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The environmental impacts of wood compared to other building materials

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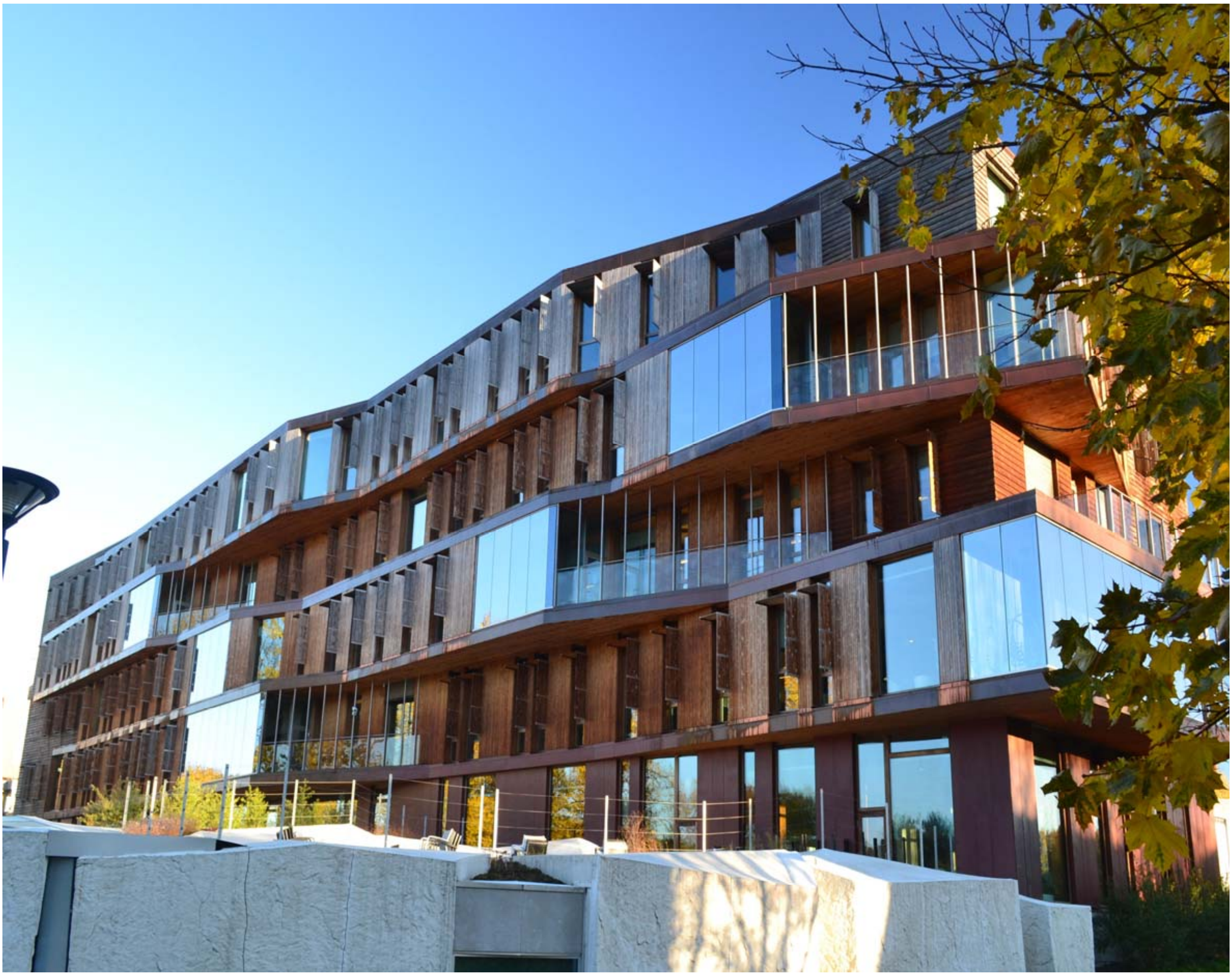

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Preface

This report was produced by order of the Norwegian Ministry for Agriculture and Food (Landbruks og matdepartementet (LMD)).

In the governmental communication 6 (2016-2017) *Values in growth* it was indicated that the documentation of wood as a building material is still challenging. Knowledge is missing on how to most comprehensively and effectively measure the environmental impact and sustainability of buildings. Such knowledge can serve as a base for political decisions as well as decisions on different building materials and building types for private companies.

After dialog with the ministry, we prepared this report by going in detail into general considerations on the different methods of environmental impact analysis. We conducted an analysis on wood LCAs that have been done in Norway and comparable countries, as well as on competing materials like concrete and steel. We conducted an analysis comparing the environmental impacts of wood and other materials, summarised the results and evaluated their importance and the use such findings can have for political decisions in the future.

Callum Hill was the main author of the report while Katrin Zimmer contributed the parts which covered the Norwegian building market, the forestry sector and the review on Scandinavian reports on the topic. The authors are very grateful for Lone Ross Gobakkens help with the translation into Norwegian and for helpful comments.

Ås, 25.04.18

Katrin Zimmer

Innhold

Sammendrag

Miljøpåvirkning av tre sammenlignet med andre bygningsmaterialer

Den norske regjeringen har satt klare mål for å redusere forbruket av fossil energi og klimagassutslipp. Byggsektoren kan bidra for å nå disse målene ved å:

- Bygge energieffektive bygg;
- Bruke materialer med lavt forbruk av grå energi (low embodied energy materials);
- Bruke byggematerialer som lager for atmosfærisk karbondioksid.

Det er foretatt en analyse av vitenskapelige artikler som foretar livsløpsvurderinger (LCA) knyttet til bygningsmaterialer. Bruk av tre i konstruksjoner spiller en viktig rolle som en del av strategien for økt karbonlagring og redusert energiforbruk i byggsektoren. I de fleste av de gjennomgåtte studiene er det enighet om at det finnes miljøfordeler knyttet til bruk av tre i bygg med tanke på å begrense klimagassutslipp og klimaendringer.

På tidspunktet for utarbeidelsen av denne rapporten, finnes det ikke LCA-verktøy som er avansert nok for bruk som beslutningsstøtte ved valg av materialer for hele bygget når målet er å minimere miljøpåvirkningen. Dette kan bare avgjøres fra sak til sak.

LCA-studier kan imidlertid brukes som informasjonsgrunnlag ved politiske beslutninger om bruk av materialer for å minimere klimapåvirkningen fra byggsektoren i Norge, dersom påvirkningskategorien for globalt oppvarmingspotensial (GWP) legges til grunn i kombinasjon med data for energiforbruk. Denne metodikken har imidlertid iboende usikkerheter.

Rapportens omfang

Utgangspunktet for rapporten var som følger:

- Generelle vurderinger av de ulike metodene for miljøpåvirkningsanalyse og -evalueringer (som LCA, EPD, HWP, BREEAM) og hva som er forskjellene mellom disse systemene;
- Gjennomføre en analyse av livsløpsvurderinger for trematerialer som er foretatt i Norge og sammenlignbare land, og en sammenstilling av dataene fra disse. Hvilke faktorer påvirker analysen og hvor mye påvirker enkeltfaktorer resultatet?
- Utføre en lignende analyse av konkurrerende materialer som betong og stål;
- Utføre en analyse som sammenligner miljøpåvirkningen fra trevirke og andre materialer. Hva er det som faktisk blir sammenlignet, og hva betyr det for det reelle klimaavtrykket?
- Oppsummering av resultatene, evaluering av betydningen av resultatene og anvendelsen slike funn kan ha for fremtidige politiske beslutninger.

Rapporten begynner med en beskrivelse av den eksisterende bygningsmassen, byggesektoren samt skog- og trenæringen i Norge, og gir deretter en oversikt over hvilke LCA-metoder som brukes, samt styrkene og svakhetene ved teknikken. LCA er et komplekst emne, og metodikkene og påvirkningskategorier er fremdeles gjenstand for drøfting. LCA har ikke det presisjonsnivået som er påkrevd innenfor mange påvirkningskategorier for å kunne foreta sammenlignende vurderinger, og det er kun påvirkningskategoriene for globalt oppvarmingspotensial og potensial for nedbrytning av ozonlag som anses å være tilstrekkelig robuste for å kunne gi nøyaktige og pålitelige data.

Det er også foretatt en gjennomgang av byggklassifiseringsordninger. LCA utgjør bare en liten del av byggklassifiseringsordninger som BREEAM (Building Research Establishment Environmental Assessment Method) og LEED (Leadership in Energy and Environmental Design), og disse sier lite om valg av byggematerialer. Disse ordningene fremmer mer miljøbevisste konstruksjonsprinsipper til en viss grad, men de er ikke tilstrekkelig robuste til å kunne anvendes som verktøy for å tilrettelegge informasjon for politiske beslutninger eller ved valg av byggemateriale.

Rapporten fokuserer på problemstillinger knyttet til opptak og lagring av karbon i skog og hvordan atmosfærisk karbon kan lagres i produkter med lang levetid i byggsektoren. En klar fordel ved å bruke tre i bygg, er potensialet for lagring av karbon fra atmosfæren i hele dets levetid. Selv om karbonlageret i treprodukter har en viktig rolle med tanke på klimagassregnskapet, har denne litteraturgjennomgangen avdekket at de fleste studier viser at substitusjonseffekten av å erstatte materialer som krever mye energi ved produksjon, samt å erstatte fossilt brensel i energiproduksjon med tre og biprodukter fra tre er langt større. Et stort flertall av livsløpsvurderingene av treprodukter har vist at mengden atmosfærisk karbon som er lagret i treet (målt i CO_2-ekvivalenter), er større enn utslippene av drivhusgasser som følger av bearbeidingen av materialet. Ytterligere fordeler oppnås når trevirket brennes og erstatter fossilt brensel ved endt livsløp. Man får størst fordeler av å erstatte fossilt brensel når kull erstattes med overskuddstrevirke og biprodukter. I en norsk sammenheng vil man oppnå størst fordeler når trevirke brukes som brensel i sementovner eller som karbonkilde for aluminiumsanoder, etterfulgt av å erstatte olje til oppvarming og deretter som erstatning for naturgass til oppvarming eller elektrisitetsproduksjon.

Studier - metodikk, variasjon, begrensninger og usikkerhet

Denne rapporten gjennomgår også den vitenskapelige litteraturen av publiserte LCA-studier av vanlige byggematerialer (tre, sement/betong, aluminium, stål, poly(vinylklorid)). Det er påvist at utfallet av en LCA avhenger sterkt av forutsetningene og systemgrensene som brukes. Det er ikke mulig å nå frem til én endelig verdi (f.eks. GWP) som er karakteristisk for et materiale, men det vil alltid være et spekter av verdier som vil ha betydning. Metodikken som brukes til å bestemme miljøpåvirkning er kompleks. Mange studier kan ikke uten videre benyttes ved en sammenligning av studier på grunn av forskjeller i funksjonell enhet, anvendte databaser, antagelser om materialenes levetid, vedlikehold, scenarier for livsløpsslutt osv. I tillegg er det vanskelig å verifisere de oppnådde resultatene i de fleste studiene. En LCA vil uvegerlig inneholde forenklinger, noe som kan påvirke dataenes nøyaktighet. De fleste studier benytter ikke følsomhetsanalyse for å vise hvordan forutsetningene og variasjonene påvirker resultatene. Det er nødvendig å ta hele livsløpet i betraktning ved valg av materialer, og den eneste måten å gjøre dette på, er å legge det totale bygget til grunn. Dette øker imidlertid usikkerheten i beregningene, og innebærer forutsetninger og innføring av scenarier som kanskje ikke er realistiske eller rimelige.

En rekke faktorer kan påvirke en LCA for byggematerialer gjennom deres levetid, og disse faktorene kan deles inn i usikkerhet og variasjon. Usikkerhet skyldes mangel på nøyaktig kunnskap om prosesser eller bruk av forutsetninger. Variasjon kan oppstå på grunn av ulike valg vedrørende bruken av materialer, som for eksempel vedlikeholdshyppighet og -type, forskjellige avhendingsmetoder, transportavstander osv. Kombinasjoner av usikkerhet og variasjon kan være vanskelig å skille fra hverandre. Det er en betydelig grad av usikkerhet som kan påvirke dataene, særlig når livsløpets bruks- og sluttstadier tas med.

Resultater og standardisering

Følgelig er det stor variasjon i metodikken som anvendes for en LCA, noe som har en stor innvirkning på resultatet. Derfor er arbeidet med å komme frem til sammenlignbare resultater ekstremt krevende. Man har imidlertid oppnådd en viss grad av konsensus ved innføring av miljøproduktdeklarasjoner (EPD) og standardisering av prosedyrer; sistnevnte er kjent som produktkategoriregler (PCR). Likevel er det fortsatt bekymring for at sammenligninger mellom produkter ikke er pålitelige på grunn av usikkerhet og variasjoner i forutsetningene som ligger til grunn, bruk av forskjellige databaser osv. Den største fordelen med en EPD som er utarbeidet i samsvar med europeisk standard EN 15804, er at påvirkningen må rapporteres separat for ulike livsløpsfaser. Av disse er livsløpsstadiet «fra vugge til fabrikkport» (modul A1-A3) sannsynligvis det mest pålitelige, siden denne delen av livsløpet innebærer færrest antagelser og har de mest nøyaktige dataene.

Globalt oppvarmingspotensial og grå energi

Denne rapporten har i stor grad fokusert på data knyttet til 1) energiforbruket ved produksjon av materialer (grå energi) og 2) miljøpåvirkning knyttet til globalt oppvarmingspotensial (GWP), ettersom disse kategoriene har de laveste usikkerhetene. GWP-dataene er sterkt påvirket av tidsrammen for de respektive studiene og av en rekke ulike faktorer som må tas i betraktning i sammenlignende studier:

- utslipp av drivhusgass (GHG) knyttet til produksjon av byggematerialer, vedlikehold, utskifting og avhending;
- utslipp av GHG knyttet til operasjonelle energibehov, hvis disse er relevante og realistiske og ikke er innført for å favorisere ett materiale fremfor et annet;
- karbonutslipp og karbonlagring fra skogbruk og binding av karbon gjennom økende biomasse;
- substitusjonseffekter knyttet til bruk av tre i forhold til andre byggematerialer
- scenarier ved livløpsslutt, som ombruk, materialgjenvinning eller forbrenning med energiutvinning.

Den grå energien som går med til å produsere byggematerialer, har en viktig betydning når man analyserer miljøpåvirkningen. Den innledende grå energien ved produksjon må skilles fra den gjentatte grå energien, som oppstår i forbindelse med vedlikehold av materialene, og driftsenergien, som er energien som forbrukes på grunn av bygningens driftskrav (f.eks. oppvarming). Etter hvert som driftseffektiviteten i bygningene forbedres, vil den grå energien utgjøre en større andel av det samlede energibehovet. Den grå energien representerer dessuten en større andel av sektorens samlede energiforbruk i et voksende marked. Her krever produksjon av en funksjonell enhet trematerialer mindre energi i sammenligning med en funksjonell enhet av ikke-fornybare gjennomgåtte studiene er det enighet om at det finnes miljøfordeler. Økt bruk av trevirke i bygg vil gi en økt mengde lagret karbon i det samlede karbonlageret av treprodukter ('harvested wood products - HWP') på et gitt tidspunkt. Dette kan bli en del av en mer omfattende strategi for å gå over til bioøkonomien, en økonomi basert på redusert bruk av fossile karboner.

Fremtiden

Skogen i Norge tar i dag opp CO_2 tilsvarende 40% av de årlige klimagassutslippene, men dette vil gå ned etter hvert som aldersstrukturen i skogen blir eldre. For å opprettholde lagerøkningen, er det nødvendig å øke utnyttingsgraden av norsk skog. Lagring av karbon i treprodukter (HWP) bør skje i produkter med lang levetid for å begrense klimaendringene. Trevirke er det dominerende materialet i eneboliger og mindre rekkehus, men er lite brukt i blokker og høyhus. Bruk av tre i høyhus til næring- og boligformål vil gi fordeler sett i et utslippsperspektiv. Den norske skog- og trenæringen bør utnytte potensialet for trevirke i fler-familieboliger og høyhus til å videreutvikle en eksportindustri for prefabrikkerte bygningsdeler. Det er avgjørende å oppnå en verdiøkning for skog- og trenæringen. Ved å oppmuntre til produksjon av massivtre og andre trebaserte elementer i Norge, skapes potensiale for eksport av modulære boliger og byggkonstruksjoner til utenlandske markeder, som for eksempel Storbritannia.

Executive summary

The Norwegian Government has set ambitious goals for the fossil carbon intensiveness of the Norwegian economy. The built environment can make an important contribution towards achieving those goals by:

- Building energy efficient buildings;
- Using low embodied energy materials;
- Using construction materials as stores of atmospheric carbon dioxide.

An analysis of life cycle assessment (LCA) studies published in the scientific literature has been undertaken. The use of timber in construction has an important role to play as part of an energy reduction and carbon storage strategy for the built environment. In the majority of studies analysed there is agreement that there are environmental advantages associated with the use of timber in construction from a climate change mitigation perspective.

At the time of writing this report there is no LCA-based tool that is sophisticated enough to be used at the whole building level to assist in decision-making processes for materials to minimise environmental impacts. This can only be determined on a case-by-case basis.

However, LCA can be used to inform policy decisions regarding the use of materials to minimise the climate change impacts of the built environment in Norway, if the GWP (global warming potential) impact category is used in combination with the embodied energy data. But the methodology does have inherent uncertainties.

The original terms of reference for the report, were as follows:

- General considerations on the different methods of environmental impact analysis and evaluations (LCA, EPD, HWP, BREEAM….) and what the differences are between these systems;
- Conduct an analysis on wood LCAs that have been done in Norway and comparable countries, and a compilation of these data. Which factors influence the analysis and how much do single factors affect the result?
- Conduct a similar analysis on competing materials like concrete and steel;
- Conduct an analysis comparing the environmental impacts of wood and other materials. What is actually being compared and what does it imply for the real climate footprint?
- Summarise the results, evaluation of their importance and the use such findings can have for political decisions in the future.

The report begins with a description of the Norwegian built environment and forest products' sectors and then gives an overview of the methodologies used in LCA and the strengths and weaknesses of the technique. LCA is a complex subject and there is still debate about the methodologies and impact categories. LCA does not have the level of accuracy needed in many impact categories in order to make comparative assessments and only the impact categories global warming potential and ozone layer depletion potential are considered to be sufficiently robust to give accurate and reliable data.

A review of building assessment schemes has also been undertaken. LCA comprises only a minor part of building assessment schemes, such as the Building Research Establishment Environmental Assessment Method (BREEAM) and Leadership in Energy and Environmental Design (LEED) and these have little to say about the choice of materials for construction. These schemes have some value in promoting more environmentally-conscious designs, but they are not sufficiently robust to be used as tools to inform policy-making, or building material choices.

The report focuses on issues surrounding carbon sequestration in forests and how atmospheric carbon can be stored in long-life products in the built environment. One of the advantages of using timber in construction is the potential for the storage of biogenic carbon (derived from atmospheric carbon dioxide) in long-life structures. Although this does have a role to play in climate change mitigation, this literature review has revealed that most studies show that the effects of substitution for high embodied energy materials and for fossil fuels for energy production are much more significant. The overwhelming majority of LCAs of timber products have shown that the amount of atmospheric carbon stored in the wood (measured as CO_2 equivalents) is always larger than the GHG (greenhouse gas) emissions associated with the processing of the material. Additional benefits arise when the wood is incinerated at the end of the life cycle, with substitution of fossil fuels. The highest fossil fuel substitution benefits arise when coal is replaced with timber wastes/by-products. In a Norwegian context, the highest benefits will arise if wood is used as a fuel for cement kilns, or as a carbon-source for aluminium anodes, followed by a replacement of oil for heating then natural gas for heating or electricity production.

This report also reviews the scientific literature of published LCA studies of commonly-used building materials (timber, cement/concrete, aluminium, steel, poly(vinyl chloride)). It is shown that the outcomes of the LCAs are very heavily dependent upon the assumptions made and the system boundaries used. It is not possible to arrive at definitive a value of (for example, global warming potential, GWP) that is characteristic for a material, but there is a range of values. The methodology used to determine the environmental impacts is complex and many studies are not readily amenable to comparative studies. This is because of differences in functional unit, supporting databases, assumptions regarding material life, maintenance, end-of-life scenarios, etc. In addition, most studies lack sufficient transparency to allow for proper verification of the results obtained. LCAs also inevitably contain simplifications, which may affect the accuracy of the data. Most studies do not employ a sensitivity analysis to show how the assumptions and variabilities affect the results. It is necessary to consider the whole life cycle when making materials choices and the only way to do this is at the whole building level. However, this increases the degree of uncertainty in the calculations and involves assumptions and the introduction of scenarios which may not be realistic or reasonable.

A variety of factors can affect the LCA of building materials over their lifetime, which can be divided into uncertainties and variabilities. Uncertainties arise from lack of precise knowledge regarding processes or the use of assumptions. Variabilities can arise due to different choices regarding the use of materials, such as frequency and type of maintenance, different disposal methods, transport distances, etc. Combinations of uncertainty and variability can be difficult to separate. There is considerable scope for uncertainty to affect the data, especially when the in-service and end-of-life stages of the life cycle are included.

Consequently, there is considerable variability in the methodology applied for LCAs, which has a significant influence on the output and hence the task of making comparative assertions is extremely difficult. However, there has been some degree of consensus reached with the introduction of environmental product declarations (EPDs) and standardisation of procedures; known as product category rules (PCRs). Nonetheless, there is still concern that inter-product comparisons are not reliable, due to uncertainties and variations in the assumptions made, the use of different databases, etc. The main advantage with EPDs which are produced in conformity with the European standard EN 15804, is that the impacts have to be reported separately for different life cycle stages. Of these, the cradle to factory gate life cycle stage (modules A1-A3) is likely to be the most reliable, since this part of the life cycle involves the least assumptions and the most accurate data.

This study has largely focussed on data concerned with the embodied energy associated with materials and the global warming potential (GWP) environmental impact category, because these have the lowest uncertainties. GWP data is strongly influenced by the time-frames of the study and by a range of different factors that have to be taken into account when making comparative studies:

- Greenhouse gas (GHG) emissions associated with the manufacture of construction materials, maintenance, replacement and disposal;
- GHG emissions associated with operational energy requirements, if these are relevant and realistic and have not been introduced to favour one material over another;
- Carbon emissions and storage from forestry operations and sequestration by growing biomass;
- Substitution effects associated with the use of timber in comparison to other building materials;
- End-of-life scenarios, such as recycling, or incineration with energy recovery.

The embodied energy used to produce construction materials is an important consideration when analysing the environmental impacts. This initial embodied energy is to be distinguished from the recurring embodied energy which arises due to maintenance of the materials and the operating energy, which is energy consumed due to the operational requirements (e.g., heating) of the building. As the operating efficiency of buildings improves, the embodied energy will be a larger proportion of the overall energy requirements. The embodied energy also represents a greater proportion of the overall energy consumption of the sector in a growing market. Sawn timber products are lower embodied energy materials when compared, on a functional unit basis, with non-renewable construction products. The increased use of timber in construction will result in more carbon storage in the harvested wood products carbon pool at a critical time. This can form part of a wider strategy to move to a low fossil carbon economy.

Although timber is the dominant material used in single-family dwellings, it is little used in multiple-occupancy buildings. The Norwegian forests are currently absorbing levels of carbon dioxide which are equivalent to about 40% of the annual emissions, but this will fall as the age structure of the forests matures. In order to maintain these high levels of sequestration it is necessary to increase the harvesting intensity of Norwegian forests. The carbon in the HWPs produced should be stored in long life products in the built environment for the maximum climate change mitigation effect. The use of timber in high-rise non-residential and multiple-occupancy residential construction would give benefits from a climate change mitigation perspective. The Norwegian forest products sector should use the opportunity provided by the increased use of timber in multi-occupancy and multi-storey buildings to develop an export industry in pre-fabricated structures. Adding value to the forest products sector is essential. By encouraging a cross laminated timber industry in Norway, there will be potential for export of multi-occupancy buildings using modular construction methods to exterior markets, such as the UK.

1 Abbreviations

AGWP	Absolute global warming potential
BIM	Building information management
BRE	Building Research Establishment
BREEAM	Building Research Establishment Environmental Assessment Method
CLT	Cross laminated timber
DALY	Disability adjusted life years
EC-JRC	European Commission – Joint Research Centre
EE	Embodied energy
EPD	Environmental product declaration
FSC	Forest stewardship certification
GHG	Greenhouse gas
GtC	Gigatonnes carbon
GWP	Global warming potential
HWP	Harvested wood product
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LEED	Leadership in Energy and Environmental Design
LULUCF	Land use, land use change and forestry
LVL	Laminated veneer lumber
MDF	Medium density fibreboard
MtC	Megatonnes carbon
OSB	Oriented strand board
PAS	Publicly available specification
PCR	Product category rule
PEF	Product environmental footprint
PEFC	Pan-European forest certification scheme
ppm	Parts per million
RH	Relative humidity
SETAC	Society of Environmental Toxicology and Chemistry
ToSia	Total sustainability impact assessment
TAWP	Time-adjusted warming potential
UNEP	United Nations Environment Program
USGBC	United States Green Building Council
WBCSD	World Business Council for Sustainable Development
WMO	World Meteorological Organization
WRI	World Resources Institute

2 Background

2.1 Introduction

In 2012, Norway reformulated its climate policies in a parliamentary communication or white paper on Norwegian climate politics (Miljøverndepartementet (2012): Meld.St. 21 (2011-12)) Norsk Klimapolitikk), where four main tasks were emphasised:

- Norway should exceed its Kyoto-commitment by 10% within the 1st period of commitment;
- Norway has an obligation to reduce global greenhouse gas (GHG) emissions by reducing Norwegian emissions until 2020 by 30% of its 1990 emissions;
- Norway should be carbon neutral by 2050;
- Norway has an obligatory task to be carbon neutral by 2030 if other countries take up large GHG commitments.

In order to fulfil these commitments, active forest policy and 'climate friendly' building strategies are among the stated policy activities. An increased use of wood in buildings, especially in public buildings, is also established in the white papers on agriculture and food politics (Landbruks- og matdepartementet (2011) Meld. St. 9 (2011-2012)) and on building politics (Kommunal- og moderniseringsdepartementet (2012): Meld. St. 28 (2011-2012) Bygg for eit betre samfunn).

For the 5-year period (2008 – 2012) of the Kyoto-commitment, Norway had a total allocated quota of 250.6 million tonnes CO_2 eq. The total Norwegian emissions for this period, however, were at 266.5 million tonnes CO_2 eq. Norway chose to exceed the commitment for the first period by resorting to the trading of carbon credits. According to Statistics Norway, GHG emissions for 2016 were 53.4 million tonnes CO_2 eq. and 53.9 million tonnes CO_2 eq. in 2015.

Terrestrial ecosystems in Norway balance approximately 40% of the national greenhouse gas emissions (De Wit et al. 2015). The Norwegian forests represented a sink of 24.3 million tonnes CO_2 eq. in 2015 (The Norwegian Environment Agency 2017). According to the Norwegian Environment Agency, the CO_2 sink of Norwegian forests has increased from 14 million tonnes in 1990, to 32 million tonnes in 2011. The main reason for these high levels of CO_2 sequestration is due to the relatively large areas of forest with young trees. As the trees mature, the size of the CO_2 sink will fall to about 19 million tonnes of CO_2 by 2020. This trend can be reversed by increasing felling of timber and further climate mitigation benefits can be obtained by using the harvested wood in long-life products in the built environment. At the end of life of these products they can either: be cascaded down the value chain, followed by ultimate incineration with energy recovery, or incinerated with energy recovery after one life cycle. Climate change mitigation benefits are also potentially obtained by incineration of thinnings, harvesting residues and processing residues.

2.2 The Norwegian built environment

As of January 1st 2017, Norway had a population of 5 258 317, with a growth in population of 44 332 in 2016 (Statistics Norway 2017b). Norway has seen a population increase of 577 183 registered people in the ten years from 2007.

Extrapolated future projections give estimates on how the population may develop in Norway until 2100, where three scenarios (low, medium, high) are described based on different assumptions with respect to fertility, life expectancy, domestic migration, and immigration. The medium population projection is the most likely alternative, where the population continues to grow and reaches 6 million inhabitants shortly after 2030 and a population of about 8.5 million in 2100. The projection with the highest national growth gives an extrapolated number of inhabitants of over 14 million in 2100. These figures are higher than population projections made a decade ago. Bergsdal et al. (2007a) discussed population scenarios for Norway; where the extrapolated numbers for high population growth resulted in a projected population of ca. 7.5 million inhabitants in 2100.

Accurate prediction of the national population is important for planning and development of the national infrastructure, including the building sector. Population is predicted to grow mostly in central areas around Oslo, Bergen and Trondheim and the surrounding municipalities, while rural areas are likely to show a decrease in population.

According to data published in February 2017 by Statistics Norway, the building sector comprises just over 4 million buildings in total, with an increase of building stock by 28 000 buildings in 2016 (Table 1).

Table 1 Building stock in Norway by building type

Building type	2013	2017	2013-2017
Total	***4 015 718***	***4 141 421***	***125 703***
Residential buildings	1 488 979	1 534 929	45 950
Non-residential buildings	2 526 739	2 606 492	79 753
Residential buildings			
Detached house	1 143 509	1 163 426	19 917
Semi-detached house	154 092	164 114	10 022
Terraced house, linked house	151 289	163 355	12 066
Multi-dwelling building	35 437	38 922	3 485
Residence for communities	4 652	5 112	460
Non-residential buildings			
Holiday house	1 777 862	1 855 047	77 185
Industrial building	103 919	108 788	4 869
Agriculture and fishery building	506 765	501 801	-4 964
Office and business building	38 781	38 876	95
Hotel and restaurant building	10 629	11 609	980
Building for education, research, public entertainment, religion	31 388	31837	449
Hospital or institutional care building	5 630	5 659	29
Prison or emergency services building	4 795	4 879	84

Oslo and the metropolitan county Akershus had the highest building activity with 2 900 and 3 500 new dwelling units, respectively, in 2016. The high building activity is reflected in the high population increase especially for these two counties (Statistics Norway 2017).

In 2015, there was a total of 2 446 686 dwellings in Norway and 48% of all households lived in detached houses (Table 2). From 2015 to 2016, the number of dwelling units (flats) in apartment blocks had the largest increase with nearly 13 000 new units, of which 6 400 were built in the metropolitan area Oslo / Akershus.

Table 2 Dwelling units (inhabited and uninhabited) by building type and year (Statistics Norway 2016b)

Building type	2015	2016	2015 – 2016
Total	***2 446 686***	***2 476 519***	***29 833***
Single houses	1 197 046	1 204 350	7 304
Semi-detached house	211 978	214 766	2 788
Row houses, terraced house, linked house	281 826	286 178	4 352
Appartment block	557 523	570 482	12 959
Buildings for residental communities / assisted living communities	50 191	51 994	1 803
Other building types	148 122	148 749	627

In 2016, building start permits for non-residential units were given for 5.8 million m^2 of utility floor space (an increase of 12% compared with 2015). Approximately 25% of these building start permits were given to holiday houses, garages for residential buildings, while the rest (4.2 million m^2) were attributed to the industrial sector (office, industrial buildings, etc.) (Statistics Norway 2017d).

Detailed data on material statistics and material use for the different building types, residential and non-residential is lacking, but some reports give information on wood consumption for the different building types (Fossdal 1995, SFT 2001, NAL 2004, Rambøll 2012a, Sand and Stene 2016). In Norway, Sweden and Finland, the market share of timber as a construction material for single family and small houses is approximately 90% (Thelandersson et al. 2004, Rambøll 2012a). A collection of the available data is presented in Table 3.

Depending on the data source and details of the construction, the use of timber in single family and other small houses varies between 0.19 m^3/m^2 and 0.3 m^3/m^2. The wood consumption in apartment buildings (multi-storey buildings) is usually lower compared to single family houses. In Finland, the share of timber is as low as 1% in multi-storey apartment buildings (Rambøll 2012a). In bigger urban buildings, steel and concrete are the dominant building materials for load bearing constructions (Denizou et al. 2007). Other metal materials and glass are also used, often as cladding material. Wood has a small share in large load-bearing constructions in urban areas; examples are given in Table 3, for apartment blocks, but also commercial buildings.

The Norwegian Pollution Control Authority published data in 2001 on the carbon stock in buildings, where for 1998, there was estimated to be 8.19 million tonnes of carbon stored in residential buildings and for all buildings, a total carbon stock of 8.34 million tonnes. In non-residential buildings, the wood content is very low, as is the resulting carbon stock (SFT 2001).

Table 3 Wood consumption in different building types

Building type	area	Total wood use	Wood consumption		References
	[m^2]	[m^3]	[kg/m^2]	[m^3/m^2]	
Residential buildings					
Single family houses	103	20		0.19	Fossdal 1995
Small house dwellings			150	0.3	SFT 2001, NAL 2004
Apartment houses			15	0.03	SFT 2001, NAL 2004
CLT buildings				0.4	Average of several recent CLT student housing projects
Non-residential buildings					
Commercial buildings			17.5	0.035	SFT 2001, NAL 2004
Other buildings			100	0.2	SFT 2001, NAL 2004

There were changes in the technical guidelines (TEK) which were introduced in Norway in 1997 for the use of wood in constructions with more than 4 storeys. Since then, designs for multi-storey timber buildings have been developed by Norwegian industry and research facilities that fulfil the building authorities' requirements on fire safety, sound insulation, load bearing and stability/durability. However, ten years later, only a few wood dwellings had been constructed with more than 4 storeys (Denizou et al. 2007). Due to low demand, a cross-laminated timber (CLT) plant in Norway (run by Moelven) closed in 2010.

In Norway, changes in the building codes (TEK) are resulting in improved building energy efficiency, with the intention of reaching passive house level (TEK 10, 01.01.2016). A new building code (TEK 17) is supposed to enter into force as from July 2017; this however, does not include new regulations on energy use in buildings. The requirements on energy use in TEK 10, increasing requirements on environmental properties and GHG emissions during production of building materials as well as national and municipal strategies on building with wood in public sector, encourage increased use of wood in the building sector. Due to the new energy requirements, embodied energy in building materials is becoming more prominent when considering material choices.

There have been some initiatives to increase the use of timber in construction in Norway. Municipal strategies were already partly established in 2005 such as Wooden city (trebyen) in Trondheim (Kommune Trondheim 2015), Norwegian wood (2008) in Stavanger and a wood based innovation program (trebasert innovasjons program, Innovasjon Norge 2006).

Example buildings in Trondheim within the municipal strategy are (Kommune Trondheim 2015):

- Borkeplassen, 9 500 m², 2007;
- Haukåsen kindergarten, 930 m², 2013, BREEAM certified;
- Moholt student housing, 23 400 m², 2012-2016, 63 00m3 CLT;
- Svartlamoen apartment building, 1 040 m², 2005
- Åsveien school, 11 318 m², 2015, CLT/glulam;
- 4 fire stations in wood construction, 11 300 m², 2013-2015;
- 5 modular kindergartens, 3 727 m², 2007-2008;
- Nardo school and kindergarten, 6 600 m², 2008, CLT;
- Ranheimsveien assisted living, 752 m², 2010, passive house standard.

A legal restriction on multi-storey timber buildings was imposed in Sweden in 1888 and repealed in 1994 (Mahapatra and Gustavsson 2008). According to a study by Rambøll (2012a) multi-storey apartment buildings with wood construction have a market share of 10-15% in Sweden. The adoption of timber as a construction material for multi-storey buildings was slow. In 1997 the Swedish Government launched the 'Wood, Construction and Furniture Program' which funded research and development projects, as well as marketing to boost the growth of the HWP sector. This was followed by the 'Wood Cluster' program from 2002-2005 and the 'National Strategy on Wood Construction' program (2004-2008), which was launched to promote the use of timber in multi-storey constructions (Mahapatra et al. 2012). The market of apartment units in Sweden is, in comparison to Norway, more under the control of public authorities rather than the private building sector. Therefore, strategic decisions can be made for the use of wood in construction. In the public building sector in Sweden, construction of multi-use buildings increased to a share of 40% build in wood (Rambøll 2012a). However, Sweden lacks statistics on material consumption in the building market, especially for the different building types.

