$AGWP_g^i = \int_i^{TH} RF_g(t)dt \quad \text{equation 4.9}$$

While for biogenic carbon AGWP is calculated as:

$$AGWP_{bio} = \int_0^{TH} RF_{bio}(t)dt \quad \text{equation 4.10}$$

with TH as the end of the time horizon of the analysis (2050) and 0 the year 2011. To note that by integrating in such a way any temporal inconsistency in the characterization of the GHG emission which typically occurs in LCA based analysis is overcome. AGWP is then converted to global warming potential (GWP) based on the AGWP of CO_2 calculated from 2011 to 2050 ($AGWP_{CO_2}^{TH}$)

$$GWP_g^i = AGWP_g^i / AGWP_{CO_2}^{TH} \qquad \text{equation 4.11}$$

Once the GWP_g^i of each gas g emitted at time i is calculated, the overall GWP due to forest management operations is simply obtained summing the impact of all the gases emitted each time as:

$$GWP = \sum_{i \in i} \sum_{g \in g} GWP_g^i \qquad \text{equation 4.12}$$

All the LCA analysis has been performed using the software Temporalis (Cardellini et al. 2018a) which, for methane emissions, estimates also its decay into CO_2.

4. Results

4.1 Timber harvesting

The continuation of the current management would lead to a harvesting level of 620 Million m^3 yr^{-1} by 2050 in Europe (Figure 4.3). The increase in the harvesting levels is due to the increase in the increments over time due to the warmer future climate conditions. By the end of the simulation period, the intensified management of BioEne would raise the yearly harvesting level with 15% in comparison to the BAU. This is mostly due to the to the fact that in this scenario harvesting is done at maximum mean annual increment. The ecoregion which experiences the highest boost in harvesting levels is sw_eu, with an increase in the harvesting rate of 29%, while the lowest increase in harvesting levels would take place in cw_eu (+7%). This behaviour is driven by the differences in the current (i.e. BAU scenario) management in the two ecoregions. While in the former the current low intensity of the management allows for an increase in the harvesting levels, in the latter the management intensity is already high and thus difficult to further increase the harvesting levels. The biodiversity scenarios, on the contrary, would reduce the harvesting rates by 14% with cw_eu as the ecoregion showing the highest relative reduction in harvesting levels (-18%) and only se_eu would slightly increase its harvesting levels in comparison to BAU (+2%).

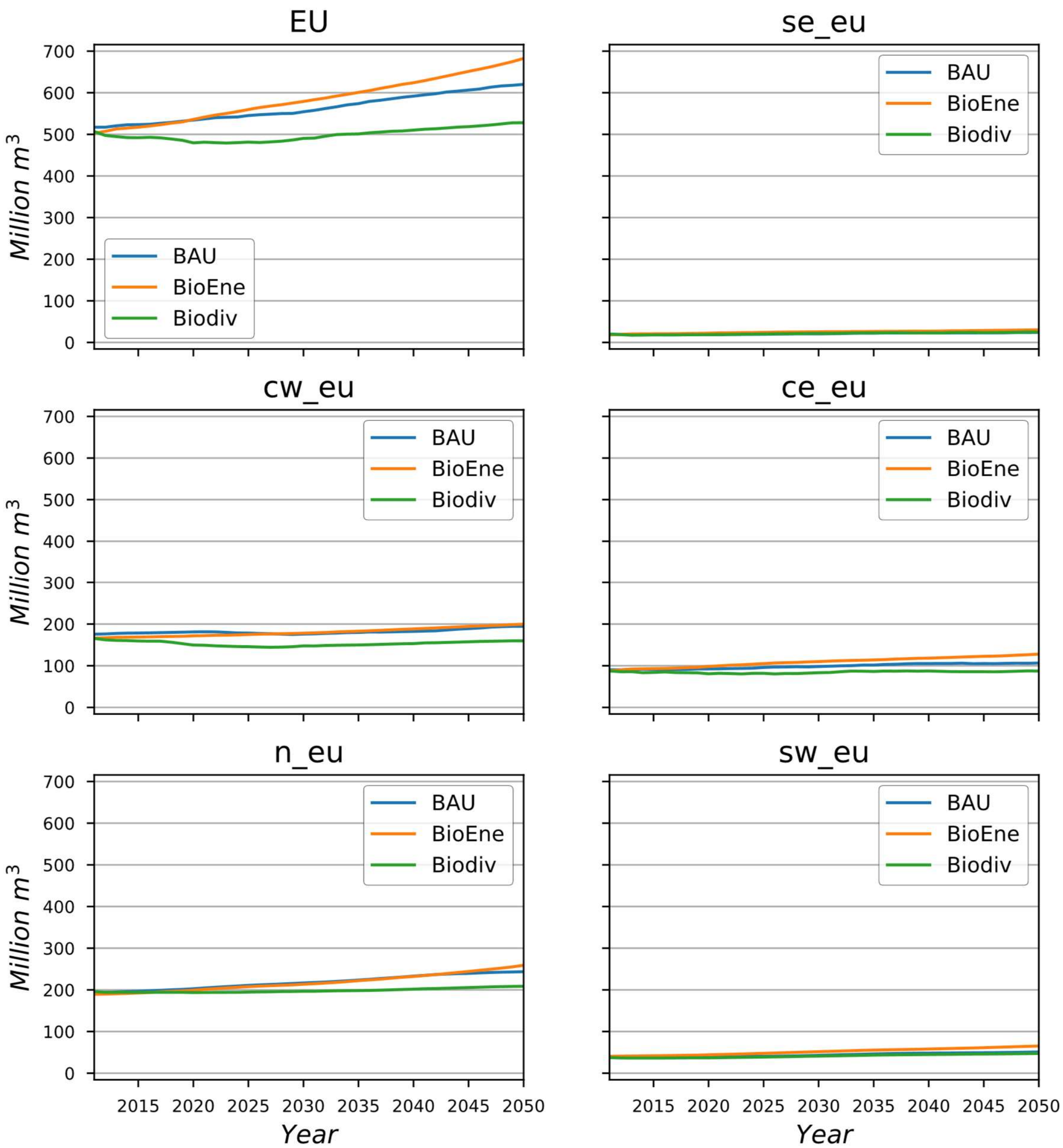


Figure 4.3 Timber harvested volume (million m^3 y^{-1}) from 2011 to 2050 for the three simulated scenarios (see Table 4.3 for codes) in Europe and by Ecoregion (see Figure 4.1 for codes). To improve clarity the results were smoothed using a 10-year moving average

4.2 Forest carbon fluxes

The continuation of the current management in the BAU scenarios results in a net increase in the net ecosystem production (NEP=NPP-Rh) of 11% in 2030 and of 14% by the 2050 (Figure 4.4c). The overall increase in NEP it is mostly driven by the increased NPP, which rise of 15% by 2050 (Figure 4.4a). Also in this case the future warmer climate is the main factor influencing this behaviour. Both alternative scenarios show a slight reduction in NEP in comparison to the reference one, which is up to 6% in Biodiv by 2050 (Figure 4.4c).

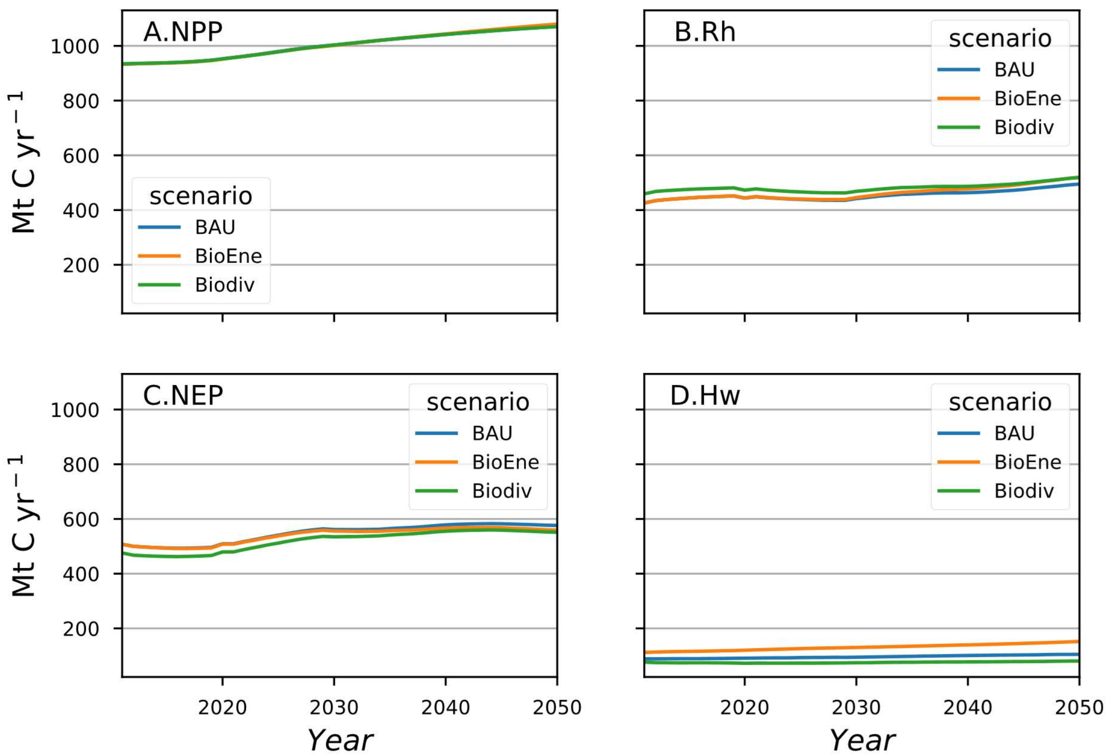


Figure 4.4 Annual effects of forest management on NPP (A), Rh (B), NEP (C) and emissions from harversted wood oxidation (D) for each scenario (see Table 4.3 for codes) from 2011 to 2050. To improve clarity the results were smoothed using a 10-year moving average.

These slight differences between scenarios are mostly due to the differences in Rh rather than NPP, with the former that shows essentially a similar behaviour among the different scenarios.

4.3 Climate change impact

Figure 4.5 presents the realized climate change impact over time for the three scenarios. European forest management contributes to the global warming reduction in an order of magnitude that ranges from –1324 Mt CO_2eq to -1532 Mt CO_2eq depending on the year and the scenario. Over the TH analysed the continuation of the current management (BAU) is the scenario with the lowest cumulative GPW impact while BioEne results to be the highest, with a GWP that is 11% higher. The climate benefits due to wood production stay rather constant for all the scenarios, except for Biodiv for which the GWP decrease of 7% from 2011 to 2050.

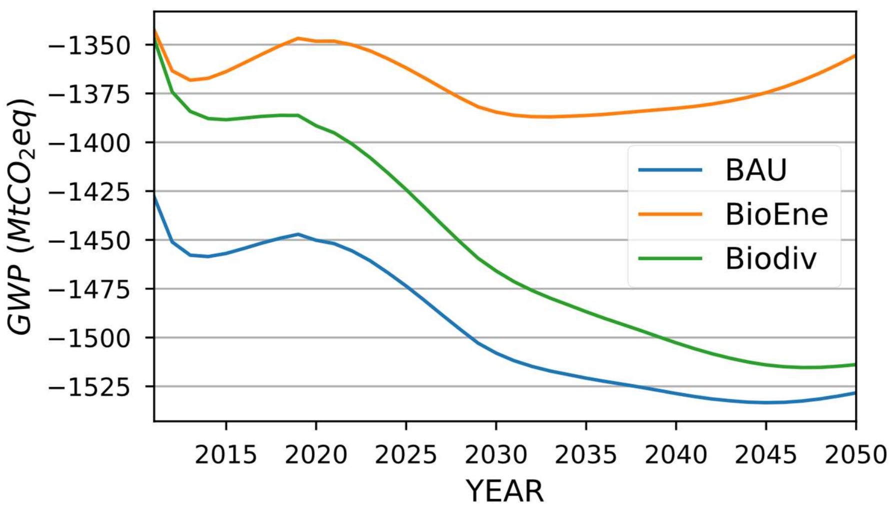


Figure 4.5 Cumulative Global warming impacts (GWP) for each scenario over the time horizon considered (2011 to 2050).

The realized changes in management do not sensibly affect also the relative contribution of each process to the total GWP impact (Figure 4.6). Biogenic forest exchanges are responsible for the lion's share of the GWP impact of timber production (Figure 4.6). Regardless of the scenario, the relative climate importance of the three forest fluxes groups (NPP+Rh+Hw) is sensibly higher than the three forestry processes (T+P+H), with a gap that tends to reduce over time. In 2011 the impact of the forestry processes in absolute term is between 0.6 % and 0.7 % of the impact hold by the forest fluxes, with the difference that is reduced to a value that ranges between 8 % and 10 % by 2050 depending on the scenario.

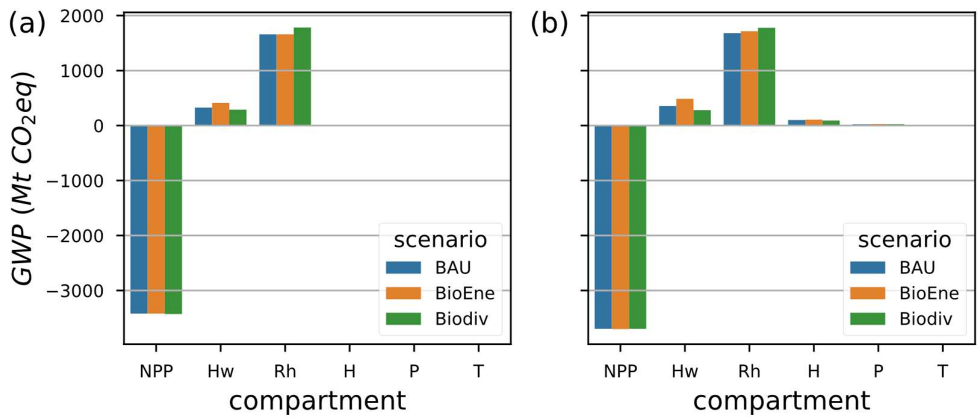


Figure 4.6 Cumulative global warming impacts (GWP) in 2011 (a) and 2050 (b) for each compartment. NPP= net primary production, Rh= heterotrophic respiration, Hw= wood harvesting oxidation, P= site preparation, T=tending, H= harvesting operations.

Looking at the relative contribution to the total impact of management, species and ecoregion at the end of the time horizon (Figure 4.7) it is visible how the majority of FoUs have a negative climate change impact. While the modification in management goals change the relative contribution played by the different FoUs, most of them still have a negative impact, except for some FoUs managed with clear-cut system (SS 3). In Biodiv, there is a sensible increase in the relative role played by unmanaged forests (SS 1) mostly at the expenses of shelterwood (SS 3).

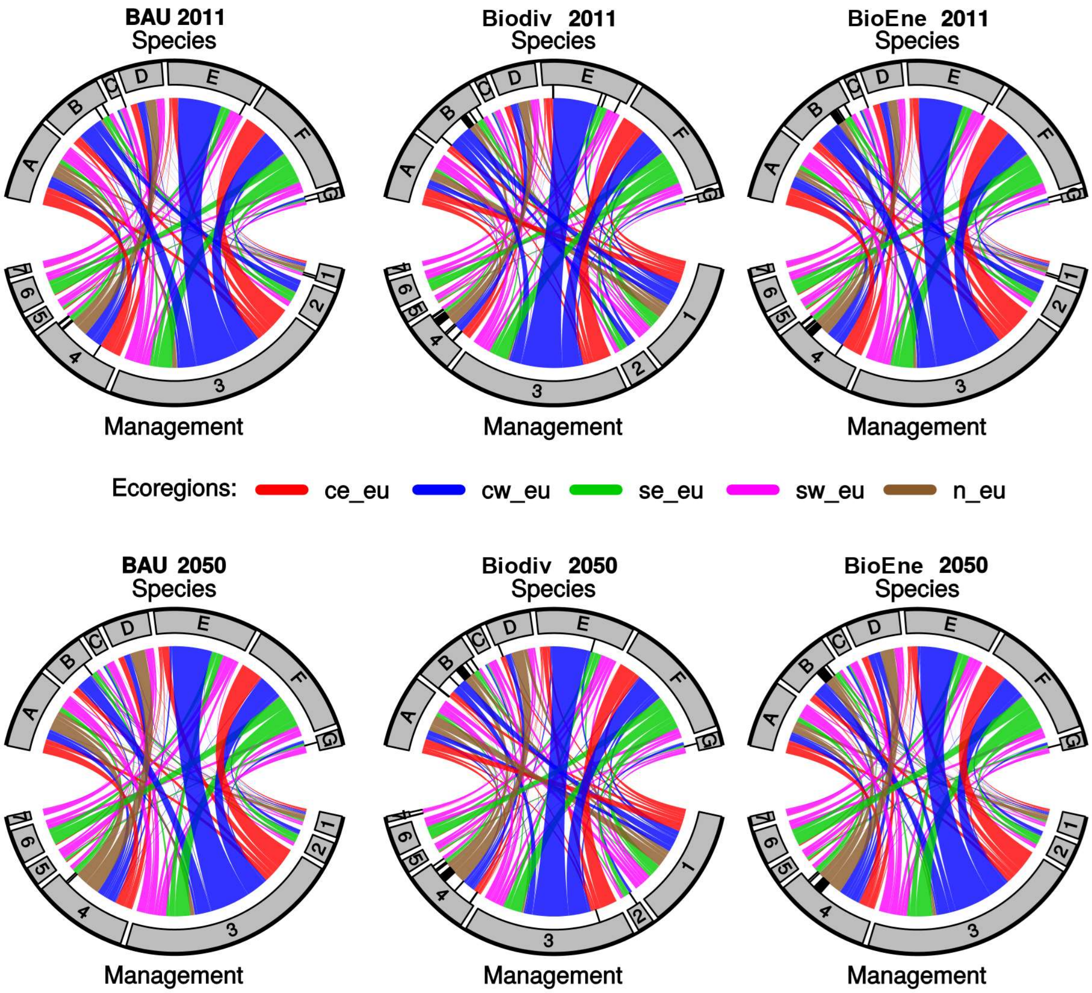


Figure 4.7 Chord diagrams showing the relative contribution to the cumulative global warming impacts (GWP) for each management (numbers), species groups (letters) and ecoregions (colours) at the beginning (year 2011) and the end of the time horizon (year 2050) for each scenario. The width of each link indicates the relative contribution and their basis is in white when the impact is negative (i.e. climate benefit) and in black when positive (i.e. climate change impact).