In 2012, Statsbygg ordered a report on decision mechanisms and how to increase the use of wood in public buildings. The report concluded that it was mainly lack of knowledge on using wood in large building projects by both contractors and knowledge gaps on handling and construction that hinder the use of wood. The low level of prefabrication and industrialised solutions on the market were a factor that negatively affected the decision makers (Rambøll 2012 b).

Changes in regulations on student housing (ICG 2015, TEK 10) opened the market for CLT and prefabricated buildings in this market. The first CLT student apartment buildings were built in 2013 in Ås (Table 4). The collaboration between project management, consultants and contractors reduced the risk of building high rise buildings with CLT and the demand rapidly increased afterwards.

Table 4 Increased demand after Palisaden student housing project in Ås 2013 (after Flindall et al. 2016)

Year	2013	2014	2015	2016	2017
student housing units constructed	200	700	1 000	2 100	4 100
student housing area total [m^2]	4 000	14 000	20 000	42 000	82 000

Assumption for calculation of built area: 20 m^2 per student housing unit

Knowledge from building high rise student housings with CLT with prefabricated solutions increased the demand in multi-storey and the public building sector. There is currently an increase in interest in CLT, glulam and prefabricated timber frame constructions. There are several smaller suppliers of CLT on the Norwegian market, although the majority is currently imported from Austria. In the beginning of 2017, three Norwegian companies (Hunton, Stangeskovene and Massiv Lust) went public introducing their plans on a new CLT production plant (skog.no, 27.01.2017) in Norway.

Table 5 Generated waste arising from construction, rehabilitation and demolition of buildings in 1000 t (Statistics Norway, 2016c)

	Construction activities (waste production in 1000 t)			
	Total	**Construction**	**Rehabilitation**	**Demolition**
2012	1 880	615	702	563
2013	1 819	621	629	570
2014	1 867	629	567	671
2014				
Wood waste	262	117	83	63
Paper and cardboard	25	14	9	2
Plastics	5	3	2	0
Glass	9	2	5	2
Metals	87	22	36	29
Gypsum	66	38	25	3
EE-waste	10	2	5	2
Hazardous waste	23	2	12	9
Hazardous waste. Asbestos	3	0	2	1
Hazardous waste. Impregnated wood	5	0	4	1
Hazardous waste. Other	5	1	4	1
Bricks, concrete, heavy building materials	777	106	202	469
Polluted bricks and concrete	22	0	6	16
Asphalt	238	181	33	24
Mixed waste	320	136	136	48
Other waste	23	5	14	4

During the 20th century, the weighted average use of concrete in new buildings increased from nearly zero to about 0.6 tonnes per m^2, whereas wood use decreased from about 0.137 to 0.11 tonnes per m^2 (Bergsdal et al. 2007a). Nonetheless, Sand and Stene (2016) concluded that the demand for wood in buildings will increase. Some analyses show possible scenarios on added value of 8 billion NOK in the wood building sector until 2030, with a consumption of 1.5 million m^3 per annum in 2015 to a consumption of a possible 3.5 million m^3 wood in new buildings in 2030. In Trøndelag this could mean an increased demand in wood harvest of 546 000 m^3 per year. Due to the already high content of wood in small family houses (single family, detached and terraced houses) the biggest potential for increased use of wood lies with buildings in the multi-occupancy, public, office and industry sector. Hovdan Molden (2011) conducted an analysis of the gap between increasing number of households and completed dwelling units. It was shown, that the gap for 2010 was especially large (>5000 dwelling units) in the metropolitan region Oslo and Akershus. Increased building activities are necessary, especially in the metropolitan areas. The Norwegian student organisation (NSO) claims in its study from 2015 on student housing that the national coverage for student housings is 14.54%. In order to reach the target of 20% coverage, more than 13 000 student housing units need to be built. In the national budget for 2016, over 700 million NOK were allocated for building 2 200 student housing units (ICG 2015). There is still a high demand for new student buildings.

In 2014 the Norwegian construction sector produced 1.87 million tonnes of waste, of which 55% was sent for recycling. Heavy building materials (mainly bricks and concrete) comprised 43% of the waste. The annual per capita consumption of mineral products in Norway amounted to 13 tonnes in 2014 (Source: Geological Survey of Norway). There has been very little research conducted into the flows of materials through the Norwegian building sector. Bergsdal et al. (2007a) presented a study on flow dynamics of the Norwegian building stock, comparing the demand for floor area to the demolition activity in an input output analysis. The scenarios show that starting in 2030, increasing demolition activities due to renewal of the old building stock will increase the demand for new dwellings drastically. This trend is similar for all population scenarios and shows the challenges and opportunities for the building market in the decades to come. By 2020, 70% of the non-hazardous waste must be recovered (Waste Framework Directive in the EU, valid in Norway under the EEA agreement). In 2014, 55% of the total waste was sent for recycling, while 31% was sent for energy recovery and 11% to landfill. Most paper, metals, glass, gypsum, brick, concrete and other heavy building materials and EE waste were sent for recovery. In July 2009, a ban was imposed on landfilling biologically degradable waste, such as paper, wood waste and food waste (Avfallsforskriften 2009).

2.3 The Norwegian forest products' sector

Norway has a total forest area of 121 908 km^2 forest, with 86 536 km^2 of productive forest and a total standing volume of more than 900 million m^3 in 2010 (Tomter and Dalen 2014).

In the period from 2011 to 2012 the annual average harvest was about 8.7 million m^3 (Tomter and Dalen 2014). In 2016, 10.345 million m^3 of wood was felled (Statistics Norway 2017a). This was the largest felling volume since the season 1989/1990. According to Granhus et al. (2014), sustainable logging can be expanded to 15 million m^3 per year. The harvested wood volume is distributed into spruce (74%), pine (24%) and broad-leaf (BL) (2%). The counties in Østlandet (Østfold, Akershus, Hedmark, Oppland, Buskerud, Vestfold and Telemark) represent 74% of the volume harvest. The three counties with the by far largest harvested volume are Hedmark (2.8 million m^3), Oppland (1.3 million m^3), and Buskerud (1 million m^3) (Table 6). The counties in Vestlandet have large areas and standing volumes of ready to harvest forests. Complicated and expensive harvesting in steep terrain, and challenging transportation logistics, however, result in relatively low harvested volumes in these areas (Table 6, Vennesland et al. 2013). The official goal is to fourfould the value creation of the Norwegian forest wood value chain sector, compared to 2012 levels (Johansen et al. 2017).

The forestry sector and the forest-based value chain, have a total number of approximately 25 000 employees and the wood mechanical industry has approximately 12 000 employees (Government 2014). The production of structural timber employs ca. 5 000 people (Trelastindustrien 2017, Statistics Norway 2015a) and approximately 3 000 employees work within the paper products sector (Statistics Norway 2017e).

Table 6 **Commercial removals of industrial roundwood, by wood species and county. [1 000 m^3] (Statistics Norway 2017a)**

	2015				2016			
	Spruce	Pine	BL	*total*	Spruce	Pine	BL	*total*
Total	7551	2441	166	*10158*	7622	2524	199	*10345*
Østfold	575	163	22	*760*	507	171	18	*696*
Akershus	570	142	12	*724*	595	152	17	*764*
Oslo	30	2	1	*33*	28	0	0	*28*
Hedmark	1890	902	83	*2875*	1857	890	98	*2845*
Oppland	1028	241	6	*1275*	1055	270	9	*1334*
Buskerud	592	395	11	*998*	595	406	23	*1024*
Vestfold	337	20	8	*365*	317	20	9	*346*
Telemark	393	210	5	*608*	394	215	8	*617*
Aust-Agder	188	168	4	*360*	196	196	4	*396*
Vest-Agder	195	61	1	*257*	209	53	1	*263*
Rogaland	119	18	0	*137*	136	18	0	*154*
Hordaland	216	12	0	*228*	309	14	0	*323*
Sogn and Fjordane	112	2	0	*114*	140	4	0	*144*
Møre and Romsdal	255	24	1	*280*	305	17	2	*324*
South-Trøndelag	380	47	3	*430*	333	51	2	*386*
North-Trøndelag	504	19	5	*528*	484	18	4	*506*
Nordland	165	2	4	*171*	157	15	2	*174*
Troms	3	6	1	*10*	2	7	1	*10*
Finnmark	-	8	-	*8*	-	7	-	*7*

BL = broad-leaved

The timber and wood industry had an added value of 10.78 billion NOK in 2016. In the timber industry in Norway, approximately 90 companies are involved. Usually, the production is located very close to the raw material, which makes them important employers in rural areas.

Table 7 **Trade of Norwegian roundwood [1000 m^3] (Statistics Norway 2017f)**

	2010	2011	2012	2013	2014	2015	2016
industrial roundwood	8.32	8.58	8.79	8.89	9.81	10.16	10.35
of which							
saw logs	4.22	4.12	4.38	4.63	5.35	5.50	5.45
pulpwood	3.74	3.91	4.19	4.20	4.00	4.13	4.39
unsorted saw logs and pulpwood	3.55	0.48	0.22	0.18	0.44	0.49	0.50

Of the Norwegian roundwood for sale on the market in 2014, 5.35 million m^3 (Table 7) went to sawmills as building material and 2.4 million m^3 sawn wood, 1.3 million m^3 dressed timber and 0.41 million m^3 impregnated wood were produced (http://www.treindustrien.no/om-treindustrien/

nokkeltall, accessed 10.02.2017). Important products in the timber industry are timber, glulam, impregnated wood, building elements, and packaging, semi-finished goods.

Secondary products from sawmills, such as sawdust, particles and clippings but also low quality roundwood are raw materials for the wood industry, and get further processed to paper and cardboard products, insulation and fibre boards as well as chemicals in bio-refineries or energy generation.

In 2015, Norway had a total use of sawn wood of 2.9 million m^3. Of the 1 million m^3 imported timber, approximately 90% was imported from Sweden, with more than half of the imported material being dressed and impregnated.

The most important export markets for Norwegian sawn wood export (2015: 565 522 m^3) are Germany, Denmark, Sweden, Belgium, Netherlands and Great Britain. With the export of processed timber, the added value remains in the Norwegian companies. The large amount of imported sawn wood, however, shows that there is potential for the production of more Norwegian produced wooden building materials.

Of the industrial roundwood harvested in Norway which was on the market in 2015, nearly 4 million m^3 of the roundwood was exported without added value. A huge potential for creating added value in Norway is lost here.

Processing waste supplies 72% of the energy production for sawmills, with the remaining 28% supplied externally (25% hydro energy and 3% fossil energy). Burning of production residues from the timber and wood industry as well as heat production of long distance heating plants generated approximately 37 000 t ash in 2014. Due to shutting down of some pulp and paper factories in 2013/2014, the share of ash from the wood industry (approximately 30 000 t ash) is assumed to be reduced by today. The amount of ash from pure wood is by about 4 000 tonnes per year (Horn et al. 2016). Earlier, ash was put to landfill, since however, ash is rich in valuable minerals, it can also be used as fertiliser. Horn et al. (2016) studied these possibilities and mapped barriers for the use of ash in forestry and agriculture. Another possibility is use as a clinker substitute in cement production.

The Norwegian forests store a considerable quantity of atmospheric carbon and the white paper on Norwegian climate politics (Meld. St. 21 (2011–2012) Norsk Klimapolitikk), emphasises the important role of forests as a tool for climate mitigation, where the protection of forests is highlighted in order to save and store carbon. However, the magnitude of the forest carbon sink will decrease as the forests mature.

The international trade in HWPs and biofuels is increasing and there is considerable potential for forest-rich countries in northern Europe to export timber products to regions that predominantly use brick or concrete construction. The competitive position of the European forest sector for timber products is rather weak due to high production costs. Exporting higher value timber products, such as modular CLT dwellings, is a potential solution to this problem. Adding value to the Norwegian timber resource is essential.

3 A review of environmental analyses and evaluations

3.1 LCA

3.1.1 Background to LCA

Life cycle assessment (LCA) is a decision support tool that has been developed in order to analyse the environmental burdens associated with the production, use and disposal of a product and is arguably the best way of quantifying this information (Hill 2011). The term product includes both goods and services. Interest in LCA grew rapidly during the 1990's and it generated high expectations, but also increasingly became the focus of criticism (Udo de Haes 1993, Ayres 1995, Ehrenfeld 1998, Krozer and Viz 1998, Finnveden 2000). However, since that time there has been considerable progress made, with the development of international standards (ISO 14040, ISO 14044). There are also several international initiatives taking place with the aim of building consensus and developing robust methodologies. These include the Life Cycle Initiative of the United Nations Environment Program (UNEP), the Society of Environmental Toxicology and Chemistry (SETAC), the European Platform for LCA of the European Commission (EPLCA) and the International Reference Life Cycle Data System (ILCD). Although a useful tool, LCA does have its limitations. There has been criticism of the practicability of the use of LCA for the construction sector, due to the lack of availability of input data and the complexity of the LCA process, resulting in a large amount of time being taken to analyse even a small percentage of the tens of thousands of construction products available (Taratini et al. 2011). The production of an LCA does involve value choices and it has to be accepted that LCA is an imperfect tool to inform decision-making processes and that other considerations may also apply when deciding on policy instruments. Werner et al. (2007) state that LCA is to some extent subjective in nature and they refer to the mental models employed by the decision maker when conducting the analysis. This process can involve:

- Making a distinction between products, co- or by-products and waste when allocating environmental burdens;
- The choice of an appropriate allocation factor;
- The selection of appropriate substitutions or additional processes if system expansion is employed to avoid allocation;
- How to handle lack of knowledge about processes when this occurs

LCA is inevitably a simplification of an extremely complex subject and it is important to realise that it does not capture all of the environmental aspects associated with the product system or process under study. For example, *'LCA may capture the global aspects of the environmental impacts by reporting impact categories such as global warming potential, ozone layer depletion, but it does not inform the analyst about more local or transient impacts and is of limited value when considering biological impacts, such as biodiversity, habitat alteration or toxicity'* (Owens 1997).

In order to conduct an LCA it is first necessary to determine the goal and scope (i.e., what is the purpose behind conducting the LCA and what is being included in the study). The scope must define what the system boundaries are in the study and the functional unit must be declared. For many purposes, the system boundary can be defined as 'cradle to gate', that is the manufacture of a specific product in a factory to the point at which it leaves the facility (modules A1-A3 in EN 15804). This gives the most accurate LCA, because this stage of a product life cycle involves the fewest assumptions and the data gathering process is relatively straightforward. However, a low impact product, as determined through a cradle to gate analysis, may prove to require a lot of maintenance during the in-service phase of the life cycle, or there may be serious environmental impacts associated with disposal. A full appreciation and understanding of the environmental impacts associated with a product choice therefore requires the whole life cycle to be considered. This invariably introduces a higher level of uncertainty into the process, because there may be aspects of the life cycle that are not well understood and this requires assumptions to be made. These assumptions may have a very significant impact upon the LCA and there may be bias introduced if comparisons are made between competing products.

Life cycle assessment is not static and there are ongoing programmes dealing with improving various aspects of this methodology (Finnveden et al. 2009). It is important that the correct decisions are made regarding the choice of materials for the built environment and LCA can be used as a means for informing those choices. This requires that LCA is used correctly and that the decision support tools allow for comparability between products (Forsberg and Malmborg 2004, Haapio and Viitaniemi 2008a, b, Ding 2008, Audenaert et al. 2012). There are several LCA-based building assessment tools available (Bribián et al. 2009), e.g.:

- ECO-QUANTUM www.ecoquantum.nl
- LEGEP www.legep.de
- EQUER www.izuba.fr
- ATHENA www.athenaSMI.ca
- OGIP www.ogip.ch/
- ECO-SOFT www.ibo.at/de/ecosoft.htm
- ENVEST 2.0 envestv2.bre.co.uk
- BECOST www.vtt.fi/rte/esitteet/ymparisto/lcahouse.html
- BEES www.bfrl.nist.gov/oae/software/bees.html
- GREENCALC www.greencalc.com
- ECOEFFECT www.ecoeffect.se
- LEGEP www.legep.de
- EQUER www.izuba.fr

3.1.2 Goal and scope definition

The goal and scope stage comprises the writing of a series of statements at the beginning of the process which tell the reader the reason why the LCA was performed, who is doing the study, who the client is and what is covered in the LCA. It is at this stage that the system boundary is defined. For example, the purpose may be to undertake an LCA of the manufacturing process only (cradle to factory gate), or the whole service life may be included. Additional parts of the lifecycle, such as recycling and disposal may also be included. The purpose of the LCA may be simply to report the environmental burdens associated with a product or process (referred to as an attributional LCA), or it may examine the consequences of changing various parameters or assuming different scenarios (called consequential LCA) (Frischknecht and Stucki 2010, Gala et al. 2015). It is also necessary to specify what the subject of the LCA is. This is referred to as the declared unit, if cradle to factory gate is being analysed, or the functional unit, if other parts of the lifecycle are also being studied. Another important consideration when studying the environmental impacts associated with a product or process is the timescale involved and it is important that this is also defined at this stage. It is also a requirement to specify what allocation procedures were used during the analysis.

3.1.3 Life cycle inventory

This phase of the analysis requires the assembly of all of the information about the process. In order to do this, an imaginary system boundary is drawn around the process and all of the material and energy inputs and outputs are quantified. This process is usually divided into the different life cycle stages, manufacture, service life, end of life, disposal. Once the life cycle inventory (LCI) phase of the analysis is complete then data gaps are identified. In some cases, it is possible to collect the missing data, but where this is not possible, 'reasonable' assumptions have to be made. During this phase, mass balance calculations are also performed. This is a very useful tool for identifying data gaps and is based upon the principle that the mass of all matter going into the system under study should equal that of all the matter exiting the system. At some stage, the data gathering process has to be terminated and the point at which this occurs is determined by cut-off criteria. Data falls into two principal categories: primary (foreground) and secondary (background) data. Primary data is that which has been gathered by the LCA practitioner and may include utility bills, delivery notes and other information that is directly linked to the process. Secondary data is that which has not been directly obtained, but is more generic in nature; for example, if wooden pallets are used to ship the product, then it is highly unlikely that a full inventory of the pallet would be made.

Ultimately, what should result from such an analysis is a table (called an input-output table) that represents flows of materials and energy to and from nature (the ecosphere). All of the foregoing is complicated enough, but if the factory in question also produces other products (co-products) then the question of allocation of the environmental burdens to the different components in the inventory to the declared unit must be considered. For example, a utility bill for a factory will give the total electricity consumption for a year, but if the factory makes ten products then a means of correctly allocating the electrical energy (and associated environmental burdens) to the analysed product must be derived. The collection and analysis of data invariably leads to issues regarding commercial confidentiality, which can cause problems, especially when the LCA has to meet adequate levels of transparency in order to be credible.

3.1.4 Life cycle impact assessment

Once the LCI phase has been completed, it is then necessary to quantify the environmental burdens, during the life cycle impact assessment (LCIA) phase. At this stage there are several further complications that have to be considered. There is still discussion as to how to do this in order to properly report the environmental burdens, but a consensus has been developing over the past decade. The principle is to aggregate the environmental implications associated with the flows to and from nature into a small (but nonetheless meaningful) set of indicators. This methodology has essentially distilled down into two main approaches, referred to as midpoint and endpoint indicators (Bare et al. 2000, Jolliet et al. 2003, 2004, Ortiz et al. 2009, Hauschild et al. 2013). In the midpoint approach, the environmental burdens are grouped into similar environmental impact categories (e.g., global warming potential, ozone layer depletion, freshwater eutrophication, etc.). The endpoint approach seeks to model the chain of cause and effect to the point of the evaluation of damage, which makes for simpler reporting with fewer indicators, but has a much higher level of uncertainty. Midpoint is preferred because of the higher level of accuracy, but can be more difficult to interpret (Dong et al. 2014). Endpoint impact categories are reported in terms of impact on human health (e.g., DALY, disability adjusted life years), or on ecosystems (e.g., species loss). Some systems have even gone so far as to aggregate all of the impacts into one category (e.g., ecopoints), but the data reported using this approach has very high uncertainties associated with them. The environmental impacts are calculated using a variety of models (over 150) which attempt to determine the impacts of processes upon the environment. Examples of such models include:

- Midpoint: TRACI, CML, EDIP, Ecopoints;
- Endpoint: Eco-indicator, LIME2;
- combined midpoint and endpoint: ReCiPe (Bare et al. 2000), IMPACT 2002+ (Jolliet et al. 2003).

In IMPACT 2002+ the 'value' of the environmental impact is reported as an ecoindicator and measured in environmental points. The accumulated ecoindicator is composed of damage categories (human health, ecosystem quality, climate change, resources) and impact categories (carcinogens, non-carcinogens, respiratory inorganics, respiratory organics, ionising radiation, ozone layer depletion, aquatic ecotoxicity, terrestrial ecotoxicity, terrestrial acidification, land occupation, aquatic acidification, aquatic eutrophication, global warming potential, non-renewable energy, mineral extraction). This requires a weighting process to be applied, which is reliant upon value judgement.

The impact categories selected should provide useful information about the product or process, taking the goal and scope of the study into consideration. When selecting the impact categories, it is also necessary to select the characterisation factors, which are the units used to report the environmental burden. To consider the example of the climate change impact category, the characterisation factor for this is global warming potential with a 100-year timeframe (GWP100) and the characterisation factor for this is kg CO2 equivalents. The method used to calculate impacts can affect the results of the LCA study and this should always be remembered when making comparisons between products or materials in different studies (Monteiro and Freire 2012). Bueno et al. (2016) compared five different life cycle impact assessment methods (EDP 97/2003 (midpoint), CML 2001 (midpoint), Impact 2002+ (midpoint and endpoint), ReCiPe (midpoint and endpoint) and ILCD recommended practices for LCIA (midpoint)) for consistency of results. The two endpoint methods gave different answers. The midpoint methods gave consistent results for the impact categories: Aquatic and Freshwater Ecotoxicity, Ionising Radiation, Particulate Matter Formation, and Resource Depletion. Global Warming, Terrestrial Ecotoxicity, Human Toxicity (except for the Non-carcinogens impact category) and Land Use (except for Natural Land Transformation), but not for Ozone Layer Depletion, Photochemical Oxidant Formation, Acidification, Terrestrial and Aquatic Eutrophication, Marine Ecotoxicity and Water Depletion. Lasvaux et al. (2015) compared the Ecoinvent database and the Environmental Product Declaration (EPD) database in France. The environmental impacts of 28 building materials were compared using the Life Cycle Impact Assessment Indicators (LCIA) of EN 15804, for the cradle to gate part of the life cycle. The results obtained for the impact categories related to fossil fuel consumption, such as abiotic depletion potential, GWP and primary energy demand showed differences of less than 25% between the two databases, but other indicators showed much higher deviations (sometimes by more than 100%). They recommended that for some impact categories mixing LCA databases is not appropriate, but that for the main indicators used by the building sector (GWP and embodied energy) the information was reasonably comparable between the two databases.

Another important factor is the correct allocation of environmental burdens to different co-products, if the system under analysis produces more than one product. Examples of this include the allocations between cereal and straw, or meat and wool in agricultural production systems (Brankatschk and Finkbeiner 2014). Ideally, allocation should be avoided when possible, but in many cases this cannot be done and a choice has to be made regarding the allocation procedure used. Various approaches can be used for allocating environmental burdens, including mass, energy, or economic allocation. Guidance regarding allocation is given in ISO 14040 and ISO 14044, recommending a hierarchy of choice for allocation methods. In many cases, economic allocation is used, which gives a more realistic allocation of the burdens. This is because economic activity is the primary motivation for the manufacturing of products and this allocates the highest environmental burdens to the highest value products.

Jungmeier et al. (2002a) identified ten different processes in the forestry value chain where allocation issues can occur: forestry, sawmill, wood industry, pulp and paper industry, particle board industry, recycling of paper, recycling of wood-based boards, recycling of waste wood, combined heat and power production, landfill. These can be divided into multi-output processes (e.g., sawmill) or multi-input processes (e.g., landfill). A forest can produce wood for a variety of uses, including: solid wood, particle board, paper pulp, plywood, biomass for fuel. The question then arises how to allocate the environmental burdens associated with the forestry and harvesting operations to the different outputs. One way of dealing with this is to employ system expansion so that all of the different product streams are included within the same system boundary. The problem with this approach is that the functional unit that is now considered may not be very useful. For comparison purposes, the wood-based functional unit must be the same as the non-wood-based functional unit. If a timber frame building is manufactured with the result that the waste from the process is used for energy production, then it is possible to use system expansion to compare the functional unit as being the structural frame plus the production of x kWh of energy. However, if the wood waste or by-products go to the production of chipboard or paper, then the comparison becomes more difficult. It is almost inevitable that some form of allocation will have to be employed. In many cases an economic allocation may be the best way of allocating burdens, but prices can fluctuate. Furthermore, the forest can produce different product streams at different times (first thinnings, second thinnings, third thinnings, harvest) which adds to the problem of economic allocation over a time scale that can be as long as a century (Jungmeier et al. 2002b).

At the end of the LCA process, there are additional analyses that can be performed, these are normalisation, grouping (aggregation), or weighting. These are usually used to make the environmental information more understandable (Chau et al. 2015).

Normalisation is the calculation of an environmental impact relative to some reference data, in order to give some context to the information. An example of this would be comparing the carbon dioxide emissions of a process with that of an average European citizen for one year. Although this can give information regarding the importance of a particular environmental impact, uncertainties in the characterisation factors can lead to uncertainties in the results.

Grouping (aggregation) involves combining different impact categories into a few or even one. An example would be the combining of the global warming impacts due to the emissions of different greenhouse gases and reporting this in one impact category. In this case, the commonly used unit is carbon dioxide equivalents over a 100 year' time frame (GWP_{100} in kg CO_2 eq.). In this example, the science is extremely well understood, due to the huge scientific effort that has gone into researching climate change, but for other impact categories there is much greater uncertainty and debate. The challenge with LCA is to use enough impact categories to make the analysis meaningful, but not so many that it makes the LCA only of interest to a very small number of academics. It is also possible (although extremely unreliable) to group everything into one impact category, as is done with systems, such as the BRE Green Book, BREEAM, LEED, Cradle to Cradle, etc., which is a huge oversimplification of what is a complex subject. For example, if a reduction of global warming potential occurs at the expense of a huge increase in ozone depletion, then this is of dubious benefit. However, if a large reduction in global warming results in a modest increase in ozone layer depletion, then this may be a sensible choice to make. The question is how to balance a decrease in one impact category against an increase in environmental burdens elsewhere. Grouping therefore requires the assigning of different weighting factors in order of a real (or perceived) impact, or importance. The environmental impacts may be quite different globally and locally (Khasreen et al. 2009). For example, global warming and ozone layer depletion are global impacts with long time scales, whereas eutrophication tends to be much more localised and with shorter time scales. The relative importance of these impacts is therefore very different depending on the perspective of the analysis (Yang 2016).

Weighting is a process which has to be performed before the indicator results of different impact categories are combined into a single score. This is reliable when based upon strong scientific evidence (e.g. GWP, or ozone depletion), but more often it involves value judgements to be made regarding the relative importance of different impact categories. This becomes increasingly unreliable as more impact categories are included and extremely unreliable when impact categories from different disciplines (environmental, economic, social) are combined into one overall impact category to give a measure of the 'sustainability' of a process or product. This single indicator approach makes the route by which the score was obtained non-transparent and can be subject to manipulation or lobbying. There is no consensus regarding the methodology of the weighting process, meaning that different schemes are incompatible. Ultimately, it would be desirable if the outputs from LCA could be converted into a monetary value, but this is a long way from being realised (Pizzol et al. 2015).

As noted previously, the science underpinning the relationship between the release of a substance into the environment and its impact is better understood for some impact categories than it is for others. In 2011 the European Commission Joint Research Centre (EC-JRC) Institute for Environment and Sustainability published the International Reference Life Cycle Data System (ILCD) Handbook. This examined 14 impact categories at midpoint level and three at the endpoint level. Only the IPCC method for climate change and the World Meteorological Organization method for ozone depletion (midpoint) are characterised as Level I (recommended and satisfactory) (Finkbeiner 2014). However, the EC-JRC is currently in the process of updating ILCD recommendations for 4 impact categories (water depletion, resource depletion, land use and respiratory organics).

Another topic that requires consideration is the use of databases and LCA software calculation tools. In principle, an LCA would have to include all of the aspects which are related to the system under consideration. For example, pallet use and transport thereof may form part of the overall analysis. There are a very large number of contributions that would have to be in turn subjected to an LCA, with the chain of cause and effect becoming very large and complex. In order to make an LCA manageable, considerable use is made of databases; the most common of which are Ecoinvent and GaBi (in Europe). From these, the practitioner can look up the environmental impacts associated with pallet use and transport and many other products and services. One of the questions to be asked is: does the database that is used affect the results? This was the question behind the study conducted by Herrmann and Moltesen (2014) on the LCA calculation tools SimaPro (which uses the Ecoinvent database) and GaBi. They found that in many cases the answers from these two databases were the same or closely similar, but in some situations very different answers were obtained. The unit processes for which different results were obtained were: 'particles > 10 µm to air', 'sulphate to air' 'niobium 95 to air', 'thorium 230 to air', 'thorium 232 to air' and 'zirconium 95 to air'. When calculating the impacts based on the Soybean Brazil Farm unit process, differences were found in the results for all impact categories, except acidification and terrestrial ecotoxicity. A very large difference of 91.6 was found for the impact category photochemical ozone formation (human exposure) and 2.8 for (GWP) global warming potential. In an LCA study of biodiesel production, differences were found in GWP and aquatic eutrophication and in a study of palm oil production there were differences in land-use change impact. This is a potential confounding factor when attempting to compare different LCA studies. The authors emphasise that they only examined about 1% of the full population of the data in the two LCA software systems and about 10% of the LCIA data was examined. There is a need for more work on the comparability of results when different databases are used to source secondary data.

3.1.5 Embodied energy

Buildings and building materials are responsible for the consumption of nearly 40% of global energy (Dixit et al. 2012). The total life cycle energy use of a building comprises the operational (or direct) and embodied (or indirect) energy. The sum of these two (direct and indirect) can be considered to be the embodied energy of the building, but it is useful to report them separately. Embodied energy can be subdivided into initial embodied energy (the energy used for resource extraction, processing, transport and manufacture of the product) and recurring embodied energy (energy used for maintenance or replacement). Definitions of embodied energy can vary from study to study. The embodied energy of the materials in a building seldom exceeds more that 30% of the total energy demand (embodied plus operational energy), unless the building is deliberately designed to be a low energy structure. End-of-life energy requirements, demolition and disposal also make a very small proportion of the energy use of conventional houses (Winistorfer et al. 2005). As building energy efficiency increases, then the material embodied energy and energy associated with demolition and disposal will assume greater importance.

However, on a global or national scale, the materials' embodied energy can be much higher than 30% of the total energy demand of the building sector, due to a growing population and hence a growing demand for buildings (Treloar et al. 2001). This is an important point.

Pauliuk et al. (2013) modelled the transformation of the complete Norwegian building stock by 2050 to passive house standard and showed that this full transformation was not sufficient to achieve the required emission reductions to reach the 2 degrees Celsius climate target in 2050. Additional energy efficiency and life style changes in the non-heating part of the building sector, however, show a reduction potential of 75%.

The embodied energy of a material or product used in the building is often defined as the primary energy used in the manufacture, which includes all of the energy used in the production, as well as the primary energy used in the transport of materials and goods required for the production process. This definition relates to the initial embodied energy, which is related to the cradle to factory gate stage (modules A1-A3, EN 15804) of the product life cycle. In some definitions, the transport to building site (A4) and the energy used on site for the erection or installation of the product (A5) is also included. The units used are generally MJ per unit mass, or volume, or per defined functional unit, although some workers report this as kWh (=3.6 MJ). Transport of materials to the building site can have a major impact on the embodied energy of the construction materials. Morel et al. (2001) reported that the amount of energy used to manufacture and transport building materials represented nearly 8% of primary energy consumption in the UK. They showed that by using local materials, it was possible to reduce the embodied energy associated with a building by up to 215% and the impact of transportation by 435%.

In some LCAs, the energy used for the maintenance of the product is also included, although this should be reported separately as the recurring embodied energy. This is distinct from the initial embodied energy, which is constant once the product is manufactured and installed (Ramesh et al. 2010, Chau et al. 2015). Some workers also include the energy associated with the disposal of a product at the end of a life cycle, although this should be reported separately. The embodied energy is invariably reported according to the cumulative energy demand (CED) method, which states that the embodied energy is assessed as the primary energy used for the manufacture, use and disposal of an economic good (product or service), or which may be attributed to it with justification. The method distinguishes between non-renewable and renewable energy use. The cumulative energy demand (CED) represents the primary energy used (both direct and indirect) during the life cycle of a product (Huijbregts et al. 2006). This includes the energy consumed during the extraction, manufacturing and the disposal of the product and raw and auxiliary materials. Different methods for determining the primary energy demand exist. For example, the lower or higher heating values of primary energy sources may be used, the use of renewable energy resources may not be included or it may be reported separately. Huijbregts et al. (2006) found a good correlation between fossil CED and GWP and resource depletion, but the correlations with acidification, eutrophication, tropospheric ozone formation, stratospheric ozone depletion and human toxicity were much lower and for land-use they were absent.

Dixit et al. (2012) noted that some research workers do not include renewable energy in their definition of embodied energy and also found that the use of different information sources and failure to distinguish between primary or secondary energy could lead to errors as high as 40% when reporting embodied energy. They stated that there is a need to develop a common methodology to accurately determine the embodied energy associated with buildings and that there is a need to develop a complete and robust database of embodied energy information. Chambers and Mueke (2010) noted that a table of the embodied energy of materials was available from Architecture 2030, although this no longer appears to be available. There is also the widely-used University of Bath Inventory of Carbon and Energy database, which is widely used (e.g., Lee et al. 2011). However, these may not necessarily be reliable sources of information. For example, the Bath ICE database has been shown to inaccurately report data for harvested wood products (Hill and Dibdiakova 2016). Cabeza et al. (2013) and Jiao et al. (2012) note that there is a relationship between embodied energy and GWP for primary production, for some building components and that there is a link between embodied energy and cost of buildings, which is related to the energy intensity per unit GDP for that country.

It is necessary to define the meaning of primary energy, since it is not always clear that the primary energy has been used when the embodied energy is reported. The primary energy is defined as the energy measured at the natural resource level. This is the energy used to produce the end-use energy which includes the energy used in the extraction, transformation and distribution to the user (Fay et al. 2000). Measurements of embodied energy are only consistent if they are based upon primary energy but if delivered energy is used, the results are misleading. Unfortunately, there is a lack of clarity and incomparability in the reporting of embodied energy (Dixit et al. 2012). The current standards do not provide complete guidance and do not address important issues. For example, EN 15804 does not mention embodied energy, although it does require the reporting of energy inputs as primary energy and requires the reporting of the following categories describing resource use:

- Use of renewable primary energy excluding renewable primary energy resources used as raw materials;
- Use of non-renewable primary energy excluding non-renewable primary energy resources used as raw materials.

It is important to distinguish between embodied energy, which is associated with the production of a good or service and the inherent (or embedded) energy, which is a physical property of the material. The terms embodied and embedded are sometimes confused in the literature. As noted previously, the embodied energy of a material is the primary energy that is associated with the extraction, processing and transportation of that material from the cradle to the factory gate. In contrast, the embedded energy of a material is a property of that material and can be directly measured. For example, the inherent energy in a wood product can be recovered at the end of its life cycle by incineration, whereas the inherent energy of concrete is zero. The inherent (embedded) energy can be reported in EN 15804 in the following categories:

- Use of renewable primary energy resources used as raw materials;
- Use of non-renewable primary energy resources used as raw materials.

However, different LCA practitioners report data for these categories in different ways. In addition, the inherent energy is reported as primary energy in these categories, which does not necessarily represent the true value of the recoverable energy, which is usually more accurately reported for wood as the lower heating value (LHV).

3.1.6 Environmental product declarations

LCA can be a useful tool when applied to a specific product or process in order to determine where the highest environmental burdens (hotspots) occur. This attributional form of LCA can be used to identify where to improve the process to reduce the overall environmental burden of the product. Consequential LCA can be used to determine the environmental impacts arising due to changes to the production process.

However, the use of LCA to compare between different materials (such as concrete or timber in construction) is much more problematic and the use of LCA for this purpose requires several criteria to be fulfilled:

- The functional unit should be the same;
- The whole lifecycle of the material or product should be considered and there should be reasonable and realistic assumptions (e.g., about recycling);
- Reasonable scenarios about maintenance and replacement must be included;
- The databases and environmental impact calculation methods used should be stated and be comparable;
- The methodologies and inventories should be transparent (often not possible due to commercial confidentiality;
- Reasonable cut-offs should be used and justified with a sensitivity analysis;
- The impact categories used should be reliable and meaningful;
- A sensitivity analysis should be used to demonstrate the impacts of different assumptions.

In order to develop a framework that allows for comparability of environmental performance between products, ISO 14025 was introduced. This describes the procedures required to produce Type III environmental declarations. This is based on the principle of developing product category rules (PCR) which specify how the information from an LCA is to be used to produce an environmental product declaration (EPD) (Bergman and Taylor 2011). A PCR will typically specify what the functional unit is for the product. Within the framework of ISO 14025, only the production phase (cradle to gate) of the lifecycle has to be included in the EPD, but it is also possible to include other lifecycle stages, such as the in-service stage and the end of life stage, although this is not compulsory. ISO 14025 also gives guidance on the process of managing an EPD programme. This requires programme operators to set up a scheme for the publication of a PCR under the guidance of general programme instructions. There have been other standards issued that apply to the construction sector in order to ensure greater comparability of the environmental performance of products. ISO 21930 gave some guidance on both PCR and EPD development, but this was recently replaced in Europe by EN 15804, which is a core PCR for building products and it is therefore considerably more detailed and prescriptive than ISO 14025 (ISO 21930 is currently being revised).

The primary purpose of an EPD according to ISO 14025 is for business to business (b2b) communication, but an EPD can also be used for business to consumer (b2c) communication. In the latter case, there are further requirements upon the process, which apply especially to the verification procedures. In any case, ISO 14025 encourages those involved in the production of an EPD to take account of the level of awareness of the target audience. Standards are increasingly removing the flexibility (and uncertainty) that was once associated with determining the environmental performance of products and services. This should, in principle, make it much easier to compare the environmental impacts of products within a product category in the future.

The life cycle stages are the succeeding and interlinked stages of a product system from raw material to disposal and can be divided into:

- Upstream processes: involving the extraction of raw materials and transport thereof to the manufacturing facilities;
- Core processes: manufacture of the analysed product, maintenance of manufacturing infrastructure, packaging, disposal of waste;
- Downstream processes: transportation from manufacturing to construction sites, construction, maintenance, reuse, recycling, recovery, disposal.

Table 8 Different life cycle stages defined in EN 15804

Module	Life cycle stage	Description
A1	Production	Raw material supply
A2	Production	Transport
A3	Production	Manufacturing
A4	Construction	Transport
A5	Construction	Construction/installation
B1	Use	Use
B2	Use	Maintenance
B3	Use	Repair
B4	Use	Replacement
B5	Use	Refurbishment
B6	Use	Operational energy use
B7	Use	Operational water use
C1	End of life	De-construction/demolition
C2	End of life	Transport
C3	End of life	Waste processing
C4	End of life	Disposal
D	Beyond building life cycle	Reuse/recovery/recycling

The different life cycle stages are divided into modules in EN15804. Modules A1-A3 cover the production stage, A4-A5 the construction process, B1-B7 the use stage and C1-C4 the end of life stage; beyond this is the 'after-life' stage (D). These are listed in Table 8. The publication of this standard ensures harmonisation of core PCRs for building products in Europe. It is mandatory to report stages A1-A3, with the other stages optional being included for any reporting beyond cradle to factory gate.

PCRs have been developed by different organisations which have set up EPD programmes (examples in Europe include EPD Norge, the International EPD System in Sweden and the Institut Bauen und Umwelt in Germany). Since the introduction of ISO 14025, there has been a proliferation of EPD systems, with their own PCRs. ISO 14025 encourages the operators of EPD programmes to harmonise their methods and PCRs and in Europe this has resulted in the creation of 'ECO' a platform for rationalising EPDs, involving 11 EPD operators within Europe. This involves mutual recognition of

EPDs, and the creation of common PCRs, working from agreed core PCRs (such as EN 15804 in the built environment).

Table 9 Different environmental impact categories according to EN 15804

Impact category	Parameter	Unit
Global warming	Global warming potential (GWP)	kg CO_2 eq.
Stratospheric ozone depletion	Ozone depletion potential (ODP)	kg CFC 11 eq.
Acidification of soil and water	Acidification potential (AP)	kg SO_2 eq.
Eutrophication	Eutrophication potential (EP)	kg $(PO_4)^{3-}$ eq.
Photochemical ozone creation	Formation potential of tropospheric ozone (POCP)	kg ethane eq.
Depletion of abiotic resources-elements	Non-fossil resources abiotic depletion potential (ADP-elements)	kg antimony eq.
Depletion of abiotic resources-fossil fuels	Fossil resources abiotic depletion potential (ADP-fossil fuels)	MJ

In theory, the introduction of EPDs which use common PCRs means that it should be possible to compare different building materials in terms of environmental impact (environmental impact categories are presented in table 9). However, while it may be possible to make choices based upon the environmental impacts associated with the manufacture of products, the use phase (module B1-B7) and end of life phase (modules C1-C4, D) also need to be considered in order to get the whole picture. Important considerations when examining the environmental consequences of the use of different materials must include the service life of the product, maintenance requirements and performance in service, especially with respect to the impact on the operating energy of the building. This can involve assumptions being made regarding life span, maintenance, end of life scenarios, etc., which will have a critical impact upon the outcome of the LCA. Although the introduction of Type III environmental declarations theoretically allows for environmental performance comparisons to be made between different products and materials, this may not always be possible in practice.

Gelowitz and McArthur (2017) conducted a review of published EPDs for building products and came to the following conclusions:

- Discrepancies between life cycle inventory methodology, environmental indicators and life cycle inventory databases were a barrier to making comparisons between EPDs;
- There was a high level of incomparability between EPDs using the same PCR, which was unexpected and should not occur;
- There was evidence of poor verification practices, demonstrated by a high proportion of EPDs containing contradictory data;
- The EN 15804 harmonisation standard has not been entirely successful. The proportion of valid comparisons was much higher with EN 15804-compliant EPDs, but the overall level of comparability was still low.

The objective of environmental labels and declarations is to provide transparent, accurate and verifiable information on the environmental performance of goods and services, with the objective of stimulating continuous market-driven environmental improvement (ISO 14020). The international standard ISO 14024 defines Type I environmental labels, which are certificates (ecolabels) that are issued by an independent, third party verification body. Examples of Type I ecolabels include single-attribute labels about wood sourced from forests that are managed sustainably (e.g., FSC, PEFC) Cobut et al. 2013) and there are many examples listed on the ecolabel website. Type II environmental labels are defined in ISO 14021; these are self-declared environmental labels. Examples include statements regarding recyclability, compostability, etc.

3.1.7 Product environmental footprints

The EU Commission published Product Environmental Footprint (PEF) methods as part of the Communication 'Building the Single Market for Green Products' (2013/179/EU). This builds on existing LCA methodologies and aims to harmonise them. This is with the intention of allowing for greater comparability between products and services by defining methods, thereby reducing flexibility. This is precisely why EPDs were introduced and the need for yet another method of comparing the environmental footprints of products has been questioned (Finkbeiner 2013).

3.1.8 End of life and multiple lives

An often overlooked or improperly considered aspect of LCA is the inclusion of a realistic end of life scenario. For a solid timber product this could mean recycling of the wood into another product with similar utility, or more likely downcycling to a composite wood product, disposal to landfill and incineration with or without energy recovery. The time-frame over which the product lifetime is considered can have an important impact upon the results of the LCA. It is also important to consider whether the assumptions that are used for an end of life scenario are realistic. If a material is recycled, then it may be legitimate to claim credits for the displaced production of what would otherwise be production of virgin material, but where the associated credits or burdens are allocated will depend upon whether the second life of the material/product is included in the LCA or not. Double-counting can occur if this is not properly managed.

It is important to distinguish between open-loop and closed-loop recycling. When a material associated with a product is used again in a product with the inherent properties of the material conserved, then this is closed-loop recycling. Note that this does not mean that the material has to be recycled in the same or a similar product system. Dubreuil et al. (2010) give the example of nickel used to make aircraft turbines. At the end of life, the scrap metal from the turbines can be added with carbon steel scrap to manufacture stainless steel, which displaces the requirement to produce primary nickel. From the perspective of the nickel, this is closed-loop because the nickel has utility in the next product.

Open-loop recycling occurs when a material is used in a product but has undergone a decrease in its properties. An example of this is the tin in steel cans, which has an anti-corrosion function. But when the steel scrap from the cans is melted the tin goes into solution, which is still present in the steel when it solidifies, but does not fulfil any protection role. The tin can be considered a tramp element (not needed for improving the quality of the steel). This is an open-loop recycling of the tin (Dubreuil et al. 2010).

Materials such as metals can often be recycled to produce products of equal utility to those of the previous life, but other materials (timber, concrete) can only be recycled with an associated reduction in performance. This is referred to as downcycling. There also needs to be consideration given to the ultimate fate of materials. Can they really be recycled indefinitely, or is it reasonable to only consider a 'reasonable' lifetime for the scenario (100 years is often the default choice). Recycling should not be the first resort when considering the next life of a material and re-use or recovery generally have much lower associated environmental burdens. For a building this means that it may have several different functions during the lifetime of the structure.

Recovery involves the salvaging of component parts from a product, to be used again in other products. This requires higher energy inputs compared with re-use, but less than recycling. Recovery involves a fundamental change in the way that products are designed. They have to be built with ease of disassembly in mind, rather than ease of production. This concept can add to the costs of production, making a product economically uncompetitive unless all manufacturers adopt the same approach. It can also restrict or even prevent innovation, since an overemphasis on the re-use of recovered parts may limit redesign, which could otherwise lead to efficiency gains elsewhere in a production process. Similar arguments can also apply to re-use. There is a balance that has to be achieved between a reduction in environmental impact due to life extension compared with the reduction obtained with a more efficient design. The answers to questions of this nature are not always obvious and require careful analysis. In a built environment context, this would mean that the building is disassembled rather than demolished and the components (e.g., a steel frame) would be used to make another building.

These different end-of-life scenarios have to be incorporated into the LCA if the whole life cycle is to be analysed, but they should be reasonable and realistic. The inclusion of end of life stages can be used to reduce the burdens of the manufacturing stage, by distributing these impacts over a much longer time period. There is still no common agreed way of dealing with allocation issues when considering recycling in LCA (Schrijvers et al. 2016).

3.2 Measuring the sustainability of buildings

3.2.1 Building assessment schemes

There are a large number of schemes available for reporting on the environmental sustainability of buildings. This makes comparisons extremely difficult and there is a requirement for standards to be introduced which result in a harmonisation of methods.

The International Organisation for Standardisation (ISO) has been developing standardised methods for the environmental assessment of buildings. ISO Technical Committee (TC) 59 'Building Construction' and its Sub-committee (SC) 17 'Sustainability in building construction' have published two technical specifications:

- ISO/TS 21929-1:2006 sustainability in building construction – Part 1: Framework for the development of indicators for ;
- ISO/TS 21931-1:2006 sustainability in building construction – framework for methods of assessment for environmental performance of construction works – Part 1: Buildings.

Building sustainability is investigated by the European Committee for Standardisation (CEN) through the activities of CEN/TC 350 'Sustainability of construction works', which has three working groups developing standards related to the sustainability of buildings:

- WG1 environmental performance of buildings;
- WG2 building life cycle description;
- WG3 product level.

Since the sustainability of a building is determined at a building level, this requires the product information to be available in a harmonised format. CEN has developed several European Standards for determining the sustainability of buildings:

- EN 15643-1 – Sustainability assessment of buildings – Part 1: general framework;
- EN 15643-2 – Sustainability assessment of buildings – Part 2: framework for the assessment of environmental performance;
- EN 15804 – Environmental product declarations – core rules for the product category of construction products;

- EN 15942 – Environmental product declarations – communication format – business to business;
- EN 15978 - Assessment of environmental performance of buildings – calculation method.

Prior to the introduction of these standards, there were many schemes developed for measuring the sustainability of buildings. Azhar et al. (2011) and Lee (2013) list some of the rating systems that are used around the world to report the environmental performance of buildings (Table 10).

Table 10 Examples of environmental rating systems for buildings

Rating system	Country
Green Star	Australia
LEED Canada	Canada
DGNB Certification System	Germany
IGBC Rating System	India
LEED India	India
Comprehensive Assessment System for Building Environmental Efficiency	Japan
Green Star NZ	New Zealand
Green Star SA	South Africa
BREEAM	UK
LEED	USA
Building Environmental Assessment Method Plus (BEAM Plus)	Hong Kong
Evaluation Standard for Green Building (ESGB)	China

Other examples include: GB Tool, ESCALE, CASBEE, GPR Gebouw, Nature Plus, ECOProfil, Code for Sustainable Homes, Nordic Ecolabelling – Svanen.

The LEED system (Leadership in Energy and Environmental design) was developed by the US Green Building Council (USGBC) in 1998 and is currently the most commonly used method to measure the environmental performance of a building. In order to obtain LEED certification, a building must obtain a certain number of points. Depending upon the type of construction or renovation being undertaken, one of a range of LEED rating systems will apply, each of which has a slightly different weighting of points. LEED (version 4) credits are divided into eight categories:

- Location and transportation;
- Sustainable sites;
- Water efficiency;
- Energy and atmosphere;
- Materials and resources;
- Indoor environmental quality;
- Innovation in design;
- Regional priority;

Under this system, a building can earn up to 45 credits, with 110 points available in total, since in some categories multiple points can be earned for higher environmental performance. For example, for waste material diversion, one credit can be earned for recycling/salvaging 50% of waste material from the site with an extra credit awarded for 75% (Kucukvar et al. 2016). In each section of the LEED system there are certain prerequisites that must be met, even though they do not count towards a building's total points. The building is then awarded LEED certification according to the following scores (Table 11):

Table 11 LEED categories

LEED category	Points
Certified	40-49
Silver	50-59
Gold	60-79
Platinum	80-110

Within the section entitled 'Materials and Resources' there are 13 points available, with two points assigned for the provision of EPDs and two points allocated to material ingredients. The LCA aspects of the material composition of a building therefore only contribute, at most, 4 out of 110 points, in terms of considering GWP and EE footprints. By comparison, 'access to quality transport' (for example) contributes 5 points. Points are given for the use of recycled materials, which implies that it is only the first user that is responsible for the environmental impact of a material and that it is burden free thereafter, or that the impacts associated with recycling are lower than those for primary production (Dubreuil et al. 2010).

It has to be concluded that the environmental footprint of the material composition of the building is given an unjustifiably low level of importance in this scheme. The weighting of the different factors in the credits is based on value judgements and has very little scientific basis. This is a scheme that is designed to encourage best practice in building design and management, but material choice has a very small role to play.

BREEAM (Building Research Establishment Environmental Assessment Method) was first introduced in 1993 and uses a similar methodology as LEED. The assessment is made in 10 different categories (with each category divided into a range of issues):

- Energy;
- Health and well-being;
- Innovation;
- Land use;
- Materials;
- Management;
- Pollution;
- Transport;
- Waste;
- Water.

When a threshold is reached, then credits are awarded. Within the materials' category the aim is 'to recognise and encourage the use of construction materials with a low environmental impact (including embodied carbon) over the full life cycle of the building'. The points score in this category is based upon the BRE Green Guide, but it is permissible to use independently verified EPDs. Where such information is available, it is a requirement that the lifecycle GHG emissions (kg CO_2 eq.) should be reported. The Green Guide gives an environmental impact rating for a material or product on a scale from A+ (best environmental performance) to E (worst). This is a measure of the overall environmental impacts covering the following categories:

- Climate change;
- Water extraction;
- Mineral resource extraction;
- Stratospheric ozone depletion;
- Human toxicity;
- Ecotoxicity to freshwater;
- Nuclear waste (higher level);
- Ecotoxicity to land;
- Waste disposal;
- Fossil fuel depletion;
- Eutrophication;
- Photochemical ozone creation;
- Acidification;

This aggregation of impacts requires value judgements to be made about the relative importance of each impact. LCA of the building materials comprises 1.9% of the total rating, making the selection of materials almost irrelevant.

The current tools for reporting on the sustainability of buildings in Norway include Statsbyggs Klimagassregnskap (carbon calculator by the Norwegian Public Construction and Property Management) which is limited to determining greenhouse gas (GHG) emissions, and BREEAM NOR, the Norwegian version of BREEAM.

There have been attempts to incorporate LCA or LEED data into building information modelling (BIM) software, in order to allow engineers and architects determine the environmental impacts of different building choices at the design stage (Wang et al. 2005, Azhar et al. 2011, Basbagill et al. 2013, Jalaei and Jrade 2015). In the work conducted by Basbagill et al. (2013), only GWP impact was considered, although the option of including other impact categories was allowed. The GWP data was obtained from the Athena EcoCalculator. The problem of including more than one impact category is that there then has to be a decision made about the relative importance of the different impacts; an issue that has not been satisfactorily resolved. Basbagill et al. (2013) proposed a BIM-enabled decision support system to help designers determine the environmental impact at the design stage. However, although a good idea in principle, any such system is only as good as the information held in the underlying database and potentially open to influence and manipulation by competing commercial interests.

3.2.2 Building assessment schemes reviewed in the literature

LEED, BREEAM and the other building evaluation schemes have been reviewed and compared by Lee (2013), Nguyen and Altan (2011), Forsberg and von Malmborg (2004) and Erlandsson and Borg (2003). Haapio and Viitaniemi (2008a) have pointed out that comparisons between building environmental assessment are difficult, if not impossible, to make. They compared 16 assessment tools: ATHENATM Environmental Impact Estimator, Building Environmental Assessment Tool (BEAT), BeCost, Building for Environment and Economic Sustainability (BEES), BREEAM, EcoEffect, Eco-profile, Eco-Quantum, Envest, Environmental Status Model, WQUER, ESCALE, LEGEP, LEED, PAPOOSE and TEAM. Alshamrani et al. (2014) integrated an LCA model with LEED in order to achieve a sustainability score for school buildings. The LCA was conducted using an ATHENA impact estimator. The results compared concrete, masonry and steel-masonry buildings. Collinge et al. (2015) compared LCA and LEED in the assessment of building materials (carpets and roof membranes) and identified shortcomings in both approaches to determining environmental impact. Lee (2013) compared five assessment schemes (BREEAM, LEED, CASBEE, BEAM Plus, ESGB), concluding that BREEAM and LEED were the most comprehensive, based on the number of assessment criteria. Chambers and Muecke (2010) showed that the selection of biobased building products results in high scores within the LEED system.

There have been attempts to develop multi-criteria decision-making tools in order to guide product choice for building procurement (e.g., Lipušček et al. 2010). Unfortunately, all such tools are inevitably very complex, they are only as accurate as the underlying databases and they rely on weighting schemes that are based on value judgement. The complexity of such tools means that the way in which the decisions are arrived at is not transparent and consequently will lead to conflicts and lobbying when one product is favoured over another.

Danatzko et al. (2013) concluded that there is at present not any design tool that is sophisticated enough to be used at the whole building level to assist in making decisions regarding design alternatives to ensure the most sustainable form of construction.

3.2.3 Alternative approaches to measuring environmental impact

The remit of this report was to cover published life cycle assessments of construction materials. However, there has been activity covering the wider aspects of sustainability, although the subject is very far from being mature. The problem with any attempted method to measure the sustainability of a process or product is that sustainability is a concept and the definitions can mean different things depending on the perspective of the analyst (Hill 2011). In order to combine many different impact categories into a number or an index (good/bad, gold/silver, excellent/satisfactory) the evaluator has to employ non-scientific value judgements regarding the relative importance of the different impacts. The relative importance depends upon the perspective of the analyst, the time-frame considered, the geographical scale and many other considerations. This is a huge (and complex) subject area and only a few examples are given here.

In order to visualise the environmental impact of building materials in terms of monetary units (eco costs), Carreras et al. (2016) developed a model based upon multi-objective optimisation (MOO) techniques. The idea was to incorporate the environmental impact into an economic assessment. The authors noted that previous attempts to convert environmental impact into economic units used one of two approaches:

- Damage-based;
- Prevention-based.

In the damage-based approach, a monetary cost is applied at the life cycle impact assessment stage, which expresses the willingness of people to pay in order to compensate for an impact. In the prevention-based approach (also known as marginal abatement cost), the damage cost is dependent upon policy targets set by government related to an environmental problem. The method adopted by Carreras et al. (2016) was based upon a prevention method, but used the concept of a carrying capacity rather than a policy target.

In the EU project EFORWOOD, an attempt was made to develop a total sustainability impact assessment (ToSia) tool. This incorporated 22 impact assessment indicators and 9 qualitative indicators, reflecting economic, social and environmental impacts. As ever, the challenge with such an over-arching tool is to decide on a weighting factor for each of the indicators chosen (Pülzl et al. 2012). There is no consensus as to how this can be achieved for such a complex issue as measuring sustainability impact.

Wu et al. (2014a) examined five different carbon labelling schemes (Singapore Green Labelling Scheme, Carbon Free (the U.S.), CO_2 Measured Label and Reducing CO_2 Label (UK), Carbon-Counted (Canada), and the Hong Kong Carbon Labelling Scheme) and concluded that all of the schemes had unsatisfactory transparency issues, at least partially related to the allowed flexibility in defining the system boundary.

The ecological footprint is a simple way for representing human demand upon the ecosystem services of the planet. It is the area of land and sea required to regenerate the resources appropriated by humanity and to sustainably detoxify or otherwise immobilise the wastes generated. This is the method used to estimate the number of Earth's required if all of humanity adopted a given lifestyle. The concept of the ecological footprint was developed by Professor William Rees of the University of British Columbia in Canada (Rees 1992, Wackernagel and Rees 1996, Rees 2006). The footprint represents the amount of ecosystem carrying capacity of the planet that is appropriated for human needs. This concept is similar to LCA, except rather than reporting the environmental impact using a range of impact indicators, this complex subject is now reduced to one all-encompassing index. The environmental burden on ecosystem carrying capacity is calculated for the wastes (such as carbon dioxide, nitrates and phosphates) for which an assimilation area can be estimated. In the case of CO_2 it represents the area of land required to sequester this gas in a growing forest. Some types of products, such as cement and minerals, have footprints calculated on the basis of their CO_2 emissions, plus any land areas associated with the mining activities. Some classes of wastes, such as ozone depleting chemicals cannot be assigned a theoretical land area at all. Urbanised or otherwise non-productive (from the point of view of resources) land is measured directly. Based upon calculations of a global environmental footprint of 2.2 ha per capita and an estimate of 1.8 ha available per person in 2001, Rees (2006) stated that humanity had already overshot the carrying capacity of the planet by 20%, meaning that we are now drawing down on the environmental capital of the planet and not simply living off the interest. As with all methods concerned with calculating environmental impacts, ecological footprint analysis has attracted criticism either regarding the methodology employed or about its relevance to policy (e.g., Van den Bergh and Verbruggen 1999, Fiala 2008). The methodology is an over-simplification of a complex situation, but the concept of the environmental footprint does have the ability to capture the public's imagination.

4 The environmental impacts of the building sector

According to the Intergovernmental Panel on Climate Change (IPCC), the building sector is responsible for 40% of primary energy demand and 36% of energy related CO_2 emissions in the industrialised countries. However, this figure only includes operational energy and does not take account of the embodied energy and emissions associated with the materials used in construction. However, it also has the greatest potential of any sector for the implementation of low energy (and low carbon) measures through improved insulation, heating, air conditioning, lighting and ventilation measures. According to the International Energy Agency, the building sector has one of the lowest GHG mitigation costs. Moreover, by using timber in construction it is possible to make buildings stores for atmospheric carbon (Lehmann and Fitzgerald 2012, Kremer and Symmons 2015). There is also considerable potential for reducing the energy associated with the production, maintenance and disposal of construction materials. Concerns about climate change and geopolitical concerns regarding energy sources are behind the introduction of the EU Building Energy Directive, which is being implemented in member states.

Globally, the construction sector is responsible of 60% of all extractions from the lithosphere, with buildings responsible for 40% of this volume (Broun and Menzies 2011). Bribián et al. (2011) note that mineral extractions for the building sector per capita in Europe amount to 4.8 tonnes per person per year. In Spain, every square metre of a habitable building requires a total of 2.3 tonnes of more than 100 different materials. If the material intensity per unit service (total resources necessary to produce the materials) is examined, then this amounts to 6 tonnes per square metre. The sector is also responsible for the generation of large amounts of waste, with Europe producing 850 million tonnes of construction-related waste every year (De Schepper et al. 2014).

In a survey of 60 LCA studies of buildings in 9 countries, Sartori and Hestnes (2007) found that the proportion of embodied energy in the materials compared with the overall energy (embodied plus operating energy) over the lifetime of the building varied between 9-46% in well-insulated buildings and 2-38% in conventional buildings. The building lifetime for many studies was considered to be 50 years. Other investigations have shown that the embodied energy in the materials used in buildings mainly in Northern and Central Europe is about 10-20% of the total energy used (Kotaji et al. 2003). The assumed lifetimes of structures vary between country and building type. In the Netherlands, 75 years for dwellings and 20 years for offices is assumed, in the UK it is 60 years for both building types, whereas in Switzerland and Finland it is 100 years and 80 years, respectively.

Apart from the materials used in construction, it is also important to be aware of the waste that is generated during this process. Monahan and Powell (2011) reported that in the UK construction sector, contingency related over-ordering amounted to about 10% of materials transported to site and that 10-15% of materials transported to site were subsequently exported as waste. This unplanned waste can be substantially reduced by increasing the use of modular off-site construction, a technology for which timber is well suited.

Although waste is produced at the start of a building life, most is generated at the end. Norway produced approximately 1.25 million tonnes of demolition waste per year in 2002 (Bergsdal et al. 2007b) and (Bohne et al. 2008) showed that this would increase with time. According to statistics Norway, the amount of construction waste generated in Norway in 2014 was 1.9 million tonnes, of which 0.7 million tonnes was demolition waste. Just over 0.6 million tonnes was construction waste and the remainder was waste from rehabilitation work. The average amount of demolition waste generated in Spain is approximately 750 kg per person per year (González-Fonteboa and Martínez-Abella 2008). Thormark (2000, 2002, 2006) points out that it is important to consider the recycling and re-use potential of materials used in construction and not just the immediate impacts associated with their manufacture. The European Waste Framework directive (2008/98/EC) provides a framework for changes towards a European recycling society. Through this directive Norway is supposed to recover and reuse 70% (by weight) of its non–hazardous construction and demolition waste by 2020.

Optis and Wild (2010) studied 20 journal articles describing LCA energy studies of buildings finding that documentation was omitted and that calculation procedures were not stated in the majority of articles, although data sources were generally well-referenced. The embodied energy varied between 2-51% of the total life cycle energy associated with the building. It was concluded that the articles were of limited use for comparing the life cycle energy efficiency of buildings. They commented that the high level of detail required in order to ensure that an LCA is transparent, is not compatible with the length of a paper required for publication in a scientific journal. Chau et al. (2015) studied the environmental impact of buildings using LCA, life cycle energy assessment and life cycle carbon emissions and reviewed previous findings using these methods. They found that discrepancies arose between the three methods and they identified limitations in the use of these methods as decision tools for building design.

Although there have been some studies of the embodied energy and embodied carbon associated with construction, Monahan and Powell (2011) noted that most of these were not comparable, lacked sufficient detail to make comparisons and had inconsistent system boundaries. However, some consistent studies were found in the literature. Hammond and Jones (2008) reported an average of 5.3 GJ/m^2 and 403 kg CO_2 eq. per m^2 embodied carbon in a study of 14 predominantly UK-based buildings. Asif et al. (2007) reported 3.25 GJ/m^2. Other studies have also reported the embodied energy for the construction of office buildings as 8-12 GJ/m^2, with GWP impacts of 750-1140 kg/m^2 (Oka et al.1993). Nässén et al. (2007) compared the results of 20 (mainly Scandinavian) studies published before 2001. The embodied energies were in the range 1.3-7.3 GJ/m^2 primary energy. They found that there was almost a 90% higher energy use (GJ/m^2) when a top down analysis was compared with previous bottom-up analyses. Of this, only 20% of the discrepancy was due to the embodied energy of the materials, the rest being due to differences in energy in construction, transport, etc. This discrepancy was considered to be of less importance where materials choices were concerned. Langston and Langston (2008) investigated 30 case studies of building construction in Melbourne and found that there was a very strong correlation between capital cost and embodied energy, at a building level, but this correlation was very weak when examined at a materials level.

The LCA impacts from the materials used in a building assume greater importance as the energy efficiency of the building is improved (Blengini et al. 2010). The question then arises: ‘is the energy spent on efficiency measures recovered during the lifetime of the building?’ (Dutil and Rousse 2012).

Cole and Kernan (1996) showed that the operating energy of a building was a much more important factor than the embodied energy of the materials contained therein and that it was a better strategy to concentrate on the energy efficiency of the building before turning to embodied energy considerations. However, as the operating energy of the building sector decreases, it is necessary to pay more attention to the energy and carbon emissions associated with the constituent building materials (Dahlstrøm et al. 2012). It is also important to be aware that the implementation of energy-efficient measures may unintentionally lead to materials choices with higher embodied energy in manufacture, or use-phase.

5 The role of biogenic carbon in mitigation

5.1 Carbon and climate change

Anthropogenic greenhouse gas (GHG) emissions are making a substantial contribution to climate change (Intergovernmental Panel on Climate Change - IPCC 2007). Since pre-industrial times (before 1750) the concentration of carbon dioxide in the atmosphere has risen from a baseline level of 280 ppm (parts per million) to about 380 ppm at present. When the global warming effects of the other GHGs (primarily methane and nitrous oxide) are also taken into account, the level is around 430 ppm of carbon dioxide equivalents (CO_2 eq.). Levels of greenhouse gases are presently higher than they have been for any time in the past 650 000 years. These contributions to the increase in atmospheric CO_2 concentration since the industrial revolution come mainly from the combustion of fossil fuels, gas flaring and emissions associated with cement production. Other sources include deforestation, land use change and biomass burning (contributing about 20%) (IPCC 2007). Although the atmospheric GHG levels continue to increase, there are various natural processes by which atmospheric carbon dioxide is removed from the atmosphere. These are:

- Photosynthetic production of biomass (terrestrial and aquatic);
- Weathering of silicate rocks;
- Dissolution in the oceans.

The removal of atmospheric carbon dioxide by these sinks is calculated by the Bern carbon cycle model. If all human additions of carbon dioxide to the atmosphere were to cease immediately, the atmospheric concentration would gradually return to the pre-industrial levels. About 50% of the increase above the background level of 280 ppm would be removed in 30 years. This is assuming that anthropogenic interference in the climate does not lead to irreversible effects, such as melting of methane clathrates, or oxidation of peat.

Forests in the Northern Hemisphere provide a significant carbon dioxide sink. Over 80% of this sink occurs in only one third of the forest, associated with abandoned agricultural land and managed forests (Goodale et al. 2002). The net carbon balance in forests is dominated by two processes: carbon uptake during net primary production and carbon release during decomposition (heterotrophic respiration). Additional releases can occur due to natural or anthropogenic disturbance (Kurz et al. 2013). At a stand level, carbon dioxide will be released into the atmosphere as a result of harvesting activities and it takes a certain amount of time before that released carbon dioxide is recaptured through the growth of new biomass (Cherubini et al. 2011a, Bentsen 2017). Understanding this behaviour requires a dynamic accounting of biogenic carbon flows (Levasseur et al. 2010, 2013, Cherubini et al. 2011a, Brandão et al. 2013, De Rosa et al. 2017).

5.2 Biogenic carbon and forestry

According to the Kyoto protocol, national reporting on GHG emissions in the Land Use, Land Use Change and Forestry (LULUCF) sector includes an assessment of carbon sequestration by forests, which is one of the ecosystem services of forestry. The net sequestration of a forest is often reported as a carbon balance, which is the difference between sequestered and released carbon. A positive balance means that the forest is a carbon sink, whereas a negative balance means that the forest is a net carbon source. This can be confusing since in LCA, sequestered carbon is reported as a negative value. Forest carbon flows differ depending upon whether they are studied at the tree, stand, or forest level. The carbon balance also depends upon the forest management practice adopted and the rotation lengths. The management of carbon in the biosphere differs from fossil carbon management in that carbon can both be emitted from and sequestered to the biosphere. Whether there is a net radiative forcing, cooling or equilibrium depends on the balance and timing of the release and sequestration of the biogenic carbon.

The amount of carbon stored in the living biomass of the planet amounts to 600-1 000 gigatonnes of carbon, with something of the order of 1,200 Gt being locked up in dead biomass. Most of the carbon of the terrestrial biosphere is stored in forests, which contain about 86% of the above-ground biogenic carbon and 73% of the carbon stored in the soil. Boreal forests store about one third of the global terrestrial carbon (Pan et al. 2011, Hovi et al. 2016). Due to the activities of humanity, carbon stocks in global forests are decreasing by 1.1 Gt per year. However, in most of the countries of Europe, the forest utilisation rate (fellings as a percentage of the annual increment) is less than 100%, meaning that the carbon pool in European forests is increasing in size, with the sink being about 365 million tonnes of sequestered CO_2 per year; equivalent to 7% of annual EU emissions (Pilli et al. 2016). There are several reasons why the European forests are currently accumulating biomass (Linder and Karjalainen 2007):

- The forests are currently characterised by an uneven age class distribution, with fast growing young and medium age class stands dominating. This will continue to be the situation for the next 10-30 years;
- Growth rates of trees in European forests have increased in the late twentieth century, mainly due to nitrogen deposition;
- Many regions in Europe have a low utilisation intensity; harvest rates are significantly lower than increment;
- The forested area is slowly increasing due to abandonment of agricultural land;
- Forest soils are accumulating carbon because of recovery from unsustainable management practices, such as litter raking.

Carbon mitigation strategies for the forest and forest products' sector can be grouped into the following categories:

- Conservation management: The aim here is to protect existing pools to reduce emissions. This includes reducing deforestation and disturbance and increasing the area of unmanaged forests;
- Sequestration management: This involves increasing the productivity of forests to increase storage in forests, combined with harvesting and utilisation of timber in long life products;
- Substitution management: Including measures to substitute fossil fuels in energy production and replacing high embodied energy and embodied GHG materials in long life products.

Conservation management is the best short-term mitigation strategy in regions with large carbon pools in old-growth forests, such as Amazonia, Africa, Indonesia. The means by which such 'set-aside' is to be successfully achieved in these forests is outside the scope of this report. However, conservation management is not the best option for managed forests in Europe. In European forests, this would mean setting aside previously managed forests, which is also a short-term strategy because the mitigation potential will eventually saturate. In addition, unmanaged old-growth forests are more susceptible to natural disturbances, such as storm or fire damage. Sequestration management strategies involve an increase in the carbon pool and/or the rate of carbon sequestration. These are medium-term strategies because it takes time for the full potential to be realised. For example, if agricultural land is converted to forestry there may be a period of time while the stand acts as a net source of carbon dioxide, until the above ground biomass accumulation exceeds the oxidation rate in the organic matter in the soil. As with conservation management, sequestration management offers a one-time mitigation strategy, because eventually all the expanded carbon pools will saturate. Substitution management offers long-term mitigation potential because the substitutions for fossil fuels and high embodied energy/GHG materials can take place permanently. Such strategies can involve materials' cascading combined with incineration with energy recovery at the end of multiple lives.