4.4 Impact per cubic meter

Also the climate change impact per m^3 of harvested timber does not change sensibly among the scenarios and over time (Figure 4.8). At European scale each m^3 of timber harvested yield a climate benefit with an impact that ranges between -1986 kg CO_2eq/m^3 (BioEne, year 2050) and -2989 kg CO_2eq/m^3 (Biodiv, year 2030).

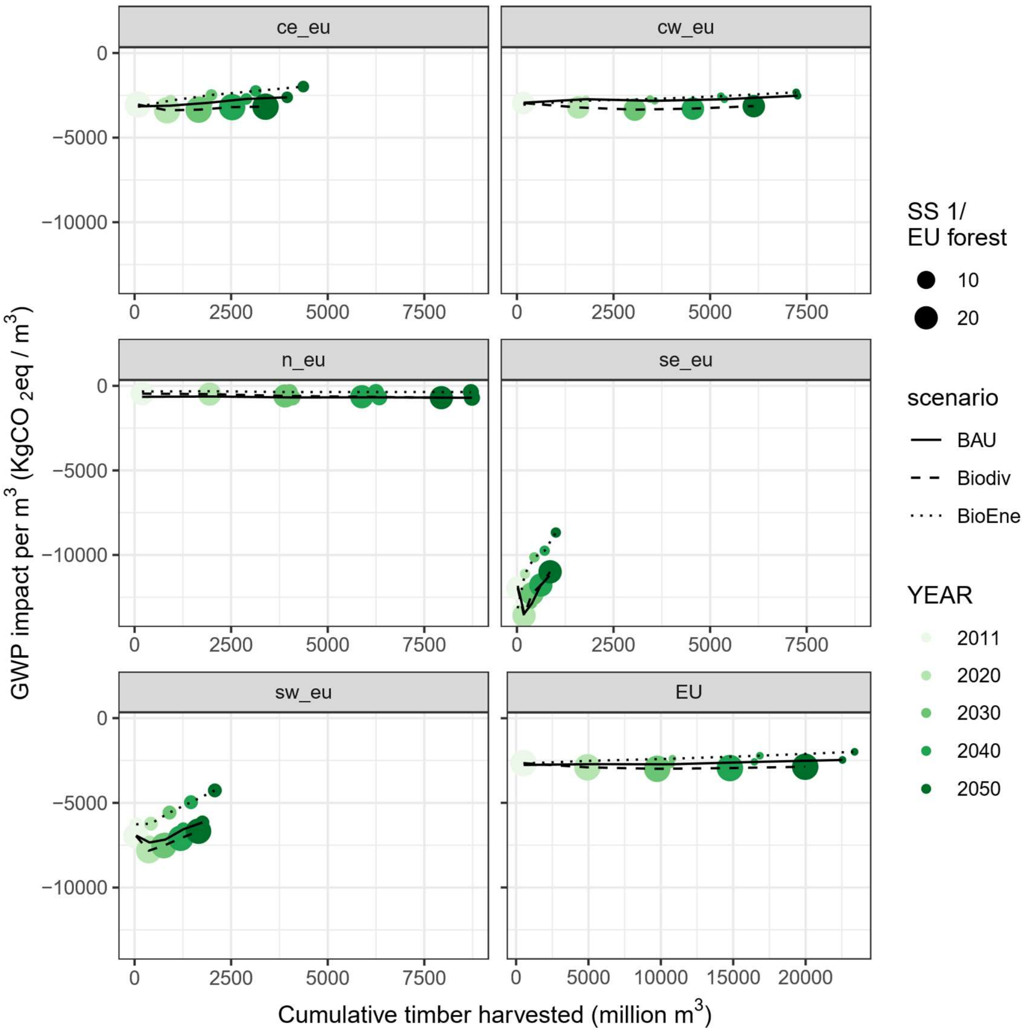


Figure 4.8 GWP impact per m³ timber harvested (y-axis) and cumulative timber harvested (x-axis) over time (colours) for the three scenarios from 2011 to 2050 for time steps of 10 years. The size of the dots indicates the relative area of unmanaged forests (SS 1) over the total area of forests. The relative impact is calculated also including the unmanaged forests. Mind the different scale of the x-axis for EU.

The pattern at regional level differs quite substantially. On the one hand, in the north of Europe timber production is almost climate neutral over time for all scenarios, on the other hand, the southern ecoregions show an important negative climate change impact that, despite the relative increase over time, it can be up to -13587 kg CO_2eq/m^3 as in se_eu for Biodiv in year 2030. This positive climate change impact is not due to the higher area of forests left unmanaged but rather to the different intensity of the management and, consequently, of the harvesting levels.

5. Discussion

The results of our work are essentially in line with and confirm what already found in other studies (Ciais et al. 2008, UNECE 2011, Böttcher et al. 2012, Pilli et al. 2017, Jonsson et al. 2018), i.e. that European forests act and will continue to act as C sink also in the future (see also NPP results comparison with other forest models in Appendix III). Based on our results the carbon budget of forests remains rather stable regardless of the goal of the management strategy applied, and only a slight reduction in the climate benefits is found when changing forest management goals. Being the biogenic carbon fluxes in the forest the greatest contributor to the climate benefits provided by wood production, also the GWP impact, which is negative in all scenarios, remain essentially steady over time for all scenarios. While the minimum effect on the overall climate change impact of timber production might look rather unexpected at a first sight, this inelastic behaviour highlights the importance of considering also economic aspects in such a type of analysis. While it is often assumed that forest management can be freely modified to achieve societal goals, local forest management decision are affected by many factors, among which the economic ones are often predominant (Sarvašová et al. 2014). It is thus rather the societal demand for timber that shapes the way forests are managed and not the other way around. This is reflected in our results where the pre-identified management strategies are, at the end of the day, slightly implemented due to the constraints imposed by the world after the forest road. Wood demand is thus an extremely important factor in determining the realized forest management and, consequently, also the relative climate role of forests.

Another important result to highlight is the remarkable difference found between the European regions in terms of climate efficiency of timber production. Our study found a clear pattern when moving from the North (lower climate benefits) to the south (higher climate benefits). While the results are certainly also influenced by the lower quality of the source data used to model the southern ecoregions, they also highlight again the well know underutilisation of southern European forests (Levers et al. 2014). While at a first sight these results might bring to the superficial conclusions that these regions perform better climate-wise, the results must obviously be interpreted from a broader perspective. The first consideration is that, in any case, our society has a defined demand for timber. A scarce domestic production of wood, or in any case a production level that cannot satisfy the demand, inevitably lead to the import of wood from abroad. Depending on the characterising of the forests from where the wood is sourced and of its production chain, this dynamic trigger a more or less big climate leakage effects. This mechanism will downstream affect the climate benefits of using wood and can even bring to a climate change impact when the wood is sourced from unsustainably managed forests. Italy, for example, can be taken as an exemplary country to explain this mechanism. In this country the relatively high demand for wood is not fulfilled domestically, also due to the relatively low harvesting intensity (Levers et al. 2014). This unmet demand triggers the import of big quantities of tropical wood (Koulelis 2015), which is also the material under the highest risk of unsustainable management practices. Ultimately, the potential climate benefits of using wood might disappear owing to the use of potentially unsustainable and climate-unfriendly timber. It must be always born in mind that the life cycle of wood does not end at the forest gate and that other stages of the life cycle have a climate role (e.g. substitution) which should be considered when looking at the relative role the whole sector can play as climate mitigator. Without considering the climate budget of the whole chain and on a broader scale (i.e. considering also trade) it has thus little sense to say what should be the adequate level of climate change impact the timber production should have, or which is the best place from where to source the wood.

The study also shows that, due to the important sink effect of European forests, the relative climate role played by the anthropogenic emissions from forestry activities is minor. If the goal is to improve the climate efficiency of timber production in the near term, it is thus more reasonable to focus on the maximization of the forest carbon sink rather than trying to reduce the emission from forestry operations. This is obviously subject to the future behaviour of forest and how big will keep on being the sink effect of European forest. The lower the sink the higher the importance of the emissions due to the forestry activities.

One aspect to pay particular attention to is the way the biogenic GWP impact has been accounted for in the study. While we avoided any climate neutrality assumption, the way the climate change impact of biogenic forest carbon must be estimated is still subject to discussion. We have, in fact, applied the approach referred in literature as gross (Stocker et al. 2014), landscape-level (Eliasson et al. 2013, Jonker et al. 2013) or constant spatial boundary (Cintas et al. 2017), which is opposed, respectively, to the net, increasing stand-level or expanding spatial boundary. In the first approach, each forest is seen as a single entity and the biogenic carbon fluxes over time are accounted for at this scale (FoU in our case). In the second, the resolution are the single stands forming that forests and the temporal carbon dynamic determined by stand harvesting and replanting are considered. The position of the scientific community on how to approach the landscape level studies is contrasting and different viewpoint exists, with some arguing for the correctness of the first (Cintas et al. 2017), and others for the second (Cherubini et al. 2013b). This has an important practical implication for this study since the two approaches can bring to rather different results. For example, in an idealized normal forest with annual growth equalling the yearly harvests (i.e. 0 net carbon fluxes), the landscape-level accounting would give no GWP impact, while the increasing stand-level accounting would result in a net impact of the landscape forest due to the regrowth of each stand. Using the increasing stand-level accounting would have certainly led to different results for our study, with also a higher relative role played by forestry practices emissions. The effect on the results of these two different accounting procedures is undoubtedly an aspect worth exploring in future studies.

The different source of data of the analysis, its complexity, together with the unavoidable use of a certain degree of assumptions and expert judgements, makes a correct quantification of the overall uncertainty challenging and basically impossible to be numerically calculated.

The use of empirical model to derive forest growth attributes is an unavoidable source of uncertainty. The forest growth functions of the FORMIT-M model were derived from NFI data collected between 2000 and 2010, with the growth functions that are estimated based on the conditions of forest at the time of inventorying. The expectable changes in forests age class distribution and structure and of the climate conditions will also affect the forest attributes and its growth, rendering the uncertainty of the model higher in the last years simulated. The biomass functions used were not always country-specific, in several case functions from other countries had to be used, reducing the precision of the estimated results.

Another limitation of the study is represented by the lack of consideration of forest disturbances. Previous studies showed the relatively little role they play in the overall carbon budget of European forests (Pilli et al. 2017), it can be thus assumed that their consideration would not have sensibly changed the results. Altering forest management practices also modify the albedo-related radiative forcing, which was not considered in this work. This effect proved to be an important factor to consider in studies like ours (Bright et al. 2014), and also this limitation affects the accuracy of the results.

One of the most important limitation of the study is linked to the difficulties in finding reliable and harmonized information on forest management data in several of the studied countries. This shortage is amplified when managements other than the traditional even-aged forest with clear-cut are studied (e.g. continuous cover forestry). Having reliable information on forest management characteristics and extension is of paramount importance both when estimating the climate role of forests and, more generally, when searching for management strategies to maximize the provision of all ecosystem services. One possible solution might be, for example, the collection of management information during the NFI campaigns in addition to the other traditional collected attributes, ideally using an internationally adopted forest management classification scheme.

Also the EFI-GTM results are obviously subject to great uncertainty driven by factors like future economic growth and the elasticity of wood demand.

Despite all the aforementioned limitations, it should be born in mind that most of these uncertainties are common for all the analysed scenarios. It is thus likely that the uncertainty of the estimated differences and trends, which have also the highest interest in a policy context, are much lower than the estimated absolute figures.

6. Conclusions

The impact of different forest management strategies on European forest in terms of timber harvesting, biogenic CO_2 fluxes and GWP impact was analysed following a life cycle-based approach by combining the results of an empirical forest model and a partial equilibrium model.

Also our study, in line with previous ones, essentially indicates that in Europe the current level of cuttings is well below its potential. The consequent increase in woody biomass stock together with the relatively stable level of carbon in soil make the NEP of EU forests positive. Due to the little importance of the anthropogenic emissions from forestry activities, the net climate change impact of timber production is negative. Future wood demand proved to be a decisive factor in shaping the realizability of the pre-identified management strategies. To achieve the planned goals strong economic incentives might be necessary to change the traditional management practices.

Chapter 5

Climate change mitigation potential of forests and forest products: a life cycle-based procedure to (better) consider time and substitution

Adapted from: Cardellini, Giuseppe, et al. " Climate change mitigation potential of forests and forest products: a life cycle-based procedure to consider time and substitution" In preparation

1. Abstract

Forests can contribute to climate change mitigation by sequestering carbon and through the use of harvested wood products (HWPs) which can substitute fossil fuels. The well-known climate trade-off of these two functions should be taken into account when designing future actions and policies for the sector. In this work, a dynamic and consequential life cycle-based assessment framework to estimate the climate mitigation potential of actions and policies in the forest-wood sector was proposed and illustrated. As numerical example, a forest with a constant production of roundwood used to produce glulam, pallet and bioenergy was modelled to estimate the effect of a hypothetical scenario in which the production of glulam is increased against the initial conditions. This case-study showed a rather high degree of variation in the estimated climate change impact of the systems over the 15 years considered. The estimated impact of the system ranged from - 274 to -111 tonnes of CO_2eq/ha/yr in function of the methodological approach followed. In addition, the estimated substitution effect was minor. The results suggested that for the maximization of the climate mitigation potential it is not enough to trying to maximize the substitution effect but the whole sector has to be looked at.

2. Introduction

Forests can affect the global carbon cycle by capturing the atmospheric carbon through photosynthesis and storing it in the forest ecosystem (Mackey et al. 2013). Without considering land use change Pan et al. (2011) estimated that the net global carbon forest sink from 1990 to 2007 was equal to 2.4 ± 0.4 Pg C year^{-1}. But forests can also contribute to climate change mitigation through the use of harvested wood products (HWPs) which can substitute fossil fuels. For example, Nabuurs et al. (2017) estimated that by 2050 the EU has the potential to attain a substitution potential of 184 Mt CO_2 yr^{-1} through HWPs use. Both approaches are suitable to mitigate climate change, but they display a trade-off relationship. For example, a reduction in harvesting level brings to an increased carbon sink effect in the forests, but the consequent decrease in HWPs use leads reduces the carbon stock in HWPs and the substitution effect. This trade-off is well known (Lecocq et al. 2011, Kallio et al. 2013, Smyth et al. 2014, Ter-Mikaelian et al. 2015, Matsumoto et al. 2016), and there is the need to take into account this effect when designing future actions and policies for the sector, as also indicated from the IPCC (2014a). This goal is generally accomplished by means of complex carbon integrated models which simulate the development of forest stands and, based on the assumed fate of the harvested roundwood, estimate also the impact of HWPs use and the potential substitution effect (Chen et al. 2014, 2018, Han et al. 2016, Matsumoto et al. 2016, Valade et al. 2017). This traditional approach presents three practical limitations: (i) it is data-intensive; (ii) it is based on C accounting models which do not consider the final climate change impact of the estimated changes in the C balances of the chain; (iii) it focuses mostly on forest development and the estimation of the role of the HWPs' use, and in particular substitution, is done in a simplified way by using pre-calculated displacement factors.

2.1 Wood products substitution

Substitution occurs when, by using HWPs, the use of other competing products fulfilling the same function and having a higher climate footprint are replaced. The competing product is a functionally equivalent product, namely a product produced using material other than wood with the same function (e.g. wooden window frame vs aluminium window frame or bioenergy vs fossil fuel). The potential energy and material substitution effect of HWPs have raised high expectations among scientists and policymakers (Van Renssen 2014). For example, in the European Union, an increased use of wood products is seen as an important avenue to mitigate climate change (European Commission 2013). However, while seen as a promising way to contribute to climate change mitigation, assessing the substitution potential of forest policies and actions is not easy. It requires a system-based approach which considers the whole life cycle of the HWPs and which should take into account: (i) the differences in climate change impact between the wood product and the competing ones (displacement effect henceforth) and (ii) how much the consumption of the competing products is affected by the changes in the supply of the HWPs (replacement intensity henceforth). In the following paragraphs, these two aspects are will be explained and discussed in more detail.

2.2 Displacement factors

In the literature, the displacement effect is normally estimated by means of displacement factors (DFs), a metric introduced in the work of Schlamadinger and Marland (1996) and improved by Sathre and O'Connor (2010). The authors define this displacement (also substitution) factors as "a measure of the amount of GHG emission that is avoided when wood is used instead of some other material" and calculate it as:

$$DF = \frac{GHG_{nw} - GHG_w}{W_{nw} - W_w} \quad \text{equation 5.1}$$

where, following the definition of the authors, GHG_w and GHG_{nw} are the GHG emissions due to the wood products and its non-wood competitor respectively, both reported in mass units of carbon (C), and where W_w and W_{nw} are the amounts of wood used in both products, reported in mass units of C contained in the wood. The DF of wood-based products is calculated through attributional life cycle assessment (LCA) by simply comparing the impact of the wood product with its competing one (Lippke et al. 2011). One of the issues with this approach is its intrinsic static nature. Time is typically not considered when calculating and using the DFs. The temporal dynamic of the emissions influences the LCA impact results and, consequently, the realizable displacement effect over time (Cardellini et al. 2018a).

Considering temporal dynamics is important also when the DFs are used and, more broadly speaking, on the general way substitution is calculated. This can be done by introducing the concept of change in the analysis and moving from an attributional to a consequential based use of the DFs. This transition requires that only for the marginal competing products affected by the action the substitution effect is calculated and that the substitution effect is not assumed to have a strictly linear behaviour. Thus, the substitution should not simply be estimated by multiplying the generic DF by the extra amount of wood produced from forests, but considering what are the products the marginal products replaced and also their replacement intensity, as explained in the next section.

2.3 Replacement intensity

While the comparison of the impacts between the competing products using displacement factors is needed, its estimation does not suffice to estimate the substitution potential of wood product use. By definition, wood product substitution takes place when the non-wood product competitor used to fulfil a function is replaced with a more climate-friendly wood one. A wood product cannot be assumed to substitute its competitor only because it is produced since, in reality, they are not perfect substitutes (Plevin et al. 2013). As well-known from economic theories, an expansion in the production of one good might only contribute to increase the overall consumption of the demanded function, bringing to the so-called Jevons Paradox (York and McGee 2016). The need to consider dynamically the effects on the market introduces features typical of consequential LCA (Earles and Halog 2011), which embeds the concept of change and requires the systems to be analysed dynamically. To estimate the substitution effect, other than knowing the DFs of the products, also the replacement intensity (RI) of the competing products should thus be assessed**.** The RI basically tells us how much a marginal difference in the wood production reduce the consumption of the competing product(s). For example, a RI of 0.6 means that every unitary change in the wood product production level reduces the consumption of the competing one with 0.6 units. Most studies (Perez-Garcia et al. 2007, Eriksson et al. 2007, Lippke et al. 2010, 2011, Knauf 2015, Knauf et al. 2015, Schweinle et al. 2018) does not follow a consequential approach and completely neglect the foregoing market effect. They incorrectly assume that all HWPs produced over time (i.e. their absolute amount and not the difference between the compared scenarios) fully displace their competing products by default at the moment of production, which is clearly in conflict with the concept of substitution. Some (Eriksson et al. 2011, Valade et al. 2017) follow the consequential logic, but they assume the full elasticity of supply, which does not consider the market-mediated effect driven by the modified supply of HWP. In other words, they assume that each extra unit of the HWPs supplied replaces one unit of the competitor (RI=1). As said the mechanism is more complicated and the indirect effects produced on the market normally makes the RI less than 1 (Rajagopal and Zilberman 2013). It has already been shown in studies applied to the agricultural sector that failing to consider this mechanism can lead to important overestimations of the attained emission reductions (Chalmers et al. 2015).