With the current rate of timber harvesting in Europe, managed forests will move into older age classes and the net increment of wood material will consequently decline (Nabuurs et al. 2002, Karjalainen et al. 2002). This provides an opportunity for increasing fellings in order to improve the carbon sequestration potential of these forests. This argument does not apply to old growth oak forests, where it has been shown that these forests continue to sequester carbon dioxide for a considerable time (Luyssaert et al. 2008). There is also evidence showing that 'primary' or 'frontier' forests hold more carbon in above- and below-ground biomass compared with plantation forests (Newell and Vos 2012). Continued, or even intensified, harvesting of managed forests will ensure that old growth forests can be dedicated to conservation measures. The best approach will prove to be a 'mixed' strategy, where some areas are optimised for timber production and others for biodiversity and amenity benefits (Triviño et al. 2016). A model of forest management and wood use developed by Werner et al. (2010) showed that in order to achieve the maximum climate change mitigation potential it was necessary to:

- Maximise the sustainable increment, taking biodiversity considerations into account, as well as soil quality;
- Harvest the increment continuously;
- Use the wood in a material cascade – use in long-life products initially (preferably in construction);
- Waste wood that is not suitable for further use should be used to generate energy.

Furthermore, it was found that forest management strategies to enhance carbon sequestration in the forests are ineffective because of an increase in consumption in fossil fuels to generate the energy not produced by biomass and to produce high embodied energy materials in place of timber. To ensure that all aspects of the forest carbon cycle are covered, it is necessary to consider the forest itself, the harvested wood products and the substitution effects (Lippke et al. 2011, Härtl et al. 2016).

The utilisation of HWPs in long-life products also allows for the carbon storage benefits of timber to be extended beyond the forest (Gustavsson et al. 2006a, Liu and Han 2009). The benefits of using HWPs are not just limited to carbon storage but also because they can substitute for materials which have a higher embodied energy and/or for fossil fuels used in the production of energy. These substitution effects may have a greater impact in climate change mitigation compared with the carbon storage benefits, but can be more difficult to determine (Marland et al. 1997, Miner and Perez-Garcia 2007, Matsumoto et al. 2016). Intensively managing forests for production has been shown to be the preferable option from the point of view of climate change mitigation. The biggest impact from this management strategy comes from the use of the biomass to substitute for fossil fuels and energy intensive materials (Poudel et al. 2012).

Gustavsson et al. (2017) showed that taking managed forests out of production and storing carbon in the forest resulted in greater emissions of GHGs to the atmosphere compared to active forest management and efficient HWP utilisation. Knauf et al. (2015) also showed that harvesting combined with the storage of carbon in wood products gave a greater benefit in terms of GHG mitigation, but this depended upon the time-scale of the study. It was important to include the effects of substitution both of materials and fossil fuels in the models. An earlier modelling study by Thornley and Cannell (2000) had shown that greater carbon storage was achieved in an undisturbed forest compared to any harvesting scenario, but in that analysis wood products were assumed to have a half-life of only 20 years. In a study of Canadian forests, a strategy of increased harvesting and thereby increasing the HWP pool was found to be the best option from the point of view of climate change mitigation (Smyth et al. 2014).

Taeroe et al. (2017) modelled the carbon fluxes between different pools for forestry energy and HWP chains for a 200-year period. They investigated three management scenarios: a traditionally managed beech forest (business as usual case), an energy poplar plantation, a set-aside forest left unmanaged for the long-term storage of carbon. Managed forests were found to have a greater climate change mitigation potential compared to unmanaged forests. However, the magnitude of the carbon savings that result from the different management practices depend on the reference fossil fuel, the material alternatives to wood, forest site growth rates, GHG emissions associated with the forest, HWP production chain emissions and energy conversion efficiencies. Triviño et al. (2016) also studied different management scenarios and concluded that optimising management practices in order to obtain the highest revenue from timber did not result in the highest levels of carbon storage. The carbon storage was estimated as the amount in the living wood (roots, stem, branches, twigs, foliage), dead wood, extracted timber (thinnings and clear-cuts) and residuals left after harvesting. The models were run for a 50-year period. However, the model did not take account of the effect of material substitution or account of the substitution of fossil fuels for energy production.

If a managed forest is taken out of production the forest biomass will continue increasing until the stand is mature. At this stage, a dynamic equilibrium is reached where the mortality and growth rate are in balance and the carbon stock is constant (ignoring natural disturbances such as storm damage and fires) (Lippke et al. 2011). The same will happen to the soil carbon, with an increase to a steady state level. A report by Flugsrud et al. (2016) recently discussed whether protection of forests (no harvest) or the use of wood (sustainable harvest) as a renewable and sustainable resource has a larger effect as a carbon sink. They concluded that by just saving forests from harvest (and using them as for carbon storage and as a sink) calamities such as forest fires, insect attack and diseases are a permanent risk for the carbon storage of the forest. Sustainable harvest and use of wood in products with a long lifetime such as building components or as substitutes for products and energy derived from fossil resources is a better strategy from the perspective of climate change mitigation.

If forests are not harvested, no forest products will be produced, which would require their replacement with more energy and carbon intensive materials. In addition, the potential for energy production by using the by-products of harvesting and processing and wood waste at the end of life cycle, is lost. The use of biomass in the built environment represents a stable and easily accountable way of storing atmospheric carbon for long periods of time, creating a new carbon pool. Furthermore, the substitution of other building materials which often have a higher carbon footprint brings additional benefits. Koch (1992) analysed the implications of a reduction of timber harvests from US forests. It was concluded that if non-wood products were used in construction instead of structural timber then overall CO_2 emissions would rise significantly. The main factors when considering the carbon balance of a forest are (Pukkala 2014):

- Change in carbon storage of living biomass (above- and below-ground);
- Change in carbon storage of dead organic matter;
- Change in carbon storage of HWPs;
- Harvesting, transportation and manufacturing emissions;
- Substitution effects of primary use;
- Substitution effects of recycling and reuse.

There are two ways on measuring terrestrial carbon losses or accumulation:

- Measuring stock changes;
- Measuring incoming or outgoing fluxes.

Forest inventory (a stock change approach) is the default method for reporting forest carbon sink data when reporting GHG information for the UNFCCC. For western European countries, these inventories are carried out every 10-15 years across over 4 million sample plots (Linder and Karajalainen 2007). This data only reports on the above ground biomass.

Nabuurs et al. (1997) estimated that the carbon sequestered each year (in the mid 1980's) by European forests (101.3 million tonnes per year) was equivalent to 9.5% of European emissions. This is somewhat higher than the 39-58 million tonnes of carbon per year estimated by Kauppi and Tomppo (1993). Nabuurs et al. (1997) also estimated that the amount of carbon entering the HWP pool was 29.2 million tonnes per year, although they did not give an estimate of the quantity leaving this pool. Terrestrial ecosystems in Norway balance approximately 40% of the national greenhouse gas emissions (De Wit et al. 2015). The Norwegian forests represented a sink of 24.3 million tonnes CO_2 eq. in 2015. According to the National Inventory Report of the Norwegian Environment Agency, the national GHG emissions were 53.9 million tonnes CO_2 eq. (The Norwegian Environment Agency 2017).

The change in carbon balance in the forestry sector requires knowledge regarding changes in the living biomass, dead organic matter and HWP pools. The amount of carbon stored in HWPs has been variously estimated in several studies. Dias et al. (2012) found that in some years more carbon flowed into the HWP pool than was accumulated in Portuguese forests, because of natural disturbances such as forest fires. A study using HWP models estimated that HWPs stored the equivalent of only 1% of GHG emissions in the EU15 (Kohlmaier et al. 2007). Similar studies have reported that this figure is 12% (Pussinen et al. 1997), or 4% (Eggers 2002, Karjalainen et al. 2003) in Finland, 6% in Europe (Eggers 2002, Karjalainen et al. 2003) and 7% in France (Fortin et al. 2014). However, this amount represents a stock, not a flow and although a significant quantity of carbon it is only important to monitor the flow (stock change) to see if this has a negative or positive impact in terms of radiative forcing. Several studies have indicated that carbon storage in the global HWP pool has been increasing (Winjum et al. 1998, Hashimoto et al. 2002, Kohlmaier et al. 2007, 2008). Pussinen et al. (1997) modelled the balance in carbon stocks between forest and HWPs over a simulation running from 1990 to 2100. They showed that it is possible to simultaneously increase the carbon stocks in both pools, but found that after 40 years the forest carbon started to decrease as a result of climate change induced temperature increases.

The HWP pool is not a sink and eventually the sequestered carbon will exit the pool with different potential fates. Oxidation will return the sequestered carbon to the atmosphere as carbon dioxide, whereas disposal to landfill will result in a proportion of the biomass being converted to methane, with the residual remaining in the ground. The amount of sequestered carbon residing in the HWP pool depends upon the product lifetime, which varies considerably depending upon application and ultimate fate. In a study of HWPs in the US, Ingerson (2011) showed that only 1% of the original sequestered carbon was still residing in the HWP pool after 100 years, whereas 13% was stored in landfill. She also claimed that in some circumstances the amount of carbon stored in long life products (measured as CO_2 eq.) may equal the emissions due to transport and harvesting of HWPs. The approach that she adopted in this analysis was to use a 'snapshot' taken at 100 years after harvest. Such an approach does not take account of the dynamic nature of the flow of carbon into and out of the harvested wood products' pool and does not give any credits for the storage of carbon in the pool. Although the potential climate change mitigation benefits of material substitution and fuel substitution by forest products were mentioned, there were not factored in to the calculations. In a study of carbon stored in forest products in Finland, Karjalainen et al. (1994) calculated that the proportion of the original carbon stored in products and landfills after 100 years was 33%. The C storage was sensitive to the landfill decay rate and to the burning of end-of-life products for energy, but the most important factor was the lifetime of products.

5.3 Soil organic carbon

Forest soils contain a large stock of carbon, some of which has a slow build up and a long turnover time. Boreal forests have accumulated large stocks of carbon in the soil and litter because low temperatures limit decomposition of organic matter (Kurz et al. 2013) and they store a large proportion of their organic carbon (around 20%) in the forest litter, which is vulnerable to wildfire disturbances (Pan et al. 2011).

Carbon release back to the atmosphere can occur due to anthropogenic or natural ecosystem disturbance. Studies of such effects in boreal and northern temperate forests do not yet provide enough information about the long-term impacts on forest ecosystems. Effects of disturbance at a landscape level are less dramatic compared to a stand level, since they are averaged over a much wider spatial distribution in the former. Early reviews of soil carbon storage in forests often yielded conflicting results, mainly due to key differences in the experimental methods adopted (Johnson 1992, Johnson and Curtis 2001, Jandl et al. 2006). There are also differences which can occur due to site type, location and management. For example, a study of carbon fluxes using the eddy-covariance technique at Douglas fir clear-cut stands found that there were significant differences at sites located only 3 km apart. Higher carbon emissions after clear-felling were associated with greater quantities of decomposing organic matter from leftover logging residues. This result showed that great caution was required when extrapolating carbon flux results from a single site (Paul-Limoges et al. 2015).

At a forest level compared to a stand level, the dynamics will look quite different and it is important to consider the whole soil, standing biomass and harvested wood products system and also the wider forest rather than the stand level. Egnell et al. (2015) noted that the effect of silvicultural management on forest carbon balances could not be based upon single pool changes (i.e., in the trees or in the soil), but had to consider the whole system. The question is whether harvesting activity will result in any measurable change in soil organic carbon over time, or will this be obscured by fluctuations due to natural disturbance.

Dean et al. (2017) noted that there was little data on the impacts of logging on soil organic carbon and that most such studies were over short time frames. In a review of the literature, they found that, as a result of conventional harvesting, there was no change (Achat et al. 2015a, Hoover and Heath 2015), a significant drop (Nave et al. 2010, Zummo and Friedland 2011, Petrenko and Friedland 2015, Vario et al. 2014, Noormets et al. 2015), a significant increase (Johnson and Curtis 2001), or no consistent trend (Nave et al. 2010, Clarke et al. 2015) in the levels of soil organic carbon in the forest. Dean et al. (2017) found that over the long term (300 years) there is a slight drop in organic carbon with each harvest, but that this is only statistically perceptible over a time frame of several hundred years. They noted that from a GHG accounting perspective, forest soil organic carbon levels are not the whole story and that forest products which reach landfill could attenuate losses from soil organic carbon. The use of whole tree harvesting silvicultural practices can reduce the input of organic carbon and nutrients into the soil (Smolander et al. 2008) and leads to a lower volume increment of trees in subsequent harvests (Helmisaari et al. 2011), although the response was observed to be weaker in Scots pine compared to Norway spruce stands (Smolander et al. 2013). The decrease in volume increment arising from whole tree harvesting can be compensated for by the application of fertilisers (Jacobson et al. 2000), although this leads to an increase in environmental burdens (Johnson et al. 2005, Klein et al. 2015). The draining of peaty soils for the establishment of production forestry can result in the release of carbon dioxide from buried organic matter to the atmosphere, due to increased aerobic respiration in the soil. This can be compensated for over time by increased above-ground biomass production combined with accumulation of forest litter at some sites (Krüger et al. 2016).

Clark et al. (2015) reviewed studies of a range of different silvicultural management strategies and their impact upon forest soil carbon stocks in boreal and northern temperate forests. They found that conifers tend to store more soil organic carbon as surface litter, whereas broadleaves store more carbon in mineral soil. During whole tree harvesting, current guidelines suggest that 30-40% of the biomass should be left on site. Good practice measures require that soil organic carbon stocks are to be protected during forest operations, but provide little direct guidance on how this is to be achieved. This is partly due to diverging results from different experiments, linked to the complexity of the processes involved and the difficulties of making measurements. In Nordic studies of tree thinning of Norway spruce and Scots pine, the removal of harvesting residues led to a long-term decrease in tree volume increment (Jacobson et al. 2000, Helmisaari et al. 2011, Kaarakka et al. 2014, 2016) and early thinning reduced above ground carbon storage (Nilsen and Strand 2008). However, the overall response of the forest ecosystem to residue removal seems to be related to site-specific effects, as some sites showed little change (Brais et al. 2002, Johnson et al. 2002, Smolander et al. 2013). Some authors have stated that there are no consistent effects resulting from the removal of harvesting residues from boreal forests (Thiffault et al. 2011). Studies of the assimilation of coarse wood debris into forest soils have shown that this is a complex phenomenon that is, as yet, imperfectly understood (Magnússon et al. 2016). Nave et al. (2010) concluded that clear cutting does result in a reduction of soil carbon, after conducting a meta-analysis of 432 data sets. Achat et al. (2015b) also concluded that removing harvesting residues reduced soil nutrients and subsequent tree growth. Differences in soil carbon and biomass carbon (including roots) have been noted between even-aged and uneven-aged stands of Norway spruce (Nilsen and Strand 2013). Liski et al. (1998) studied a 5000-year soil chrono-sequence and found that forest fires resulted in a decrease in soil organic carbon, but so did long term harvesting. Wäldchen et al. (2013) were not able to detect any significant differences in soil organic carbon stocks linked to management history of sites going back over 200 years. However, chronosequence data has to be interpreted with care, especially if the history of the site is not accurately documented (Yanai et al. 2003). Puhlick et al. (2016) showed that clear-felling resulted in the lowest C-stocks in both the above- and below ground C pools and that even when the C stored in the HWP pool was included, the total carbon stored was less than that where selective harvesting regimes were applied. However, the forest studied (Penobscot, Maine) was a mixed hardwood softwood forest, with a diverse species mix and results from this study may not be applicable to a boreal forest with relatively low tree species diversity. According to a model developed by Poudel et al. (2012) for Swedish forests, managing forests intensively for production resulted in an increase in carbon stocks in tree living biomass, forest soil and HWP pools.

A comprehensive review of the impact of whole tree harvesting upon soil productivity in boreal and temperate forests was conducted by Thiffault et al. (2011). The review covered a large range of climates, soils, forest sites and silvicultural methods. The reviewed studies showed that the impacts of biomass harvesting on nutrients (nitrogen, phosphorus and base cations) and soil pH were more often observed in the forest floor litter rather than in mineral soils. The reduced levels of nitrogen and phosphorus due to whole tree harvesting resulted in reduced tree growth with some stands. There were not found to be any consistent effects of whole tree harvesting on soil productivity. It was concluded that there needed to be long-term experiments that follow stand development through a rotation, in order to establish the identification of categories of site or stand conditions under which the negative impacts of whole tree harvesting would be likely to occur. Although soil compaction by machine traffic is to be avoided for a number of reasons, it was found to have little effect upon carbon dioxide or methane fluxes compared to a control plot (Epron et al. 2016). The effects of climate change will also have to be taken into account when modelling the changes in soil carbon, because of increased microbial respiration activity leading to loss of soil carbon arising from rising global temperatures due to climate change (Ågren and Hyvönen 2003).

5.4 Standing biomass

In plantation forests, stem wood is usually the main carbon sink in the ecosystem (Gielen et al. 2013). Increasing the rotation length can result in an increase in carbon sequestration in stands which are managed for production rather than wood quality. These stands are felled at the time of maximum annual increment, whereas stands managed for wood quality have much longer rotation periods (Kaipainen et al. 2004). At a stand level there can be a large temporal imbalance or carbon debt between biomass removal and regrowth over long rotations, which can have a significant impact upon considerations regarding the use of virgin biomass for energy compared to fossil fuel alternatives (Pingoud et al. 2012). Liski et al. (2001) found that choosing a rotation length to maximise overall carbon sequestration in the forest was not straightforward and depended upon the species and the annual increment. Shortening the rotation length from 90 years to the age of culmination of mean annual increment (66 years for Scots pine and 49 years for Norway spruce) decreased the above ground biomass carbon stock, but increased the soil carbon levels, due to litter fall and leaving harvesting residues. This effect was stronger with Norway spruce sites. Shortening the rotation times increased the amount of harvested roundwood at both types of site, but this was in the form of pulpwood and hence went to short-term products, especially with the Norway spruce. This decreased life span of products reduced the benefits of short rotation lengths and is an important consideration in such a study. Also, a shortening of the rotation length in order to bind more carbon in the forest and wood products was found to be of little benefit because of the additional carbon released due to the increased wood processing within the model time-frame. Part of the assumption made was that (with shorter rotations) there would be additional use made of wood for pulp and paper products, which has a higher carbon footprint than use for construction timber. The outcome would probably be different if the assumption was that thinnings were used to substitute for fossil fuels in energy production. At Scots pine sites, an increase in rotation length (from 90 to 120 years) was beneficial from the point of view of carbon sequestration. However, there was a financial penalty for the forest owners. Pingoud et al. (2010) found that an increase in rotation length provided GHG benefits because more of the timber could then be converted to sawlogs and hence to long-life timber products. At the landscape level, this would increase carbon sequestration in the forests, but would simultaneously decrease wood supply. As a result, lower substitution benefits would apply for the time of transition. The conversion of more timber to sawlogs would result in greater substitution benefits because of the generally lower embodied energy associated with the production of solid timber products compared with board or paper products, as well as the longer life generally associated with solid wood products.

Kilpeläinen et al. (2014) noted that many studies failed to take account of the dynamic nature of carbon exchanges with forests in the context of wood production. They accordingly calculated the atmospheric impacts for timber production and utilisation in Finnish boreal forests, compared to a reference management regime. The study covered two 100-year rotation periods for a Scots pine (*Pinus sylvestris*) stand. The model included carbon sequestration in the growing forest biomass (above and below ground), carbon emissions of humus and litter and emissions from the degradation of HWPs. In the managed forest regimes, the forest system acted as a net sink for atmospheric carbon, whereas in the unmanaged scenario the forest become a carbon source after 120 years. The wood management and production regimes emitted more carbon to the atmosphere during the first rotation. This situation reversed in the second rotation. However, no consideration was given to the substitution effect where the biomass replaced fossil fuels, which has been shown to be very important.

Although most studies generally show that longer rotation lengths result in higher levels of carbon sequestration in the forest biomass, it is important to include the effect of natural disturbance, such as storm damage, or fire, otherwise the effect of longer rotations is exaggerated (Fortin et al. 2012, 2014). Wild fires occur every 30 to 200 years in boreal forests and depending upon their intensity, they can destroy 25-75% of the standing organic matter. However, during the 20th century, humans have started to control wild fires in the boreal zone and this has allowed the soil organic carbon to increase (Liski et al. 1998). After a meta-analysis of the effects of management practices upon carbon storage in forests, Kalies et al. (2016) came to the view that over the long term, the different management practices were unimportant compared to the effects of large scale natural disturbances (drought, fire, storm damage, pests and diseases). The impact of storms on the biomass carbon stock is 5-10 times greater than that of fires, but the release of carbon is more gradual, whereas that from fires is immediate. Although harvesting of managed forests has a greater impact on carbon stock compared with natural disturbances, this analysis is complicated by the inclusion of salvage logging in the harvesting inventories (Pilli et al. 2016).

After clearfelling, a stand becomes a net source of carbon dioxide for some time until the growth of woody biomass compensates for the loss of soil carbon through respiration. In a study of a chrono-sequence of Scots pine clearcut stands in Southern Finland, it was found that the 4-year old stand was a net source of carbon dioxide throughout the year, the net exchange of carbon dioxide for the 12-year old stand was close to zero and stands aged 40 and 75 years were net sinks (Kolari et al. 2004). The differences were largely attributable to variations in soil respiration between the sites. The 4-year old stand had been clearfelled and scarified, but not replanted and consisted of grasses and dwarf shrubs, but no trees. The other sites had been re-seeded after clearfelling.

For a correct understanding of the stocks and flows of biogenic carbon in the forestry and HWPs sector it is necessary to couple wood product models with forest ecosystem models. Researchers are then able to compare alternative forest management strategies in conjunction with different wood product uses, in order to maximise the climate change mitigation potential of the sector. Such combined models can include recommendations regarding silvicultural management with respect to rotation lengths (Liski et al. 2001, Kaipainen et al. 2004, Perez-Garcia et al. 2005a, Pingoud et al. 2010), species (Pukkala 2011, Pukkala et al. 2011), planting density (Fortin et al. 2012), thinning regime (Profft et al. 2009, Pukkala 2014), transitions from clearfell to continuous cover (Eriksson et al. 2012) and allocation to different wood product pools (Klein et al. 2013, Werner et al. 2006, 2010, Eriksson et al. 2007, Fortin et al. 2012, Smyth et al. 2014). Both the geographical and time scale over which these studies are performed can affect the outcome (Helin et al. 2013, Lundmark et al. 2014).

The effects of climate change on timber production also need to be included. The climate of Norway is likely to become warmer and wetter as a result of global climate change (O'Brien et al. 2004). There is expected to be a dramatic decline in pine and spruce in Scandinavian countries in the 21st century (Kindermann et al. 2013). It has been shown that management of spruce stands and use of the timber in long life products is likely to lead to the highest climate change mitigation benefits compared to many other species. However, spruce forests are also more likely to be subject to disturbances as climate change progresses (Klein 2013).

5.5 Carbon accounting

Carbon accounting refers to processes used to measure and track the flows of carbon atoms through technological systems and how these interact with the environment (Stechemesser and Guenther 2012). The first models for carbon accounting of forest biomass were developed in the 1980's (Cooper 1983, Karjalainen 1996). Methodologies for carbon accounting are assuming greater importance due to concerns regarding the impact of the release of fossil carbon into the atmosphere, primarily as carbon dioxide and methane. It is an essential element of carbon trading schemes, such as the European Union Emissions Trading System and is also needed to report on national greenhouse gas inventories required under the Kyoto protocol. Carbon accounting can also be used as a means of supporting informed decisions about products and processes, using life cycle assessment methodologies; these are sometimes referred to as carbon footprints. Carbon trading schemes have been introduced as a way to internalise the external costs of carbon emissions and are a means by which countries are able to meet their obligations under the Kyoto protocol. The EU launched a carbon trading scheme in 2005, covering power plants, aviation and energy intensive industries. There are various carbon trading schemes around the world, but there is no global trading scheme at present. In a future carbon trading market, it is envisaged that carbon credits could be given for the storage of atmospheric carbon (as biogenic products) in buildings. The value placed upon the storage of atmospheric carbon has to be represented in the market. For example, credits would be given for the use of timber in construction because of the sequestered atmospheric carbon dioxide.

The Kyoto approach to carbon accounting assigns territorial responsibility for CO_2 emissions to the producers. This requires the forestry and forest products industry to report on emissions associated with the activities of that sector (Knauf et al. 2015). But unlike other industries, the forestry sector is also responsible for absorbing and storing carbon dioxide. When the products of that sector are used in the built environment, that carbon storage benefit continues in time. Eventually, the carbon will exit the built environment carbon storage pool, with the most likely final outcome that the material will be oxidised with energy recovery at the end of life (or multiple lives).

The role of harvested wood products in mitigating greenhouse gas emissions has only recently been recognised by the Kyoto Protocol. In 2009, the 15th Conference of Parties of the UN Framework Convention on Climate Change, Copenhagen, it was agreed that harvested wood products (HWPs) could be included as an additional carbon pool. For the first commitment period (2008-2012), it was assumed that the quantity of carbon leaving the HWPs' pool every year was equal to the annual inflow. This means that although a considerable quantity of atmospheric carbon may be stored in the wood products pool, this amount is assumed stable over time and there is not net benefit in terms of mitigation potential. For the second commitment period (2013-2020) the carbon accounting included carbon stock changes in the HWP pool.

Although the IPCC recognises the importance of the built environment, its mitigation strategies listed in the fourth and fifth assessment reports (IPCC 2007, 2014) are almost exclusively concerned with energy consumption. The use of wood as an example of a low embodied energy material is mentioned, but there is no consideration given to the potential for timber and other plant derived products to act as carbon stores in the built environment. Furthermore, the use of mitigation strategies associated with forestry is only concerned with bioenergy and does not discuss the carbon storage potential of timber products. However, the Conference of the Parties to the Kyoto Protocol in Copenhagen in 2009 did recognise the importance of including timber products as carbon sinks and the 2011 Durban and 2012 Doha conferences stated that carbon stored in wood products should be integrated into reporting procedures.

Hashimoto (2008) performed a detailed comparison of the accounting procedures that were outlined by the IPCC (2006) and found that different results could be obtained, depending upon the indirect effects of wood use changes and also the reference scenarios. The current 2006 IPCC Guidelines for National GHG Inventories (which is used for convention reporting of HWPs) allow for four accounting approaches to HWP: stock-change, atmospheric-flow, the production approach, and the simple decay approach (the production approach and decay approach are mathematically equivalent). These differ in the way that they define system boundaries and therefore, national carbon emissions that are reported differ according to the accounting approach used. They concluded that the most reliable results were obtained by using the default approach, although they admitted that this did not account for growing carbon stocks in HWPs. The different approaches can be summarised (Tonn and Marland 2007, Jasinevičius et al. 2015):

- Stock change approach = domestic HWP + imports – exports (system boundary is HWPs consumed in the reporting country). This reports the stock changes when and where they occur regardless of whether the outputs and inputs are timber or carbon dioxide;
- Atmospheric flow approach = domestic HWP + imports – exports (system boundary is the atmosphere). This method reports on carbon fluxes to and from the atmosphere;
- Production approach = domestic HWP + exports. Imports are not included (system boundary is HWPs originating from the forests of the reporting country). This requires the reporting of all stock changes deriving from the harvesting of the timber, with the party who harvested the timber originally monitoring the stock change in HWPs, regardless of who holds or owns the carbon-containing products.

The calculation methods were summarised by Hashimoto (2008):

IPCC default approach:

$$- (NG_d - H_d)$$

Stock-change approach:

$$- (NG_d - H_d) - ((P_d - UP_d) + (P_i - UP_i) - ((LW_d - DW_d) + (LW_i - DW_i))$$

Production approach:

$$- (NG_d - H_d) - ((P_d - UP_d) + (P_e - UP_e)) - ((LW_d - DW_d) + (LW_e - DW_e)$$

Simple decay approach:

$$- NG_d + (E_d - E_e) + (IW_d + IW_e) + (DW_d + DW_e)$$

Atmospheric flow approach:

$$- NG_d + (E_d + E_i) + (IW_d + IW_i) + (DE_d + DW_i)$$

Where: NG = net growth of forest, H = harvested wood, P = product, UP = used product, LW = landfilled waste, DW = emission from decomposed waste, PW = process waste, IW = emission from incinerated waste, E = emission from energy production; and the subscripts i = imported, e = exported, d = domestic.

Pilli et al. (2015) used the Tier 2 approach, described in the 2013 Revised Supplementary Methods and Good Practice Guidance Arising from the Kyoto Protocol, to determine removals and emissions of HWPs between 1990 to 2030 for the EU-28 countries, with three different harvesting scenarios (historical average, +20% and -20%). For the period 2000-2012, their results agreed with other studies (e.g., Pan et al. 2011), indicating a HWP sink of -44 million tonnes of CO_2 per year, which is about 10% of the forest pool sink for that period. This decreased to -23 million tonnes under the constant harvesting scenario, -43 million tonnes under the increased and -9 million tonnes under the decreased harvesting scenario. Maintaining a constant harvesting scenario resulted in the HWP pool becoming saturated (approaching zero in the long-term). The results showed that there was limited potential for a HWP sink in the EU, based upon analysis of the sequestered carbon only. However, there is potential for increasing the size of the HWP pool by using timber in long life products in the built environment, thereby leading to longer product half-lives (Braun et al. 2016, Dolan et al. 2012).

5.6 The harvested wood products pool

The carbon pool of HWPs can be in one of three states, stable, increasing or decreasing. Which one of these applies obviously depends upon the rate at which harvested wood enters the pool and the rate that the carbon in the pool is oxidised. Although a huge amount of atmospheric carbon is stored in wood products, this is of no significance from the point of view of mitigation if the carbon stock is stable. If the size of the pool is decreasing then this means that more biogenic carbon is being released than is entering, which will result in an increase in atmospheric radiative forcing, as is the case with the burning of fossil fuels. From this perspective, it is irrelevant whether the source of the carbon is biogenic or fossil, it is the fact that the stock in the HWP pool is decreasing that is important. Conversely, an increase in the size of the HWP pool is of benefit, since this results in a net sequestration of atmospheric carbon, provided that the amount of carbon stored in the forests from which the wood is derived is either stable or is increasing. This means that the timber has to come from sustainably managed forests. The HWP pool size can be increased by raising the amount of wood harvesting and/or by increasing the lifespan of wood products in the HWP pool (increased levels of recycling, improved durability, etc.). The best overall strategy is to increase the level of HWPs and other biogenic materials in the pool as well as increasing the retention time by extending the life of products (enhanced durability) and by adopting a cascade materials management structure. Finally, the biogenic carbon can be returned to the atmosphere by incineration with energy recovery, thereby obtaining credits by substitution of a fossil fuel source. There are at least 41 models that attempt to describe the dynamic behaviour of harvested wood product pools and these have been recently reviewed by Brunet-Navarro et al. (2016). This paper highlighted a lack of reliable data to accurately estimate the carbon stocks and fluxes. The paper concludes by stating 'if the sector wants to demonstrate the environmental quality of its products, it should make it a priority to provide reliable life cycle inventory data, particularly regarding aspects of time and location'.

Guest et al. (2013) modelled the effects of carbon storage periods of HWPs combined with different rotation periods in the forestry operations. The combinations ranged from one year rotation and 100-year storage to 100 year rotations combined with 0 years storage. This was an extension of the approach that was first introduced by Cherubini et al. (2011a). The model used the impulse response function (Bern model) to describe the decay of a pulse of CO_2 entering the atmosphere, which arose from the complete oxidation of the biomass at the end of life. Time horizons of 100 years and 500 years were investigated. This model was compared with the PAS 2050 method, a model developed by Clift and Brandão at the University of Surrey, Centre for Environmental Sustainability; and also with the approach of Moura Costa and Wilson (2000). Because these models did not include the regrowth of biomass in the forest, they were considered to be less representative of reality compared with the analysis by Guest et al. (2013). Although the benefits of carbon storage were clearly seen in the Guest et al. (2013) model, they were less apparent compared with the other approaches. Time horizons of 100 years showed the benefits of carbon storage more clearly, compared to 500 years. The general conclusions were that short rotation and long biomass storage periods were the most beneficial. By contrast, long rotation periods and 0 years of storage (this is instant oxidation – as in biomass burning) were not beneficial; although the effects of substitution of fossil fuels, or high embodied energy materials was not considered.

Lauk et al. (2012) noted that a better understanding of the global carbon cycle as well as considering potential mitigation options requires an understanding of both natural and socio-economic flows of carbon. They point out that an under-researched part of the global carbon budget is the storage of carbon in long life products, such as buildings and furniture. They therefore undertook an assessment of global socioeconomic carbon stocks and flows for the period 1900-2008. This considered the carbon stored in wood, bitumen and plastic. The calculation of carbon stocks in each year were based on the Tier 1 approach of the IPCC guidelines for harvested wood products. The total global socioeconomic stocks of carbon were 2.3 GtC in 1900, increasing to 11.5 GtC by 2008. The share of wood in the stocks decreased from 97% in 1900 to 60% by 2008, but nonetheless increased from 0.22 GtC to 6.9 GtC. The rate of gross carbon sequestration in socioeconomic stocks increased from 17 MtC/yr in 1900 to 247 MtC/yr in 2008. But they failed to distinguish between biogenic carbon in the wood, which is derived from atmospheric carbon dioxide and the fossil carbon that is stored in plastic and bitumen. They concluded that although socioeconomic carbon stocks are not negligible, there was only a modest potential to mitigate climate change by the increase in stocks.

A study of the Northern Region of the USA concluded that the HWP pool was acting as a net emitter of CO_2 due to reductions in carbon stocks linked to the reduction in harvesting from a peak in 1995 (Stockmann et al. 2012). The method used was that recommended by the IPCC, which focusses on the physical stocks of carbon in clearly defined pools. The IPCC methodology does not account for the carbon emissions arising from the processing and transportation of wood products, nor does it take account of changes in carbon fluxes associated with product substitution, nor the effect of substitution of biomass for fossil energy sources.

The effect of climate change upon the storage of carbon in wood products has not been examined in any detail. An increase in atmospheric carbon dioxide levels will lead to an increase in net primary production and hence increased carbon stocks in forests (Karjalainen et al. 2002, 2003). However, it is also necessary to take account of natural disturbances (storm events, pests and diseases) when modelling carbon stocks and this introduces an inherent level of uncertainty (Cameron et al. 2013). Fortin et al. (2014) stated that carbon stocks in forests could be over-estimated by as much as 8% if storm damage events are not taken into account. Härtl et al. (2016) interpret sustainable timber use in HWPs not as a removal, but a prevention of carbon being oxidised due to the normal cycle of growth and respiration. This means that timber contributes to climate change mitigation by a substantial delay in emissions.

Eriksson et al. (2007) found that the greatest accumulated carbon emission reduction occurred when timber products were used as a construction material, slash and stumps were used as a biofuel, with coal used as a reference energy source. They also found that the greatest soil organic carbon levels occurred with the fertilisation regime, followed by intensive harvesting, with traditional harvesting being the lowest of the three scenarios considered. This contradicts the findings of Dean et al. (2017) who concluded that harvesting resulted in a reduction of soil carbon over a long period of time (hundreds of years).

Cameron et al. (2013) reported on a study of the GHG balance for a forest products company operating in NE North America. The study involved a dynamic attributional LCA, estimating cumulative net GHG emissions over a period of 100 years. This was compared with a no-management scenario, in which the forests were not harvested and non-renewable building products replaced structural wood products. The company that was analysed operated sawmills, pulpmills and papermills. The model included 2.2 MHa of productive forest land. Carbon was tracked from the forest through raw wood conversion into product groups, in life use and disposal. Harvested wood not transferred to a wood products pool was assumed to be immediately oxidised. The analysed system acted to sequester atmospheric carbon until year 85, but became an emitter thereafter, because of projected harvest increases. Paper products had high energy demands for manufacturing and high emissions due to landfill decomposition. Both the HWP and landfill pools increased, but landfill emissions increased throughout the 100-year analysis period. Several alternative scenarios were considered. Improvements in the carbon balance were found when the landfill methane capture rates were increased, when wood destined for pulp/paper was used for bioenergy instead and where bioenergy was used for an electricity source for the wood processing. In evaluation of different strategies, it was emphasised that natural disturbance should be included in the analysis. The final conclusion of the study was that intensive forest management to produce a sustainable long-term supply of softwood products and biofuel would result in a GHG mitigation potential similar to that resulting if the forests were allowed to grow unmanaged. The outcomes were however dependent on factors such as natural disturbance risk, products produced and grid electricity emissions.

These models can result in very different conclusions depending upon what is included in them. For example, Brunet-Navarro et al. (2016) note that contradictions occur depending upon whether a bucking allocation module is included in the model or not. This module assigns log grades to different wood products. Where this is included in the model, it is concluded that long rotation lengths are beneficial (Liski et al. 2001, Pingoud et al. 2010) but when it is absent, the opposite conclusion is obtained (Kaipainen et al. 2004, Perez-Garcia et al. 2005a). This is typical of all methods to quantify sustainability. The assumptions made can have major effect on the outcomes, the models are invariably complex and this complexity can make the outputs from these models very hard to compare.

5.7 Calculating carbon storage in harvested wood products

Atmospheric carbon dioxide is sequestered by plants during photosynthesis and remains stored in the material of the plant until the carbon is subsequently oxidised at the end of life. The question of how to account for the storage of carbon in biogenic products has been the subject of much debate and the issues that this presents have still not been fully reconciled. Between 2006 and 2009, the LCA community debated how to construct methods to calculate biogenic CO_2 and eventually they came to the conclusion that it was best not to calculate it at all (Vogtländer et al. 2014). The reason for this decision was that the stored biogenic carbon will eventually re-enter the atmosphere at the end of life of the product. The inbuilt algorithms that calculated biogenic carbon storage were consequently removed from computer programs, such as Simapro, which uses the Ecoinvent database. Biogenic carbon dioxide was removed from the GWP indicators of the IPCC and systems such as CML-2 and ReCiPe. However, more recently it has been realised that the storage of biogenic carbon does have a role to play in climate change mitigation and this needs to be recognised. Gustavsson and Sathre (2011) note that much of the methodology developed for determining the GWP impacts associated with the production and use of biofuels is also useful for the comparison of HWPs with alternative construction materials. The difference is that the harvested material is not immediately oxidised and the methodology has to take account of this 'delayed emission'. The storage of atmospheric carbon in long life timber products has a greater climate change mitigation benefit, compared with immediate oxidation for energy recovery (Stewart and Nakamura 2012). Furthermore, the use of bioenergy facilities to utilise the inherent energy of processing and harvesting residues provides additional climate benefits not included in the forestry chapter of national GHG inventories.

For the calculation of the GHG emissions and storage associated with products and services, there have been several methodologies published: The World Resources Institute (WRI) and World Business Council for Sustainable Development (WBCSD) Greenhouse Gas Protocol, publicly available specification (PAS) 2050, the ILCD method and ISO 14067.

The WRI/WBCSD GHG Protocol was developed as a product GHG accounting and reporting standard in 2001. It was subsequently revised in 2011 to provide a generalised methodology for GHG reporting and quantification.

When the benefits of carbon storage are considered in LCA, this requires consideration of the time of storage to determine what the GWP impact is. The question of the temporal nature of emissions of carbon into the atmosphere and considerations of the length of time that atmospheric carbon is held in storage are extremely important when biogenic carbon is considered (Cherubini et al. 2012). Unfortunately, there is no consensus regarding the methodology for measuring and accounting for carbon in biogenic products. Although the ILCD methodology is still current, there have not been any useful developments in standardisation. The 2008 version of PAS 2050 did include methods for calculating the temporal aspects of biogenic carbon storage in annex C, but by the time that the 2011 version had been published, this was no longer present. The European Standard EN 16485 giving product category rules (PCR) for round and sawn timber featured a temporal calculation method for determining the storage of biogenic carbon in the draft form, but in the final published version this had been removed.

ISO 14067: 'Carbon footprint of products - requirements and guidelines for quantification and communication' provides protocols for the transparent reporting of GHG results. Due to objections raised by some countries, ISO 14067 was published as a technical specification rather than an international standard in May 2013. Although the WRI/WBCSD GHG Protocol, PAS 2050 and ISO 14067 all have similarities, ISO 14067 gives details on the selection of appropriate system boundaries and it also gives guidance on the simulation of use- and end-of-life phases of the life cycle (Wu et al. 2014b, 2015). The development of a GHG reporting standard was first proposed by the ISO Technical Committee 207, Working Group 2 (ISO TC207/WG2) in April 2008 and development involved more than 100 experts from over 30 countries.

ISO 14067 gives the methodology for the calculation and reporting of the carbon footprint of goods and services, due to the emissions and removals of GHG gases during the product lifetime. It was developed based upon previously published ISO standards on environmental labelling and environmental management:

- ISO 14020: Environmental labels and declarations - general principles;
- ISO 14024: Environmental labels and declarations - Type I environmental labelling - principles and procedures;
- ISO 14025: Environmental labels and declarations - Type III environmental declarations - principles and procedures;
- ISO 14040: Environmental management - Life cycle assessment - principles and framework;
- ISO 14044: Environmental management – Life cycle assessment – requirements and guidelines.

The methodology requires two main parts: a carbon footprint report and a critical review (which is not the same as a third party verification). The purpose of the critical review is to ensure that the methods given in ISO 14067 are followed correctly and that they are scientifically valid. The data that is used and interpretations should be reasonable and appropriate, given the limitations identified and the goal and scope of the study. The study should also be transparent and consistent.

The current version of the PEF guidance document (Commission Recommendation 2013/179/EU) available from the European Commission, Section 5.4.9 deals with the issue of temporary carbon storage with the statement 'Credits associated with the temporary (carbon) storage or delayed emissions shall not be considered in the calculation of the default EF impact categories. However, these may be included as "additional environmental information".

Conventional LCA methods do not assign any benefits to the temporary storage of atmospheric carbon, because the timing of emissions relative to removals is not considered (Pinsonnault et al. 2014). Although there are benefits to be gained from using timber products in long-life products as a store of atmospheric carbon, there is still no agreed way of accounting for this (Brandão et al. 2013).

The advantages of using timber and other bio-derived materials as a means of storing sequestered atmospheric carbon in the built environment has received considerable attention in the scientific literature (e.g. Pilli et al. 2015, Jasinevičius et al. 2015, Brunet-Navarro et al. 2016). Werner et al. (2005) analysed the consequences of the increased use of timber in construction on the carbon storage and emissions associated with substitution for more energy-intensive building materials. The study showed that the effects due to carbon storage were of minor importance compared with those due to substitution for more energy-intensive materials and the use of timber residues and post-consumer waste wood as an alternative energy source compared to fossil fuels.

Werner et al. (2006) argued that the climate change mitigation consequences of using increasing amounts of timber in long-life products depended upon a number of factors:

- Up to twice as much biomass is removed from the forest compared to the amount of wood being used in the built environment and it is important to consider what happens to this wood;
- What happens to the surplus wood in the forest after harvest is important with respect to the effect on CO2 emissions associated with harvesting. Using this residue as an energy source to substitute for fossil fuels appears to be the most promising strategy;
- In the alternative scenario, where no extra timber is used in construction, the results obtained depend upon what happens to the unused forest biomass;
- The overall effect is highly dependent upon the ratio between the growth rates of the trees and the lifetime of the HWPs;
- Overall negative pool effects (where emissions exceeded storage) could potentially result from the increased use of wood in some situations, but when the energetic and material substitution effects were included then the increased use of wood was always justified (from the perspective of a pool a negative value is viewed as an emission, or loss, whereas from an LCA perspective a negative GWP impact represents sequestered atmopsheric carbon dioxide).

At the product, or building level the claim that carbon storage benefits arise from the use of wood products in the built environment depends upon the embodied emissions being lower than the amount of atmospheric carbon stored in the wood product itself. Many studies have shown that this is indeed the case and that there is a measurable carbon storage benefit. Pingoud and Lehtilä (2002) studied wood products in a Finnish context and estimated that the associated GHG emissions were only 7% of the CO_2 eq. stored in sawn wood products. These percentages rise with the amount of processing that is required for the wood product and are highest for virgin paper products (30-60%), but even in these extreme cases the amount of CO_2 eq. released is lower than the amount stored in the product. Hill and Dibdiakova (2016) conducted an analysis of published EPDs of different wood products and plotted the GHG emissions associated with the processing (modules A1 to A3) compared with the amount of biogenic carbon stored in the material. The results (reproduced in Figure 1) showed that the carbon stored in the timber product always exceeds the embodied GHG emissions.

Kilpeläinen et al. (2014) noted that many studies of HWPs failed to take account of the dynamic nature of carbon exchanges with forests in the context of wood production. In their analysis they calculated the atmospheric impacts for timber production and utilisation in Finnish boreal forests, compared to a reference management regime. The study covered two 100-year rotation periods for a Scots pine (*Pinus sylvestris*) stand. The model included carbon sequestration in the growing forest biomass (above and below ground), carbon emissions of humus and litter and emissions from the degradation of HWPs. In the managed forest regimes, the forest system acted as a net sink for atmospheric carbon, whereas in the unmanaged scenario the forest become a carbon source after 120 years. However, the wood management and production regimes emitted more carbon to the atmosphere during the first rotation. This situation reversed in the second rotation. Where biomass from forestry residues and end of life timber was examined, no consideration was given to the substitution effect where the biomass replaced fossil fuels.

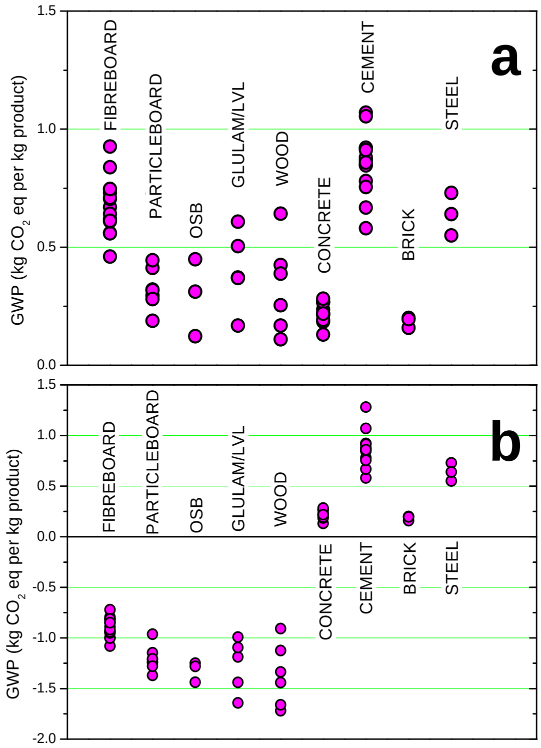

Figure 1 **A comparison of the GWP impact of different harvested wood products with concrete, brick, cement and steel. In Fig. 1a the sequestered carbon in the wood products has not been included, whereas in Fig. 1b the sequestered carbon in the wood products is included (from: Hill and Dibdiakova 2016).**

The impacts of storing atmospheric carbon dioxide are dependent upon the length of time for which the carbon is removed from the atmosphere (Levasseur et al. 2013). Clearly the reduction in radiative forcing due to the removal and storage of atmospheric carbon dioxide is only of benefit during the time that storage occurs. At end of life, oxidation of the timber products results in release of the stored carbon as carbon dioxide, whereas anaerobic degradation causes the production of methane, a much more potent greenhouse gas. End of life options are clearly important. Kirschbaum (2006) argued that the relatively limited time that carbon is stored in the HWP pool does not provide a significant benefit in terms of mitigating increases in GHGs in the atmosphere and that efforts to reduce energy consumption would provide more benefit. The reasoning behind Kirschbaum's argument is that a decrease in carbon dioxide levels in the atmosphere due to storage in wood products would reduce the concentration gradient between the air and the sea and hence reduce the rate of uptake of carbon dioxide by the ocean. At the end of this temporary storage period, the CO_2 is now released into an atmosphere which has a higher CO_2 concentration than it would be otherwise. The Kirschbaum paper has been criticised, because it disregards the cumulative climate impact and because the results of his analysis are an artefact of his particular perspective (Fearnside 2008, Dornburg and Marland 2008). Dornburg and Marland (2008) and Fearnside (2008) argue that even a temporary storage of carbon in the HWP pool is of benefit because it 'buys time' at a critical period, allowing for society to accumulate capital and develop technology to make a transition to a low carbon development path.

A much better approach to dealing with the carbon storage benefits of using biogenic materials in the built environment is to calculate the carbon storage in a product pool, which is an ecomomic approach to the problem. If the harvested wood products pool (HWP) is considered from a carbon storage point of view, then there is a flow of biogenic carbon into the the pool from the forest and there is a flow out of the pool as the timber is oxidized and the carbon is returned to the atmosphere. An increase in the HWP pool therefore provides a benefit during the time that the size of the pool is increasing, thereafter the pool will be in equilibrium with the environment. In other words, there is a sequestration benefit for as long as the carbon flow into the pool exceeds the carbon flow out of the pool. However, there are also other potential benefits arising from the use of timber as a construction material in place of a higher embodied energy alternative. Kirschbaum (2006) did not consider the potential benefits of materials' substitution and fossil fuel substitution if the wood is incinerated with energy recovery at end of life in his analysis. Gustavsson et al. (2006a) pointed out that the carbon stock in wooden buildings is actually less important than the reduction in CO_2 emissions associated with using timber as a construction material when compared to alternatives (substitution effect). They also concluded that there are sufficient timber resources in Europe to allow for a significant expansion of the use of wood for both materials and energy. Lundmark et al. (2014) also found that materials and energy substitution were more important contributions to climate change mitigation than stock change effects. Kirschbaum (2006) noted that there are benefits if the storage of carbon in HWPs is combined with landscape level policies which also include increases in biogenic carbon storage in forests. He further noted that expenditure in short term carbon storage projects is of little benefit without the obligation of the long-term maintenance of biogenic carbon stocks.

In LCA at the product level, the time of storage of atmospheric carbon dioxide has been taken into consideration in the UK PAS 2050 and the European Commission's International Reference Life Cycle Data (ILCD) Handbook. Both of these methods rely on the idea of a 'delayed pulse' of carbon dioxide into the atmosphere and use this to give a credit for the time that the atmospheric carbon is stored in the biogenic material. With PAS 2050, the benefits of carbon storage are calculated on the basis of a weighted time average approach for an assessment period of 100 years. For example, if a bio-derived product containing 1 kg of atmospheric carbon is used in a building for 50 years before disposal by incineration, then the benefit of carbon storage is calculated as (50/100) x 1 = 0.5 kg. A slightly different calculation method is used if the storage period is 25 years or less. The ILCD linear discounting methodology considers biogenic carbon sequestration as a negative value and emissions as a positive value. The carbon credits in biogenic materials arise from the effect of delayed emission over a 100-year assessment period. If emission of 1 kg carbon is delayed for a period of 50 years, this is calculated as (50/100) x 1 = 0.5 kg.

The benefits of atmospheric carbon storage in bio-derived products can only be accounted for if the material is derived from a sustainable production source. For the case of timber products, this means that there has to be regeneration of the forest after felling to produce the timber. If felling of the timber results in land use change (such as conversion to agriculture) then the benefits of atmospheric carbon storage in the HWPs are no longer present and according to the ILCD guidelines this biogenic carbon should be treated as if it were fossil carbon. ISO 14067 (2013) does not deal with the storage of atmospheric carbon in products containing photosynthetically-derived materials, other than stating that removals of atmospheric carbon associated with biogenic sinks should be treated separately in the carbon footprint report and that the storage time period should be reported if applicable.

Brandão et al. (2013) reviewed six methods (including PAS 2050 and ILCD) used for accounting for the impacts of carbon sequestration and the temporary storage and release of biogenic carbon. The paper identified that the benefits of carbon storage are highly dependent upon the time horizon adopted and that this is based upon value judgements rather than having any sound scientific basis. As such, the time frame adopted is informed by policy considerations and the commonly used 100-year period for GWP calculations is based upon the desire to bring about achievable change in a crucial period in the history of humanity. The intention is to change behaviour to a sustainable development trajectory.

Although many studies of carbon storage in harvested wood products have been conducted, there are no commonly recognised methods for determining and reporting this in bio-derived products from a time perspective. PAS 2050 and ILCD give two methods for dealing with the temporal factor, but other approaches have been suggested. The method of Moura-Costa and Wilson (2000) calculates a sequestration-based equivalence factor called the Absolute Global Warming Potential (AGWP). The AGWP is defined as the cumulative radiative forcing potential for CO_2 of unit mass over a specified time horizon. This is calculated from the following relationship:

$$\boldsymbol{AGWP = \int_0^{TH} a_x \,.\, [C(t)]\, dt} \qquad \text{(Eq. 1)}$$

Where TH is the time horizon under consideration, t is time, a_x is the radiative forcing due to the presence of unit mass of CO_2 in the atmosphere and C(t) is the concentration of a pulse of CO_2, decaying as a function of time, which is usually expressed in terms of the Bern model. Based upon these considerations, they found that removing 1 tonne of CO_2 eq. from the atmosphere and storing it for 55 years counteracts the effect of releasing a pulse of CO_2 into the atmosphere with a residence time of 100 years. This method allows for benefits greater than 100% if the 55-year storage period is exceeded. Another approach, referred to as the Lashof method, assumes that the storage of atmospheric CO_2 is equivalent to a delayed emission of fossil CO_2, but the carbon tracking is performed in the atmosphere rather than the biosphere (Fearnside 2002) (Fig. 2). Vogtländer et al. (2014) noted that the ILCD and the PAS 2050 methods both overestimate the benefits of the temporary fixation of carbon dioxide. The PAS 2050 method is a linearisation of the Lashof curve for the first 25 years, but thereafter follows the ILCD linear discounting approach.

Pingoud et al. (2012) developed a method for determining the GWP impact of forest biomass life cycles compared with functionally equivalent alternatives based upon fossil fuels and non-renewable materials, using an extension of that described by Cherubini et al. (2011b,c). In order to determine the change in radiative forcing with time, an impulse model based upon the Bern Carbon Cycle is used. An integral of this decay curve gives the cumulative radiative forcing over a desired time frame; this is called the Absolute Global Warming Potential, as with the Moura-Costa approach. The same behaviour can be assigned to any pulse of CO_2 entering the atmosphere, irrespective of whether it is fossil or biogenic in origin. However, with a pulse of biogenically-derived CO_2, the uptake of CO_2 from new growth in the forest also has to be taken into account; but this is highly dependent upon the biomass re-growth rate. Other factors, such as release of N_2O from fertiliser use, or change in albedo can also be included. In addition to this, the effect of substitution of the harvested biomass, either directly for a fossil fuel, or indirectly by the use of fossil fuels for processing a non-renewable equivalent material, also has to be taken into account (these are the displaced emissions), as well as any GHG emissions due to processing of the biomass to make the functional unit. When all of these effects are taken into account it is possible to arrive at a cumulative global warming payback time of the biomass compared with the functionally equivalent non-renewable-based reference service.

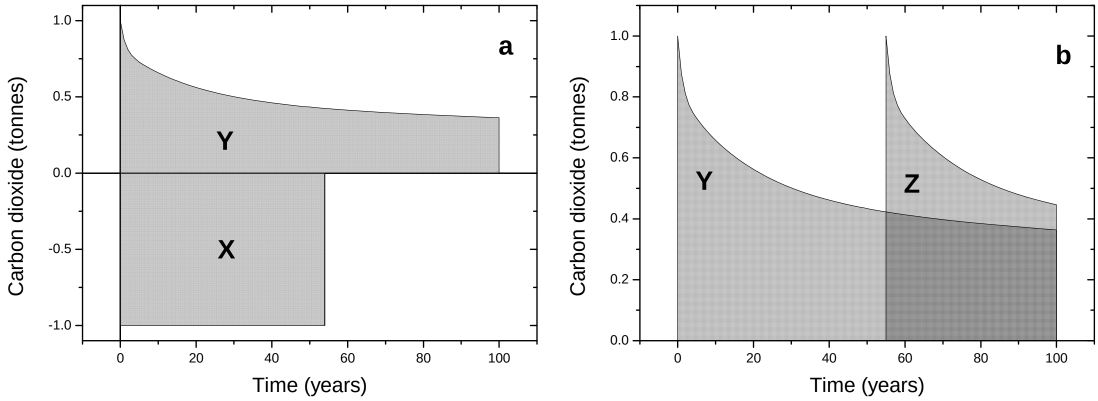

Figure 2 **Illustration of the Moura-Costa (a) and Lashof (b) methods for calculating the benefit of carbon storage. In (a) a pulse of 1 tonne of carbon dioxide is released into the atmosphere and this decays according to the Bern mechanism. The total global warming potential (GWP) over 100 years is represented by the area under the curve. The same total GWP is represented by storage of 1 tonne of CO_2 for 55 years (X=Y). In (b) the carbon is stored for 55 years and then released as a pulse of CO_2. The total GWP is the area under the curve Z, the benefit of storage is given by subtracting Y from Z.**

Kendall (2012) noted that the commonly used method of measuring all GHG emissions to the environment and then calculating the GWP based upon IPCC values could introduce distortions, because the timing of the emissions was not taken into account. Kendall describes a calculation called the time adjusted warming potential (TAWP) method, which introduced a weighting value depending upon when the emission occurred during the period of analysis.

Levasseur et al. (2013) examined the problem of GWP impact using a traditional LCA approach without including sequestered carbon, as well as a traditional approach including sequestered carbon, PAS 2050, ILCD and dynamic LCA methodologies. Each approach gave different results, with there being dramatic differences in some cases. It was concluded that the dynamic LCA approach was the preferred method for providing reliable data, although the results obtained were heavily dependent upon the assumptions made and the time horizon considered. The study also examined the problem using a functional unit of a wooden chair, which can give different results compared with studying temporal carbon storage of a pool of harvested wood products (HWPs). A pool of biogenic carbon products does not release carbon to the atmosphere in a pulse, as is the case with a single product, but in a way that is better modelled as a probability distribution (Shirley et al. 2011). Many studies investigating the release of carbon from HWP pools have modelled this behaviour as a single exponential decay (as in the IPCC guidelines) (Pingoud and Wagner 2006), but this does not adequately consider the fact that the probability of a product being taken out of service is related to the age of that item (Shirley et al. 2011). This problem was dealt with by the development of a distributed decay model (Marland and Marland 2003, Marland et al. 2010), which uses a probability distribution to determine how much of production from a particular year decays in any given time interval. This type of model is analogous to the approach adopted by the life assurance industry in actuarial mathematics. This form of modelling is very useful when attempting to adopt a realistic methodology for pricing carbon and assigning a value to the cost of emissions from the HWP pool in the future.

Røyne et al. (2016) reviewed 101 papers which reported on the LCA of forest products. They found that most of the studies excluded the dynamic features of carbon uptake and storage and that climate change impacts from land use change, aerosols and changes in albedo were often not considered. They stated that this could have important implications for decision support. The time frame of the study was found to be a very important consideration affecting the outcome. Depending upon the time perspective of the study, the authors concluded that impact factors other than GWP might be more important. The authors also noted that LCA has to be limited in order to be valuable and that it is important that it provides the correct information in order to be useful to the decision-making process. They recommended that LCA practitioners should:

- Reflect on the decision-making context;
- Use a sensitivity analysis to estimate the influence of different aspects of the study;
- Consider which climate impact aspects are important for the decision-making.

As noted earlier, the best approach is to deal with the issue of biogenic carbon storage at a product pool level, rather than within the LCA of a product.

6 Review of LCA of wood

6.1 LCA of forestry operations

Although LCA can be a useful tool for estimating the environmental impacts associated with the production of timber from forestry operations, it only makes a contribution towards understanding the sustainability of the process. The original concept of forest sustainability was related to sustained wood yield, but has now expanded to cover the diversity of goods, ecosystem services and other benefits (social and economic) demanded by society (Päivinen et al. 2012). For example, Newell and Vos (2012) noted that LCA does not capture the forestry and land use change impacts well in many cases, due to underdeveloped linkages between life cycle inventory modelling for wood and forest carbon modelling for all forest types and silvicultural practices. When assigning environmental burdens associated with the forestry operations to timber-based building products, it is essential to ensure that the correct allocations are made to (for example) thinning for pulpwood (González-García et al. 2009a).

Klein et al. (2015) undertook a review of 22 peer-review LCAs, four reports and two databases concerning life cycle assessment in the forestry sector. They noted that consistent and comprehensive LCA studies were lacking. They found a range of GWP values ranging from 2.4 to 59.6 kg CO_2 eq. per cubic metre (median = 11.8 kg CO_2 eq./m^3) over bark for site preparation to forest road and 6.3 to 67.1 kg CO_2 eq. per cubic metre over bark (median = 17.0 kg CO_2 eq./m^3) for site preparation to plant gate or consumer. Although there was a wide range of values, even with the worst-case scenario the embodied carbon emissions were lower than the amount of atmospheric carbon stored in the wood. The different reviewed studies used a variety of databases (Ecoinvent, Gemis, GHGenius, US Life Cycle Inventory Database, IDEMAT, GREET, Franklin 98), although in some cases the basic supporting LCA data was obtained from the literature. Different impact assessment methods were employed (SETAC, GHG-protocol, CML 2001, Eco-indicator, Eco-indicator 99, TRACI 2, ReCiPe Midpoint, CML2) and LCA software (GaBi, SimaPro, Umberto, BEATv2, or bespoke LCA tools). In half of the studies the methodological approach was not stated (other than IPCC 2006 guidelines). In 43% of the reviewed studies the allocation methods were not mentioned, in 21% allocation was stated as not being required, in 29% allocation was by mass in 11% (three cases) allocation by market price and in one case, allocation by energy.

Aldentun (2002) showed that the energy and associated emissions used for seedling production could not be neglected in LCA calculations of forestry operations, a major component being the use of fossil fuels for the heating of greenhouses. Obviously, impacts due to seedling production do not apply to forests that are managed using natural regeneration. Berg and Lindholm (2005) presented an analysis of energy use (in MJ/m^3) and GHG emissions (in kg CO_2 eq./m^3) for the various forestry operations in different regions of Sweden. Berg et al. (2012) reported on the activities of the EU project Eforwood, which developed a tool for sustainability impact assessment of the forestry wood chain. This paper focussed on logging operations. The project used several indicators (economic, social, environmental) which were applied in the sustainability impact tool and combined into one value expressing the sustainability. As with all such schemes, this involves value judgements to be made regarding the importance of different impacts. Mirabella et al. (2014) studied the use of LCA in order to make choices in harvesting practices in the forest, but found that LCA was not a sufficiently sophisticated tool for this purpose. They noted that many of the impacts that can occur due to forestry operations are local and that this is not captured in traditional LCA methodology, e.g., soil compaction due to the use of mechanical harvesters resulting in increased rainfall run-off from steep slopes. They also stated that it was necessary to address other aspects of sustainability by using wood produced according to certification schemes (such as PEFC, or FSC) and integrating the LCA with economic and social aspects (local supply chains, social and economic impacts on local communities) and a more ecological approach, including terrestrial and soil biodiversity. It has also been suggested that the chain of custody protocols associated with certification schemes could be combined with carbon footprint methodologies for HWPs (Sikkema et al. 2013). There have been some attempts to address the social aspects of the wood value chain with social LCA, but this is still very much a new and developing topic (Siebert et al. 2016).

6.2 LCA of wood products

Werner and Richter (2007) reviewed the results of 20 years of research into the environmental impact of the use of wood products in the building sector. They noted that wood products tended to have a better environmental profile compared to functionally equivalent products made of non-wood materials. In particular, the GWP and embodied energy associated with wood products was usually much lower than that of competitor materials.

As part of the US CORRIM initiative, Wilson and Dancer (2005a) conducted a gate to gate LCI of I-joist production, using data collected from factories in the Northwest and Southeast of the USA. Laminated veneer lumber (LVL) was used for the flanges. The functional unit was 1000 linear meters of I-joist, but other dimensions were not given. It required 1 680 kg of LVL and 1 640 kg of OSB to manufacture 1 km of I-joist in the Northwest, whereas it required 2 400 kg of LVL and 1 770 kg of oriented strand board (OSB) for the Southeast USA. LCI's were also published for LVL production (Wilson and Dancer 2005b), OSB (Kline (2005) and plywood production (Wilson and Sakimoto 2005) for the same geographical regions. Milota et al. (2005) performed an LCA on the production of sawn, planed, dry timber produced in the western and southern US. The embodied energy per m^3 of produced timber was 3.36 GJ for the west and 3.95 GJ in the south and associated CO_2 emissions were 258 kg and 354 kg. Emissions of VOCs can occur during the wood drying process (Milota 2000).

Nebel et al. (2006) found that the most energy consuming process during the production of timber for flooring was kiln drying. The embodied energy associated with harvesting represents less than 5% of the total in a cradle to gate analysis. Bioenergy also generally represents a greater proportion of the total energy used in processing since it is very convenient to use waste products (such as bark and off-cuts) as an energy source (Puettman and Wilson 2005a). For timber, more energy is available from biomass residues than is used in the processing steps (Sathre and Gustavsson 2007). Most studies show that sawn timber is both a carbon-negative and an energy-negative material.

González-García et al. (2013) quoted a GWP impact of 2.7 kg CO_2 eq. and an energy use of 39 MJ/m^3 of solid fresh Douglas fir roundwood (underbark (ub)) produced from German forests. Stand establishment and tending (thinning, roundwood forwarding and loading onto trucks) were found to have the greatest environmental impacts. The results were compared to the data of Dias and Arroja (2012), who reported a GWP of 4.8-12.2 kg CO_2 eq. and an energy use of 135 MJ/m^3 ub for pine and a GWP of 8.3-18.5 kg CO_2 eq. and energy use of 169 MJ/m^3 ub for eucalypt production in Portugal. González-García et al. (2009a,b) reported 35.5 kg CO_2 eq. and energy use of 395/m^3 ub for eucalypt in Spain, 36.1 kg CO_2 eq. and embodied energy of 370 MJ/m^3 ub for spruce production in Sweden and Michelsen et al. (2008) stated 25.0 kg CO_2 eq./m^3 ub for spruce production in Norway. Berg and Lindholm (2005) reported an energy use of 82 MJ/m^3 ub for softwood production in Sweden.

Schaubroeck et al. (2013) conducted a cradle to grave LCA of a wood processing chain for a functional unit of 1 m^3 of Scots pine sawn timber. This required the processing of 2.65 m^3 of stem softwood to make 1 m^3 of sawn timber, with the remaining 1.65 m^3 allocated to co-products. End of life of the timber was assumed to be incineration with energy recovery to make electricity for the Belgian grid. The resource and emission flows for the Scots pine stand were obtained from Schaunbroeck et al. (2012). Two LCA methodologies were applied using Simapro version 7.3 software. The resource consumption was quantified using the Cumulative Exergy Extracted from the Natural Environment indicator method, which is based upon the work of Szargut (1988) and Valero et al. (1986). The ReCiPe method was used to model ecosystem diversity, which was found to be reduced due to the intensively managed plantation forestry. There was an improvement in human health (DALY) largely due to the filtering and removal of particulate matter by the forest. The impact on climate was deemed to be less important, since this represented only 17% of the endpoint impact on human health and 8% of ecosystem diversity. The GHG emissions were 0.91 tonnes of CO_2 eq. per FU, which was nearly all attributed to the burning of wood co-products. The total GWP impact was -1.96 tonnes of CO_2 eq./m^3 of sawn timber for the whole life cycle, which included the -2.6 tonnes of CO_2 sequestered by the Scots pine stand during the life cycle. This only included the above-ground biomass, but not the carbon stored in the roots, soil, or litter, nor their associated flows of GHGs. The inclusion of the forest stand in the LCA was important to understanding the whole system.

Murphy et al. (2015) reported on the embodied energy and GWP impacts of a variety of timber products (sawnwood, wood chip, wood-based panel boards and wood pellets) manufactured in Ireland. A mass allocation method was used for assigning environmental burdens. The LCA was conducted using Simapro 7.3 with the foreground data collected from Irish manufacturing facilities. It was found that electricity use was responsible for most of the emissions in sawmilling. Allocation was made on a mass basis. The impacts of forestry operations involving seedling production, site establishment, harvesting and haulage were included. The functional unit was 1 m^3 of sawn softwood and three different scenarios were modelled:

- Scenario 1: A conventional sawmill, logs are delivered and debarked, sawn into planks with chips and sawdust as co-products. A proportion of the timber was treated with a preservative, such as Tanolith E. The bark and sawdust was used to supply heat to the processing plant. This had a GWP of 40.2 kg CO_2 eq. and an embodied energy of 761 MJ/m^3.
- Scenario 2: This was a sawmill with an integrated CHP plant. This had a GWP impact of 23.7 kg CO_2 eq. and embodied energy 1460 MJ/m^3.
- Scenario 3: A sawmill integrated with a pellet plant. This had a GWP impact of 31.7 kg CO_2 eq./m3 and an embodied energy of 914 MJ/m^3.

The study also considered medium density fibreboard (MDF) and OSB production in Ireland, with the following results:

- MDF: GWP 896.7 kg CO_2 eq. and embodied energy of 17 901 MJ/m3.
- OSB: GWP 235.6 kg CO_2 eq. and embodied energy of 5 569 MJ/m3.

Suter et al. (2016) conducted a material flow analysis of wood use in Switzerland. The environmental impacts related to the material flows were analysed using LCA environmental impact indicators. For GWP they concluded that there was an overall average benefit of 0.5 tonnes of CO_2 eq. per m^3 of timber used, when replacing non-renewable materials. Environmental impact calculations were based upon Ecoinvent 3.1. The model representation of wood use in Switzerland consisted of 52 processes producing 40 wood-based products. The study also investigated potential cascade use of wood in Switzerland, concluding that cascading would result in less wood extraction from the forest unless there was an increase in wood demand. In Switzerland, most of the wood from forests (about 50%) is used for energy production and for paper production (about 25%); with only about 10% being used for buildings. For each of the services, the environmental impacts for wood and non-wood products were compared (except for paper). A materials' flow analysis model for a wood supply chain was created to study the effects of introducing a wood products cascade chain on the overall environmental impact (Taskhiri et al. 2016). This showed that there were significant reductions in GWP impact arising from the introduction of cascading.

Asdrubali et al. (2017) recently published a review of the structural and environmental properties of wood products in building applications. When analysing the embodied energy and carbon emissions of timber products they came to the conclusion that the drying process was the most energy consuming stage of the manufacturing process. Transport obviously become more significant as the distances travelled from forest to building site increased.

With glulam manufacture, the drying of the wood and the adhesive make major contributions to the environmental burden of the product. The embodied energy associated with glulam manufacture was found to be between 6.8-7.2 GJ/m^3 (Puettmann and Wilson 2005b). The production of 1 m^2 of a massive wood (CLT) element was found to have a GWP impact of approximately 35 kg CO_2 eq. (Santi et al. 2016). A 1 m^2 element of this CLT weighed 128 kg at a moisture content of 13%, with a wood density of 480 kg/m^3.

At the time of writing this report there are over 70 published EPDs of HWPs. As already noted, an analysis of these was published in a paper by Hill and Dibdiakova (2016). The relationship between GWP impact and embodied energy for published EPDs of different HWP product categories (modules A1-A3) using updated published EPD data is shown in Figure 3, for a declared unit of 1 m^3 of timber.