2.4 Dynamic life cycle analysis of the forest sector

Life cycle thinking is often used to estimate the expected climate change mitigation potential of alternative policies and actions (Farrell et al. 2006, EU 2009, IEA 2009, Manfredi et al. 2011), also in in the forest sector (Helin et al. 2013, Kilpeläinen et al. 2014, Klein et al. 2015, Alam et al. 2017). To know the complete life cycle performance of the forest sector and its climate change mitigation potential, the substitution effect must be considered together with the climate effect owing to the use of HWPs and forest management. Using wood can be seen as a way of prolonging the storage of the carbon fixed through the photosynthesis before it is released back in the atmosphere. When the stock of the HWPs rises, the HWPs pool acts as a carbon sink. For example, Pilli et al. (2015) estimated that from 2000 to 2012 the HWPs sink in EU-28 equals

44 Mt CO_2 yr^{-1}, which is approximately 10% of the forest sink. Such studies rely on carbon accounting models. They estimate the temporal changes in the carbon pools in the forest and the HWPs as a result of simple input-output balances but give no insight on how these changes are translated into their relative warming impacts, as normally done in life cycle-based studies. The net climate change impact of the forestry-wood sector is, in fact, not equivalent to the net carbon flux balance between the biosphere and the atmosphere. From the perspective of carbon accounting, for example, stocking carbon in the HWPs is only a way to buy time and postpone an emission that will eventually occur in the future (Kirschbaum 2003). But when the effect of this temporary storage is translated into its effect on climate, it is seen that the longer the time the carbon is stocked in wood products, the higher the benefits are. This is due to the positive response of the climate system to the postponed emissions, which occurs also when the stock does not increase (Guest et al. 2012, Guest and Strømman 2014). Also when looking at the climate role of forest management, if the changes in the forest carbon pools are translated into their climate change impacts, the story changes. In managed forest, the year-by-year change (i.e. net C stock change) in forest carbon stocks is different from its gross change. This is due to the transient depletion of carbon caused by the harvest of wood and the consequent regrowth of forest stands (Shevliakova et al. 2009, Wilkenskjeld et al. 2014, Stocker et al. 2014, Arneth et al. 2017). Already several studies showed the importance of considering the temporal dynamic of these biogenic C fluxes and in general, of all the GHGs fluxes, when assessing the impact of bio-based systems (Levasseur et al. 2010, Cherubini et al. 2011a, 2012, 2013a, Guest et al. 2012, Bright et al. 2012, Almeida et al. 2015, De Rosa et al. 2017a, Demertzi et al. 2017). Their findings can be summarized in three main points. (i) The climate change impact of each GHG emission is dependent on the time of its emission and the analytical time horizon considered. (ii) The temporary loss of carbon in sustainably managed forests (i.e. the biogenic carbon) does influence the climate change impact of raw wood produced by managed forests. Forest regrowth, in fact, exerts an impact on climate that is depending on the re-growth rate of the forest, the harvesting intensity and the length of the rotation. This makes the biogenic C neither neutral (as still often assumed in life-cycle studies) nor equivalent to the anthropogenic C. (iii) Postponing the emission of the biogenic carbon when the harvested wood is transformed into HWPs delays the moment in which the biogenic carbon sequestered in the forest is emitted, lowering its climate change impact.

As already suggested (Helin et al. 2013), only by moving to a dynamic and consequential climate change impact assessment and translating the temporal dynamic of the GHGs fluxes into their climate change impact over time will allow capturing the climate mitigation potential of the forestry and wood sector. The assessment of the climate mitigation potential associated with alternative actions or policies will consequently need to consider the dynamic climate change impact of (i) forest management, (ii) HWP production and use and (iii) substitution. These impacts eventually need to be compared against what would have happened without the studied changes (i.e. the reference). An action can be considered to have a net positive climate change mitigation effect at time t [$I_{net}(t)$] when the impact of the studied system at year t after the change [$I_{sc}(t)$] plus its substitution benefits[1] [$I_{sub}(t)$] is less than the impact the system would have had without the change [$I_{ref}(t)$]:

$$I_{net}(t) = I_{sc}(t) - I_{ref}(t) + I_{sub}(t) \qquad \text{equation 5.2}$$

With equation 5.2 it is possible to consider both the differences in the impact between the scenario and the reference, the additional benefits (if existing) due to the substitution effect generated from the scenario and consider explicitly their evolution over time.

In this introduction, we have briefly discussed the need of considering the existing trade-off between the forest-sector mitigation approach of sequestration and HWPs use when planning actions and policies for the sector. We especially focused on some of the shortcomings of the current practice used to assess the substitution effect of HWPs and, particularly, on how the temporal aspects are relevant when assessing the climate mitigation potential of the sector in a life cycle perspective. With this work, we want to address these issues by proposing a dynamic and consequential life cycle-based assessment framework that can be used to estimate the climate mitigation potential of actions and policies in the forest-wood sector. We will illustrate our approach with a simple fictitious case-study and show how the results can sensibly change depending on the way the study is performed. The case-study will serve solely as a didactical example to highlight the issues discussed so far and should not be used to draw conclusions on the validity and merits of a change in the use of HWPs pool but rather as a means to stimulate a more robust approach to the analysis of the problem.

3. Development of the approach

3.1 Goal and scope

Studies aiming at estimating the climate mitigation potential of forest-based actions presents one important methodological problem. They have a supply-side perspective with an input-based functional unit and the goal is to estimate the attainable climate benefits given a change in the absolute or relative supply of roundwood (i.e. the input). This requires the unit by which the impacts are calculated to be the mass (or volume) of roundwood produced. In contrast, the impacts of the value chains need to be estimated using a life-cycle approach which is a demand-

[1] In this study a negative value for substitution indicates a reduction in the climate change impact and a positive an increase.

side type of study with an output-based functional unit. In this value chain approach, the functional unit is the function provided by the final product (i.e. the output), and the analysis is performed upstream to all the sources of impact needed to fulfil this function. While the former approach looks at the supply of material, the latter focuses on the function provided by the product. This is a non-trivial aspect which complicates the analysis for a great deal. In most of the cases, equation 5.1 is used including only the wood contained in the end-use product following either a gate-to-gate or gate-to-grave approach. This neglects the impact hold upstream by the biomass used to produce the product (the cradle-to-gate part), which needs to be separately calculated with the consequent reduction of the practical usability of the LCA analysis done to derive the DFs. In addition, the DF calculated as such does not consider the resource efficiency of wood processing all over the chain. For example, two wood products might have the same DF when calculated with equation 5.1, thus apparently provide the same climate benefits, but they might use an intermediate wood products with different environmental load, with an overall substitution effect that is different between the two. Also the efficiency of roundwood utilization is not considered with equation 5.1 i.e. how much roundwood is needed to produce the final product. Being wood a limited resource is essential to also be able to estimate the substitution benefits achievable per amount of harvested wood. One way to overcome these issues is to identify all end-products made from the harvested batch of roundwood, the relative functions delivered by this wood resource (i.e. the reference flows) and the alternative technologies (i.e. the competing products) first. Then, calculate the cradle-to-grave life-cycle climate change impact at the level of each single reference flow p. Finally, reporting the results relatively to the roundwood equivalent (RWE) contained in the wood products (see equation 5.7 below). This approach allows to easily perform a life cycle analysis of the products and scale-up the results to the whole chain and use the results to calculate the impact of the system before and after the studied change. The results of these life cycle analysis can also be easily used to derive the DFs. The advantage of using this impact-based DFs calculation is that they can be readily used also when the competing products contain a certain amount of wood, which is not the case for the approach of equation 5.1 that is normalized for the amount of wood in the two competitors.

3.2 Life Cycle Analysis

A dynamic cradle-to-grave LCA was performed which included forest operations, above ground forest carbon dynamic, transportation, transformation, use and end of life (see Appendix IV for details) and the four most important GHGs were considered (i.e. CO_2, CH_4, N_2O, SF_6). Timber harvesting, products production and consumption were assumed to happen the same year. Glulam and its competing product were modelled with an average service life λ of 50 years (Wernet et al. 2016) and with a discard rate estimated following a gamma distribution (Marland et al. 2010). This distribution has been parameterized with $a = k/2$ and $b = 2$, where k corresponds to the mean lifetime of the product λ (i.e. the year of maximum oxidation) as proposed by Cherubini et al. (2012). This parameterization of the gamma distribution brings to a Chi-squared distribution with k degrees of freedom (equation 5.3).

$$\chi^2(t;k) = \frac{(1/2)^{k/2}}{\Gamma(k/2)} x^{k/2-1} e^{-t/2} \qquad \text{equation 5.3}$$

Where t=time and Γ(k/2) is the gamma function in equation 5.4.

$$\Gamma(k/2) = \int_0^{-\infty} t^{k/2-1} e^{-x} dx \qquad \text{equation 5.4}$$

Bioenergy and its competing products were assumed to have am an average lifetime of 0 years and their emissions simulated as a single pulse emission using a delta Dirac function.

The airborne decay of biogenic C was calculated following equation 5.5

$$IRF_{CO_{2bio}}(t) = \int_0^t e\,(t')y(t-t')dt - \int_0^t g\,(t')y(t-t')dt \qquad \text{equation 5.5}$$

with the right part of the integral representing the concentration of CO_2 in the atmosphere due to the biogenic emission response to distributed CO_2 emissions, and the left part the CO_2 removals due to biomass regrowth (negative). For all the anthropogenic GHGs emissions the standard IPCC approach to derive the impulse response function (IRF) was used and the GWP impact over time was calculated following the procedure reported in the chapter Dynamic climate IA methodology in Temporalis of Appendix II. We solved the LCAs dynamically using Temporalis (Cardellini et al. 2018a) setting as t_0 the year of timber harvesting to calculate the GWP impact at relative time t- t_0 for the wood product *ws* [$GWP_{ws}(t-t_0)$] and the non-wood products *nws* [$GWP_{nws}(t-t_0)$] for each unit of the function *p* delivered in all sub-systems *s*.

3.3 Scenario impact calculation

We first calculated the difference in the unitary GWP impact over time $\Delta GWP_{ps}(t-t_0)$ between the two competitors for each sub-system as:

$$\Delta GWP_{ps}(t-t_0) = \frac{GWP_{nws}(t-t_0)}{p} - \frac{GWP_{ws}(t-t_0)}{p} \qquad \text{equation 5.6}$$

To test the impact of the accounting methodology used, the GWP was calculated for three cases: taking into account the effect of all anthropogenic GHGs as well as the role of biogenic carbon (*bio+fos*), only considering the anthropogenic GHGs (*fos*) and following a carbon balance approach[2] (*Cfos*). We then derived the relative dynamic displacement factors for the sub-system *s* [$DF_{ws}(t-t_0)$] for each m^3 of roundwood equivalent contained in the wood product *w* (RWE_{ws}) as follows:

$$DF_{ws}(t-t_0) = \frac{\Delta GWP_{ps}(t-t_0)}{RWE_{ws}} \qquad \text{equation 5.7}$$

while the impact of the product *ws* relative to the roundwood equivalent contained RWE_{ws} was calculated as:

$$I_{ws}(t-t_0) = \frac{GWP_{ws}(t-t_0)}{RWE_{ws}} \qquad \text{equation 5.8}$$

The results of this first step are used to calculate the $I_{net}(t)$ of the system for which the (dynamic) functional unit is defined as an increased production of glulam over 15 years. From the year T_0 (set to 2015 in this example) to T_0+15 the roundwood production level of the forest remains thus constants and equals 265 m^3/yr. On the contrary, the relative share between the three final products changes as a consequence of a hypothetical policy which aims at boosting the use of wood in construction. As a consequence, the relative wood flow changes and the amount of roundwood used to produced glulam increases. This trend was assumed to follow the results of Hildebrandt et al. (2017), which estimated the realizable average growth rates of glulam production of +9% (rising from +5% in year T_0 to +14% in year T_0+15) for Europe. Being roundwood production constant over time, the relative increase in glulam production determine a reduction in the other two produced products (Table 5.1).

Table 5.1 Relative share of roundwood used to produce each product, amount of service produced (p_s) and the replacement intensity (RI) in the *construction* (CON) scenario at the beginning (T_0) and end (T_0+15) of the time horizon studied.

Product	**Roundwood fate**		**Amount of service produced (p_s)**		**RI[3]**	**Source RI**
Year	**T_0[1]**	**T_0+15**	**T_0[1]**	**T_0+15**		
Glulam	30%	76%	40 m^3	100 m^3	0.68	(Bird 2013)
Pallet	20%	0%	496 units	0 units	1[2]	Own assumption
Bioenergy	50%	24%	139023 MJ	67843 MJ	0.75	(Bird 2015)

[1] In the *BAU* scenario, the roundwood fate remains constant over the 15 years considered and as such also the amount of service produced p_s.

[2] We report the results in kg CO_2eq to simplify the comparison but, being this a CO_2 flux and not the GWP impact, should be kg CO_2. Nevertheless, since the biogenic carbon is not included in this case the results are numerically equivalent i.e. kg CO_2=kg CO_2eq.

[2] Based on the assumption that global pallet consumption (plastic+wood) is driven only by the number of goods traded and not by the price of pallet.
[3] The RI is assumed to be constant over time.

Based on the results of the previous steps the absolute impact of the system for the scenario *s* in the year t in both the *BAU* and *CON* scenarios is calculated as:

$$I_s(T) = \sum_{i=0}^{15} \sum_{ws \, \epsilon \, WS} P_{ws}(T) \times \; GWP_{ws}(T-i) \qquad \text{equation 5.9}$$

The actual substitution effect due to a change in the supply of the wood product *ws* in year T [$S_{ws}(T)$] thus depends on the absolute change in production levels between subsequent years ($P_{ws}(T) - P_{ws}(T-1)$) its $DF_{ws}(t\text{-}\ t_0)$ and the replacement intensity *RI* and is calculated as:

$$S_{ws}(T) = \sum_{i=0}^{15} [P_{ws}(T) - P_{ws}(T-1)] \cdot RI_{ws} \cdot DF_{ws}(T-i) \qquad \text{equation 5.10}$$

$S_{ws}(T)$ gives the substitution effect (both negative or positive) for a single product, the overall substitution effect of the system is obtained summing the effect for all the products *ws:*

$$I_{sub}(T) = \sum_{w=1}^{n} S_w(T) \qquad \text{equation 5.11}$$

Eventually, the overall GWP impact of the change of the system is calculated following equation 5.2.

4. Case-study application

4.1 Case-study description

Based on the previous considerations we have elaborated our fictitious case-study. It consists of a forest providing a yearly amount of roundwood used to produce three different products: glulam, pallet and bioenergy (Figure 5.1). The continuation of current management (BAU) is evaluated against a scenario in which the production of glulam is increased (CON). It is assumed that the roundwood come from an even-aged forest management managed with a sustained-yield harvest rate where the annual harvest is equal to the annual growth (i.e. the stand is C neutral) and where each stand of one hectare has a rotation length of 100 years and is clear-cut every year.

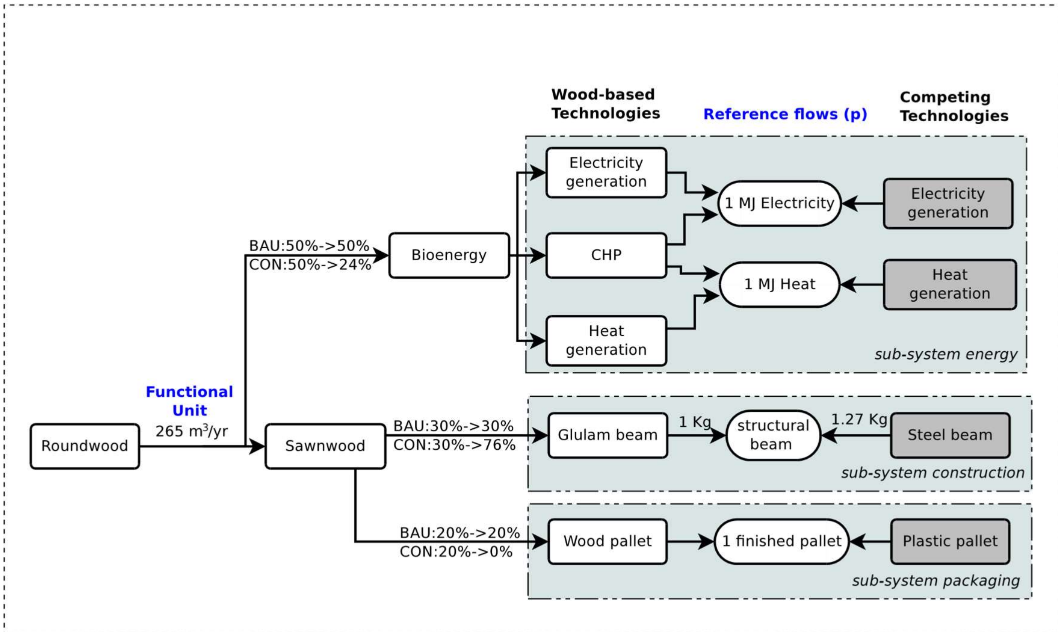


Figure 5.1 Schematic description of the system studied with the input-based dynamic functional unit (yearly supply of roundwood) and the reference flows on which both the impact of the systems and the displacement factors are calculated. BAU is the reference scenario, which assumes the continuation of the current management of the wood chain. CON represents the construction scenario which assumes an increased glulam production over 15 years. The respective percentage indicates the yearly amount of roundwood equivalent flowing to each product in the two scenarios at the beginning (left of the arrow) and the end (right of the arrow) of the 15 years studies. The grey boxes indicate the sub-systems with the relative reference flow (see Appendix IV).

The felling activity at the end of the rotation resets the living biomass to zero and the regrowth during the next rotation sequesters the same amount of CO_2. Due to its minor effect on LCA results (De Rosa et al. 2017a), the below-ground carbon is assumed to remain steady and thus is not considered in this study. On purpose, this example was built hypothesizing that the wood came from a C neutral forest (i.e. harvest=regrowth) to focus only on the role of the accounting procedure and cancelling out the potential climate change impacts of the changes in the C sink of the forest. The stand growth rate was estimated using the Schnute growth function (Schnute 1981, Yuancai et al. 1997):

$$G(t) = \left[y_1^b + (y_2^b - y_1^b) \frac{1 - e^{-a(t-t_1)}}{1 - e^{-a(t_2-t_1)}} \right]^{1/b} \qquad \text{equation 5.12}$$

where t is the age of interest, *G(t)* is growth at time t and with the other parameters that are explained in Table 5.2. The function is parametrized for a softwood stand of average productivity which, at the end of the 100 years of rotation (Kaipainen et al. 2004), stores 160 t of dry matter/ha in the above-ground biomass, as by IPCC default factors for managed temperate oceanic forest (IPCC 2006). The value of the parameters *a* and *b* were obtained via numerical approximation assuming that half of this amount is reached at half rotation period and in two rotations the IPCC default value for natural forests in the same ecological zone (180 t of dry matter/ha) is reached (Bright et al. 2012).

Table 5.2 Schnute model growth (Schnute 1981) parameters used in this study and explanation.

Parameter	Value	Explanation
t_1	100	age at the beginning of the interval
t_2	200	age at the end of the interval
$y1$	160	value of the function G at t_1
$y2$	180	value of the function G at t_2
a	0.0367	constant acceleration in growth rate
b	0.2137	incremental acceleration in growth rate

After harvesting, 80% of the total wood harvested is removed (Pilli et al. 2017). Of the removed wood 50% is used for bioenergy as reported in the Joint Wood Energy Enquiry for Germany (UNECE/FAO 2015), 20% for producing pallet (UNECE 2016) which is used here to represent packaging, and the remaining as construction material, which is here represented by glulam. All the woody bioenergy is used to produce energy with the energy conversion technology reported in Table 5.3.

Table 5.3 Relative type of energy produced and substituted from the bioenergy produced in our study.

Energy source	Main activity producer			Data source
	Heat	Electricity	CHP	
Roundwood[1]	31%	9%	60%	(UNECE/FAO 2015)
CHP[2]	91%	9%		(Wernet et al. 2016)
Energy replaced	**Energy source (total 100%)**			**Data source**
	Heat	**Electricity**		
Gas	25%	100%		(Bird 2015)
Coal	16%			(Bird 2015)
Hydro	2%			(Bird 2015)
Oil	56%			(Bird 2015)

[1] Wood directly entering the energy production without any further treatment or conversion. Total=100%
[2] Relative amounts of heat and electricity produced from CHP plant. Assumes that CHP from UNECE/FAO (first row) follows the fate from Wernet et al. (2016).

For both heat and electricity, the type of marginal fossil fuel replaced reported in Table 5.3 were considered as the competing products. For glulam, the competing product was assumed to be steel beam as previously done in other studies (Paulitsch and Barbu 2015) and for the wooden pallet its plastic equivalent is substituted. The studied system was divided into three sub-systems *s* based on the function *p* provided by the HWPs (subscript *ws*) and the non-wood competitor (subscript *nws*): the construction sub-system (glulam beam vs steel beam), the packaging sub-system (wooden vs plastic pallet) and the energy sub-system (bioenergy vs fossil energy). The reference flow on which they were compared were respectively the function provided by one kg of structural glulam beam (see section "Functional equivalence structural glulam vs steel" in Appendix IV), one finished pallet and one MJ of energy produced.

4.2 Results

Figure 5.2 presents the cumulative GWP impact over time for the unitary production (1 *p*) of the 6 products studied over 100 years. The carbon impact (*Cfos*) of the *nws* of both the energy and the packaging sub-systems are essentially steady over time. In the former, the impacts equal to 0.3 kg CO_2eq/p and in the latter slightly fluctuates around 25 kg CO_2eq/*p* over the years considered. In the *nws* of construction, the impacts peak the value of 887 kg CO_2eq/*p* in year four to diminish till 699 kg CO_2eq/*p* at the end of the TH considered.

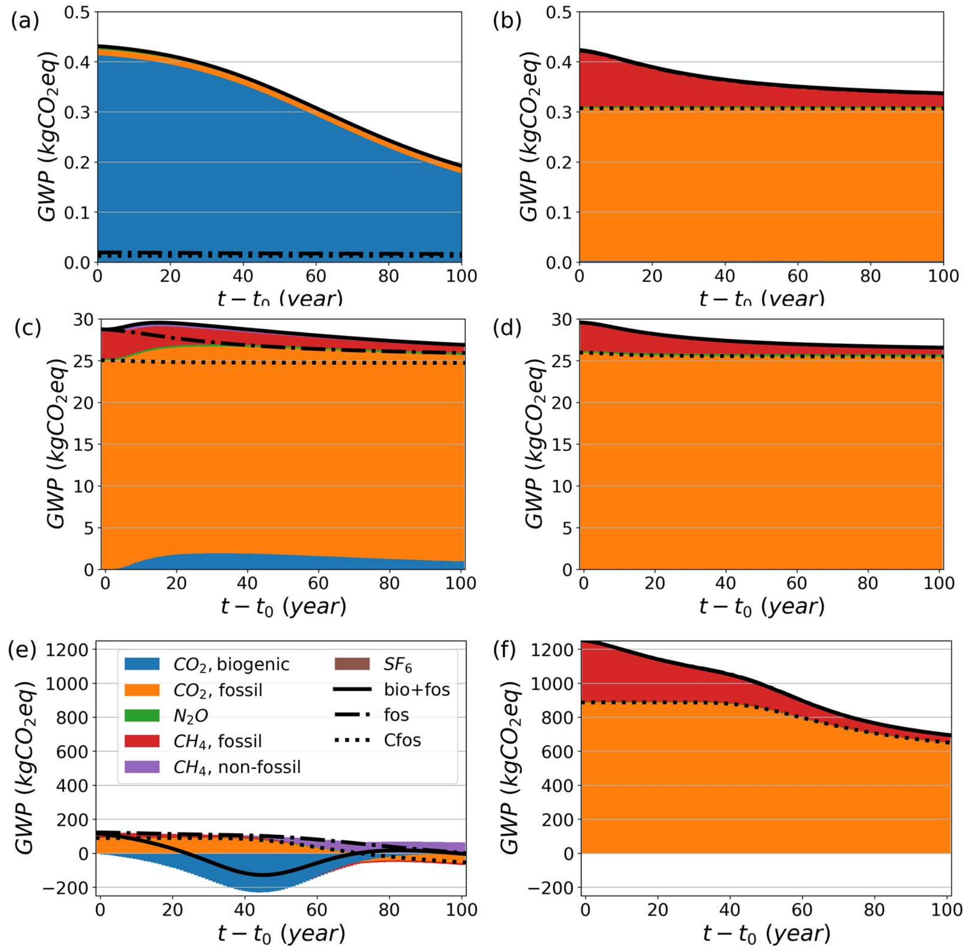


Figure 5.2 Cumulative GWP impact over time with the relative contribution of each gas for the unitary consumption of the reference flow p of the six products studied: bioenergy (a), fossil energy (b), wooden pallet (c), plastic pallet (d), glulam beam (e) and steel beam (f). The results are relative to the year of production (t_0) and differentiated by the way the impact is calculated (*bio+fos*: GWP of fossil and biogenic emissions; *fos*: GWP of only fossil emissions; *Cfos*: C balance).

When the other GHG are considered the differences in the results over time are more pronounced, particularly for the construction *nws* where the impact in the *fos* case decreases from 1255 to 712 kg CO_2eq/*p* over one century. In the first years, the estimated *fos* impact is up to 41% higher than the *Cfos* one. In all three cases, the role of biogenic carbon (*bio+fos*) is negligible. For the *ws* products, the relative role of the biogenic carbon impact is more relevant. In the bioenergy *ws* the *bio+fos* impact goes from the 0.4 kg CO_2eq/p in year 0 to 0.1 after one century while in the other two accounting approaches the impact is around 0.01 kg CO_2eq/*p* over time. Due to the relatively high recycling rate of pallet, in the packaging *ws* the impact of biogenic C is minor and only in the short term some slight differences can be found between the accounting procedures. In the construction *ws*, when the biogenic carbon impact is considered, the results drastically change over time. 44 years after production the maximum differences are observed and the *fos* case registers an impact of 102 kg CO_2eq/p while the *bio+fos* one of -127 kg CO_2eq/p. In the long term the realized displacement effect $DF_{ws}(t-\ t_0)$ in three sub-systems tend to asymptotically converge while, in the short term, the accounting procedure can substantially change the results (Figure 5.3). For the energy sub-system, the estimated $DF_{ws}(t-\ t_0)$ ranges from -380 (*fos*, $t-\ t_0=0$) to 21 (*bio+fos*, $t-\ t_0=0$) kg CO_2eq/RWE. In the *bio+fos* case only after 22 years the displacement factor becomes negative. For the packaging sub-system the $DF_{ws}(t-\ t_0)$ goes from -8 (*Cfos*, $t-\ t_0=0$) to 12 (*bio+fos*, $t-\ t_0=30$) kg CO_2eq/RWE. Also in this sub-system the *bio+fos* give opposite results over time, being the estimated impacts negative for the first 8 years, to then turn positive and peak the value of 12 kg CO_2eq/RWE after 30 years. For the construction sub-system the estimate $DF_{ws}(t-\ t_0)$ is always negative and ranges from - 527 (*bio+fos*, $t-\ t_0=39$) to -289 (*bio+fos*, $t-\ t_0=500$) kg CO_2eq/RWE.

For the bioenergy sub-system (Figure 5.4a), the accrued substitution effect S_{ws} in the year 2030 can range from -0.5 (*bio+fos* without RI) to 26 (*fos* without RI) t CO_2eq, from -0.2 (*bio+fos* both with and without RI) to 0.4 (*Cfos* both with and without RI) t CO_2eq for the packaging sub-system (Figure 5.4c), and from -61 (*bio+fos* both without RI) to -30 (*Cfos* with RI) t CO_2eq for the packaging sub-system (Figure 5.4e). The change in the climate change impact of each sub-system is rather similar between the accounting approaches, with the only exception of the bioenergy system where by 2030, the estimate GWP is reduced of 104 t CO_2eq in *bio+fos,* contrarily to the other two cases where it is minor (Figure 5.4b).

The total estimated substitution benefits for the system consequently bring to rather different results with an estimated reduction in the cumulative GWP impact by 2030 that ranges from - 62 (*bio+fos* without RI) to -15 (Cfos with RI) t CO_2eq (Figure 5.5a). The net climate change impact of the systems (I_{net}) is estimated in to range from - 274 (*bio+fos* without RI) to -111 (Cfos with RI) t CO_2eq by 2030. The substitution effect (I_{sub}) at the end of the 15 years considered has an estimated relative importance that can be from 14% (*Cfos* with RI) to 25% (*fos* without RI) of the total impact I_{net}.

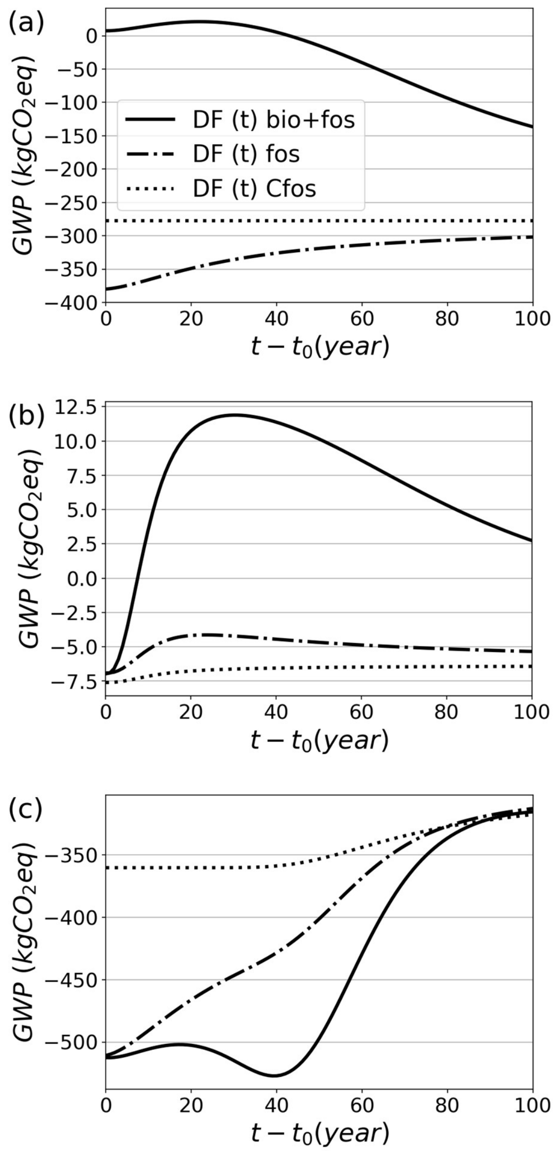


Figure 5.3 Displacement factor over time [$DF_{ws}(t\text{- }t_0)$] for the three sub-systems: bioenergy (a), packaging measured and (b) construction as cumulative GWP per m^3 of roundwood equivalent contained in the wood product and relative to the year of production (t_0). Results are differentiated by the way the impact is calculated (*bio+fos*: GWP of fossil and biogenic emissions; *fos*: GWP of only fossil emissions; *Cfos*: C balance)

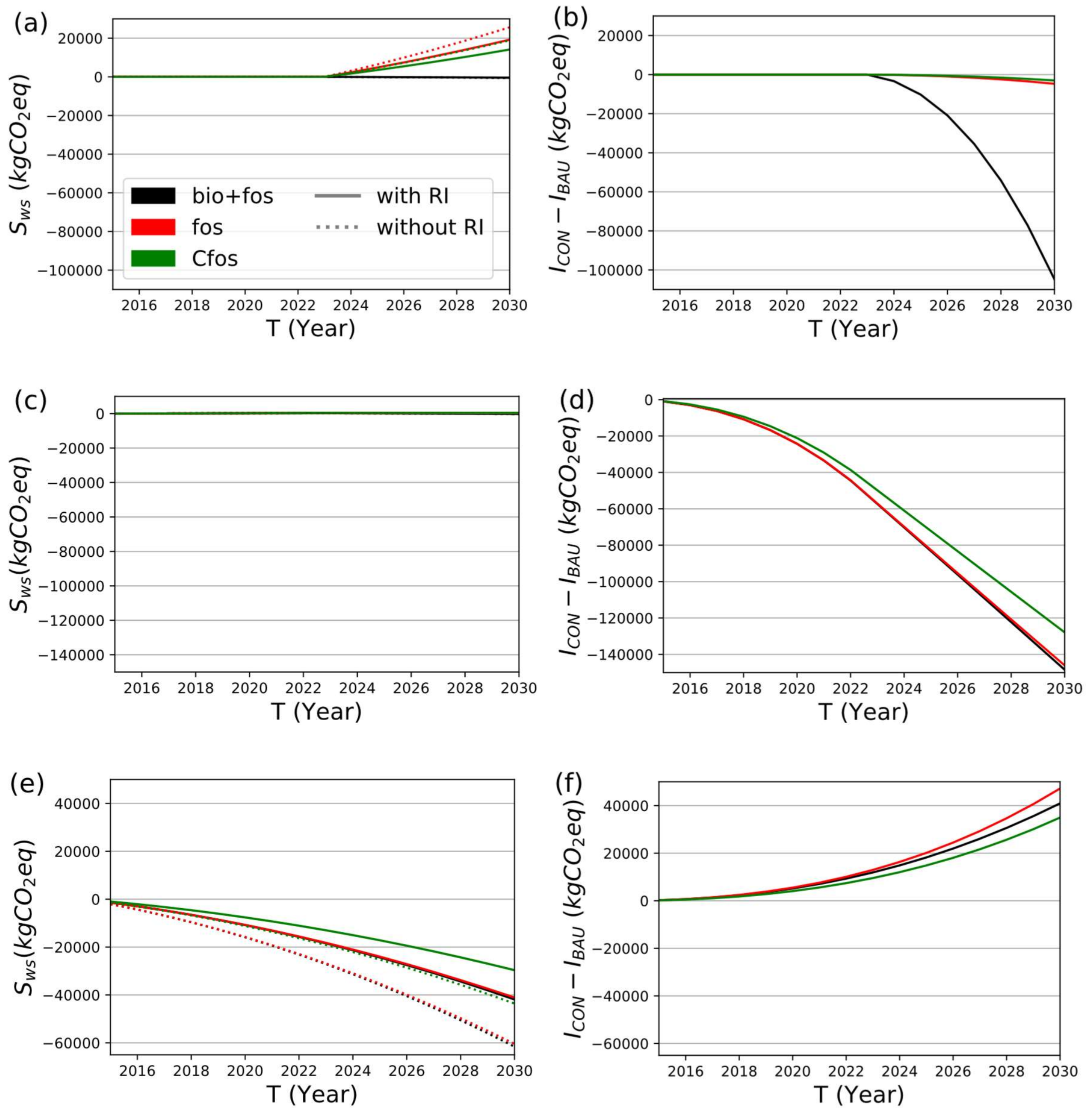


Figure 5.4 Realized substitution benefits [$S_{ws}(t)$] and net impact [$I_{CON}(t)$-$I_{BAU}(t)$] from 2015 to 2030 separately for the energy (respectively a and b), packaging (c and d) and construction (e and f) sub-system measured as cumulative GWP. Results are differentiated by the way the impact is calculated (*bio+fos*: GWP of fossil and biogenic emissions; *fos*: GWP of only fossil emissions; *Cfos*: C balance) and by when the replacement intensity (RI) is considered (case \`with RI\`) or not (case \`without RI\`).