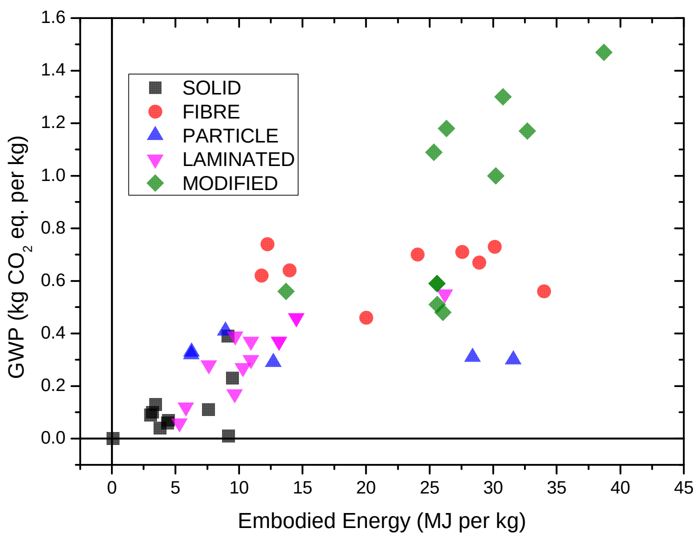

Figure 3 **Relationship between GWP impact and embodied energy (per kg of dry wood) for different harvested wood product categories: Solid wood, fibreboard, particleboard, laminated wood products and modified wood (Kebony, Accoya, thermally modified).**

In a sensitivity analysis of an LCA of MDF manufacture, it was found that both the final transport of products and the electricity generation profile (grid mix) had a significant influence upon the results (Rivela et al. 2007). A study of medium density particleboard production in a Brazilian context showed that the use of heavy fuel in the manufacturing process (including forestry operations) was the hotspot in all impact categories except ecotoxicity (Silva et al. 2013). Benetto et al. (2009) conducted an LCA of OSB production with emphasis on evaluating the environmental impact associated with a new wood drying process that had reduced emissions of VOCs. The study concluded that the environmental gains resulting from the new drying process were largely negated by changes required in the adhesive formulation. This shows the need to consider the entire process when considering the environmental impact of production and not focusing on making improvements of one part of the process.

The combination of an OSB production plant with a biorefinery for the production of acetic acid and methanol has been studied from an LCA perspective recently (Earles et al. 2011). Significant reductions in human toxicity potential and freshwater ecotoxicity potential were recorded for the combined plant compared to a conventional OSB production process. However, the economic viability of small scale production of commodity chemicals is a considerable barrier to the adoption of these combination plants. Garcia and Freire (2014) reported in the carbon footprint of particleboard production in Portugal. They compared the results obtained using the methods described in ISO/TS 14067, GHG Protocol (WRI/WBCSD), Climate Declaration (International EPD System), or PAS 2050. The functional unit was 1 m^3 of particleboard (density 640 kg/m^3) and both cradle-to-gate and cradle-to-grave lifecycle scenarios were studied. Six methods to assess delayed emissions due to carbon storage in the particleboard were analysed. Consequently, a wide range of carbon footprints were obtained: - 939 to + 188 kg CO_2 eq./m^3 for cradle-to-gate, +107 to + 201 kg CO_2 eq./m^3 for cradle-to-grave with incineration, -692 to 433 kg CO_2 eq./m^3 for cradle-to-grave with landfill. The atmospheric carbon stored in the particleboard was equivalent to 1098 kg CO_2 eq./m^3. The results were highly sensitive to the inclusion or exclusion of biogenic carbon storage. Emissions and removals of biogenic carbon is included in ISO/TS 14067, the GHG Protocol and PAS, but not in the Climate Declaration. There were also differences in the way that the environmental burdens from incineration with energy recovery were dealt with. González-García et al. (2009c) performed an LCA of hardboard manufacture, reporting on the abiotic depletion, global warming potential, ozone layer depletion potential, human toxicity, ecotoxicity, photochemical oxidant formation, acidification and eutrophication impact categories. The functional unit was 1 m^3 of hardboard, with a density of 987 kg/m^3. The raw material was wood chips from Norway spruce and European beech. The GWP impact was calculated to be 350 kg CO_2 eq./m^3.

There have been a few LCA studies of wood polymer composites reported in the literature. Xu et al. (2008) undertook a study of wood fibre reinforced polypropylene composites, reporting that the majority of the environmental burdens were associated with the polypropylene rather than the wood.

6.3 End of lifecycle

Because wood is a renewable material, it can be incinerated with energy recovery at the end of its lifecycle, releasing the biogenic carbon into the atmosphere as carbon dioxide. The carbon dioxide can then be captured by photosynthesis to make more trees. This is a complete materials cycle, because the use of wood in the technosphere is coupled directly into the natural biogeochemical carbon cycle. However, before the end of life, there is the option to use the wood for multiple lives.

Incineration of wood products at the end of life yields environmental benefits, because substitution of biomass for fossil fuel results in a reduction in the emission of fossil carbon. The environmental benefits depend upon the origin of the wood (e.g., virgin wood or demolition wood) and the primary energy fossil fuel substitution (e.g., coal or gas) (Petersen Raymer 2006). Using incineration of wood also significantly reduces the amount of waste generated by demolition (Peuportier 2001). The use of woody biomass as a feedstock for biofuel production avoids the food vs. fuel conflict, which makes it more attractive from the environmental perspective. Hall and Scrase (1998) provided a literature review concerning greenhouse gas and energy balances of bioenergy. The LCA study revealed that results may differ due to the type and management of raw materials, conversion technologies, end-use technologies, system boundaries and reference energy systems with which the bioenergy chain is compared. Lindholm et al. (2010) modelled and calculated the environmental performance from an LCA perspective of different procurement chains of forest energy in Sweden. One of the conclusions of the study was that uncertainties and use of specific local factors for indirect effects (like land-use change and N-based soil emissions) may give rise to wide ranges of final results. Cherubini and Strømman (2011) performed a review of the bioenergy LCA literature, concluding that most LCAs found a significant net reduction in greenhouse gas emissions (GHG) and fossil energy consumption when bioenergy replaces fossil energy. Cherubini et al. (2009) explained that the initial use of biomass for products, followed by use for energy at end of life can further enhance climate change mitigation.

At the end of the building life, a decision must be made regarding the fate of the materials in the structure. Recovery of the calorific value of the wood at the end of one lifetime is not the only option.

With timber, that choice can be re-use in a similar form, re-use in a product with inferior properties, incineration with energy recovery, formation of biochar, or landfilling. Re-use in a series of products where the material properties of the wood are degraded with each life cycle is called cascading and may be the most efficient use of the material because it keeps the sequestered carbon out of the atmosphere for the longest time possible. However, determining whether cascading is always the best option requires careful analysis. The question is whether down-cycling leads to an overall reduction in process energy for the whole system or whether it is better to use virgin material for manufacture and incinerate at the end of life. In the paper by Morris (1996), it was concluded that in almost all the scenarios studied, recycling was more beneficial in energy terms for 24 of the 25 waste materials studied. However, a proper analysis of this issue requires detailed information in order to make the appropriate choices. Höglmeier et al. (2014) examined the environmental impacts of wood cascading compared with utilisation of primary wood using life cycle assessment. LCAs were performed for different cascading options of waste wood, compared with making the functionally equivalent products from primary wood. It was found that the cascading options created lower environmental impacts than the primary wood systems. Rivela et al. (2006a,b) investigated the most suitable option for the use of wood wastes. The study concluded that based on the environmental, economic and social considerations, the use of forest residues in particleboard manufacture is more sustainable than their use as fuel. Cascading through several life cycles prior to incineration may be a better option, although the arguments can get complex and careful analysis is required for each individual case. The re-use, or recycling of a wood product means that the opportunity for incineration with energy recovery is lost and that this energy must now be obtained from another source. A decision has to be made about how the environmental burdens of the initial product are allocation to the second, third, etc. products in the cascade chain.

Cascading only makes sense if there is sufficient industrial capacity and a large enough demand for board products made from wood waste. Contamination of wood waste with preservative treated wood is an issue that must be addressed and can involve investment in expensive technology to allow for sorting of different waste streams. This is unlikely to be a significant problem when dealing with large timber elements, such as cross-laminated timber panels and glue-laminated timber beams at end of life, when compared with smaller post-consumer items, such as painted joinery products. Although the recycling of timber residues involves a loss of quality of the material and is classified as down-cycling, there may be an argument for cascading the biomass through several product lives before final incineration with energy recovery (Haberl and Geissler 2000). It has been shown that the environmental footprint associated with particleboard production can be reduced by using recycled wood (Saravia-Cortez et al. 2013). Sathre and O'Connor (2010) studied this problem by comparing the energy and carbon balances of chains of cascaded wood products to those of products made from virgin fibre, or non-wood material.

The study explored different scenarios, including change in land use due to a decrease in demand for virgin timber, due to cascade use. Reuse constraints caused by preservative treated wood were not considered in the models. Four different cascade chains were analysed:

- A comparison of particleboard production using recovered or virgin wood;
- A comparison of material substitution for building construction using recovered wood, virgin wood, or reinforced concrete, with the forest limited and not limited;
- The sequential use of wood as a building frame material and particleboard, using either virgin or recovered wood, compared to providing the same services with non-wood materials, with the forest limited and not limited;
- A cascade chain of construction timber, particleboard, pulp, then energy recovery, with the forest not limited.

The results obtained showed that one of the most significant factors was the effect of forest land-use alternatives and the substitution effects in energy use and GHG emissions for different non-wood materials. When forest land was modelled as a limited resource, the available material options were limited to using recovered wood in a cascade chain and using non-wood substitute materials. Substitution effects assume greater importance as forest land becomes more limited. However, if forest production is not a limiting factor then a lower demand for wood products allows more biomass to potentially be used as fuel and substitution effects become less important. This depends upon the type of fossil fuel being replaced as an energy source. Cascade chains can be beneficial by allowing for greater substitution benefits per unit of harvested biomass. If the harvested biomass is not limited in supply, then cascading gives relatively minor benefits. However, if forest land is limited, then cascading allows for greater wood use and substitution of non-wood materials, which provides environmental benefits.

There have been some studies of the manufacture and properties of particleboard manufactured from wood waste (e.g., Yang et al. 2007; Wang et al. 2008). Recycled paper sludge has also been investigated as a potential source of raw material (Taramian et al. 2007). One of the problems with the recycling of waste wood, is the potential presence of wood preservatives, such as copper chromium arsenic (CCA), pentachlorophenol, tributyl tin oxide and other toxic substances that were used in the past (Yu and Kim 2012). This requires a rapid means of determining the presence of such substances, or a robust chain of custody system for materials with a known provenance. The production of cement-bonded particleboards from recycled CCA-treated wood has been suggested as a potential use for this waste stream (Chen and Cooper 2000). Recycling of the wood material derived from particleboard using a hydrothermal process has been shown to result in a significant deterioration in the mechanical properties of the particleboard made from recycled material, particularly after the second cycle (Lykidis and Grigoriou 2008). In order to prevent a deterioration in mechanical properties, a limit on the amount of recycled material in the furnish would be advisable. The drying of wood is the most energy intensive part of the particleboard manufacturing process and using waste wood can result in savings of up to 80% in drying energy (Sathre and Gustavsson 2006).

LCAs of wood products sometimes assume that the timber is disposed to landfill at the end of the lifecycle and that the material is converted to methane over a period of time. This can have a major impact upon the outcome of LCAs. In one such study, where wooden and concrete railway sleepers were compared, it was found that wooden railway sleepers were a far worse choice from an environmental perspective if they were landfilled at the end of life, but the opposite conclusion was drawn if the timber was incinerated with energy recovery (Engberg and Eriksson 1998).

These assumptions regarding methane emissions are based on laboratory studies of the decomposition of wood under anaerobic conditions, rather than field experiments but actually, very little is known about the degradation of wood products under landfill conditions (Barlaz 2006). Indeed, some workers state that landfill can have benefits in terms of carbon storage. Barlaz (2006) estimated that the global carbon storage of wood in landfill amounted to 119 million tonnes. Christensen et al. (2009) estimated that the carbon storage in landfills amounted to an offset of 141-261 kg CO_2 eq. per tonne of waste disposed. De la Cruz et al. (2013) measured the fraction of biogenic carbon in excavated landfill waste and found that the average biogenic carbon content of these samples was 64.6 +/- 18.0%. Wang et al. (2013) found that lignified materials exhibited low levels of degradation under anaerobic conditions, although paper products exhibited much higher levels of degradation. Micales and Skog (1997) estimated that only 30% of carbon from paper and 0-3% of carbon from wood products is ever emitted as landfill gas. In Europe, disposal of waste wood to landfill is prohibited and it must either be reused or incinerated (Krook et al. 2004, Carpenter 2013). In Norway, landfilling of biogenic material has been banned since 2009.

The combustion of 1 kg of wood with a moisture content of 20% and a carbon content of 41% has a lower heating value of 13.1 MJ/kg, and produces 0.115 kg CO_2/MJ (Jungmeier et al. 2003). Concerns have been expressed regarding the levels of heavy metals in ash from incinerated wood waste. Ash from incinerated Swedish wood waste contained high levels of arsenic, chromium, zinc, nickel and copper, whereas the ash from imported wood waste contained elevated levels of lead, mercury and cadmium (Krook et al. 2004). This material has to be disposed of as hazardous waste. A report on the metal content of ash from bioenergy plants in Norway shows that whereas fly ashes had a high concentration of heavy metals, the bottom ash had potential to be used as a fertiliser (Horn et al. 2016). There is potential for wood ash to be used as a clinker substitute in cement production. The ash also contains a proportion of unoxidised carbon, which represents an opportunity for the permanent storage of atmospheric carbon.

7 Review of LCA of cement and concrete

7.1 The cement life cycle

In terms of volume, cement products are the second largest commodity of material consumed by society after water. In 2010, the world's cement production was about 3.3 Gt, expected to rise to 3.5 Gt by 2020 (Akashia et al. 2011) and 3.7-4.4 Gt by 2050 (Earth Institute 2012), but these figures are low. Dossche et al. (2016) (quoting Cembureau) reported that the global production of cement was 4.3 Gt in 2014. Cembureau state in their 2015 report that this figure was 4.6 Gt in 2015. Imbabi et al. (2012) state that cement production may be as much as 5.7 Gt per annum by 2050. The cement market is dominated by ordinary Portland cement.

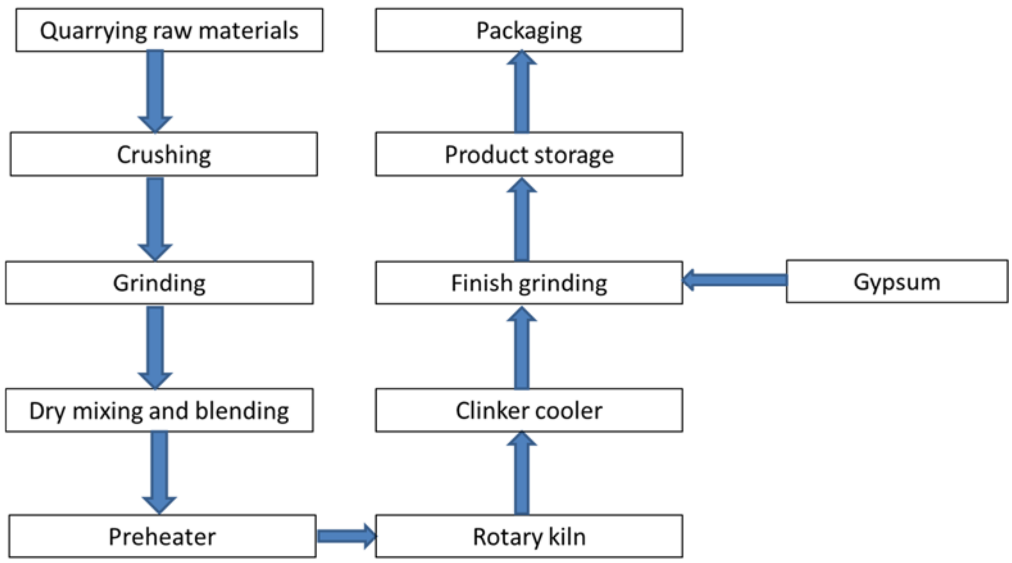

Figure 4 Flow diagram for the production of cement

The cement manufacturing process consists of three main stages:

- raw material preparation;
- clinker production (pyro processing);
- clinker grinding and processing.

For Portland cement production, raw materials mainly consist of limestone, magnesium carbonate, silica, alumina and iron oxide. The raw materials are quarried and transported to the manufacturing site where they are crushed and ground before entering a pre-heater and then a kiln. After size reduction, the powdered ingredients are sent to a pre-heater where spent flue gas from the calciner is used to heat the mixture to approximately 550°C, from here the mixture is sent to the calciner (kiln) where it is heated initially at 600°C and finally to 1400-1600°C. Portland cement requires approximately 1.5 tonnes of raw materials to make one tonne of cement (Gao et al. 2016a,b). Portland cement is largely composed (93-97%) of a material known as clinker. Clinker is produced when limestone is heated to high temperatures in a cement kiln. During this process, the calcium carbonate ($CaCO_3$) in the limestone decomposes to form calcium oxide (CaO) accompanied with the emission of carbon dioxide. The amount of material required to make one tonne of clinker varies from 1.5-1.69

tonnes (Li et al. 2014). The processes in a kiln can be divided into four zones: the decomposition, transition, sintering, and cooling zones (Ishak and Hashim 2015). The clinker exiting the kiln is cooled and the excess heat is routed to the pre-heater unit. The chemical constituents of clinker are shown in Table 12.

Table 12 Chemical constituents of clinker and main properties (Ishak and Hashim 2015)

Constituent	Name	Chemical formula	Properties
C_2S	Belite	Ca_2SiO_4	Slow hardening, slow strength development
C_4AF	Ferrite	Ca_4AlFeO_5	Slow hardening, causes grey colour
C_3A	Aluminate	$Ca_3Al_2O_6$	Quick setting, which is retarded by gypsum
C_3S	Alite	Ca_3SiO_5	Rapid hardening, giving early strength

Within the pre-heater and kiln systems there are filters to capture the fine particles of unburnt and partially burnt material entrained in the flue gases. This material is known as cement kiln dust (CKD), which is either stored in stockpiles on site or disposed to landfill. It has been estimated that the CKD represents something of the order of 15-20% of clinker production by mass (Van Oss and Padovani 2003). Some facilities recycle some or all of the CKD back into the raw material entering the kiln, but the degree to which this is possible depends upon the trace metal composition and upon the potential for alkali-silica reactions within the kiln, which can vary widely from plant to plant (Huntzinger and Eatmon 2009). CKD is potentially a hazardous waste because of its alkalinity and small particle size, making it a respiratory, eye and skin irritant, as well as the tendency of contaminants to concentrate in this material. The detrimental environmental and health impacts can potentially be reduced by reaction of the CKD with carbon dioxide process emissions (Eatmon 2009).

Norcem (Heidelberg cement group) is Norway's only producer of cement, producing a total volume of 1.7 million tonnes of cement per year in 2 production plants (Brevik, Kjøpsvik). The entire chain of custody of concrete in Norway has about 5 000 employees (BEF 2015) working in the areas of ready mixed, site mixed and precast concrete. Production took place in 200 ready-mix concrete stations and 50 precast concrete plants (BEF 2015). In 2014, 5 million m^3 concrete were produced in Norway, consuming 2 million tonnes of cement. Compared with many other European countries, Norwegian produced concrete has the highest average cement content (382 kg/m^3), resulting in a production of concrete in high strength classes. Usually, 55 kg/m^3 fly ash are added to the ready mixed concrete (0.03 million m^3, 2014) (REMCO 2015).

The total turnover of the ready mix and precast concrete industry in 2014 was together by about 8.5 billion NOK (REMCO 2015, BEF 2015). Statistics Norway reports a total turnover for the production of concrete, cement and gypsum products of 14.8 billion NOK in 2015. According to the Norwegian Construction Products Association (CPA), Cement and concrete products with a value of 1 713 million NOK were imported, in contrast to an export value of 169 million NOK. The most important import countries are Sweden and Germany, while exported goods go to neighbouring countries such as Denmark, Finland and Sweden (CPA 2015). For the sector of precast concrete products, only 12.7% (11.3% in 2015) were imported in 2014.

7.2 Cement production and the environment

It is usually stated that the cement industry is responsible for around 5-7% of all CO_2 emissions globally. Ekolu (2016), Feiz et al. (2015a,b), Hasanbeigi et al. (2012) quote a value of 5%, whereas Shen et al. (2015) and Teklay et al. (2015) state 8%. In 1990, cement production was responsible for emitting 0.576 Gt of CO2; this had risen to 1.88 Gt by 2006 and is expected to reach 2.34 Gt per year by 2050, according to Benhelal et al. (2013). Gursel et al. (2014) state that 3 Gt of Portland cement was produced in 2011, resulting in carbon dioxide emissions of approximately 2.6 Gt. Using the IPCC values for cement process carbon dioxide emissions and Cembureau production figures (4.6 Gt), this places the global CO_2 emissions from cement production at a conservative value of 3.76 Gt in 2015, which represents slightly more than 10% of total global anthropogenic emissions in that year.

Cement production is both energy- and emission-intensive. The production of 1 tonne of cement consumes 60-130 tonnes of fuel and 110 kWh of electricity (Shen et al. 2014). The cement industry consumes 12-15% of total global industrial energy use (Ali et al. 2011). In Europe, typical emissions for production are 700 kg CO_2 eq. per tonne of cement and 900-935 kg CO_2 eq. per tonne in China, India and the US (IPCC 2007), although these do not include GHG emissions associated with quarrying and transportation (Imbabi et al. 2012). Li et al. (2015) quote a value close to 800 kg CO_2 eq. per tonne of Portland cement manufactured in China and about 780 kg CO_2 eq. per tonne in Japan. The embodied energy of cement ranges from 3.5 MJ/kg to 5.5 MJ/kg, averaging 4.1 MJ/kg in Europe (Ali et al. 2011), 4-5 MJ/kg (Gursel et al. 2014). Dossche et al. (2016) state that the production of one tonne of cement requires 1.5 tonnes of raw materials and between 4.0-7.5 GJ of energy and involves the emission of approximately one tonne of CO_2. The electricity demands for a cement plant in Europe range from 90 to 150 kWh/t of cement (García-Gusano et al. 2015).

Carbon dioxide emissions are both indirect, due to use of energy for production and direct, due to the release of CO_2 during the calcining process. Calcination is the process whereby the calcium carbonate in the limestone is converted to calcium oxide with the emission of carbon dioxide. This is typically responsible for around 530 kg of the CO_2 emitted per tonne of clinker (Habert et al. 2010). The grinding processes are responsible for CO_2 emissions of approximately 100 kg CO_2 eq. per tonne of cement; but this depends upon the electricity generation fuel mix. Clinker is a mixture of di- and tricalcium silicates, tricalcium aluminate and tetracalciumaluminoferrite (Galvez-Martos and Schoenberger 2014). Clinker production has different embodied energy values depending upon the process used (Table 13).

Table 13 Clinker embodied energy values (Galvez-Martos and Schoenberger 2014)

Process	Embodied energy (MJ/kg clinker)
Wet kiln	5.0 - 7.0
Semi-wet kiln	3.5 - 5.0
Semi-dry kiln (Lepol kiln)	3.1 - 3.8
4-stage cyclone preheater kiln	3.0 - 4.2
5-stage cyclone preheater kiln with pre-calciner	2.9 - 4.0

A simple mass balance calculation shows that for each tonne of CaO produced in the clinker 786 kg of carbon dioxide will be released to the atmosphere, without taking any other factors into account such as GHGs emitted due to the energy required, or transportation, quarrying, etc. However, the clinker raw material is not derived solely from calcium carbonate and the actual figure is closer to one third of the raw material weight (Gao et al. 2016). The IPCC default value for reporting CO_2 emissions from the calcination process is 520 kg/t of clinker, while the Cement Sustainability Initiative recommend using

547 kg per tonne of clinker produced (Gao et al. 2015). Emissions from kilning have been reported by different studies to be in the range 450-553 kg per tonne of clinker (Mohammadi and South 2017). Gursel et al. (2014) state that the production of one tonne of clinker for Portland cement manufacture results in the emission of 870 kg of CO_2. The process of converting $CaCO_3$ to CaO is called calcination, which typically causes more than 50% of the carbon dioxide emission from the whole cement production process. Within the kilning process, calcination is responsible for 63% of carbon dioxide emissions and fuel emissions. One way of reducing the carbon footprint of cement is, therefore, to reduce the amount of clinker (Feiz et al. 2015a,b).

Other pollutants are produced in the kilning process: NOx, 3.8%; SO_2, 1.7%; CO, 2.5% and VOC, 0.15% (García-Gusano et al. 2015, Kim and Chae 2016a). SOx emissions are derived mainly from the combustion of sulphur-containing fuels, although there can be some contribution from non-fuel raw materials. However, the clinker production process also acts to absorb about 70% of the emitted SOx. This ability to handle high sulphur content fuels is one of the positive benefits of the cement industry. High temperature combustion in the lime kiln generates NOx, with 90% of the emissions being NO and the remainder NO_2 (Van Oss and Padovani 2002). Valderrama et al. (2012) give an example of a cement manufacturing facility in Spain where design improvements have resulted in a reduction of 20% in NOx and over 50% in SO_2. Cement production also results in emissions of heavy metals (Hg, Cd, Ti, Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V), particulates and polyaromatic hydrocarbons (PAHs) (Strazza et al. 2011).

The calcination stage of the process uses about 90% of the total energy for cement production (Worrell et al. 2001). Lime plus clay and clay-like minerals are added to rotary kilns where calcination takes place at 1450°C, which requires a theoretical energy between 1590-1840 kJ per kg of clinker (Galvez-Martos and Schoenberger 2014). However, the real energy demand is considerably higher, since energy is used to evaporate water and to compensate for radiation losses. A highly efficient coal-fired kiln using dry raw materials consumes about 3 GJ per tonne of clinker and produces 3.6 tonnes of CO_2 per tonne of clinker (Gartner 2004). This is close to the practical efficiency limit for a coal-fired kiln. Kiln energy consumption dropped from close to 4 GJ per tonne of clinker in the 1970's to about 3.3 GJ per tonne in 2000. It is unlikely that this value will fall below 3 GJ per tonne of clinker (Habert et al. 2010). Van Oss and Padovani (2002, 2003) quote a range of 3.2 to 6.3 GJ per tonne of clinker. Wet process kilns consume about 5-7 GJ, semi-dry between 3.1-5.4 GJ, pre-heater kilns 3.0-4.2 GJ and kilns with pre-heater and pre-calciner about 2.9-4.0 GJ per tonne of clinker (Galvez-Martos and Schoenberger 2014). Gas fired-kilns will produce lower levels of CO_2 for each unit of energy produced, and it is possible to use 'carbon neutral' fuels, such as agricultural and forestry biomass, biodegradable municipal waste and paper waste (Mikulčić et al. 2016). The calcination process has associated GHG emissions of 0.97-1.24 tonnes CO_2 eq. per tonne of clinker, 60% of which is chemically generated (Zhang and Mabee 2016). This footprint can be reduced by using low carbon fuels. For example, Zhang and Mabee (2016) reduced the GWP of calcination from 971.4 kg CO_2 eq. to 582.6 kg CO_2 per tonne of clinker by substituting fossil fuel energy sources with wood waste. Waste materials derived from fossil fuels, such as solvents, plastics and used tyres are not considered carbon neutral (Habert et al. 2010, Gäbel and Tillman 2005). The best fuel efficiency for a cement kiln is of the order of 54 g CO_2 per MJ of energy and many kilns exceed this consumption (Habert et al. 2010). A variety of technological changes can be applied to improve the energy efficiency of cement production, including improvements in raw material preparation, clinker production, cement grinding, finish grinding and raw material substitution; leading to energy savings as high as 3.4 GJ and emissions savings of 212.54 kg CO_2 eq. per tonne of cement (Madlool et al. 2013). New efficiency measures for the recovery of waste heat include the use of the Organic Rankine cycle, which allows for the generation of electricity from relatively low temperature flue gases (Schneider et al. 2011).

The clinker that is produced is cooled and ground to a fine powder, with other ingredients, such as gypsum, added to improve the properties of the cement. Grinding requires approximately half the electricity consumption of a cement plant (Gartner 2004), with the associated carbon emissions depending upon the local grid primary energy mix. There are many initiatives taking place to reduce the GHG emissions associated with cement production. These include recovering waste heat from flue gases, clinker substitution, changing from wet to semi-wet or dry processes, using waste or biomass as an energy source, or carbon capture and storage (CCS).

CCS may double the price of cement (Benhelal et al. 2013). CCS is a capital-intensive process, which has been estimated to cost between 24-75 Euro per tonne of captured CO_2, depending on the CO_2 concentration of the emissions, the process used and the amount of captured CO_2 (Schneider et al. 2011, Ni et al. 2016). It is also an immature technology which requires further development before deployment and carries with it an energy penalty (McLaren 2012, García-Gusano et al. 2013). The main impact in reducing CO_2 emissions comes from a decrease in the amount of calcination through clinker substitution (Damtoft et al. 2008, Ammenberg et al. 2015).

There are many different types of cement, which have different clinker contents, with the remainder being referred to as supplementary cementitious materials (SCMs); the standard EN 197-1 classifies 27 different types of cement in five groups. According to EN 197-1 a CEM I (Portland) cement must contain 95% clinker, a CEM II Portland composite cement (65-94%) clinker, a CEM III blast furnace cement (5-64%) clinker, CEM IV is a Pozzolanic cement with 45-89% clinker and CEM V a composite cement with 20-64% clinker. The SCMs act either as latent cements or pozzolans. The latent cements are weakly hydraulically cementitious materials that become strongly cementitious when reacted with lime and can partially replace clinker. Pozzolanic materials are non-cementitious silicate-based materials that become hydraulically cementitious when reacted with free lime. The GWP impact of a cement decreases almost linearly with the substitution ratio for the clinker (clinker factor) (Galvez-Martos and Schoenberger 2014). Clinker manufacturing is responsible for about 85% of the environmental impact of cement and by substituting this for other materials the GWP impact can be substantially reduced, to as low as just over 200 kg CO_2 eq. per tonne of cement at a clinker factor of 0.2. Reducing the clinker factor is the most effective way of reducing the GWP impact of cement.

Examples of clinker substitutes are industrial by-products, such as ground granulated blast furnace slag (GGBFS) from the iron and steel industry, silica fume from the semiconductor industry, fly ash from coal-fired power plants (Meyer et al. 2009, Liu et al. 2012, Miller et al. 2015), fly ash from biomass power plants (Teixeira et al. 2016), or bottom ash from municipal incineration facilities (Schneider et al. 2011). The use of GGBFS is considered to lead to greater reductions in environmental burdens, compared with the use of fly ash (Tait and Cheung 2016). However, there are concerns that the use of additives such as slags and pulverised fuel ash results in increased carbonation depths compared to unblended cements, leading to concerns about the life expectancy of reinforced cements and concretes due to corrosion of the steel reinforcing elements (Pacheco Torgal et al. 2012).

Type I cements typically have associated GHG emissions of 800 kg per tonne of cement, although some estimates as low as 355 kg per tonne are undoubtedly not accounting for some aspect of associated emissions, due to differences in the associated system boundaries (Josa et al. 2007). Kim et al. (2016b) gives a GWP value of 948 kg CO_2 eq. per tonne of cement. Mikulčić et al. (2016) quote a global average value of 830 kg CO_2 eq. per tonne of cement.

There have been several EPDs (published by the Norwegian EPD foundation) for cement manufacturing in Norway covering modules A1-A3 (cradle-to-gate). The GWP and embodied energy data per tonne of cement is reported in Table 14.

Table 14 Published EPD data for Norwegian cement production (per tonne of cement)

Declaration no.	Cement type	GWP (kg CO_2 eq.)	EE (MJ)
NEPD-211-199-NO	CEM II/B-M	694	4654
00151N rev1	Portland with fly-ash	503	3853
NEPD 00023N rev1	CEM I	748	5484
NEPD-1218-383-NO	CEM I	859	5559
NEPD-24-201-NO	CEM II	604	4342
		637	4799

A survey of published EPDs for cement production was also undertaken as part of this project, with the data reproduced in Figure 5 (for modules A1-A3, EN 15804). The data is shown for a functional unit of 1 kg of cement and the cement production has been broken down as CEM I, CEM II, CEM II and CEM IV, where this was reported in the EPD, otherwise it is shown in the category ‘other’. Also included is the embodied energy data for the cement, which was calculated by summation of the categories (EN 15804):

- use of renewable primary energy excluding renewable primary energy resources used as raw materials;
- use of non-renewable primary energy excluding non-renewable primary energy resources used as raw materials.

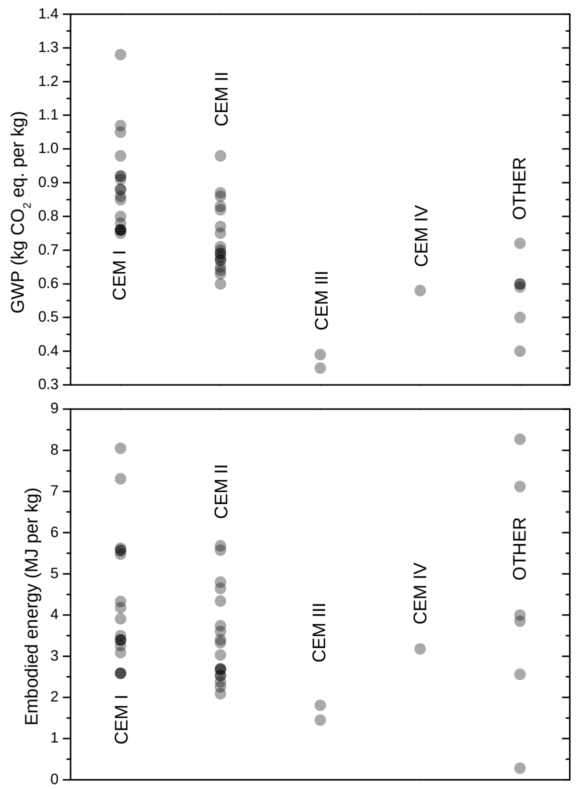

Figure 5 Comparison of GWP impact and embodied energy for cement from published EPDs

The relationship between the GWP impact and the embodied energy for cement production is shown in Figure 6, with the Norwegian cement production shown in magenta.

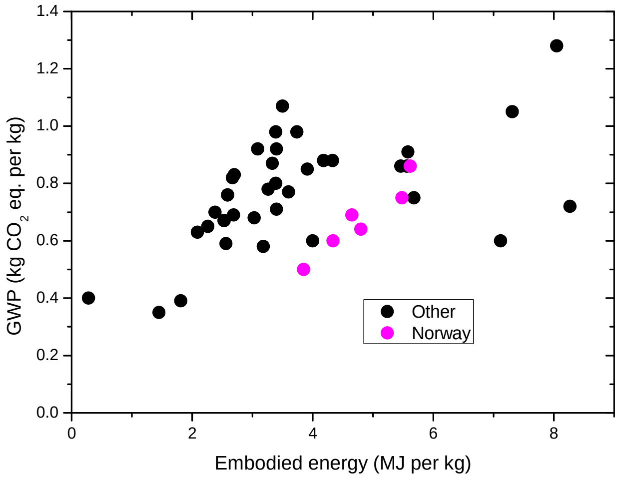

Figure 6 Relationship between embodied energy and GWP impact for cement production.

The Norwegian production (shown in magenta) generally exhibits the lowest GWP impacts for any given embodied energy. There are two data points which have a very low GWP impact for high embodied energies, which do not appear to be representative of the whole data set. These are for an IBU EPD (MR-ENV-EPD-ICG-20160002-EN) associated with two products from the Italcementi Group of Italy, which use high levels of recycled aluminium-based materials and lower kilning temperatures.

There is some potential for replacing Portland cements with magnesia-based cements in specialist applications, but due to the lower pH of these cements they are not suitable for use with reinforcing steel (Walling and Provis 2016). Magnesia cements use lower production temperatures (800^{o}C) compared with Portland cement, but release more CO_2 per unit mass compared to calcium carbonate. However, when carbonation of magnesia cements is taken into account their net CO_2 emissions are 73% lower than Portland cements (Ruan and Unluer 2016). Their potential for replacing Portland cements is not large.