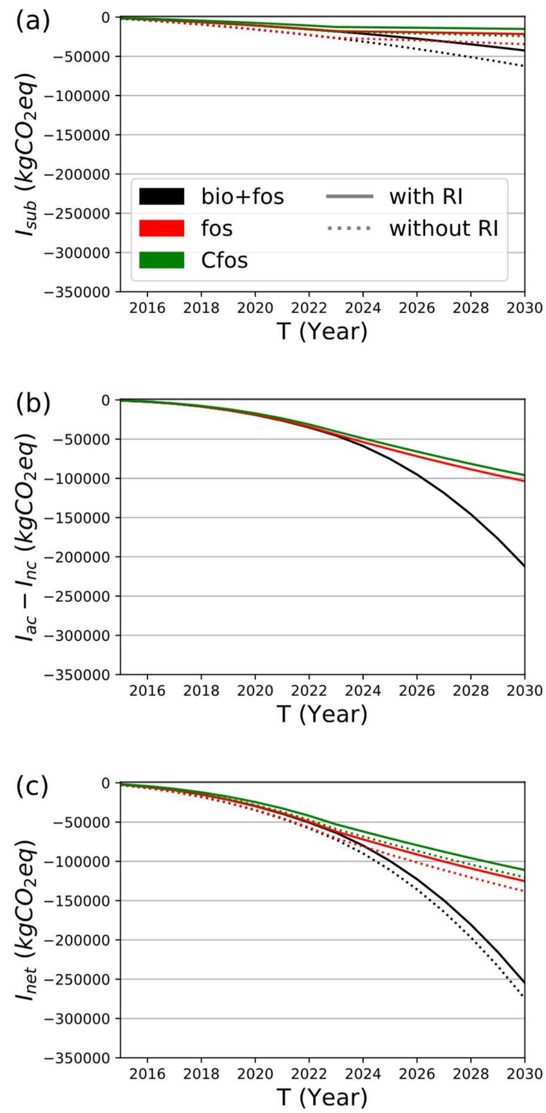


Figure 5.5 Total substitution effect I_{sub} (a) difference in impact between scenarios (b) and net climate change impact of the system I_{net} (c) measured as cumulative GWP from 2015 to 2030. Results are differentiated by the way the impact is calculated (*bio+fos*: GWP of fossil and biogenic emissions; *fos*: GWP of only fossil emissions; *Cfos*: C balance) and by when the replacement intensity (RI) is considered (case \`with RI\`) or not (case \`without RI\`).

5. Discussion

In this work, we proposed a dynamic and consequential life cycle-based approach to assess the mitigation effect of actions and policies in the forest-wood sector which considers the net impact of biogenic carbon dynamics, the substitution effect, the market effects and the temporal evolution of the impacts. With the help of a simple example, we have shown how the magnitude of the estimated impacts can substantially change in function of the followed methodology.

With regard to the estimated substitution benefits, they proved to be rather sensitive to the way they are calculated. Although HWPs are generally assumed to have a constant displacement effect over time (Sathre and O'Connor 2010), their temporal explicit accounting and the consideration of the climate change impact of biogenic C, shows that this effect can vary over the lifetime of the products. This behaviour is rather product-specific, and it might be difficult to identify generalized patterns. In our case, for example, the three analysed products present contrasting behaviour. While for glulam the displacement factor over time is always negative, for bioenergy 22 years must be waited before a positive displacement effect is found and, for pallet, positive effects are exerted only along the first 8 years. This fluctuating behaviour is to be attributed to the climate role played by the dynamic of biogenic carbon in managed forests.

With regard to the general methodology used to estimate substitution, this is normally done by simply taking the DF calculated in the other study and use them into C balance models which quantify the temporal changes in the forest carbon pools. Here we have shown that, for the same case, substantially different conclusions can be drawn depending on which GHG is considered and if the fluxes are just inventoried or are characterized into their climate change impact. When DFs from other studies are used, it would thus be important to ensure that the way they were estimated is consistent with the C balance models using them, to avoid any incorrect calculation and/or form of double counting.

Borrowing the jargon used in carbon offset projects, we can say that substitution must meet the additionality principle, meaning that substitution (both positive or negative) should be credited or debited only to the marginal changes in the production levels. Ideally, also the assumption of the full elasticity of supply should be abandoned. While theoretically correct, in our example the consideration of the replacement intensity does not prove to influence sensibly the results. How much neglecting the market-mediated effects can affect the overall results depends on the relative importance substitution plays. In our case-study, for example, the relative weight of the substitution benefits of glulam is rather high, and as such is taking into account its RI while for pallet, which has a minor substitution effect, the consideration of its RI would be of little relevance. Despite its potential importance, capturing the actual RI owing to a change in the supply of HWPs is rather complex since it depends on many factors, among others economical and societal, and its estimation is of tremendous difficulty. When not possible to use robust approaches (e.g. econometrics), we suggest following a conservative approach and simply assume a RI lower than 1. In following this marginal approach, identifying the marginal products affected by the increased wood products consumption and assessing their RI, features typical of consequential life cycle assessment are introduced. Nevertheless, the approach cannot be really considered a consequential life cycle assessment. This is because consequential aspects are introduced only for the changes occurring in the foreground system, namely the wood sector and the other directly affected sectors (i.e. the competing products), without considering all possible indirect effect triggered by the studied changes. Furthermore, the DFs are still estimated following traditional attributional LCA principles.

In our example, the relative importance of the substitution effect is essentially secondary. It can still contribute to up to a quarter of the total realized benefits, but the climate viability of the assessed scenario is mainly driven by the reduced climate change impact of the value chain. Although secondary, still substitution plays its role. Regardless of the way it is calculated, it can be seen that the augmented impact due to the increased production of glulam is essentially cancelled out from the substitution benefits it yields. For the other two products, the dynamic is quite different and is very dependent on the accounting produced followed. Particularly relevant is the case of bioenergy. Here, when all GHGs fluxes are characterized, the net reduction in the impact of the sub-system is driven by the reduced production levels and the substitution effect is negligible. Contrarily, when only anthropogenic GHGs are considered, the tables are turned, and the substitution plays the lion's share in determining the climate benefits of this sub-system.

So far the discussions focused mostly on substitution. When trying to identify the best actions to maximize the mitigation effectiveness of the sector, the relative impact old by the production and end-of-life phases can be rather important and, in relative terms, also bigger than the one played by substitution. These aspects are generally neglected when the analysis is performed using carbon integrated model and only the C stock changes in HWPs are considered. An enlargement of the system boundaries with the inclusion of these two stages would thus be advisable, being the relative impact of these two stages often substantial (Börjesson and Gustavsson 2000, Gustavsson and Sathre 2011, Sandin et al. 2013, Taylor et al. 2017, De Rosa et al. 2017a). Also the results of our case-study suggest that, rather than focusing on the maximization of the substitution effect, an effective climate policy for the sector should focus on the improvement of the climate efficiency of the whole chain. This would, in turn, also increase the (potentially) delivered substitution benefits.

Eventually, the climate change impact of the sector does not only depend on the GHG balance of the system. Although not considered here forest management, for example, can drive changes in albedo which can have a non-negligible effect on the life-cycle impact of the sector (Bright et al. 2015, Arvesen et al. 2018). We deem that, to fully estimate the mitigation effectiveness of forest and wood-based actions, a transition from a simple C balance assessment to a more exhaustive climate change impact assessment is necessary.

Last but not least, we would like to stress again that it is thus crucial to know the wood flow of the raw wood produced until its disposal to realistically estimate the impact of the sector. Despite this need, there is a tremendous paucity of material flow data for the wood sector. The main and almost unique source of information available to the scientific community is the international forest products statistics derived from the Joint Forest Sector Questionnaire. The problem with these data is that they report the wood flow up to the semi-finished products stage. In many cases it is basically unknown what is the fate of these intermediate products and their post-consumer destiny. The main priority of the future researches on the topic of forests and HWPs mitigation potential should be to improve the availability of data on the material flow of wood-based resources and their end-of-life management.

The importance of knowing the effective wood flow is certainly a priority, but this cannot solve all the problems. For example, even when the final use of wood is known, establishing a direct functional equivalency between the wood products and their competitors to assess substitution is not always straightforward. While for certain product this might be rather simple (e.g. pallet in our case) for others indirect ways might be needed. In the construction sector, for example, the same products can provide different functions, or the same function be provided by a combination of materials. Structural glulam, for example, can have different uses like wall and roof framing and, also within the same use, different architectural choices can be made which ask for different properties.

6. Conclusions

The expectations on the potential climate benefits of forest and HWPs use is high (Grassi et al. 2017, Nabuurs et al. 2017). Despite this, the consensus on which is the best way to maximize the climate mitigation potential of the sector between sequestering carbon in the forest and increasing the climate benefits of using HWPs has not been reached yet. With this work wanted to stimulate a more comprehensive and robust approach to the analysis of the problem and especially on the role of HWPs. It was shown that the estimated substitution effect is very sensitive to the accounting procedure used. Furthermore, its role might be much lower than expected and might be more an effective climate policy for the sector to focus on the improvement of the climate efficiency of the whole chain rather than aiming at the increase of the substitution benefits.

Chapter 6

Conclusions

1. General conclusions

The overall goal of the dissertation was to contribute to the improvement of the assessment of the climate change mitigation role of the forest sector, with focus on the European context. To achieve this object the issue was tackled from a more general LCA perspective in the first part of this manuscript addressing methodological and data availability issues.

Chapter 2 was driven by a sense of shortage of harmonized and regionalized data on the characteristics of forest management in Europe. This brought us to the development of a survey which landed in the development of the EFO-LCI database. The construction of the databases was mainly driven by the need for such a type of information for life cycle-based analysis. However, it can serve also as a source of information for the members of the forestry community interested in the knowledge of the status of European forest management. The first scientific question (**research question 1**) this chapter aimed to answer was "*to what extent the forest management practices differ in Europe?*"

It was found that the situation among European countries can be rather different in terms of type of forest regeneration, rotation length, amount and assortments of wood products harvested and machinery used in the interventions. Other than presenting an inter-country variability, it was shown that the management of the European forest is also rather diverse in function of the type of silvicultural system and species involved.

The **research question 2** wanted to answer the question: *are these differences (if existing) affecting the climate change impact of wood production?*

The differences found in the management were also reflected in the LCA analysis. Climate change was taken as exemplary impact category and it was shown how the variability of the input data translates into LCA results with a relatively high variability. The anthropogenic Global Warming impact of raw wood production was estimated to be between 0.4 and 73.1 kg CO_2eq/m^3, while the biogenic one between 1.6 and 451.9 kg CO_2eq/m^3.

Due to the relatively high differences found between management and ecoregions, it is recommended that regionalized life cycle inventories start to be regularly used when performing life cycle analysis in the forest-wood sector.

On the one hand, the use of regionalized life cycle inventories can help to improve the accuracy of life cycle studies trying to assess the relative role of wood production. On the other hand, it is difficult to clearly estimate how much this work can contribute to a better estimation of the environmental role of raw wood production. This is essentially dependent on the nature of the product studied, but also on the methodological choices made (e.g. system boundary) and the assessed environmental impact.

All the data were disclosed under *Creative Commons* license and made freely available for the community. The hope is this way that the purpose of improving the assessment of the environmental impact of forestry activities will be better served. It must be acknowledged that a limitation of this work is its non LCA software-friendly structure. All the data were solely collected, harmonized and codified into a big spreadsheet which is not (yet) readily importable and usable in LCA software. This was essentially a forced decision. Although some initiatives to tackle this issue are ongoing (Canals et al. 2016, Kuczenski et al. 2016) interoperability is still an important issue in LCA. This is translated into the difficulty of creating a database with a data format that is readily importable by the different LCA software available. The hope is that this problem will be quickly solved being a tremendous concern for the LCA community which experiences the frustrating situation of not being able to exchange projects between LCA software (Mutel 2015). To facilitate the use of regionalized LCI it is further recommended that the issue of data interoperability that affects the LCA community will be tackled jointly from LCA database providers and LCA software developers.

In Chapter 3 the methodological aspect of temporal considerations in LCA has been addressed due to the relative importance of this issue in forest-related LCA studies. Despite the acknowledged importance of this limitation and the previous works trying to address this issue, no complete and operationalized approaches were developed so far. This work addressed this gap answering the **research question 3**: *can the traditional static LCA methodology be modified to consider time in both the LCI and LCIA?*

We based our work on the traditional matrix-based LCA structure and, by using network analysis and convolution, we achieved the goal of making it possible to solve both the life cycle inventory dynamically and consider time also in the impact assessment. Particular attention was paid to not only translate this work into a scientific paper, with its consequent relative practical value for both the scientific and practitioners' community but also to make it really usable by translating it into a readily and freely available open source software.

Other than developing the model, it was used to perform a dynamic LCA of glulam to demonstrate its functioning and the advantages of using a dynamic approach and answer the **research question 4**: *does the previous methodological advance provide added value for the life cycle accounting of the climate change impact of wood products?*

With a real-case example, it was shown how the consideration of temporal information can provide new insights into the role played by wood products use as climate mitigators and, more generally, on their environmental impact. The analysis showed how the climate change impact of wood products vary over time, an aspect which is not possible to capture using a static approach. Furthermore, the dynamic results proved to be very sensitive to the to the temporal parameters (i.e. rotation length and product lifetime) used to model the dynamic of biogenic carbon fluxes. When dealing with wood products, especially long-living ones, it is thus of paramount importance and is thus recommended to pay particular attention during data collection to the quality and reliability of such data to maximize the representativeness of the results.

Despite the advance of this methodological work, the lack of temporalized data still remains a major constraint toward the widespread use of dynamic LCA. A small step toward this was done with the work performed in the previous chapter where also temporal information (i.e. when what was reported occurred) was includes in EFO-LCI. The systematic consideration of temporal aspects in LCA is still difficult not only for the relative shortage of data, but also for the lack of LCI database data formats that can explicitly consider temporalized information. For example, ecospold2 (Meinshausen et al. 2016) and ILCD (Wolf et al. 2011), the two most widely used data format from LCI databases, allow reporting only the time representativeness of the unit processes and not the temporal occurrences of biosphere and technosphere flow exchanges. It is thus recommended that such a LCI format are updated with the possibility to include also information on the temporal profile of the exchanges.

In the second part of the work, a more applied approach was used to address the general objective.

Chapter 4 had the goal of assessing the expectable effect of different management strategies on the carbon fluxes, timber harvesting and climate change impact of European forests. The work combined different data source and modelling approaches. An empirical model was used to simulated future forest development and the EFI-GTM partial equilibrium model to estimate future wood demand. The results were integrated with the EFO-LCI data and analysed in Temporalis to estimate the GWP impact over time.

The work addressed the **research question 5**: *What can the use of an integrated approach in which dynamic LCA and partial equilibrium model are integrated with forest model tell us about the effect of different forest management practices in Europe?*

The study essentially confirmed the current sink effect of European forests, which is translated in a net climate benefit of forest management and timber production. From our results, this effect is maintained also when different strategies management strategies are tried to be implemented. Only a slight reduction in the climate benefits of the system was found (11% at most by 2050) when the current management is changed, which is mostly driven by the increased emissions of soil carbon. In all cases, timber production proved to be a climate efficient production chain, with an estimated GWP impact ranging from -1986 kg CO_2eq/m^3 for the BioEne scenario in the year 2050 and -2989 kg CO_2eq/m^3 for the Biodiv scenario in the year

2030. These results present a relatively high inter-regional variability, and a clear pattern in the climate efficiency is found when moving from the lowest to the highest latitudes. Timber production in Nordic countries, in fact, yield much lower climate benefits in comparison with the southern European regions due to the higher intensity of the management (i.e. harvest intensity). The inclusion of economic considerations produced what is maybe the most interesting result of this work. We showed that, when the role of forest management is studied also from the demand side and not only from a supply perspective, the implicit constraints imposed by the free-market reduce the theoretical results obtainable in a regulated market. While other aspects like legal requirements and professional experience play an important role in shaping forest management decisions (Sarvašová et al. 2014), still economic considerations are maybe dominant in guiding silvicultural choices. Translated into policy recommendations, this discovery suggests that to obtain the planned goals, strong economic incentives might be needed to modify the traditional profit-oriented forestry practices, e.g. in the form of economic subsidies.

Chapter 5 addressed the issue of wood products substitution. It dealt with the approach used to account for it and how to integrate this assessment into a framework that can be used to estimate the climate mitigation potential of actions and policies in the forest-wood sector. A dynamic and consequential life cycle-based assessment framework was proposed to answer **research question 6:** *What is the effect of considering temporal aspects in the assessment of the substitution effects of wood products?*

While using wood products is normally assumed to have a constant substitution effect over time, in the study was shown that, when accounted dynamically, this effect can be delayed in time or not occurring at all. The realized substitution effect can also change substantially depending on the accounting approach used, namely if the GHG are only inventoried or if the effect of these fluxes in terms of warming impacts is also estimated. The studied scenario evaluated the effect of an increased use of wood in construction over 15 years. Ceteris paribus, the estimated climate benefits of the scenario could change from 274 to 111 t CO_2eq/ha/yr solely in function of the approach used. While important, the substitution effect proved to be secondary contributing at most for a quarter of the total realized climate benefits. The consideration of the market-mediated effects did not change drastically the estimated impacts. The results essentially suggested that looking solely at the maximization of the substitution potential is not enough, and climate policy for the sector should aim at boosting the climate mitigation role of the whole chain. The correct estimation of the climate change impact of the forestry-wood chain requires a detailed knowledge of the path followed by the wood resource along all the value chain. This is still very superficially known, at least in Europe, and its study is certainly the highest future priority for the sector it if the goal is to have more realistic estimates of its climate role. We recommend that future works start to abandon the use of generic displacement factors for the assessment of the substitution effect of wood products and that a more integrated and holistic approach in which the substitution effect is integrated with the assessment of all the other impacts occurring along the wood value chain is used.

2. Use of LCA in the forest-wood sector

Undoubtedly the production circumstances for the forestry-wood sector are quite peculiar due to e.g. the hygroscopic nature of the material, the relatively long production cycles and the complexity of the chain in terms of flows. While LCA has been widely used in this sector, still many issues remain. This brings up the question of whether LCA can be considered a suitable tool to be used in this context. A simple and unique answer is difficult to give, and different aspects should be considered. LCA is inherently an extremely complex and data-intensive methodology owing to its holistic nature that tries to assess all direct and indirect effects of actions and decision. While originally conceived as a tool to assesses product level impacts (ISO 2006a), LCA is more and more used to answer sector or even economy-wide questions (Farrell et al. 2006, EU 2009, IEA 2009, Manfredi et al. 2011). Someone said that “the scope of LCA is rather modest: it tries to model the entire modern economy and its interaction with the natural environment” (Mutel 2017a). Although slightly ironic, this statement can be essentially considered true, and let also understand how the complexity of LCA analysis increases exponentially with the scale of the analysis. While some scholars consider the use of LCA at big scale out of its scope (Frank Werner, personal communication), in the writer's opinion this is not necessarily true. As demonstrated in chapter 4 LCA, or at least life cycle thinking, can be used to perform broad-scale analysis, also combining this with other modelling tools. The ability to use LCA to answer “big” questions essentially dependents on two aspects: data availability and the representativeness of the modelling approach used. Quality of data is essential in modelling works, and the availability of good data is essential also for LCA. This holds true for all the LCA analysis, and even more when this tool is used in the context of forestry and wood systems. For example, the allocation issue should not be considered a problem per se, but it can become so when the effective flow of the wood resource is not known. Indeed, knowing the details of the production and use chain allows to explore and compare the impact of different allocation procedures on the results and thus better interpret the results. With the development of EFO-LCI was contributed to the end of increasing the quality of LCI data for the forestry sector in Europe. Having good data is not enough. To make LCA working in the forest-wood sector it is also necessary to use representative models that can capture the complexity of the sector. The development of Temporalis will help the scientific and LCA community to better understand the implication of the temporal dynamic of GHGs emissions and sequestration, which are quite peculiar of the forest-wood sector.

To sum up, in the writer's opinion LCA is a suitable tool to be used in the forestry-wood sector, provided that good data are available. With the work of chapter 2 a small piece to the puzzle of trying to improve the data quality and availability for LCA in the forestry sector was added. Obviously, good data needs to be efficiently and realistically processed in order to provide results that can represent the reality at their best. With the work of chapter 3 the issue of temporal consideration in LCA, which allows dealing better with the long production cycles of the chain, was addressed. The goals achieved in the previous two works were used in chapter 4 to show how life cycle thinking can also be used to perform big scale, integrated assessment analysis. Eventually, life-cycle analysis was used in chapter 5 to also address dynamically the issue of wood products substitution and show how this tool can help in assessing its climate role.

3. Policy implications and recommendations

It is important that all sectors commit themselves to the goal of limiting global warming below 2.0 °C. The use of wood products may certainly contribute to mitigate climate change through the benefits provided by the temporary stocking of carbon in wood products and the reduction of emissions due to the substitution for energy-intensive materials such as concrete and steel. While it is important that all sector help fight climate change, an increased consumption of wood products it is certainly not the definitive solution to climate change. It is thus important to acknowledge that replacement of wood with fossil-based product, while contributing to mitigate climate change, cannot be considered a priori as the as the best mitigating action for the sector. The substitution effect is not as big as expected and, when found, the benefits of an increased use of wood can present a very peculiar temporal dynamic. The fact that the reduction in the climate change impact can be found only decades after the production of the product can lead to unintended effect in the short term. It is thus important that, when designing mitigation policies, are explicitly considered also their temporal horizon and the undertaken actions are aligned accordingly.

Furthermore, a holistic approach that considers the whole value chain and its interaction with the other sectors, the economy and the society is necessary to ensure that the actions undertaken are really effective and able to obtain the predefined goal. At the same time, policymakers must be aware that the inherent inelastic structure of the sector can require stronger effort than expected to drive changes that can help fight climate change. It is thus important that this is taken into account, for example, stimulating changes not only with policy recommendations and guidelines but by means of fiscal and economic incentives that could stimulate the society to use more wood and the sector to improve its efficiency.

4. Toward the assessment of the climate change impact of the forest-wood sector

The study on the climate change mitigation role of forests has tremendously advanced since the second half of the 20th century. It moved from the very first simplified forest carbon budget studies of the seventies (Bolin 1977, Woodwell et al. 1978) to much more exhaustive studies, which take advantage of the great technological (e.g. remote sensing and lidar) and scientific (e.g. eddy covariance techniques) advances of last decades. These new techniques have also been integrated into complex, global scale, general circulation models, which allow studying also the interactions between the biosphere and the atmosphere. Also the amount of forest models developed in the last years has grown exponentially (EFIATLANTIC 2018). These advances in the understanding of the role of forests in the climate change regime has not been followed by a more accurate modelling of the wood production and use chain. This can be traced back to both the scarce methodological advances in the field and the tremendous shortage of data. Some wood sector models have been developed (Brunet-Navarro et al. 2016), but the data used to feed them are still scarce. With regard to the methodological aspect, a striking example is represented by the IPCC approach used to model the carbon stock changes in HWPs (IPCC 2006, 2014b). Tier 2 method is still the prominent and almost exclusive approach used, also from the scientific community (Eggers 2002, UNECE 2011, Donlan et al. 2012, Pilli et al. 2015). While the dynamic of the wood sector is much more complex, here wood products are simply grouped into the three big semi-finished products families (sawnwood, panels, and paper). Their decay is assumed to follow an unrealistic (Marland et al. 2010) first-order decay, with lifetime values that are assumed to be equal within each of the three groups. Regarding data availability, the situation is even more critical, and the paucity of basic data might also be considered the trigger for the limited methodological development. Why spend time building a better model if the quality of data used to feed them it is still extremely low? Continuing with the IPCC example the lifetimes are, for example, simply assumed based on best guesses. To the best of this author's knowledge, only one work exists in the literature that estimated the real lifetime of HWPs (specifically single-family houses) based on hard data (McFarlane et al. 2012). Interestingly, the study found an average half-life for the investigated HWP of 113 years, which is substantially larger than the IPCC default value of 30 years. More generally speaking, data on wood use and post-consumer fate are almost inexistent or not publicly available, at least in the European context. In the extremely rare case they are found their geographical scope is limited (Ratajczak et al. 2018), and the broader picture of what is going on at European scale is still essentially missing. This lack of basic data has been indicated also from other scholars as the main constraint toward the comprehensive assessment of the European wood resource flow and the assessment of its climate role (Mantau 2015, Jasinevicius et al. 2018). One reason for this shortage of data is certainly the lack of collaboration from the wood industries and, more generally speaking, from the wood sector, which is well known for being rather conservative and innovation adverse (Štěrbová et al. 2016). This non-cooperative behaviour was also experienced by other researchers of this same institution (Brunet Navarro 2017) and responsible of other projects having the goal of assessing the climate change impact of the sector. This non-cooperative behaviour is counterproductive for the sector first, as it makes it impossible to show the qualities and potentiality of wood as a renewable material to the outside world. Other more powerful and organized lobbies, like the ones of the steel and concrete industries, are very active in showing their alleged environmental strengths.

5. The way forward

The highest priority for the future stands in the collection and disclosure of more realistic wood sector data. This is of paramount importance if the role the wood sector as a whole can play in the climate change regime wants to be robustly assessed. The concern about this aspect comes also from the profound frustration experienced over the years of this PhD research work were, sadly enough, most of the time has been spent in searching for data that were either unavailable or simply way too difficult to obtain. This is undoubtedly an ambitious goal, and it might be considered a difficult task to achieve, but time is ripe for it. The wood sector, at least in Europe, consists mainly of small and medium enterprises, with the small ones representing one third of the total (EUROSTAT 2016). This fragmented nature makes more difficult to capture its details in official statistics due to the cut-off thresholds applied during data collection. This lack of representativeness is downstream reflected in the scarce quality of the public data produced (Kallio and Solberg 2018). In spite of these objective difficulties, we are living in the “big data” era, which offers tremendous possibilities to overcome these issues, making the data collection more efficient, powerful and of better quality. Luckily, some initiatives to fully utilise the potentiality offered by internet to build official statistics are already in place (UN 2018), and the hope is that the foreseen actions will soon be implemented also in the forestry-wood sector.

To speed up the achievement of the goal funding institution, at least public ones, should promote actions which ensure that, not only the results, but also the underlying data used to obtain them are disclosed. This transition has started in many parts of the world (Huijboom and Broek 2011), and actions like the adoption of the EU Open Data Strategy represents a big milestone toward the goal of extending the right to knowledge and helping science in developing. It is simply a shame that the society still must struggle to obtain readily available public data. Emblematic is the case of National Forest Inventories which, despite built and developed with public funds, are often not disclosed and firmly kept inaccessible to create a dominant position in the market. The achievement of the ambitious goal of better data availability would also require strong efforts from the scientific community. We should start paying more attention toward and put more effort in the dissemination of data, and not only of the results (i.e. scientific papers) they produce. It is certainly more interesting for a scientist to enjoy the modelling work, but scientific milestones can only be achieved when methodological progress are accompanied by progress in the quality of the used data. Science is not made of assumptions but of solid and realistic data. Last but not least, a strong transition toward a more cooperative behaviour with the private sector should be pursued.

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Appendices

Appendix I

Github repository with EFO-LCI

The github repo with EFO-LCI can be accessed at: https://github.com/cardosan/EFO-LCI

Questionnaire

The questionnaire can be downloaded from:

https://github.com/cardosan/EFO-LCI/tree/master/questionnaire

Database

The database can be downloaded from:

https://github.com/cardosan/EFO-LCI/tree/master/database

Source code of the the analysis

All the data analyses have been done using Jupyter notebooks which can be downloaded from: https://github.com/cardosan/EFO-LCI/tree/master/paper_notebooks

LCA impact analysis

Table A.1 EFO-LCI intervention and classification into generic and specific intervention

Intervention in EFO-LCI	Type of intervention	Operational productivity used
beating-up	generic	h/ha
browsing control	generic	h/ha
Building game protection fence	generic	h/ha
cleaning	generic	h/ha
clearing	generic	h/ha
disking	generic	h/ha
Enrichment planting	generic	h/ha
exotic species control	generic	h/ha
Forest care	generic	h/ha
Individual game protection	generic	h/ha
Mound Plowing	generic	h/ha
partial schreding	generic	h/ha
pest control	generic	h/ha
planting	generic	h/ha
Pre-commercial thinning	generic	h/ha
Protection	generic	h/ha
pruning	generic	h/ha
scarification	generic	h/ha
shredding	generic	h/ha

Intervention in EFO-LCI	Type of intervention	Operational productivity used
site preparation	generic	h/ha
stump destruction	generic	h/ha
stump lifting	generic	h/ha
weed control	generic	h/ha
clear cutting	specific	m^3/h
final felling	specific	m^3/h
harvesting	specific	m^3/h
plentering	specific	m^3/h
preparatory felling	specific	m^3/h
regeneration felling	specific	m^3/h
secondary felling	specific	m^3/h
seedling felling	specific	m^3/h
selection felling	specific	m^3/h
tending	specific	m^3/h
thinning	specific	m^3/h

Table A.2 Background processes and unit used

Machinery in EFO-LCI	Background process	Database	Unit	Note
brushing_saw	power sawing, with catalytic converter, RER	(modified according to doi.org/10.1016/j.biombioe.2010.11.022)	hours of use	
chainsaw	power sawing, with catalytic converter, RER	Ecoinvent 3.3	hours of use	
cultivator	Soil decompactation (with Decompacting soil, 5 shank subsoiler)	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
excavator	Indentation of pots, with tractopelle (i.e. digger/Back hoe)m FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
feller_buncher	harvesting, forestry harvester, RER	Ecoinvent 3.3	hours of use	
feller_buncher	harvesting, forestry harvester, RER	Ecoinvent 3.3	hours of use	
forwarder	forwarding, forwarder RER	Ecoinvent 3.3	hours of use	
forwarder and ripper	Soil decompactation (with Decompacting soil, 5 shank subsoiler), FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
harrow	Harrowing, with small tractor, FR)	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
harvester	harvesting, forestry harvester, RER,	Ecoinvent 3.3	hours of use	
harvester	harvesting, forestry harvester, RER,	Ecoinvent 3.3	hours of use	
machine for mechanized planting	Sowing or planting, trees, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI

Machinery in EFO-LCI	Background process	Database	Unit	Note
manual activity (all like coating, manual pruning, manual hoeing etc.)	transport, passenger car, large size, diesel, EURO 4	Ecoinvent 3.3	hours of use	
motor_hoe	small self motorized machine with 2l/hour of consumption, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
mulcher	Crushing, with shredder, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
planter	small self motorized machine with 2l/hour of consumption, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
ripper	Soil decompactation (with Decompacting soil, 5 shank subsoiler), FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
self walking seed planter	small self motorized machine with 2l/hour of consumption, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
shredder	agribalyse Crushing, with shredder or chipper', FR		hours of use	
skidder	skidding, skidder RER	Ecoinvent 3.3	hours of use	
sprayer	agribalyse Plant protection, spraying, with knapsack sprayer,FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
stump grinder	delimbing, with excavator-based processor, RER	Ecoinvent 3.3	hours of use	
tractor	Pushing wood, with small tractor, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
tractor and disk	Soil preparation, with disc harrow, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI

Machinery in EFO-LCI	Background process	Database	Unit	Note
tractor and disk	Sowing or planting, trees, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
tractor and plough	Tillage, ploughing, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
tractor and ripper	Soil decompactation (with Decompacting soil, 5 shank subsoiler), FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
tractor and shredder	Crushing, with shredder, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
tractor and skidder	Pushing wood, with small tractor, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
tractor with attachement for seedlings transportatio	Pushing wood, with small tractor, FR	Agribalyse 1.3	hours of use	similar consumption (l/hr) and use of what reported in EFO-LCI
wood_chipper	wood chipping, terrain chipper, diesel, RER	Ecoinvent 3.3	hours of use	
yarder	yarding, mobile cable yarder on trailer, RER	Ecoinvent 3.3	hours of use	
all manual activities	transport, passenger car, large size, diesel, EURO 4	Ecoinvent 3.3	km	

Appendix II

Github repository with analysis of chapter 3

At the link https://github.com/cardosan/dLCA the complete Jupyter notebook used for the analysis and the detailed system graph of the glulam case study can be found.

Temporalis repository

The open source repository containing Temporalis can be found at https://bitbucket.org/cardosan/brightway2-temporalis while the page http://temporalis.readthedocs.io/en/latest/ hosts its documentation.

Temporalis algorithm

The algorithm used by Temporalis is shown in Figure A1 and is explained more in detail in the following. The first thing the user has to do is to choose the appropriate worst-case CF as explained in section 3.2 and in equation 3.3 of the main paper. Before the traversal starts it is initialized performing a static LCA for the FU, and the results are added to the priority queue. To this priority queue are added all the technosphere exchanges (i.e. nodes) as they are traversed, and they are prioritized based on their importance (i.e. exchanges that contribute the most to the overall LCA impact of a FU receive the highest priority). After this initialization, the traversal starts removing the first element from the queue, which at this step is the FU. The first operation performed is the calculation of the environmental interventions $G_{j,p}(t)$ due to the traversed exchange node. These interventions are temporally resolved and added to the timeline $T_{t,i,p}$. To reduce running times and avoid unnecessary calculations, the environmental interventions are calculated following a different approach depending on if the databases is dynamic (i.e. containing temporal parameter) or static (i.e. missing temporal parameters)

$$G_{j,p}(t) = \begin{cases} B\widehat{A^{-1}a_{i,p}},\ t = TD_{ip}(t),\ \forall j\colon b_j\ , & if\ static\ database \\ TD_{ip} * TD_{jp},\ \forall j\colon b_j\ , & if\ dynamic\ database \end{cases} \qquad \text{equation A}\mathbf{1}$$

In the former case, the conventional matrix-based approach is used to calculate the environmental interventions $G_{j,p}$ and the results are added to $T_{t,i,p}$ with t=TD_{ip}(t) (first case in equation A**1**). In the latter, a convolution product between the product-process temporal distribution TD_{ip}, and the biosphere-process temporal distribution TD_{bp} is applied for each biosphere flow $b_{j,p}$ and again the resulting $G_{j,p}(t)$ are added to $T_{t,i,p}$ (second case in equation A**1**). After the environmental interventions, also the temporal distribution TD_{ip} of all the upstream technosphere exchanges $a_{i,p}$ need to be scaled and temporarily propagated into TD'_{ip}. TD'_{ip} depends on the amount consumed of the process p from the downstream consuming process d.

In this case, a different approach is used based on the temporal nature TD_{ip}(t), i.e. if the exchanges are either in the form of absolute or relative time.

$$TD'_{ip} = \begin{cases} TD_{ip} \cdot a'_{p,d},\ \forall i{:}\, a_i & if\ absolute \\ TD_{ip} * TD_{pd},\ \forall i{:}\, a_i & if\ relative \end{cases} \qquad \text{equation A2}$$

In the case of absolute time, being TD_{ip}(t) already expressed in absolute date, the rescaled exchange amount TD'_{ip} is simply obtained by multiplying TD_{ip} with the exchange amount demanded by the rescaled downstream node $a'_{p,d}$ (first case of equation A2). For relative temporal data, a convolution product between TD_{ip} and the temporal distribution of the downstream consuming process TD_{pd} is applied (second case of equation A2). After having rescaled the product-process TD of the node, the edges are added to the priority queue and the traversal continues with the next most impacting node. This process is iteratively repeated until the whole supply chain is traversed or one of the stop conditions is encountered, namely when the impact falls below the LCA cutoff indicated or the maximum amount of calculations is achieved.