Portland cement is the primary source of embodied CO_2 emissions in concrete, being responsible for 74-81% of the total emissions. The next highest impact comes from the production of coarse aggregates, of which 80% is attributable to the grinding/size reduction. Fine aggregates require far less energy because they are only graded and not crushed (Flower and Sanjayan 2007). The use of recycled concrete aggregate in new concrete, or replacement of some of the Portland cement by fly-ash, or blast furnace slag has been investigated (Berndt 2009, 2015) and this is discussed in more detail later. The production of Portland cement is responsible for 74-81% of the total CO_2 emissions associated with concrete and coarse aggregates contribute 13-20% (Dossche et al. 2015). The reinforcement steel can also make a significant contribution.

7.3 Carbonation

Carbonation is the reaction of carbon dioxide with hydrated (hardened) cement. Atmospheric carbon dioxide dissolves in the pore water of the cement where it forms weak carbonic acid. The acid dissociates and reacts with calcium hydroxide and then calcium-silicate-hydrates in the cement to form calcium carbonate. Carbonation is now receiving increasing attention due to the claimed environmental benefits that it confers. The degree of carbonation is the ratio between the carbonated and available calcium oxide in the cement. Carbonation in concrete results in a reduction of pH leading to corrosion of steel reinforcement, reducing the life of reinforced concrete structures. In addition, carbonation leads to weathering degradation of the cement phase, reducing the durability of the concrete (Jang et al. 2016, Lo et al. 2016, Šavija and Luković 2016).

Carbonation reactions of cement based materials occur at a very low rate in the natural environment due to the low concentration of atmospheric CO_2. Due to the slow rate of reaction, laboratory studies use accelerated carbonation reactions, which are obtained by using higher CO_2 concentrations, higher temperatures and higher relative humidity (RH). The extrapolation of these accelerated studies to real life behaviour is in some doubt, however. Ekolu (2016) states that: 'there is a severe lack of research data on the effects of CO_2 levels on natural carbonation; extensive and detailed investigations are needed'.

The carbonation rate is highest at 40-80% RH, Ekolu (2016) states 50-70%. The reaction of carbon dioxide with calcium hydroxide in the cement will take place provided that sufficient water is present. Since carbonation is controlled by the rate of diffusion of CO_2 through the structure, only partial carbonation of concrete structures can occur. This is advantageous, since carbonation deep within a structure will lead to corrosion of reinforcing elements and will compromise structural integrity (Ahmad 2003, Ashraf 2016). For this reason, concrete structures are designed to minimise carbonation reactions.

The rate of carbonation is very dependent upon cement or concrete composition and is reduced for higher strength concrete (which is of lower porosity than low strength concrete), concrete in outdoor unsheltered exposure conditions (Shi et al. 2016a, Ta et al. 2016, Leeman and Moro 2017), or concrete with additives such a blast furnace slag or fly ash (García-Segura et al. 2013). Carbonation of an exterior concrete wall protected from the rain will proceed faster than on an interior wall with a wall covering. The rate of carbonation can vary greatly, Börjesson and Gustavsson (2000) report that the carbonation front travels into the concrete at a rate of 0.1-1.0 mm/yr. Gustavsson et al. (2006b) estimated that the CO_2 re-absorbed by carbonation is around 8% of that emitted from the calcination operation, based upon a study by Gajda and Miller (2000) and Gajda (2001). They studied carbonation depth in over 1 000 samples of different ages during the use phase of the lifecycle and considered that the CO_2 taken up in carbonation accounted for 3-4% of the emissions from the production of cement. Lee et al. (2013) stated that the CO_2 uptake did not exceed 3% of emissions from the production of concrete. Kim and Chae (2016b) put this figure at close to 11%. Kjellsen et al. (2005) claimed that the amount of CO_2 sequestered by concrete during a 100-year lifetime (70-year service life and 30 years of exposure after demolition) of the material is equal to 25% of that emitted by the calcination process. The theoretical carbon dioxide uptake has been estimated to be in the range 0.08-0.55 tonnes CO_2 per tonne of material and the experimentally measured range is in the region of 0.02-0.43 tonnes CO_2 per tonne (Jang et al. 2016). Wu et al. (2014c) argue that since many LCA studies are cradle to gate, they do not take proper account of the carbonation reactions of concrete, which occur during the use-phase and end-of-life phase. They conducted a review of LCA's of concrete and concluded that the cradle to gate studies could be biased and either overestimate or underestimate the environmental impact of concrete use, depending on the circumstances.

In many environmental comparisons between timber and concrete, the carbonation of concrete during the end of life phase is generally not considered (Kjellsen et al. 2005, Collins 2010). The focus instead has been on carbonation during the in-service phase of the lifecycle. It has been argued that the CO_2 uptake of concrete has been underestimated in such studies, leading to an overestimation of the lifecycle GHG emissions of concrete (Pade and Guimares 2007, Engelsen et al. 2005, Wu et al. 2014c). Carbonation will take place much faster in crushed concrete than in concrete structures due to the increased exposed surface of the former, although even under optimal conditions, only 75% of the calcium oxide in cement is estimated to be carbonated (Engelsen et al. 2005). Pade and Guimares (2007) determined that between 33-57% of the CO_2 released by calcination was re-absorbed by carbonation, during a service life of 70 years, followed by crushing and exposure to air for a further 30 years. However, Andersson et al. (2013) concluded that only 11% of the theoretical level of carbonation occurs in concrete in the end of life phase. This is because the crushed concrete is stored in piles through which there is limited air flow. To ensure that there is exposure of the surfaces of the crushed material to atmospheric carbon dioxide requires an energy input with associated environmental impact, because the piles would have to be turned many times during the storage period (Butera et al. 2015). It is also possible that carbonation would lead to increased oxyanion leaching (As, V and Sb) and Butera et al. (2015) state that more research is needed to understand this. Typical uses for crushed concrete are construction landfilling, road bases, or building foundations. There is limited CO_2 access in such situations. Dodoo et al. (2009) point out that the carbon emissions associated with the crushing operation significantly reduces the benefits associated with the subsequent carbonation of the material and that the emissions associated with the crushing operation can exceed the absorption of CO_2 due to carbonation of the crushed material. Transportation of the recycled concrete aggregate can also have a significant environmental impact, particularly for long transport distances (Ding et al. 2016).

Pade and Guimares (2007) calculated the CO_2 uptake in buildings in Norway, Sweden, Denmark and Iceland due to carbonation over a 100 year period. This was for a 70 year service life followed by 30 years storage after demolition. They obtained information regarding concrete production from the European Ready Mix Concrete Organisation database. Andersson et al. (2013) estimated that carbonation in concrete in existing structures in Sweden amounted to about 300 000 tonnes of CO_2 uptake, which was 17% of the total emissions (calcination and fuel) from the production of new cement in that country in the same year. Xi et al. (2016) calculated that the global concrete carbonation sink had increased from 0.10 Gt C per year in 1998 to 0.25 Gt C per year in 2013, concluding that carbonation represented a significant carbon sink that was not currently included in inventories. The carbonation during service life, demolition and reuse phases was calculated. During the demolition phase, it does not appear that the limited exposure to atmospheric CO_2 due to storage in piles was assumed.

7.4 LCA studies of cement and concrete

A comprehensive review of the published LCAs of cement was conducted by Van den Heede and De Belie (2012). These show that the most significant environmental impacts are due to the calcination process and that a reduction of clinker content of the cement will be the most effective strategy for reducing the impact. The use of ground granulated blast furnace slag (GGBFS), coal combustion fly ash, or foundry sand to reduce the environmental impact of cement will have different effects upon the final LCA depending upon the allocation procedures used (Chen et al. 2010a,b). Although both materials are wastes from another industrial sector, if purchased for use in cementitious products, then an allocation procedure must be used, which pulls through some of the environmental burdens associated with the source of those materials. If a mass allocation procedure is used, then this has the advantage of being stable, but can be viewed as being advantageous to the steel or coal sectors producing the wastes, whereas the cement sector would take a disproportionate share of the environmental burdens, for using what is essentially a waste. An economic allocation might be a better way of assigning the burdens, but this is not stable due to price fluctuations. Chen et al. (2010a,b) take the view that no allocation is preferable, but that allocation is to be preferred over system expansion if a choice must be made. Flower and Sanjayan (2007) found that the use of GGBFS and fly ash both resulted in reductions in embodied carbon emissions, although allocation procedures were not reported. Substitution of clinker will avoid the environmental burden of the clinker that is not used and a reduction in environmental impact will be the result. Lee and Park (2005) advocated a system expansion approach, where the environmental loads associated with the recycling process are included in the product system under investigation and the loads associated with the substituted system are subtracted. This would involve subtracting the burdens associated with the replaced cement and only including burdens for the GGBFS or fly ash associated with transport or material preparation for incorporation. Any burdens associated with the steel manufacture or coal firing would not be included in their approach. Pushkar and Verbitsky (2016) note different outcomes from an LCA study of blends of cements with different man-made pozzolanic additives depending upon the allocation method used, although a Portland-slag blend was found to possess the best environmental profile in all cases.

Habert (2013) points to a recent EU Directive European Union Directive (2008/98/EC) (mis-referenced by Habert 2013), which states that:

'*a substance or object, resulting from a production process, the primary aim of which is not the production of that item, may be regarded as not being waste but as being a by-product only if the following conditions are met: a) further use of the substance or object is certain; b) the substance or object can be used directly without any further processing other than normal industrial practice; c) the substance or object is produced as an integral part of a production process; and d) further use is lawful, i.e. the substance or object fulfils all relevant product, environmental and health protection requirements for the specific use and will not lead to overall adverse environmental or human health impacts*'

This means that GGBFS and fly ash can no longer be considered waste products, but must be treated as by-products. It is therefore necessary to allocate environmental burdens to their use in cement. As an alternative to allocation, it is possible to use system expansion coupled with the avoided burdens approach (Saade et al. 2015, Turk et al. 2015). This considers that the use of the GGBFS by product removes environmental burdens that would otherwise occur if the material was landfilled. Teixeira et al. (2016) used an economic allocation for fly ash from a coal burning plant, where 0.03% of the burdens from the coal plant were allocated to the fly ash. However, no burdens were assigned to biomass-derived fly ash, since it has no economic value.

Whatever choice is made, be it mass allocation, economic allocation, system expansion, or treating the material as a waste with zero burdens, this will affect the outcome of the LCA study and this needs to be recognised and the allocation method clearly stated.

LCA studies have shown that the use of GGBFS and fly ash do result in a reduction of environmental burdens of concrete (Gursel et al. 2014, Jiménez et al. 2015, Miller et al. 2016a,b, Salas et al. 2016). However, Crossin (2015) points out that the GHG reductions often claimed as a result of adding ground blast furnace slag only apply when the supply of this material is unconstrained, which was shown not to be the case in a market analysis; in addition, the steel industry is making efforts to reduce slag production (Matsumiya 2005). It is possible to transport fly ash for clinker substitution over long distances and still reduce overall GHG emissions (O'Brien et al. 2009). In 2005 the world production of BFS was around 150 million tonnes and clinker production was 2 500 million tonnes, so substitution is not going to be a major contributor to reducing CO_2 emissions globally (Habert et al. 2010). In any case, substitution rates above 50% result in a product with inferior properties (Habert 2013). Substitution of fly ash by rice husk ash has been suggested (Gursel et al. 2016). There is about 156 million tonnes of rice husk produced globally and the ash content is about 20%, so although there is some local potential for use, this would make little difference with a global production of 4.6 billion tonnes of cement in 2015 (Cembureau). In a growing global market for cement, there will not be an overall reduction in clinker production, even with substitution.

Huntzinger and Eatmon (2009) conducted LCA upon four different types of cement: (a) a traditional Portland cement, (b) blended cement (natural pozzolans), (c) cement where 100% of waste cement kiln dust is recycled into the kiln process, and (d) Portland cement produced when cement kiln dust (CKD) was used to sequester some of the process related CO_2 emissions. Recycling of CKD had no overall effect on the environmental impact, but using natural pozzolans to replace some of the clinker and using CKD to sequester some of the CO_2 emissions both reduced the impact.

Moretti and Caro (2016) conducted an LCA of cement production in Italy based upon the core product category rules (PCR) of EN 15804 and the International EPD System PCR 2010:09 version 2.1 of the International EPD System. The following GWP results were obtained from this study for different cement types produced in Italy (Table 15).

Table 15 **GWP (kg CO_2 eq. per tonne) (Moretti and Caro 2016)**

	Clinker	Cement	CEM I	CEM II	CEM IV
Mean	964.3	801.6	972.3	794.1	710.9
SD	42.2	125.0	47.9	82.6	73.6
Maximum	1032.4	1045.5	1045.5	985.1	802.7
Minimum	901.8	581.5	861.6	630.4	585.4

This compares with an earlier study by Strazza et al. (2010), which studied 7 cement plants in Italy and reported a GWP of 769 (+/-74) kg CO_2 eq. per tonne of cement, with predominantly CEM II production. Salas et al. (2016) reviewed 18 LCA studies of cement manufacture, giving GWP values as low as 265 kg CO_2 eq. per tonne of cement up to 987 kg CO_2 eq. per tonne (Table 16).

Stafford et al. (2016) quoted 632 kg CO_2 eq. per tonne of cement, this was compared to a study by Chen et al. (2010a,b) who quoted 782 +/-141 kg CO_2 eq. per tonne of cement. García-Gusano et al. (2013) reported 799 kg CO_2 eq. per tonne of cement for production in Spain in 2010, reducing to a projected 626 kg CO_2 eq. per tonne by 2030. This was reduced to a projected 534 kg CO_2 eq. per tonne of cement in 2030, if post combustion CO_2 capture (CCS) was employed.

Table 16 **GWP for the manufacture of 1 tonne of cement (from review of Salas et al. 2016), Portland cement = PC, BAT = best available technology, GBFS = ground blast furnace slag**

Reference	Material	GWP (kg CO_2 eq.)
Li et al. (2014)	PC - China	799
Valderrama et al. (2012)	PC old production	987
	PC – best available technology	938
Chen et al. (2010a)	Case study	782
	France mean	899
Feiz et al. (2014a,b)	Clinker	850
	PC 92% clinker	779
	Blended 50% clinker w. GBFS	452
	Blended 27% clinker w. GBFS	265
Chen et al. (2014)	PC dry kiln large production	734
	PC dry kiln moderate production	801
	PC dry kiln small production	693
	PC shaft kiln	1000
García-Gusano et al. (2013)	2010 base case	799
	2030 base	620
	2030 PCC	530
García-Gusano et al. (2014)	Clinker – Spanish industry	929
	cement	799

Song et al. (2016) give a breakdown of the environmental impacts associated with different parts of the cement production chain for typical pre-heater dry cement process Portland cement production in China (Table 17). This shows very clearly that the clinker kiln calcination process has the highest impact.

Table 17 GWP impact associated with different stages in cement production (Song et al. 2016)

Stage	GWP Impact (kg CO_2 eq. per tonne cement)
Mining	16.5
Transportation	1.16
Raw meal preparation	18.9
Calcination	599
Waste gas treatment	0.119
Grinding	33.5
Packaging	0.153
Others	8.97
Total	**678**

Li et al. (2014) included the emissions of heavy metals in their LCI of Chinese cement production, which they noted were not usually reported in most studies. They captured data for production of cement at a manufacturing facility.

Biogenic CO_2 emissions are excluded from the formal CO_2 emissions reporting of cement producers in Germany (Feiz et al. 2015a,b). However, it is recommended that biogenic emissions should be included, but reported separately. Additionally, the assumption that bioenergy carbon neutral is not correct and depends upon the source of the material (Cherubini et al. (2011a,b).

Ingrao et al. (2014) investigated the use of basalt as an aggregate in concrete and reported that this was not a good choice of material from an environmental perspective. Kua and Kamath (2014) reported that the GWP associated with the cradle to grave lifecycle of 1 kg of concrete was 0.107 kg CO_2 eq. with an embodied energy of 1.67 MJ. This analysis did not include carbonation.

The reinforcing steel makes the largest contribution to the embodied energy and associated GHG emissions of reinforced concrete (González and Navarro 2006). They quoted values of 1.2 MJ/kg for the embodied energy of concrete and 10.1 MJ/kg for concrete reinforced with recycled steel, with associated GWP impacts of 0.0194 kg CO_2 eq. per kg and 0.1631 kg CO_2 eq. per kg, respectively (data was obtained from the New Zealand Institute of Architects).

Higher strength concrete has a larger environmental impact, but this can potentially be offset if less concrete is required for a construction. Habert et al. (2012) looked at an example of two concrete bridges located in France, one of which was built using high performance concrete and the other using 'traditional' concrete. Although the bridges did not have the same spans (51 m (traditional) and 46 m (high performance)), it was considered that they were similar enough that the comparison would be valid. The functional unit was 'the crossing of a four-lane divided highway with a two-lane road over a 100 year period'. It was found that the uncertainties in calculating the environmental impacts were about +/- 20% for most impact categories, but that uncertainties regarding the maintenance were the most problematical aspect of the analysis. Despite these difficulties, it was concluded that the use of high performance concrete resulted in a reduction of GHG emissions by about 50%.

The main ingredients of concrete are cement and aggregates. Aggregates are materials in concrete that take no part in the chemical reactions associated with hardening. There is an environmental burden associated with the production of aggregates which is higher for crushed stone compared to quarried pebbles. One potential source of reduction of environmental burdens is to use crushed post-consumer glass as an aggregate material (Meyer 2009). Hájek et al. (2011) reported on the environmental burdens associated with the production of different types of cements (CEM I, CEM II, CEM III) and concretes. Rather surprisingly, they stated that the embodied energy associated with one tonne of crushed gravel (35.6 MJ) was less than that associated with the production of sand/gravel (38 MJ). This data was obtained from Czech companies producing the materials. Other data was obtained from the GEMIS database. Kim et al. (2016b) illustrated how the carbon emissions associated with concrete manufacture increased along with the compressive strength, largely due to the difference in cement content. Transport of wet concrete to site and the formwork used for construction results in higher impacts when compared to using pre-cast concrete elements (Dong et al. 2015), but this is not always an option and removes one of the advantages of using concrete.

7.5 End of life issues

Metals can be recycled to give products with more or less the same function (closed loop recycling), but this is not possible for concrete. One of the options for crushed concrete is to use the material as substitute for fresh aggregate in new concrete products. However, there can be technical limitations with this approach and products with inferior mechanical properties can result, requiring the addition of higher than normal levels of cement, resulting in a higher environmental impact (González-Fonteboa and Martínez-Abella 2008, Shi et al. 2016b). This is because the recycled aggregate contains numerous small cracks that can act as failure initiation points and there is also an additional layer of mortar attached to the aggregate which means that there are two interfaces (Xiao et al. 2013). The requirement for adding additional cement effectively cancels out any advantage of using recycled aggregate from an environmental perspective. Other options include use as engineering fill, as sub-base materials in road construction, or in building or bridge foundations (Poon and Chan 2006). Recycled concrete is an option for replacing the primary aggregates in the production of low to middle strength concrete (Weil et al. 2006, Rao et al. 2007, Marinković et al. 2010). However, the recycling of concrete and brick wastes as aggregates can result in the incorporation of other mineral materials (such as plasters, floor screeds, lightweight mineral materials) which can compromise the strength of the concrete. Weil et al. (2006) showed that the use of recycled concrete or brick aggregates resulted in a reduction in the use a primary mineral aggregate by up to 44%. However, the quantity of cement needed in concrete with recycled aggregates is higher than that with natural aggregates, resulting in an increase in embodied energy by 36% and GWP by 39%. The main reason for this is that cement and mortar adheres to the surface of the aggregate particles, which increases water sorption and reduces abrasion resistance (De Schepper et al. 2014). Using coal combustion fly ash and concrete plasticisers, instead of extra cement, resulted in an environmental impact that was comparable with concrete made using fresh aggregates. Knoeri et al. (2013) investigated the differences between recycled and conventional concrete using LCA endpoint indicators. This study showed that although the cement was a significant contributor to the overall environmental impact, reductions were found due to the recycling of steel scrap, avoided transport of the demolition waste to a disposal site and avoided impacts due to disposal. Concrete crusher sand, the fine material (less than 2 mm) that is obtained from the concrete crusher at the end of life has potential to be used as a feedstock for the calciner to make clinker in the cement production process (Schneider et al. 2011). The uncarbonated portion of the concrete crusher sand would produce less CO_2 in the kilning process. However, this material has a high silicon content and can only be used if the other raw materials used as feedstocks allow for this.

8 Review of LCA of aluminium

8.1 The aluminium lifecycle

Primary aluminium is obtained from bauxite and secondary aluminium from scrap. Bauxite ore contains high concentrations of aluminium hydroxide (40-60%) plus impurities of iron, silicon and titanium oxides. Bauxite is usually obtained by opencast mining and approximately 4-5 tonnes of bauxite are used to produce 1 tonne of aluminium metal. The main sources of bauxite are Australia, Guinea, Jamaica, Brazil, India, Venezuela, China, Suriname, Greece and Hungary.

Bauxite is converted into aluminium oxide (alumina), via the Bayer process where the aluminium hydroxide is dissolved in sodium hydroxide solution and the impurities removed by filtration. The aluminium hydroxide is then precipitated from solution and the unused sodium hydroxide solution is recycled. The residue from this process is mainly composed of oxides and hydroxides of iron and silicon. This sludge is referred to as red mud due to its colour. About 0.3-2.5 tonnes of red mud are produced per tonne of aluminium, depending upon the purity of the bauxite (Moors 2006). Alumina refineries are usually located close to bauxite mines. About 4-7 tonnes of bauxite are required to make one tonne of alumina (Tan and Khoo 2005).

The main aluminium production route (Bayer process) uses the Hall-Héroult electrolytic process in which alumina (Al_2O_3) is dissolved in molten cryolite (Na_3AlF_6) and electrolytically decomposed at temperatures below 1 000^oC. This process takes place in large carbon-lined steel electrolytic vessels, where carbon cathodes form the bottom of the vessel. Molten aluminium collects at the bottom of the vessel, where it is periodically siphoned off. Carbon anodes are located at the top of the vessel and are consumed during the process, where they react with oxygen (to form carbon dioxide) which is released from the alumina during the electrochemical reduction process. About 1.8 tonnes of CO_2 is released by the consumption of the 0.5 tonnes of carbon used for the production of 1 tonne of aluminium metal. If the emissions from foundry operations and the production of anodes is included, this figure is closer to 2 tonnes of CO_2 eq. per tonne of aluminium. The potential for reductions in these emissions is limited. Emissions of fluorides in particulate and gaseous form can occur during the process, requiring aluminium production facilities to be fitted with scrubbers, which has reduced fluoride emissions to 0.5-1.1 kg per tonne of aluminium. The greenhouse gases tetrafluromethane and tetrafluorethane are produced when the alumina concentration in the vessels drops too low and are reduced by efficient control measures, such as alumina point feeding. The PFC (perfluorocarbons) emissions per tonne aluminium produced in Norway was 4.48 kg CO_2 eq. in 1990 and was reduced to 0.12 kg CO_2 eq. by 2015, a reduction of 97.3% (NIR 2017).

Polycyclic aromatic hydrocarbons (PAH) are released during the production or curing of carbon anodes, which are largely removed by scrubbers, so that emissions are now around 0.05 kg per tonne of aluminium, or even lower with modern plants. Emissions from Søderberg plants (which use wet, 'self-baking' anodes) are somewhat higher (0.25 kg per tonne), although these plants are converting to dry 'pre-baked' anode technology. Søderberg plants also produce more CO_2 from electrode consumption compared with dry anodes. Sulphur dioxide is released due to the burning of fossil fuels as part of the production chain and from anode combustion in the smelters. These emissions can be reduced through the use of scrubbers. The old linings of the electrolytic vessels (which consist of refractory bricks, carbon, electrolytic bath materials and a small amount of cyanide) have to be disposed to suitable landfill sites (Moors 2006).

Molten aluminium from the smelter is cleaned, alloyed and cast into ingots. The ingots are then further processed by rolling, extrusion, or casting to form a variety of semi-finished products.

Aluminium production represents the major part of the Norwegian metals' industry sector and it is the largest producer in western Europe with primary aluminium (1.2 million tonnes per year) being produced at seven sites. Norsk Hydro is the largest company in this sector. The Norwegian primary aluminium production employs about 3 000 people and 1 000 more are employed in further processing. The production of primary aluminium is very energy intense and is one of the largest consumers of electricity in Norway (20 419 GWh, 2015, aluminium and other non-ferrous metals). In 2015, aluminium and aluminium products with a value of 10.83 billion NOK were imported in contrast to an export value of 27 billion NOK (Statistics Norway 2016e).

8.2 Environmental impact

It requires 4 MWh of electricity to produce one tonne of alumina (Olivieri et al. 2006). Typically, the production of 1 tonne of aluminium requires 1.9 tonnes of alumina and consumes 0.4-0.5 tonnes of carbon. Electricity consumption for the electrolysis process is in the range of 13-18 MWh per tonne of aluminium produced, but this does not include transformation losses and operation of ancillary equipment. Overall electricity consumption for new plants is about 14.7 MWh per tonne of aluminium. This figure was closer to 21 MWh per tonne in the 1950's (Moors 2006). In contrast, the production of secondary aluminium (from scrap) requires only 0.7 MWh of electrical power per tonne (Olivieri et al. 2006).

The quoted embodied energy values for the production of aluminium vary. Theoretically, 31.1 GJ per tonne of aluminium is required to produce aluminium from alumina, compared with 6.8 GJ per tonne for iron. However, the real values for industrial production are inevitably going to be higher than this. The electricity requirements are in the region of 47-54 GJ per tonne. Norgate and Haque (2010) give 180 MJ/kg for the smelting process, plus 30 MJ/kg for ore extraction and processing, giving 210 MJ/kg in total. Liu and Müller (2012) reviewed LCAs of aluminium. They found that GWP impact per kg of primary aluminium ranged from 5.92 to 41.10 kg CO_2 eq. with a typical range of 9.7-18.3 kg CO_2 eq./kg. GHG emissions from bauxite mining, alumina production have remained static over time, but GHG emissions from primary aluminium smelting have fallen since the 1990's (Ciacci et al. 2014) (Table 18). With secondary aluminium production, the GWP impact was 0.32-0.74 kg CO_2 eq./kg of metal with a typical range of 0.3 to 0.6 kg CO_2 eq./kg. These values are heavily influenced by the electricity grid mix used for the process. For example, China with a very large contribution from coal fired generation to the grid mix, has a much higher GWP impact for aluminium production compared with Europe, which has a substantial contribution from hydro and nuclear to the grid mix for aluminium production (Nunez and Jones 2016). Nunez and Jones (2016) conducted a generic cradle to gate LCA for aluminium production, using the LCI data of the International Aluminium Institute. They divided the results into global (GLO) and global minus China (rest of the world, RoW) production. They reported a GWP of 16.5 kg CO_2 eq./kg Al (GLO) and 10.8 kg CO_2 eq./kg Al (RoW). These values were higher for the Søderberg process, which was 18.3 kg CO_2 eq./kg Al (GLO) and 11.8 kg CO_2 eq./kg Al (RoW). The electrode pre-bake process values were slightly lower than the global average, at 16.3 kg CO_2 eq./kg Al (GLO) and 10.6 kg CO_2 eq./kg Al (RoW).

Table 18 Changes in GWP impact for primary aluminium production over time (Ciacci et al. 2014)

Decade	kg CO_2 eq. per kg Al
1960-1969	16.50
1970-1979	15.90
1980-1989	15.80
1990-1999	13.40
2000-2009	9.98

Norgate et al. (2007) quote an embodied energy of 211 MJ/kg and a GWP of 22.4 kg CO_2 eq./kg for aluminium production. They note that these figures could be reduced by about 15% if new technologies, such as the vertical electrode cell are introduced. However, they also point out that the impacts from metal production will increase as the metal content of ore grades decrease.

Koltun et al. (2010) undertook an LCA study of aluminium smelter cast house operations in Australia, which are a small fraction of the impacts associated with the smelting process, but are still significant compared to other industries. Energy consumption is typically 300 MJ per tonne, where the Al scrap content is 1% and rises with the scrap content (500 MJ per tonne at 7% scrap content). The theoretical figure for melting aluminium is 1.1 GJ per tonne. They found that the total GHG emissions were 221.7 kg CO_2 eq. per tonne for remelt ingot cast house activities and 623.2 kg CO_2 eq. per tonne for a wrought alloy cast house. These represent 1% and 3% of the overall GHG emissions to produce liquid aluminium. The total embodied primary energy was 2856 MJ per tonne for a remelt cast house and 7988 MJ per tonne for a wrought alloy cast house.

Table 19 **Environmental impacts associated with different manufacturing stages of primary aluminium production in Australia (Koltun 2010) (per kg of primary Al)**

Manufacturing stage	GWP (kg CO_2 eq.)	EE (MJ)
Bauxite mining	0.08	0.32
Anode production	0.13	9.30
Alumina production	2.27	28.94
Al smelting	2.85	53.28
Casting	0.06	3.36
Auxiliary operations	0.0	7.33
Limestone, caustic soda production	0.61	0.18
Electricity production	14.4	116.45
Transport	2.56	2.65
Total	**22.51**	**221.81**

Aluminium production is very energy intensive. The embodied energy associated with the production of 1 kg of primary aluminium is 221.8 MJ, but for secondary aluminium is 29.3 MJ (Koltun 2010). In Koltun's paper, the environmental impacts associated with primary aluminium production and secondary aluminium production in Australia are broken down, as shown in Table 19 and Table 20, respectively.

Table 20 **Environmental impacts associated with different manufacturing stages of secondary aluminium production in Australia (Koltun 2010) (per kg of secondary Al)**

Manufacturing stage	GWP (kg CO2 eq.)	EE (MJ)
Scrap separation	0.0	0.29
Shredding and decoating	0.0	0.89
Casting	0.0	8.35
Electricity production	2.48	19.35
Transport	0.37	0.43
Total	**2.85**	**29.31**

8.3 End of life

Recycling of aluminium has associated GHG emissions of 360-1260 kg CO_2 eq. per tonne of metal, representing a saving of 5 040 to 19 340 kg CO_2 eq. per tonne of primary virgin metal (Damgaard et al.

2009). Recycling of 1 kg of aluminium can save about 8 kg of bauxite, 4 kg of chemical products and 14.7 MWh of electricity (Moors 2006).

Although metals have the highest potential for systematic recycling, the contamination of the metals with alloys and residuals with each lifecycle, especially for those for which removal from the melt is problematic and makes reprocessing more difficult. There are over 450 alloys of aluminium, with typical alloying elements being silicon, copper, zinc, magnesium and manganese. Due to the uncontrolled mixture and accumulation of alloying elements during the lifecycle stages, they are considered as contaminants. Iron occurs as a tramp element (Paraskevas et al. 2015). Conventional LCAs tend to ignore the down-cycling aspect of aluminium and account for the primary metal and secondary metal as equivalent in quality (Amini et al. 2007).

9 Review of LCA of steel

9.1 The steel life cycle

Although of huge economic importance, the iron and steel industry is also responsible for high carbon emissions and energy consumption, being responsible for 6-7% of global anthropogenic CO_2 emissions (Vadenbo et al. 2013). Steel production in 2014 was 1 665 million tonnes (Renzulli et al. 2016) and 1 630 million tonnes in 2016, according to the World Steel Association. Global CO_2 emissions in 2020 for the predicted production of 1 781 million tonnes of steel have been estimated to be 3 169 million tonnes, with an embodied energy of 14.43 GJ per tonne of crude steel (Yellishetty et al. 2010). Steel production is responsible for 31% of the CO_2 emissions of the global manufacturing sector. The only strategy for significantly reducing the CO_2 emissions associated with primary steel manufacture is to use carbon capture and storage (CCS) (Quader et al. 2016). Production of recycled steel has lower embodied environmental impacts compared to virgin steel production.

Input materials for the production of iron include iron ore, coke and limestone. The production of iron from iron ore involves the reduction of the iron oxide by heating with a carbon source and then removal of most of the remaining carbon in the molten iron in a blast furnace (BF). The molten iron can then be converted to steel by reducing the carbon content. Two processes currently dominate steel manufacturing, the basic oxygen furnace (BOF) which is used for 74.3% of global production and the electric arc furnace (EAF), which is used for 25.3% of global production, with a very small amount of production (0.4%) using the open-hearth furnace. The electric arc furnace process relies almost entirely upon electricity. Recycling of steel normally uses the EAF route, which can use 100% scrap. This contrasts with the basic oxygen furnace (BOF), which is limited to 30% scrap input (with the remainder pig iron) because of the need to maintain a thermal balance in the process (McMillan et al. 2012). The basic oxygen furnace is used to convert molten pig iron to steel, by blowing oxygen through the molten iron in the presence of lime. The molten iron is transferred from the blast furnace into a refractory ladle into which high purity oxygen is introduced through a water-cooled lance. The basic oxygen furnace is much more energy efficient compared to the EAF, but the BOF must be used in combination with the blast furnace. The BOF can even be used as a source of energy, because the reaction with oxygen is exothermic. The GWP impact of the EAF is very dependent upon the local grid primary energy mix and only direct emissions of CO_2 are shown in the table (Flues et al. 2015) (Table 21).

Table 21 The environmental impact of the BF-BOF steel making route compared to the EAF route (one tonne of steel) (Flues et al. 2015)

Process	GWP kg CO_2 eq.	EE (GJ)
Coke oven	250	3.6
Sintering	250	1.6
Blast furnace	360	13.8
Basic oxygen furnace	70	0.1
Total	**930**	**19.1**
Electric arc furnace	90*	2.5

*Direct emissions only

The EAF can be used to produce steel from iron ore, when used in combination with a direct reduction furnace, in which the iron ore is reduced by reaction with a mixture of CO and H_2.

Approximately 38% by weight of the total virgin raw material input into steel-making are by-products and waste; of this about half is blast furnace slag (Lee and Park 2005). According to Habert (2013), approximately 0.24 kg of slag is produced for each kg of pig iron. The blast furnace slag can be cooled by air cooling with occasional water spray, or by rapid cooling with a continuous jet of high pressure water. The former can only be used as road foundation material, but the high pressure water jet cooled ground blast furnace slag (GBFS) can be used as a clinker substitute in Portland cement.

Norway has one EAF plant (Celsa Steel Service AS) located at Mo i Rana, producing steel from scrap iron (NIR 2017) consuming 1.44 million tonnes scrap iron steel billets in 2012 (KFD 2013). The iron and steel production in Norway in 2015 had GHG emissions of about 28.3 kt CO_2 eq. There were 12 plants producing ferroalloys in Norway in 2015 (NIR 2017) with total GHG emissions of 2 516.5 kt CO_2 eq. The production of steel, iron and ferroalloys had in total 1 637 employees with a turnover of 10.77 billion NOK (Statistics Norway 2016d). In 2015, the production of steel, iron and ferroalloys had an electricity consumption of 4 978 GWh. In 2015, steel, iron and ferroalloy materials and products were imported with a value of 14.9 billion NOK, and exported with a value of 12.73 billion NOK (Statistics Norway 2016e).

9.2 LCA studies of steel

The blast furnace and coke oven operations have by far the greatest contribution to GWP impacts from the steel making process, whereas that of the electric arc furnace has a lower environmental impact. This is because the blast furnace relies on coke as a source of energy and as a reducing agent. The impact of the EAF is determined largely by the local electricity grid primary energy mix (Gomes et al. 2013, Renzulli et al. 2016, Zhang et al. 2016). Bribián et al. (2011) quote an embodied energy value of 24.3 MJ/kg and 1.5 kg CO_2 eq./kg of steel. Bawden et al. (2016) showed that depending on the type of steel, country of origin and recycled content, the life cycle CO_2 eq. emissions of steel can vary from 0.7 to 5.9 kg CO_2 eq./kg. Regional variations in production technologies can make a difference, e.g., the grid mix in the US used coal as a fuel for 37% of the total electricity generation in 2012, whereas in China this figure was 65%. Broadbent (2016) quotes a value of about 1.9 kg CO_2 eq. for basic oxygen furnace steel with a 0% scrap content. This value reduces as the scrap steel content is increased. In a study conducted by the World Steel Association, Broadbent (2016) reported that recycling 1 kg of steel at the end of product life resulted in savings of 1.5 kg CO_2 eq. emissions and 13.4 MJ of primary energy, compared with the production of 1 kg of virgin steel. Bribián et al. (2011) quote an emission reduction of 1.2 kg CO_2 eq/kg of recycled, compared to virgin steel. The embodied energy (cumulative energy demand) ranges from around 36 MJ/kg at 0% recycled content to about 15 MJ/kg at 100% recycled content.