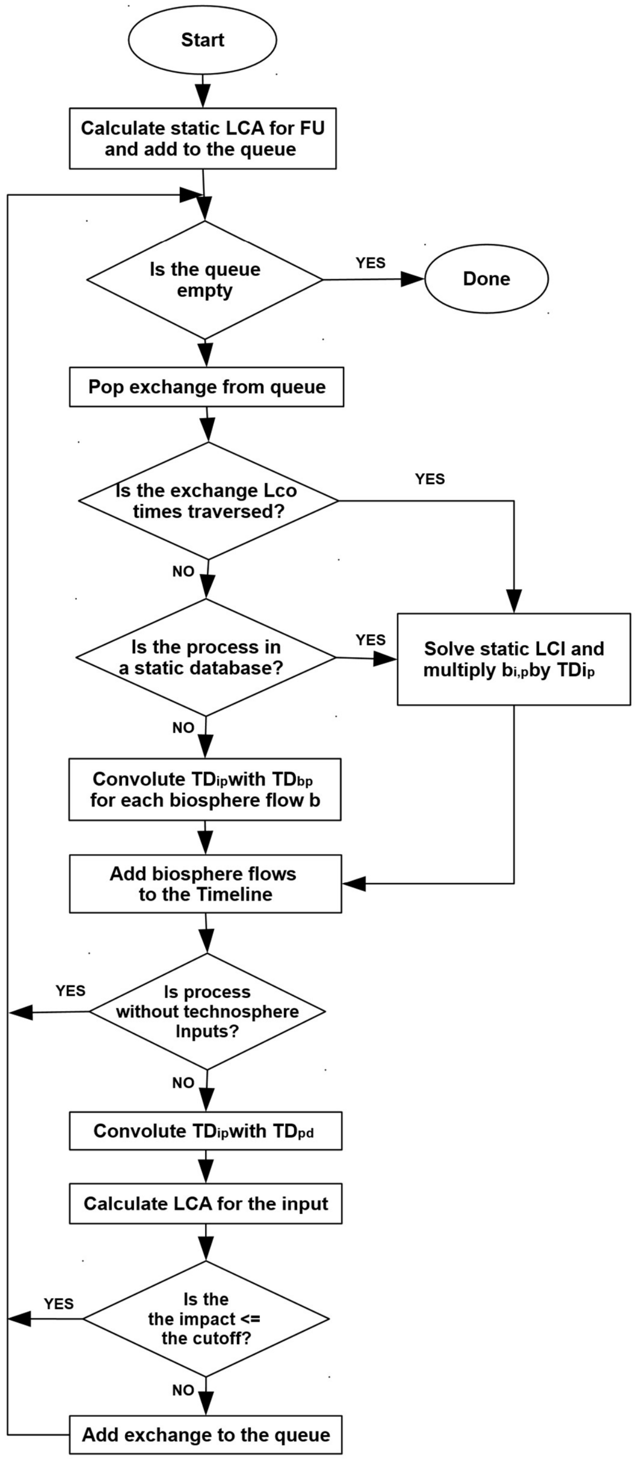

Figure A1 Flow chart of the algorithm used to solve dynamically the inventory matrix

Complexity of the graph traversal algorithms

Time and space complexity of the graph traversal algorithms

	Breadth-first	Depth-first	Best-first
Time complexity	$O(b^{m+1})$	$O(b^m)$	$O(b^m)$
Space complexity	$O(b^{m+1})$	$O(bm)$	$O(b^m)$

m=maximum depth of the network (i.e. longest path)

b= branching factor (i.e. number of children at each node)

Dynamic climate IA methodology in Temporalis

All the methods, metrics and parameters used in Temporalis to calculate the climate change impact of GHG emission are presented in this chapter and are taken from the 5th Assessment Report (AR5) of IPCC (2013). All these metrics are derived from the impulse response function (IRF) of CO_2 and the other Greenhouse Gases (GHG) g.

$$IRF_{CO_2}(t) = a_0 + \sum_{i=1}^{N} a_i\, exp\left(-\frac{t}{\tau_i}\right) \qquad \text{equation A3}$$

$$IRF_g(t) = exp\left(-\frac{t}{\tau_i}\right) \qquad \text{equation A4}$$

These IRF describe the decay of any GHG pulse emission from the atmosphere over time based on its decay τ. It can be noted that IRF for CO_2 is more complex and represented by a superposition of exponentials. This is due to the fact that about a fifth of the emitted CO_2 remains in the atmosphere for millennial due to the equilibrium response of the ocean-atmosphere system.

The IRF for biogenic CO_2, from which the GWPbio is derived, is calculated as:

$$IRF_{CO_{2bio}}(t) = \int_0^t e\,(t')y(t-t')dt - \int_0^t g\,(t')y(t-t')dt \qquad \text{equation A5}$$

where $g(t')$ is the atmosphere rate of removal of CO_2 due to biomass regrowth (i.e. the growth function of the stand), $y(t)$ is the IRF of CO_2 and $e(t')$ is the emission function of the oxidized biogenic carbon, which is normally based on probability distribution functions (Marland et al. 2010). In the equation the first integral represents the atmospheric CO_2 concentration response to distributed or delayed CO_2 emissions (e.g. from the harvested wood products' oxidation), while the second integral accounts for the CO_2 removals due to re-sequestration of biogenic carbon (e.g. due to forest regrowth), which is modeled as a negative emission. Any growth function can be modelled in Temporalis (so far normal and Schnute growth model are implemented) and for the biogenic carbon oxidation rate Dirac, Uniform, Delta, Chi2 and Exponential functions have been already implemented. In the software the effect of forest regrowth on atmospheric concentration CO_2 is by default accounted for also when data from static databases (e.g. Ecoinvent) are used. In these cases, since the forest growth functions are unknown, the forests are assumed to be carbon neutral, with a rotation length of 100 years and a growth rate that is assumed to follow a normal distribution. While certainly not perfect, this approach at least allows avoiding the carbon neutrality assumption and account for the effect of forest regrowth. Despite we consider this a reasonable approximation when databases which lack specific temporal information are used in the background system, we strongly encourage the use primary and more realistic data when modeling the dynamic of biogenic carbon in the foreground systems. We further include the indirect effects of methane emissions and its decay to CO_2 in Temporalis.

The instantaneous radiative forcing over time for each GHG (RF_g) is calculated multiplying its atmospheric mass by the relative radiative efficiency of the gas (RE_g) :

$$RF_g(t) = RE_g IRF_g(t) \;\; ; \;\; RF_{bioCO_2}(t) = RE_{CO_2} IRF_{CO_{2bio}}(t) \qquad \text{equation A6}$$

From the instantaneous radiative forcing the absolute global warming potential (AGWP) for each GHG emitted at time i is calculated as:

$$AGWP_g^i = \int_i^{TH} RF_g(t)dt \qquad \text{equation A7}$$

with TH representing the time horizon of the analysis.

Knowing the AGWP of the gas, its time i of emission and the TH of the analysis the global warming potential (GWP) is calculated as:

$$GWP_g^i = AGWP_g^i / AGWP_{CO_2}^{TH} \qquad \text{equation A8}$$

Once the GWP_g^i of each gas g emitted at time i is calculated, the overall GWP of the system is simply obtained summing the impact of all the gases emitted each time as:

$$GWP = \sum_{i \in i} \sum_{g \in g} GWP_g^i \qquad \text{equation A9}$$

To note that by integrating in such a way, the temporal inconsistency in the impact assessment phase due to the application of static climate metrics is avoided.

In Temporalis it is possible to account also for the global temperature potential (GTP). This metric estimates the global change in surface temperature at a certain TH given the RF profile, also in this case relative to CO_2. From the instantaneous RF_g of the gas and knowing its temperature response function $\delta T_g(i)$, the absolute global temperature potential (AGTP) is calculated using a convolution integral:

$$AGTP_g^i = \int_i^{TH} RF_g(t)\delta T_g(TH - t)dt \qquad \text{equation A10}$$

$AGTP_g^i$ gives the instantaneous temperature impact at a given time i due to the gas g. The GTP of the gas g emitted at time i is obtained as:

$$GTP_g^i = AGTP_g^i / AGTP_{CO_2}^{TH} \qquad \text{equation A11}$$

Analogously to GWP, the overall GTP of the systems is calculated as:

$$GTP = \sum_{i \in i} \sum_{g \in g} GTP_g^i \qquad \text{equation A12}$$

Appendix III

Simulated forest areas

Table A.3 Forest area (1000 ha) by Ecoregion (ER), Silvicultural System (SS) and Species Group (SG) in the three scenarios

ER	SS	SG	BAU	Biodiv	BioEne
ce_eu	1	A	76.7	2326.7	76.7
		B	245.5	1238.9	245.5
		D	93.6	858.4	93.6
		E	7.4	712.4	7.4
		F	320	1149.5	320
	2	B	118.6	91.6	118.6
	3	A	1696.1	912.9	962
		B	2126.1	1537.1	2860.2
		D		4.3	809.9
		E	1986.8	1746	1751.9
		F	4957.2	4182.5	4652.3
	4	A	4926.9	2348.3	2928.9
		B	1654.5	867.2	3652.5
		D	3041.8	1853.4	3082.9
		E	342.9	1473.2	304.9
		F	8.2	523.1	5
	5	E	345.6	225.9	345.6
	6	E	304.7	287.2	304.7
	7	A		5.9	
		B	273.4	87.2	273.4
		D		2.1	
		E		58.2	
		F		86.5	
cw_eu	1	A	150.5	1722.8	154.2
		B	287.3	2830.5	287.8
		C	10.6	10.6	10.6
		D	50.1	394.3	50.1
		E	112.5	698.5	112.5
		F	143.6	1058.6	143.6
		G	6.2	6.2	6.2
	2	B	2550	1868.7	2550
		F	945.5	652.6	945.5
	3	A	1388.8	691.9	915.9
		B	7399.1	4421.7	7872.1
		D	358.2	246.1	3473.4
		E	8267.2	7420.2	6268.8
		F	4401	5808.3	3415.4
		G	521.9	528.3	390.6

ER	SS	SG	BAU	Biodiv	BioEne
	4	A	5403.4	2852.4	3349.4
		B	984.1	545	3612.1
		C	1097.6	620.7	523.6
		D	1964.1	1322.5	2035.5
		E	471.7	1610.6	411.2
		F	147.5	1259.1	130.6
		G	108.6	249.3	100.7
	5	D		0.9	12.4
		E	54.9	29.4	46.6
		F		6.8	
	6	D			8.6
		E	50.7	22.7	45
		F		2.2	
n_eu	1	A	3247.3	7485.8	3247.3
		B	1325.2	3687.6	1325.2
		D	415.9	2556.7	415.9
		E		2.6	
		F		41	
	2	B	179.4	131.7	179.4
		F	86.7	68.1	86.7
	3	A	8119.5	6475.8	5470.5
		B	187.4	561.5	2836.4
		D		318	
		E	12.2	9.9	12.2
		F	109	88.8	109
	4	A	24112.8	19318.3	16278.6
		B	17550.6	17133.1	25393.8
		D	13891.7	11467.4	13882.7
		E	0.7	0.4	0.7
		F	9.4	7.2	9.4
	5	E	0.1	0.1	0.1
	6	E	0.1		0.1
se_eu	1	A	116.4	482.3	116.4
		B	29.1	287.1	29.1
		C		150.1	
		D		3.5	
		E	69.2	777.8	69.2
		F	263.6	1555.1	263.6
		G	12.7	255.5	12.7
	2	B	825.2	660.3	825.2
		F	1348.3	1111.9	1348.3
	3	B	575.5	465.1	575.5
		E	1952.4	1566.3	1952.4
		F	3637.2	3055.7	3637.2
	4	A	1947.6	1604.6	1947.6

ER	SS	SG	BAU	Biodiv	BioEne
		B	31.6	26	31.6
		C	750.3	600.2	750.3
		D	0	0	0
		E	49.1	70.7	49.1
		F	31.2	25.7	31.2
		G	0.6	0.6	0.6
	5	D	17.5	14	17.5
		E	163.1	102.6	163.1
		G	1271.9	1029.2	1271.9
	6	E	1681.8	1398	1681.8
		F	2656.7	2188.6	2656.7
sw_eu	1	A	416.6	1785.4	416.6
		B	13.4	40.8	13.4
		C	418	3103.7	418
		D	40.6	481	40.6
		E	307.6	939.7	307.6
		F	84	184	84
		G	232.3	1306.7	232.3
	2	B	457.4	429.9	457.4
		F	471.4	471.4	471.4
	3	A	3236.2	1903.2	3236.2
		B	311.3	311.8	311.3
		C	8948.8	5410.4	8948.8
		D		68.4	
		E	1441.4	1829.6	1441.4
		F	760.3	660.1	760.3
		G	3453.4	2991.3	3453.4
	4	A	1287.8	1279.1	1287.8
		B	6.8	6.8	6.8
		C	737.2	653.4	737.2
		D	2.3	2.3	2.3
		E	13.5	16.5	13.5
		F	6.8	6.8	6.8
		G	170.7	132.8	170.7
	5	A		13.2	
		B		0.5	
		D	480.5	481.7	480.5
		E	1951.3	1237.2	1951.3
		F		29.2	
		G	701.5	754.3	701.5
	6	E	1921.8	1921.8	1921.8
		F	933.8	933.8	933.8
	7	A	450.1	244.6	450.1
		B		0.5	
		D	1521.1	879.6	1521.1

ER	SS	SG	BAU	Biodiv	BioEne
		E		249.5	
		F		11	
		G		195.4	

Countries simulated with the FORMIT-M model

Table A.4 List of countries included in the study. In bold, the countries with NFI data.

Ecoregion	Countries
Northern Europe	Denmark, **Estonia**, **Finland**, Latvia, Lithuania, **Norway**, **Sweden**
Central-West	**Austria**, Belgium, **France**, **Germany**, Ireland, Liechtenst., Netherlands, Switzerland, UK, Andorra, Monaco
Central-East	**Czech R.**, Hungary, **Poland**, Romania, Slovakia
South-West	**Italy**, Portugal, **Spain**, San Marino, Vatican
South-East	Albania, Bosnia and Herzegovina, Bulgaria, Croatia, Cyprus, Greece, Malta, Montenegro, Serbia, Slovenia, The former Yugoslav Republic of Macedonia

For the European countries without NFI data the simulated results were generalized for the non-NFI countries by multiplying their forest unit areas by the neighbouring countries' simulated average result in the corresponding FoU.

The simulations from the countries listed below were used to generalize the results for the non-NFI countries:

- **Estonia:** Latvia, Lithuania
- **Germany:** Denmark, Netherlands, Belgium, UK, Ireland
- **Austria:** Slovakia, Slovenia, Romania, Hungary, Bulgaria, Switzerland, Liechtenstein
- **Spain:** Portugal
- **Italy:** Croatia, Bosnia, Montenegro, Macedonia, Serbia, Albania, Greece (Spain was used as the reference instead of Italy due to better coverage of forest types and species in its NFI data)

Identification of proxy FoUs

The following rules were considered in choosing the FoUs' proxy, by application order:

- For the same country and management system, the proxy is done assuming the same species family (example: Austria, even-aged forest-uniform clear-cut system, slow growing light demanding deciduous simulated with Austria, even-aged forest-uniform clear-cut system, fast growing deciduous.

- For the same management system and species groups consider a different country inside the ecoregion (example: Albania, continuous cover forest management, shade tolerant conifers simulated with Slovenia, continuous cover forest management, shade tolerant conifers).
- For the same country and same species groups consider the same family of management type (example: Bulgaria, even-aged forest uniform clear-cut system, shade tolerant conifers simulated with Bulgaria, even-aged forest with shelterwood, shade tolerant conifers).
- For the same ecoregion assume the same management system regardless of the species group (example: Albania, even-aged forest uniform clear cut system, Mediterranean evergreen tress simulated with Serbia, even-aged forest uniform clear cut system, light demanding conifers).
- For counties in the ecoregion South-East of Europe, a given management system and species group, consider a country from South-West Europe (example: Albania, coppice, Mediterranean evergreen trees simulated with Italy, coppice, Mediterranean evergreen trees).
- For a given management system and species group use information from any available country (example: Slovakia, coppice with standards, slow growing light demanding deciduous simulated with Austria coppice with standards, slow growing light demanding deciduous).

Business as Usual scenario (BAU) management rules description

The BAU forest management regimes (i.e. thinning rules) were defined for different silvicultural systems and TSG based on national expert opinions. These management regimes were defined separately for the North (Table A.5), the Central East (Table A.6), the Central West (Table A.7) and the South Europe (Table A.8). The silvicultural systems for NFI plots to be simulated were selected as following:

1. **Unmanaged forests:**
    - **Selection criteria:** Those plots, which were located on protected area, were selected first. If the share of unmanaged stands in the country was higher (in statistics), than the share of plots selected this way -> random set of other plots were added to silvicultural system 1 (no management) so that the announced share was fulfilled.
    - **Management:** Unmanaged forest are not thinned or cut during the simulation.
2. **Continuous cover forestry**:
    - **Selection criteria:** a random set of plots was selected out of remaining plots so, that the statistics (announced share of continuous cover forestry in the country*) were fulfilled.

- **Management:** Continuous cover forestry is simulated by applying constant annual averages of stand mean characteristics. These come from BAU simulations for the corresponding plots and different averages were used for 2000-2050.

3. **Even-aged uniform forests with shelterwood:**

    - **Selection criteria:** a random set of plots was selected out of remaining plots (only in the South and Central Europe, only for certain tree species) so, that the statistics (announced share of this silvicultural system in the country*) were fulfilled.

    - **Management:** Thinning rules are explained in Table A.7. At the moment of final cutting there is assumed to be already a new set of 10-15 year old trees, which form the new forest stand.

4. **Even-aged uniform forests with clearcut:**

    - **Selection criteria:** a random set of plots was selected out of remaining plots (in the North all species, in the South and Central Europe only certain tree species) so, that the statistics (announced share of this silvicultural system in the country*) were fulfilled.

    - **Management:** Thinning rules are explained in Table A.7.

5. **Coppice:**

    - **Selection criteria:** a random set of plots was selected out of remaining plots (only in the South and Central Europe, only for certain tree species) so, that the statistics (announced share of this silvicultural system in the country*) were fulfilled

    - **Management:** Thinning rules are explained in Table A.7. Living root biomass stays in the forest despite of final cutting.

6. **Coppice with standards:**

    - **Selection criteria:** a random set of plots was selected out of remaining plots (only in the South and Central Europe, only for certain tree species) so, that the statistics (announced share of this silvicultural system forestry in the country) were fulfilled

    - **Management:** Thinning rules are explained in Table A.7. Living root biomass stays in the forest despite of final cutting.

7. Short rotation:

- **Selection criteria:** a random set of plots was selected out of remaining plots (only for certain tree species) so, that the statistics (announced share of this silvicultural system in the country*) were fulfilled
- **Management:** Thinning rules are explained in Table A.7.

*In case the share of silvicultural system 1 became higher than announced in the statistics, the shares of silvicultural systems 2-7 were scaled down equally.

Thinning rules

Table A.5 BAU thinning rules for the Northern Europe

		Species group	
		1 - 2	**4 - 6**
Silvicultural system	**1**	No management	
	2	Each year: average annual values from 100 years of BAU management for that plot	
	3	Not applied	

	4	<u>Thinnings:</u>
		if site class = 3:
		if (H<20 & BA> -0.0893*H^2 + 4.0071*H - 11.343):
		BA = -0.0536*H^2 + 2.7643*H - 9.6857
		else if BA> 33: BA = 24
		
		if site class = 2:
		if (H<20 & BA> -0.125*H^2 + 4.95*H - 20.9):
		BA = -0.1071*H^2 + 3.9286*H - 15.771
		else if BA>28: BA = 20
		
		if site class = 1:
		if (H<20 & BA> -0.1071*H^2 + 4.2286*H - 15.571):
		BA = -0.0714*H^2 + 2.7857*H - 9.1143
		else if BA>26: BA = 18
		
		<u>Final cutting:</u>
		if SP GROUP = 1 & (D>26 | age = 90): BA = 0
		if SP GROUP = 2 & (D>29 | age = 80): BA = 0
			<u>Thinnings:</u>
			if (H<20 & BA> -0.0179*H^2 + 1.2214*H + 3.7714) :
			BA = - 0.0536*H^2 + 2.4643*H - 12.886
			
			if BA> 21:
			BA = 15
			
			<u>Final cutting:</u>
			if (D>27 | age =60): BA = 0

	5	-	
	6	-	
	7	-	

Table A.6 BAU thinning rules for the Central East Europe

		Species group				
		1	**2**	**4**	**5**	**6**
Silvicultural system	**1**	No management				
	2	Each year: average annual values from 100 years of BAU management for that plot				
	3	Thinnings: if BA>10: BA = 0.0143267762601335*H^3 - 0.614927051837991*H^2 + 9.08632900819839*H^1 - 22.5383283711983 if H>22: BA=30 Final cutting: if D>40: old forest is cut, new trees of age 10 remain	Thinnings: if BA>10: BA = -0.000449565934865019*H^3 - 0.0647381648173298*H^2 + 3.57770708458383*H^1 - 8.62935488762737 if H>22: BA = 33 Final cutting: if D>40: old forest is cut, new trees of age 10 remain	Same as SP 6	Thinnings: if BA>10: BA = 0.0026787284262556*H^3 - 0.142666627785475*H^2 + 3.07535121266828*H^1 - 1.92762917567757, BA))) if H>22: BA = 26 Final cutting: if D>40: old forest is cut, new trees of age 10 remain	Thinnings: if BA>10: BA = 0.000384281194291039*H^3 - 0.0216390579886234*H^2 + 1.25889296446695*H^1 + 3.50818742704763, BA))) if H>29: BA = 30 Final cutting: if D>40: old forest is cut, new trees of age 10 remain
	4	Thinnings: if BA>10: BA = 0.0143267762601335*H^3 - 0.614927051837991*H^2 + 9.08632900819839*H^1 - 22.5383283711983 if H>22: BA=30 Final cutting: if D>30: BA=0. New seedlings assumed to be planted next year.	Thinnings: if BA>10: BA = -0.000449565934865019*H^3 - 0.0647381648173298*H^2 + 3.57770708458383*H^1 - 8.62935488762737 if H>22: BA = 33 Final cutting: if D>30: BA=0. New seedlings assumed to be planted next year.	Same as SP 6	Thinnings: if BA>10: BA = 0.0026787284262556*H^3 - 0.142666627785475*H^2 + 3.07535121266828*H^1 - 1.92762917567757, BA))) if H>22: BA = 26 Final cutting: if D>30: BA=0. New seedlings assumed to be planted next year.	Thinnings: if BA>10: BA = 0.000384281194291039*H^3 - 0.0216390579886234*H^2 + 1.25889296446695*H^1 + 3.50818742704763, BA))) if H>29: BA = 30 Final cutting: if D>30: BA=0. New seedlings assumed to be planted next year.
	5 - 6	Same as silvicultural system 4, roots remain & planting is not needed				
	7	Final cutting: if age>25, BA = 0				

Table A.7 BAU thinning rules for the Central West Europe

<table>
<tr><td></td><td></td><td colspan="7">Species group</td></tr>
<tr><td></td><td></td><td>1</td><td>2</td><td>3</td><td>4</td><td>5</td><td>6</td><td>7</td></tr>
<tr><td>Silvicultural system</td><td>1</td><td colspan="7">No management</td></tr>
<tr><td></td><td>2</td><td colspan="7">Each year: average annual values from 100 years of BAU management for that plot</td></tr>
<tr><td></td><td>3</td><td>Thinnings:
if age>=30:
BA = 15.7
if age>=35 |
D>11.9: BA = 20
if age>=50 | D>17:
BA = 23
if age>=60 |
D>19.9: BA = 23
Final cutting:
if age>=85 | D>50:
old forest is cut,
new trees of age 10
remain</td><td>Thinnings:
if age>=25:
BA = 20.15
if age>=40 |
D>23.4:
BA = 25
if age>=60 |
D>31.6:
BA = 30
if age>=80 |
D>36.8:
BA = 35
Final cutting:
if age>=85 |
D>50:
old forest is cut,
new trees of age
10 remain</td><td>Final cutting:
if age>=60 |
D>50:
BA = 0</td><td>Thinnings:
if age>=30:
BA = 12.9
if age>=50 |
D>31.6:
BA = 20
if age>=70 |
D>36.8:
BA = 23
Final cutting:
if age>=90 |
D>50:
old forest is cut,
new trees of age
10 remain</td><td>Thinnings:
if age>=25:
BA = 10.02
if age>=35 |
D>11.5:
BA = 13
if age>=55 |
D>19.5:
BA = 17
if age>=80 |
D>30.3:
BA = 19
Final cutting:
if age>=95 | D>50:
old forest is cut,
new trees of age 10
remain</td><td>Thinnings:
if age>=30:
BA = 12.66
if age>=35 |
D>23.4:
BA = 16
if age>=60 |
D>31.6:
BA = 21
if age>=100 |
D>36.8:
BA = 24
Final cutting:
if age>=105 |
D>50:
old forest is cut,
new trees of age 10
remain</td><td>Final cutting:
if age>=60 |
D>50:
BA = 0</td></tr>
<tr><td></td><td>4</td><td>Thinnings:
if age>=30: BA =
15.7
if age>=35 |
D>11.9: BA = 20
if age>=50 | D>17:
BA = 23
if age>=60 |
D>19.9: BA = 23</td><td>Thinnings:
if age>=25:
BA = 20.15
if age>=40 |
D>23.4:
BA = 25
if age>=60 |
D>31.6:
BA = 30</td><td>Final cutting:
if age>=60 |
D>50: BA=0. New
seedlings assumed
to be planted next
year.</td><td>Thinnings:
if age>=30:
BA = 12.9
if age>=50 |
D>31.6:
BA = 20
if age>=70 |
D>36.8:
BA = 23</td><td>Thinnings:
if age>=25:
BA = 10.02
if age>=35 |
D>11.5:
BA = 13
if age>=55 |
D>19.5:
BA = 17</td><td>Thinnings:
if age>=30:
BA = 12.66
if age>=35 |
D>23.4:
BA = 16
if age>=60 |
D>31.6:
BA = 21</td><td>Final cutting:
if age>=60 |
D>50: BA=0.
New seedlings
assumed to be
planted next year.</td></tr>
</table>

		<u>Final cutting:</u>
		if age>=85 | D>50: BA=0. New seedlings assumed to be planted next year.
			if age>=80 | D>36.8:
			BA = 35
			<u>Final cutting:</u>
			if age>=85 | D>50:
			BA=0. New seedlings assumed to be planted next year.
					<u>Final cutting:</u>
					if age>=90 | D>50:
					BA=0. New seedlings assumed to be planted next year.
						if age>=80 | D>30.3:
						BA = 19
						<u>Final cutting:</u>
						if age>=95 | D>50: BA=0. New seedlings assumed to be planted next year.
							if age>=100 | D>36.8:
							BA = 24
							<u>Final cutting:</u>
							if age>=105 | D>50: BA=0. New seedlings assumed to be planted next year.	

	5 - 6	Same as silvicultural system 4, roots remain & planting is not needed						
	7	Final cutting: if age>25, BA = 0						

Table A.8 BAU thinning rules for the South Europe

<table>
<tr><td></td><td></td><td colspan="7">Species group</td></tr>
<tr><td></td><td></td><td>1</td><td>2</td><td>3</td><td>4</td><td>5</td><td>6</td><td>7</td></tr>
<tr><td rowspan="4">Silvicultural system</td><td>1</td><td colspan="7">No management</td></tr>
<tr><td>2</td><td colspan="7">Each year: average annual values from 100 years of BAU management for that plot</td></tr>
<tr><td>3</td><td></td><td><u>Thinnings:</u>
if age>=30|
BA>30:
N = 700

if age>=70|
BA>40:
N = 400

<u>Final cutting:</u>
if (age>120 |
BA>50):
N = 0</td><td><u>Thinnings:</u>
if (age>=15:
N = 1000

if (age>=20 | D>20:
N = 850

if (age>=40 | D>24:
N = 700

if (age>=60 | D>30:
N = 500

<u>Final cutting:</u>
if ((age>=60 & D>=30)
| BA>50):
N = 0</td><td></td><td><u>Thinnings:</u>
if (age>=40
|D>=21:
N = 500

if (age>=60
|D>=35:
N = 370

if (age>=80
|D>=37:
N = 290

if (age>=100
|D>=40:
N = 255

<u>Final cutting:</u>
if (age>=120
|D>=42 | BA>50):
N = 0</td><td><u>Thinnings:</u>
if (age>=40
|D>=20:
N = 550

if (age>=60
|D>=30:
N = 400

if (age>=80
|D>=39:
N = 360

if (age>=100
|D>=46:
N = 300

<u>Final cutting:</u>
if (age>=120 |D>53
| BA>50):
N = 0</td><td></td></tr>
<tr><td>4</td><td><u>Thinnings:</u>
if (H>=11:
N = 550

if (H>=25:
N = 300</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

<table>
<tr><td></td><td></td><td>Final cutting:
if (age>100 \|
BA>50):
N = 0</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td></td><td>5 - 6</td><td></td><td></td><td></td><td></td><td></td><td></td><td>Final cutting:
if (age>=50 \|
BA>50):
N = 0</td></tr>
<tr><td></td><td>7</td><td colspan="7">if (age >= 15): N = 0</td></tr>
</table>

NPP results comparison with other forest models

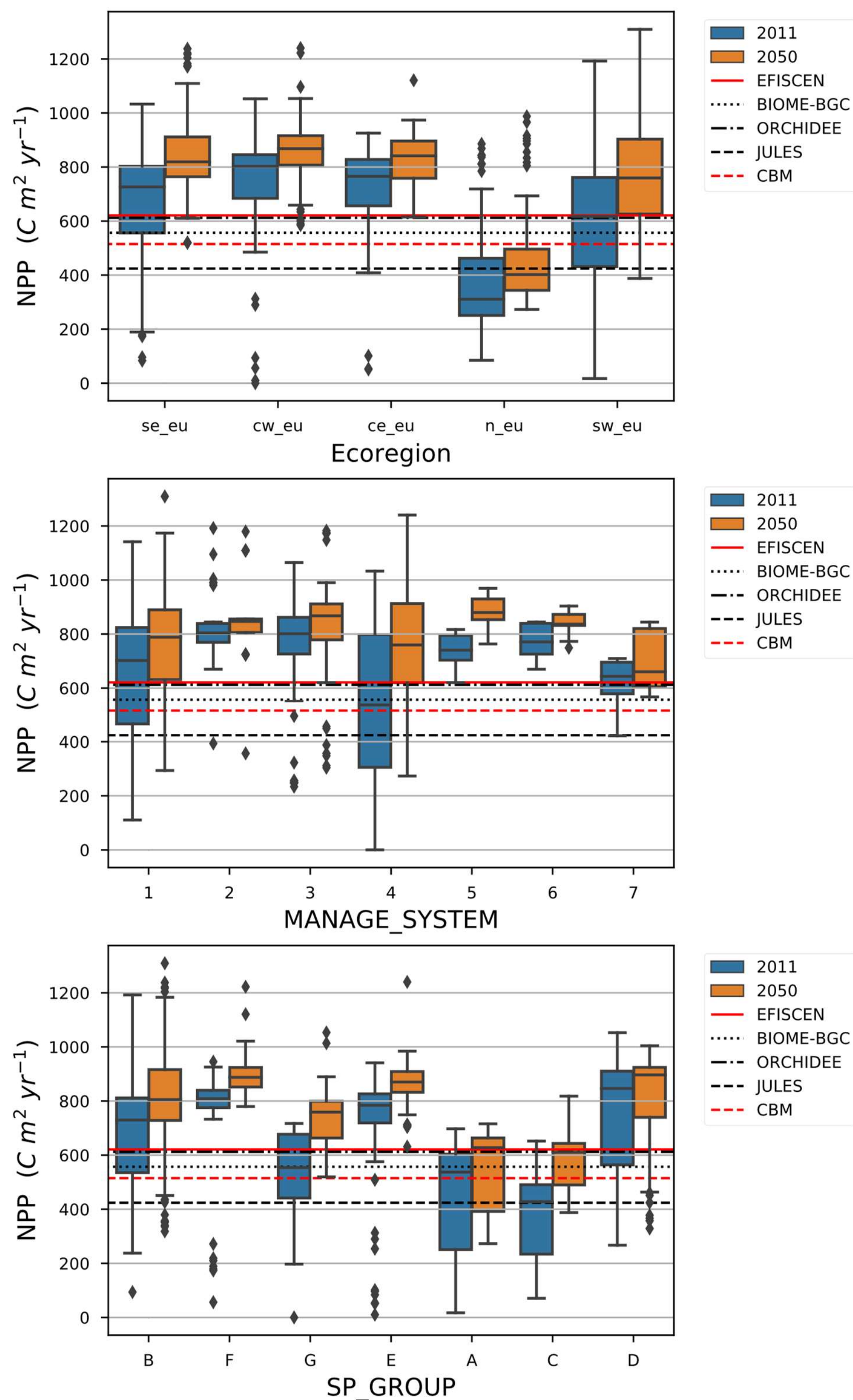

Figure A2 Comparison of our NPP estimate (reported in g C m^2 yr^{-1} FoU^{-1}) with the average European level estimates of other models (EFISCEN, BIOME-BGC, ORCHIDEE, JULES and CBM). The values are taken from Pilli et al. (2017)

Appendix IV

Modelling forestry operations

Life-cycle emissions of forestry operations are taken from the Ecoinvent 3.3 (Wernet et al. 2016) unit process ` softwood forestry, mixed species, sustainable forest management ,CH`.

Functional equivalence structural glulam vs steel

In Table A.9 LCA studies that compare functionally equivalent buildings using different materials have been gathered and studied to compare the consumption of wood vs steel to build the structural framework and derive the functional equivalence between the two materials. In all the studies included in the analysis the wooden beams are made of glulam.

Table A.9 Equivalence ratios for glulam vs steel frames used to build the structural framework

Source	Comment	Ratio use of steel in competing structure/ use of wood in wood structure (kg steel / kg glulam)
Albrecht (2008)		1.9
Petersen and Solberg (2002)		0.78
Courard et al. (2001)		1.7
The Engineered Wood Association (2013)	Average	1.0
	Minimum	0.66
	Maximum	1.6
Our study	Average of previous	1.27

Construction modelling (glulam and steel beams)

The life cycle of the glulam has been modeled in accordance to the Environmental Product Declaration (EPD) standard EN 15804 (CEN 2012). The life cycle inventories of both first and second transformation have been modeled mostly based on Ecoinvent 2.2. In accordance with the aforementioned standard in both stages economic allocation was applied. Also steel fittings are included in the modelling of the glulam production. At the end-of-life the glulam beam was assumed to be partially recycled, partially landfilled and partially used for energy recovery according to the figures reported in Mantau et al. (2010). Following the EPD standard, system expansion is applied in this stage and substituted impacts for recycling and energy recovery are included in the calculation. It was assumed that the electricity and heat recovered substitute respectively the current European electricity and heat production grid. The part that is recycled is assumed to replace the production of wood panels from virgin wood.

Pallet Modelling

The main data source used for modelling reusable wood pallet and reusable plastic pallets was the study RDC (2010). Some modifications were made and are described in the following section to be consistent with our case. The functional unit is to transport and protect goods during 28 rotations which is the average rotation number for both the plastic and the wood pallet in the RDC (2010) study. The average load of a truck, when it is transporting goods, is 13,6 tonnes (European Commission 2010), so the weight of the pallet does not change the number of trucks needed to transport a certain amount of goods. However, a heavier pallet increases the fuel consumption of the truck and this supplementary consumption is accounted for in the calculation of the substitution factor. The system boundaries include the production phase, the transport phase, the use phase and the end of life phase. The use phase consists in using the pallet to transport goods, transporting the pallet where it is needed next and, for the wood pallet, repairing it. For the plastic pallet (18,5 kg), the use phase consists in either transporting the pallet from one client to another or sending back the pallet to the pooler. For the wooden pallet (26 kg), the use phase consists in transporting the pallet to a sorting platform where in 25% of the time, the pallet must be repaired (4 dm^3 of wood are then used). As a result, over the lifetime of the pallet, considering a 26 kg pallet, 15 kg of additional wood is used for repair. This additional amount of wood is accounted for in the analysis. The transport from client to client is also accounted for in the evaluation and three transport distances are considered: 1250 km (RDC 2010) and 610 km which is the average bilateral international transport distance in Europe for goods (European Commission 2010) and 150 km (European Commission 2010), 76% of goods are transported less than 150 km). Regarding the other transport phases during the use phase (relocalisation and transport to sorting/repair site), the distances considered in RDC (2010) study are smaller for the wood pallet as compared to the plastic pallet to account for the fact that the wood pallet industry has a very tight network of sites. A sensitivity analysis was performed to consider the same network for the plastic pallet as for the wooden pallet. The end of life assumptions for both wood pallets and plastic pallets follows RDC (2010) study: 95% recycling for the plastic pallet, 95% recycling and energy recovery for the wooden pallet, the rest being incinerated (50%) and landfilled (50%). The shares of recycling and energy recovery for the wooden pallet are respectively 60% and 40% (study own estimation). The end of life assumptions for plastic pallets follows RDC (2010) study: 90% recycling, the rest being incinerated (5%) and landfilled (5%). For the wooden palled we followed the data reported in Vis et al. (2016) with 32% of pallet landfilled, 33% recycled, 30% sent to boiler for energy recovery and 5% incinerated.

Bioenergy modelling

For bioenergy we used data from Ecoinvent 3.3 (Wernet et al. 2016). The unit process `heat and power co-generation, wood chips, 6667 kW, CH` was used to model the impact bioenergy produced from CHP plant. For the of electricity the allocated unit process ` heat and power co-generation, wood chips, 2000 kW, state-of-the-art 2014` was used while heat production was modelled based on the unit process `heat production, softwood chips from forest, at furnace 1000kW, state-of-the-art 2014,CH`. The datasets include the infrastructure, the wood input, the emissions to air and the disposal of the ashes. Also included are substances needed for operation: lubricating oil, organic chemicals, sodium chloride, chlorine and decarbonized water.

Appendix V

Author's contribution in the chapters of the thesis.

	Ch.2	**Ch.3**	**Ch.4**	**Ch.5**
Conception and design	GC+BM+WA+LV	GC+CM	GC	GC
Planning	GC	GC	GC	GC
Data collection	All co-authors	GC+EV	GC+TV+SA+AM	GC+EV
Data analysis	GC+TV	GC	GC+SA+AM	GC
Writing the article	GC	GC	GC	GC
Editing and reviewing	All co-authors	All co-authors	GC+LV+BM	GC+WA+BM
Overall responsibility	GC	GC	GC	GC

GC = Giuseppe Cardellini, BM = Bart Muys, WA= Wouter Achten, LV = Liesbet Vranken, CM = Chris Mutel, EV = Estelle Vial, TV = Tatiana Valada, SA = Sanna Härkönen, AM = Alexander Moiseyev.