When using LCA to inform decision making processes, the recycling level in the steel used can have a major impact, but this may not always be known at the time of procurement (Du et al. 2014). The Ecoinvent generic value assumes that 37% of steel is manufactured using the EAF. Gomes et al. (2013) calculated that the steel production contributes about 50% to the total GWP impact, with hot rolling about 40% and the remainder transport. They calculated a GWP impact of 1.12 tonnes CO_2 eq. per tonne of steel reinforcing product sold in France, which compared with an Ecoinvent value of 1.54 tonnes CO_2 eq. per tonne. The reinforcing steel was composed of a very high level of recycled material, which reduced the impact. It was found that the result was relatively insensitive to the allocation method used for recycling, transport distance and electricity grid mix.

The World Steel Association has been developing life cycle inventories associated with steel production for over 20 years. They represent about 45% of global steel production and have produced embodied energy and GWP data for virgin and recycled steel production (Table 22).

Table 22 Environmental impact data for steel production according to World Steel Association (2011) (per tonne of steel)

Product	EE virgin [GJ]	EE recycled [GJ]	GWP primary [kg CO_2 eq.]	GWP recycled [kg CO_2 eq.]
Sections	19.6	16.4	1 600	1 200
Hot-rolled	21.6	11.9	2 000	900
Hot-dip galvanised	27.5	17.5	2 500	1 300

Iosif et al. (2010) developed a physicochemical model combined with LCA for investigating the environmental burdens associated with the steel making process. The physicochemical models were developed for each element of the steelmaking process: coke plant, sinter plant, blast furnace, basic oxygen furnace and hot-rolling. Based on knowledge of the chemical reactions and thermodynamics, it was possible to model the production of each pollutant released by the process. This approach was considered by the authors to be superior to conventional methods for conducting LCI of the steelmaking process. According to this model the CO_2 emissions were 1.59 tonnes CO_2 eq. per tonne of hot rolled steel sheet in an integrated steelmaking plant. Patel and Seetharaman (2013) quote 1.8 tonnes of CO_2 per tonne of steel. Renzulli et al. (2016) give an embodied energy of 19.8 GJ per tonne of steel (blast furnace 13.4 GJ, coke oven 3.7 GJ, sintering plant 1.9 GJ, basic oxygen furnace 0.8 GJ per tonne) for the integrated steel mill in Taranto, Italy. The GWP impacts were 660 kg for the blast furnace, 360 kg for the coke ovens, 330 kg for the sintering plant and 240 kg CO_2 eq. for the basic oxygen furnace; giving a total of 1.6 tonnes of CO_2 eq. per tonne of steel produced. Norgate and Haque (2010) report an embodied energy of 23 MJ and a GWP of 2.3 kg for the production of 1 kg of steel using an integrated BF-BOF route.

Recycling of metals is always beneficial because there is always an energy saving. This is because the recycled metal offsets the material that would be produced by the primary production process. The recovery of waste metals for recycling results in low levels of emissions compared with reprocessing (13-53 kg CO_2 eq. per tonne of recovered metal). Recycling of steel has embodied emissions of 400-1020 kg CO_2 eq. per tonne of recycled steel, which represents a saving of 560-2360 kg CO_2 eq. per tonne of metal (Damgaard et al. 2009). Although there are undoubted environmental benefits arising from the recycling of steel from buildings, the re-use of steel in next generation buildings has much greater benefit (Zygomalas and Baniotopoulos 2016).

GWP impacts and embodied energy values from EPDs for Norwegian steel production and manufacturing are listed in Table 23.

Table 23 Published EPD data for Norwegian steel production (per kg of steel)

Declaration no.		GWP (kg CO_2 eq.)	EE (MJ)
NEPD 00255E	Norwegian Steel Association	2.55	26.10
NEPD-326-206-EN	Ferrometall AS	2.68	30.18
NEPD-348-237-EN	Norsk Stål AS	0.33	8.81
NEPD-321-200-EN	E.A. Smith AS	0.52	10.53
NEPD-434-305-EN	Celsa Steel Service AS	0.36	5.61
EPD-CEL-20130219-IBD1-EN	Celsa Steel Service AS	0.63	9.59
S-P-00305	Celsa Steel Service AS	0.37	5.95
S-P-00306	Celsa Steel Service AS	0.36	5.61
S-P-00307	Celsa Steel Service AS	0.42	6.29
S-P-00308	Celsa Steel Service AS	0.45	6.39

Celsa Steel Service AS manufacture steel using an electric arc furnace from 100% steel scrap at Mo i Rana. The Norwegian Steel Association EPD (NEPD 00255E) only covers modules A1 and A2 for the manufacture of hot-rolled steel plates. The module A1 corresponds to average European steel manufacture. The Ferrometall AS EPD (NEPD-326-206-EN) uses Worldsteel data for steel manufacture for use in the manufacture of stressed steel concrete reinforcement. The E.A Smith EPD uses data for European steel manufacture, but gives no further details (the values appear to be for electric arc furnace with a low carbon grid mix using scrap steel). The Norsk Stål AS EPD also uses European steel manufacture (again, this appears to represent electric arc furnace with a low carbon grid mix using scrap steel). All other EPDs are for indigenous Norwegian steel manufacture using scrap steel and electric arc furnace. The carbon footprint for the Norwegian grid is given as 0.024 kg CO_2 eq. per kWh.

The relationship between the embodied energy and GWP footprint for steel manufacture is shown in Figure 7 (Norwegian steel production is shown in magenta).

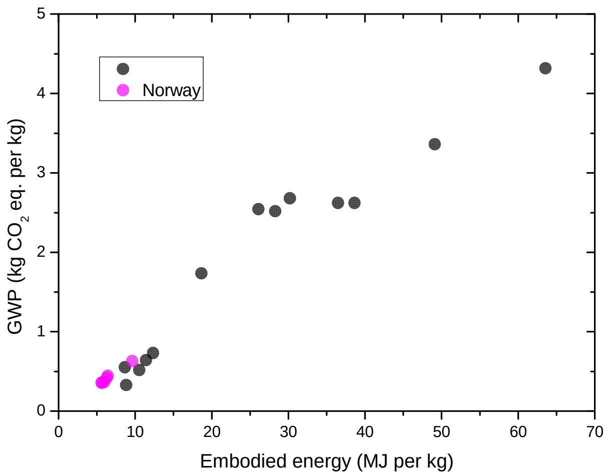

Figure 7 Relationship between embodied energy and GWP impact for steel production from EPD data (per kg of steel).

10 Review of LCA of bricks

Bricks are typically composed of 90% clay, with much of the remainder being sand. The raw materials are kneaded and mixed, then stored for some time before shaping in a mould or by extrusion. After this, the brick is dried at a temperature of 75-90°C, usually using exhaust heat from the kiln. After drying, the bricks are fired in tunnel kilns in an oxidising atmosphere. The unfired bricks are placed on tunnel kiln cars that pass through the kilns at temperatures of 800-1000°C, for a total firing time of 17-25 hours.

Brick production has a large impact related to energy use and carbon emissions (Buchanan and Honey 1994). Koroneos and Dompros (2007) gave a very detailed account of brick manufacture in Greece along with an LCA. For the life cycle of brick production, they quoted a GWP impact of 221 kg CO_2 eq. per tonne of bricks. Bribián et al. 2011 reported a GWP of 271 kg CO_2 eq. and an embodied energy of 3.6 GJ per tonne for the manufacture of a clay brick with a density of 1 800 kg per m^3. Although there have been a few LCA studies of brick manufacturing, comparisons are difficult because they have used different system boundaries, data sources, impact indicators and impact assessment methodologies. Almeida et al. (2015) reported on a cradle to grave LCA of ceramic brick manufacturing in Portugal, using EN 15804 as the core product category rules for the study. Because different fuels are used for the firing process, three different energy sources were included in a comparative study (natural gas, biomass, petroleum coke). The LCA study also investigated the impact of using different cut-off criteria. The study was for bricks of dimension 30 x 20 x 11 cm, but the kilns produced other sizes of bricks as well. A mass allocation procedure was accordingly adopted. The Ecoinvent database was used for secondary data, EN 15804 impact categories were reported and calculations of impacts were performed using the CML database, except for respiratory inorganics, where the Impact 2002+ method was used. The methods were run using Simapro software. GWP impact varied from 132 to 303 kg CO_2 eq. per tonne of bricks, depending on the fuel source employed, with the lowest impact being associated with biomass and the highest with petroleum coke. Kua and Kamath (2014) undertook a consequential LCA study of brick manufacture compared to concrete, focussing on GWP. In the attributional LCA for bricks it was reported that about 2.9 GJ was required per tonne of bricks for the cradle-to-grave life cycle and that 87% of this was used for the drying and firing of the bricks. The GWP for the raw material extraction, manufacture and transportation of one tonne of bricks was 188 kg CO_2 eq.

One EPD for Norwegian brick manufacture (Wienerberger AS) has been published by IBU (EPD-WIE-20130206-IAB1-EN). The declared unit is 1 m^3 of bricks with a density of 1 395 kg/m^3. The reported GWP impact is 277.6 kg CO_2 eq. and an embodied energy of 4 586 MJ (equivalent to 0.199 kg CO_2 eq. and 3.29 MJ/kg).

11 Review of LCA of uPVC

11.1 The uPVC life cycle

Poly(vinylchloride) (PVC) is produced by the polymerisation of vinyl chloride monomer, which is in turn derived from ethylene dichloride. Ethylene dichloride is obtained from the reaction of ethylene with chlorine, with the ethylene being obtained by steam cracking of hydrocarbons derived from fossil oil reserves. Chlorine is produced from brine (salt solution) by the chlor-alkali industry.

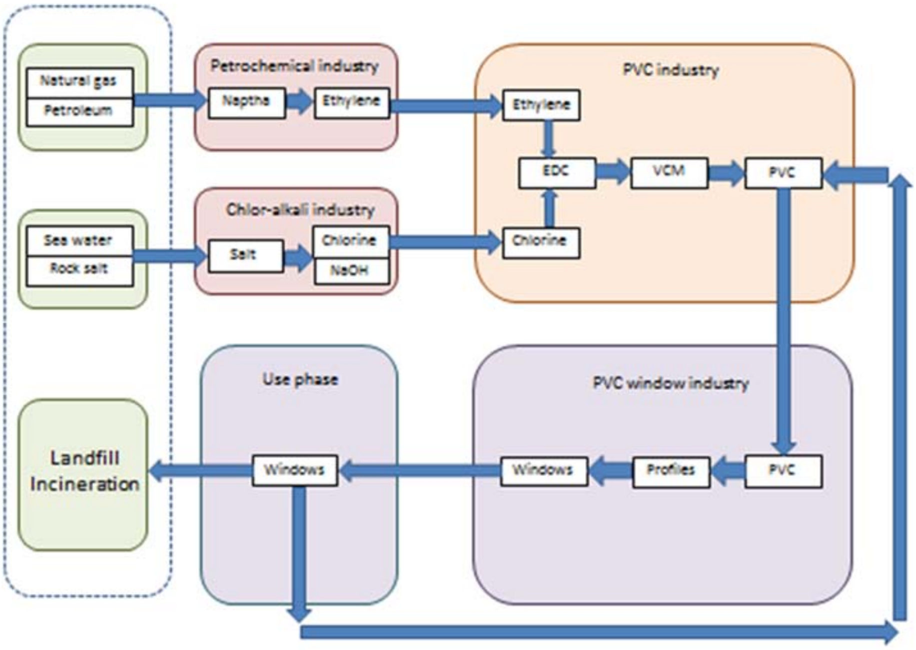

Figure 8 The poly(vinyl chloride) lifecycle

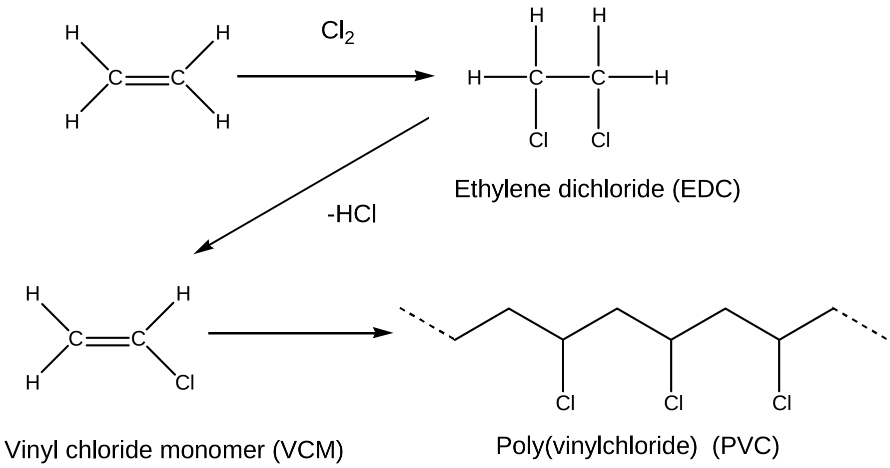

Figure 9 Chemistry of poly(vinyl chloride)

There is some residual VCM in the polymer, which is set at below 1 g VCM per tonne (1 ppm) PVC by voluntary agreement by members of the European Council of Vinyl Manufacturers. Exposure of workers to VCM during production was a serious safety issue (Mundt et al. 2000), but industrial exposure levels have been reduced substantially in North America and Europe due to stringent safety regulations. Steam stripping is now used to reduce residual levels of VCM in PVC resin by 99% compared to levels previously present.

PVC is a rigid polymer and is used in this way in products such as windows and water pipes, this is referred to as unplasticised PVC (uPVC). Flexible PVC products, such as bottles, vinyl flooring, electrical cabling and toys require the addition of a plasticiser (usually a phthalate). Since PVC is colourless, a white pigment (titanium dioxide) is added to impart colour and opacity to the rigid PVC used in window profiles. The presence of chlorine in PVC contributes to the fire retardancy of the material, but PVC is sensitive to heat and UV exposure (resulting in loss of hydrogen chloride) and for this reason requires the addition of stabilisers.

Rigid PVC experiences higher processing temperatures and higher shear than flexible PVC and for this reason requires more effective heat stabilisers. Until recently, this relied upon the use of cadmium and lead-based stabilisers. The use of cadmium stabilisers in the European market was phased out in 2001 and lead-based stabilisers were phased out in 2015. Some tin stabilisers continue to be used in rigid products, but calcium and calcium/zinc-based stabilisers are now used for most applications and use of tin stabilisers in uPVC windows is limited in Europe, with some use in the cellular core of window profiles. The metal stabilisers are present as soaps (usually Cd stearates or laurates, lead stearates), a form that is non-volatile and not water-soluble. Lead may also be present as lead sulphate, lead phosphite or lead phthalate. Lead sulphate is sparingly soluble in water (0.0032 g/100 mL) as are all lead stabilisers (Grossman 1999). There is no evidence that metal stabilisers migrate to the surface of the PVC in uPVC. Heat stability in some applications requires enhancement when calcium/zinc, or barium/zinc stabilisers are used and organic co-stabilisers (polyols, epoxidised soya bean oil, antioxidants, and organic phosphites) will often be added to the formulation. Global annual consumption of PVC for window frames is estimated to be around 3 million tonnes or 8% of global production (Stichnothe and Azapagic 2013).

11.2 PVC manufacture and environmental impact

The main sources of environmental impact associated with the production of PVC arise from energy usage and with fugitive emissions associated with the chlor-alkali industry. The chlor-alkali industry uses three electrolysis technologies for producing chlorine and caustic soda from brine, referred to as membrane, mercury and diaphragm cells. The European industry has committed to closing all of its mercury-based electrolysis plants by 2020 which accounted for 20% of capacity in 2016, with membrane technology accounting for 64%. Total chlorine production in Europe was nearly 9.7 million tonnes in 2016, with the largest single end use (33%) being for PVC production. Because the production of chlorine is energy intensive, it is also emissions intensive, linked to the energy mix of the electricity grid.

Vinyl chloride monomer (VCM) can be produced from three feedstocks: ethylene, acetylene, or ethane. Acetylene is produced by the reaction of water with calcium carbide and is the route use in the manufacture of PVC in China. European production uses the ethylene route, with reaction via ethylene dichloride. The ethylene dichloride is produced by reaction with chlorine (chlorination route) or by reaction with hydrogen chloride and oxygen (oxychlorination route). Oxychlorination is the preferred technology because it consumes HCl which is a by-product of vinyl chloride synthesis. The vinyl chloride is produced from ethylene dichloride by thermal cracking. There is no commercial production of VCM from ethane at this time, due to technical difficulties. Emissions from VCM production plants can occur from leakage losses in the pipework, pumps, etc., losses in storage and handling, combustion emissions and flaring disruptions.

There is a possibility that freshly produced PVC could be contaminated with dioxins from some of the processes in the production chain, although this has not been reported. Dioxins can be generated at several stages of the PVC manufacturing process (Zhang et al. 2015):

- Brine electrolysis – this is associated with graphite electrode sludge, but graphite electrodes are no longer used in Europe. Dioxins can be generated when carbon is present in the equipment (such a rubber lining);
- Oxychlorination of ethylene to EDC using a copper chloride catalyst;
- Thermal oxidation of chlorinated production residues.

Evers et al. (1996) concluded that vinyl chloride production was a major source of dioxins in the sediments of the River Rhine. However, the levels of dioxins emitted by the PVC industry have been significantly reduced over the past 20 years (Zhang et al. 2015). Mercury emissions per tonne of chlorine produced in Europe were 0.68 g/tonne in 2015; 8.06 million tonnes of chlorine was used to manufacture PVC in 2015, out of a total of 9.691 million tonnes of chlorine produced (source: Chlorine Industry Report 2015-2016 EuroChlor).

11.3 LCA studies of PVC

Ye et al. (2017) reported on the environmental impacts of primary and secondary PVC production in China. The primary data was taken from two production factories in China. The PVC was derived from the ethylene route. The GWP impact for primary PVC production was 2 820 kg CO_2 eq. and for secondary PVC production was 687 kg CO_2 eq. per tonne of PVC. The main contribution to the environmental impacts of PVC production was from VCM production.

Stichnothe and Azapagic (2013) conducted an LCA study of the manufacture of a rigid PVC window frame and showed that there was a considerable advantage to making the windows from recycled material. There was a saving of 2 tonnes of CO_2 eq. per tonne of recycled PVC. The results obtained were sensitive to the transport distances involved and the truck payload factors.

Plastics Europe – the Association of Plastics Manufacturers have produced EPDs for emulsion-polymerised PVC. The declared unit was 1kg of PVC resin. The GWP was 2.5 kg CO_2 eq. and the embodied energy was 66 MJ.

The Norwegian EPD Foundation has published a generic EPD for PVC storm water and sewage pipes, based on US and Canadian manufacturing data. But this gives data per 100 feet of pipe.

The Australian EPD Program had published an EPD for low pressure PVC pipes (S-P-00716), which reports a GWP of 3.6 kg CO_2 eq. and an embodied energy of 76.2 MJ/kg of product for life cycle stages A1-A3.

11.4 End of life of PVC

Recycling of rigid PVC involves grinding into particles small enough for further processing (Braun 2002). It has been shown that uPVC can withstand repeated recycling and still be used for window profiles (Braun 2002, Kelly et al. 2005), although some workers report on a reduction in mechanical properties with repeated mechanical recycling (Sadat-Shojai and Bakhshandeh 2011). The issue of legacy additives remains to be dealt with when considering long-term recycling potential. Recycling of uPVC from windows takes place through mechanical grinding, combined with mechanical contaminant removal (melt filtration, tribo-electric separation, air classification). The total amount of PVC consumed globally since the early 1960's is estimated at some 400 million tonnes, of which half is still in use as long life products, such as window frames and water pipes (Sadat-Shojai and Bakhshandeh 2011).

At the beginning of the 21st Century there was effectively no PVC recycling in Europe (Leadbitter 2002), but this is now 0.5 million tonnes per annum, with nearly 50% of this (232 757 tonnes) being window and other profiles (such as cladding) (source: Vinylplus). The target is 0.8 million tonnes by 2020. Demand for PVC products in Europe is close to 5 million tonnes per annum (source: Plastics Europe). Profiles accounted for 28% (1.4 million tonnes) of PVC used in 2014 in Europe. The presence of cadmium and lead-based stabilisers in PVC products has resulted in concerns with respect to recycling, since these 'legacy' stabilisers will continue to appear in products even though they have been theoretically phased out. It is not known at present how this situation will be resolved. There is an EU derogation in place allowing for cadmium levels of 0.1% in rigid PVC products containing recovered PVC, which is due to be reviewed at the end of 2017. No doubt the industry will argue for an extension of the derogation. Ten years ago, the vast majority of uPVC waste was disposed of to landfill (Kelly et al. 2005). In 2011, end of life profiles comprised 25 480 tonnes of the total 48 544 tonnes of recycled PVC waste, slightly down on the 49 343 tonnes of PVC recycled in 2010.

The main issue with incineration of PVC is the generation of large amounts of HCl, which causes technical problems. Although incineration of PVC waste can result in the emission of dioxins (Katami et al. 2002), this is by no means the only such source (Dyke et al. 1997). For example, the combustion of waste wood burnt in the presence of chlorides or treated with chlorinated organic preservatives is a source of dioxins (Yasuhara et al. 2003). Blomqvist et al. (2007) found that large contributions to dioxins in the atmosphere came from fires in landfills, waste plastics and tyres. Uncontrolled burning is much more likely to result in dioxin release compared to the controlled conditions present in an incinerator. The presence of copper acts as a catalyst for dioxin formation and PVC coated copper wires were a source of historic dioxin emissions in incinerators and when there was uncontrolled burning of PVC coated copper wire (Carroll 2001). Hatanaka et al. (2000) found that no dioxins were produced by a laboratory scale fluidised bed reactor in the absence of a chlorine source (PVC or NaCl). It is known that decreasing chlorine input into incinerators reduces the production of dioxins (Costa et al. 2012). The typical chlorine content of mixed solid waste is approximately 0.5%, which does not cause problems in incineration, but is at the limit for cement or lime kilns, or for co-firing in coal-fired power stations (Zhang et al. 2015). Landfill fires involving PVC represent a potential source of dioxins (Ruokojärvi et al. 1995, Roots et al. 2004, Blomqvist et al. 2007).

The environmental impacts associated with the recycling of PVC waste from windows was analysed by Stichnothe and Azapagic (2013). They found that significant reductions in environmental impacts of 1.8 to 2.0 tonnes of CO_2 eq. per tonne of resin could be achieved by recycling waste rather than using virgin PVC resin.

12 Review of LCAs where wood has been compared to other materials

12.1 Introduction

The study of wood substitution for other building materials can take place at a number of different levels:

- Product level (such as a window, or door, or wall element);
- Building level;
- Sector level (an economic sector, such as residential buildings);
- Country level (including landscape implications);
- Global level.

Examining the environmental impacts associated with the use of HWPs requires knowledge of material and energy flows in different economic sectors including forestry, timber processing, construction, energy and waste management.

Although considerable progress has been made in evaluating the environmental impacts associated with the use of construction materials, the study of timber products presents additional challenges. These include the long timeframe involved and the range of different products that can be obtained at different times (e.g. forest thinnings for pulpwood or bioenergy, co-products from timber processing and harvesting, bioenergy products or co-products).

Gustavsson and Sathre (2011) discuss the complexities associated with comparing the use of wood with other materials in construction. They conducted a review of the literature and examined the definition of an appropriate functional unit and the establishment of system boundaries in terms of activity, time and space. For comparison purposes, the functional unit can be defined at the level of a building component, a complete building, or in terms of services provided by the built environment. They pointed out that comparisons in terms of a declared unit (e.g., carbon emission per unit volume, or mass) are inadequate because equal mass or volumes of different materials do not fulfil the same function. Furthermore, a functional unit such as a wall element may perform more than one function (e.g., insulation, fire protection, sound insulation as well as support) and some materials may not be able to provide this multiple functionality. Consequently, a more comprehensive analysis must be undertaken at a building level, which can involve the study of a hypothetical building (e.g. Björklund and Tillman 1997), or of completed buildings (e.g., Gustavsson et al. 2006b, Lippke et al. 2004). Where different buildings are compared, then the unit of comparison can be on the basis of a service, such a 1 m^3 of living space, or 1 m^2 of floor area.

A variety of factors can affect the carbon dioxide and energy profiles of building materials over their lifetime, which can be divided into uncertainties and variabilities. Uncertainties arise from lack of precise knowledge regarding processes or the use of assumptions. Variabilities can arise due to different choices being made regarding the use of materials, such as frequency and type of maintenance, different disposal methods, transport distances, etc. Combinations of uncertainty and variability can be difficult to separate. The allocation procedure can also affect the outcome of the study. This is very important for HWPs because many co-products are produced from the same raw material and the HWPs may themselves be used as an energy source at the end of product life cycle. If possible, allocation should be avoided by using system expansion. This is achieved by adding additional functions to the system under study. An example would be the comparison of a timber frame building where the material is incinerated with energy recovery at the end of life cycle, compared with a concrete equivalent where fossil fuels would be used for that purpose.

When attempting to compare timber with concrete use in buildings the LCA has to consider the whole life cycle of the building materials. The GHG balances are related to:

- The primary energy used in the production of the building materials (embodied energy);
- Changes in the biogenic carbon stocks in the forest;
- How the demolition waste is processed;
- The CO2 released from cement production and the rebinding of this CO2 through the process of carbonation.

There is a growing body of literature showing that using wood-based materials results in lower energy use and CO_2 emissions when compared to other building materials (e.g., Koch 1992, Buchanan and Honey 1994, Buchanan and Levine 1999, Börjesson and Gustavsson 2000, Lippke et al. 2004, Gustavsson and Sathre 2006, Petersen and Solberg 2002, 2003, 2004, 2005, Pingoud et al. 2010, Sathre and O'Connor 2008, 2010, Lippke et al. 2011). These studies vary in terms of their focus, transparency and completeness.

The substitution of wood for other building materials can often result in a reduction in the embodied energy of a building and associated GHG emissions. This reduction can be expressed in terms of a displacement factor. The conceptual basis of the GHG benefits of using HWPs was examined by Schlamadinger and Marland (1996). They used computer modelling of carbon flows in systems representing different land use strategies. They concluded that using biomass for direct substitution of fossil fuels as an energy source was an important means of GHG reduction. At the time of their study, there was insufficient data to determine the benefits of using timber products as a means of carbon storage. There was also not enough data to determine a 'displacement factor'; this is the GHG or energy benefit of substituting timber for an alternative building material, or using wood as an alternative energy source to a fossil fuel.

Sathre and O'Connor (2010) undertook a meta-analysis of 21 international studies where the effect of substituting wood for other building materials was reported. This analysis of the scientific literature concluded that the average displacement factor when substituting wood products for other materials was 2.1 (Sathre and O'Connor 2010). This means that for every 1 tonne of carbon in a wood product that is used to substitute for a non-wood product, 2.1 tonnes of CO_2 eq. is saved. The displacement factors in the study ranged from -2.3 to 15, with most values lying in the range of 1.0 to 3.0. The average displacement factor corresponded to 3.9 tonnes of CO_2 eq. emissions saved per tonne of dry timber used. The authors concluded from their study that there was a clear rationale for using timber in construction to mitigate climate change and to enable a transition to a low carbon economic development path.

The magnitude of the displacement factor depends upon a number of assumptions, including the product, the lifecycle and the fossil-fuel based reference system that is substituted (Pingoud et al. 2010). The value of the displacement factor also depends upon the quantities of material that are considered to be equivalent to make the same functional unit. For example, Perez-Garcia et al. (2005b) assumed that 4 tonnes of reinforced concrete was an equivalent to 1 tonne of wood for the same structural performance. This always resulted in a lower embodied energy and GHG impact when timber was used for functionally equivalent buildings. Petersen and Solberg (2005), found that the quantity of avoided GHG emissions due to substitution of steel with wood were in the range 36-530 kg CO_2 eq./m^3 of timber, depending on waste management and how carbon sequestration on forest land was included. The amount of GHG savings when substituting wood for concrete was between 93-1062 kg CO_2 eq./m^3 for timber used. The use of wood was also found to result in lower SO_2 emissions, although the results in other impact categories (acidification, eutrophication, photochemical ozone creation) were more mixed and depended upon the scenario modelled. They noted that a weakness of LCA is that costs are not included and that it was necessary to include LCA with an economic analysis, in order to make such studies more relevant to policy decisions. In their modelling of substitution benefits of timber, Eriksson at al. (2007) and Perez-Garcia et al. (2005a) used detailed inventories of materials' use based upon real life situations, Hennigar et al. (2008) used theoretical displacement factors. Koch (1992) undertook a study at a regional level and showed that reducing the amount of timber used in construction and using non-renewable materials instead would result in a substantial increase in energy consumption and carbon dioxide emissions to the atmosphere.

12.2 Whole building comparisons

The choice of functional unit is clearly a significant factor in making a fair comparison between building materials. The best way of comparing is with the materials integrated in a system at the whole building level, with a functional unit comparison of 1 m^2 of floor area (Paleari et al. 2016). But as noted earlier, the comparison also needs to take account of the recoverable energy in the wood wastes from processing and at end of life, which requires system expansion. As with any LCA study, the results obtained can be influenced by the assumptions that are used when conducting the analysis. This includes the assumed lifetimes of buildings and components, maintenance cycles and replacement intervals.

One of the first studies of the environmental performance of timber compared to other building materials was undertaken by the US National Academy of Science/National Research Council in 1976 (Perez-Garcia et al. 2005a,b). This study was rudimentary, since the science and protocols of life cycle assessment had not been developed very far at that time. In 1996 the US set up a research consortium called 'The Consortium for Research on Industrial Renewable Materials' (CORRIM) in the USA which performed a major study on the environmental impacts of building materials. This has shown that the use of timber in construction gives significant reductions in environmental impacts in most categories (GWP, embodied energy, air emission index, water emission index, solid waste) compared to materials derived from non-renewable resources (Lippke et al. 2004, Lippke and Edmonds 2006). CORRIM undertook an in-depth analysis of the embodied energy and carbon emissions of timber processing and compared timber, plywood and oriented strandboard (OSB) floors with a concrete equivalent. Only the OSB floor had higher carbon emissions (CO_2 eq.) than the concrete floor, but half of those emissions were attributable to biomass derived energy. Furthermore, when the atmospheric carbon stored in the wood was included in the calculations, all of the timber products had a negative carbon footprint (Lippke et al. 2010, 2011). An analysis was also presented in this paper of the total carbon sequestration in different carbon pools in the forest and the harvested wood products, including all manufacturing emissions, end of life of the products and substitution for fossil fuels due to energy recovery from the biomass. This analysis, developed by Perez-Garcia et al. (2005a), showed a continuous increasing trend in the average amount of carbon stored in or saved by the whole system over the 150 years of the dynamic analysis. The effect of product substitution was found to be more important than the physical storage of carbon in the HWPs. This whole system approach showed very clearly the benefits of using timber in long life products in construction, whereas an approach where only one product was considered would not necessarily be so clear in its conclusions. This work showed that it was quite possible to produce carbon negative structures by utilising timber products in construction. In this study, the highest level of avoided net emissions occurred when slash and stumps were removed, forests were fertilised, stem wood was used as a construction material and coal was the avoided fossil fuel. The lowest avoided net emissions occurred when the slash and stumps were left in the forest, stem wood was used as a biofuel, there was no fertilisation and natural gas was the avoided fossil fuel (Lippke et al. 2011).

Buchanan and Honey (1994) studied a comparison of a timber-framed building with equivalents using steel of concrete and concluded that the use of timber resulted in lower CO_2 emissions arising from material processing. Buchanan and Levine (1999) concluded that using timber in construction required lower process energy and resulted in reduced CO_2 emissions, when compared to the use of other construction materials (brick, aluminium, steel, concrete). They used energy coefficients taken from Alcorn (1998) of 0.6 GJ/m^3 for sawn wood, 0.25 GJ/m^3 for other industrial wood, 5 GJ/m^3 for plywood and veneer, 6 GJ/m^3 for particle board, and 10 GJ/m^3 for fibreboard. These values have also been used by the Bath Inventory of Carbon and Energy (ICE) database, which is used in many other studies because it is freely available, although the data is of questionable validity (Hill and Dibdiakova 2016).

Fossdal (1995) compared the embodied energy and GHG emissions associated with the production of materials for single-family dwellings. One of which was constructed using wood and the other made from lightweight aggregate concrete blocks. The results showed that the embodied energy for the materials used in the timber construction was 41-16% less (per m^2 of floor area) than that used for the concrete block structure. Lower embodied CO_2, SO_2 and NOx emissions were also recorded.

Cole and Kernan (1996) examined a generic structure made using a timber, concrete or steel frame. In their paper, they reviewed earlier studies of the embodied energies of structures, reporting the data as GJ per m^2 of floor area. Some of this data was referenced and some of it was not. In general, timber structures were found to have a lower embodied energy when compared to steel or concrete. This contrasts with the later paper of Cole (1999), where the steel frame building had a slightly lower embodied energy. They found that the recurring embodied energy for the lifetime of the building over 50 years was equal to the initial embodied energy of the materials used in construction. They also found that the operating energy was by far the largest component of life cycle energy use (as is commonly stated).

A study of the environmental impacts associated with the building frame (concrete or timber) of a multi-storey dwelling house was undertaken by Björklund and Tillman (1997). The functional unit for comparison was 1 m^2 of floor area and included all stages of the life cycle, with a building life time of 50 years. The concrete framing was either precast or cast in-situ. The manufacturing of the timber frame resulted in an environmental impact that was 50-80% lower, compared to the concrete equivalents, although this became less significant over the whole life cycle because of uncertainty regarding the use phase.

Buchanan and Levine (1999) conducted a study of the consequences of using timber in construction in New Zealand. They concluded that a 17% increase in wood usage would result in a 20% decrease in carbon emissions associated with the manufacture of building materials. An average lifetime for wood products in buildings was assumed. The study represented materials comparisons in terms of embodied energy and GWP impact per m^2 of floor area, showing clear advantages of using wood in construction. They reported that the carbon storage in wood products was relatively unimportant compared to the total carbon emissions associated with the manufacturing; the replacement of more energy intensive building materials was a much more important consideration.

Cole (1999) conducted an analysis, for the ATHENA project funded by Natural Resources Canada, of building elements made from timber, steel or concrete, which were constructed on-site. The analysis only considered the construction process (worker journeys, transportation of materials and equipment, use of on-site equipment and supporting processes). Several different scenarios were considered, which showed that concrete had the highest construction energy demands (reported as MJ/m^2 for the functional units), steel the lowest and timber slightly higher. Plots were shown of the construction energy and GHG emissions against the initial embodied energy of the materials, which showed that no simple relationship existed between the two. The embodied energy values were obtained from ATHENA.

Kristensen (1999) compared a glulam warehouse frame with functionally equivalent steel, or concrete frames. It was found that the glulam frame was responsible for 58% lower GHG emissions compared with steel and 64% lower compared to concrete.

Börjesson and Gustavsson (2000) calculated the CO_2 and CH_4 emissions and the embodied energy associated with the construction of a multi-storey dwelling which either had a timber or a concrete frame. The primary energy use (embodied energy) associated with the production of building materials was about 60-80% higher when a concrete frame was used compared to a timber equivalent. The net GHG balance depended strongly upon how the demolition wood was used at the end of the building life cycle. It was found that the GHG balance was slightly positive if the waste wood was incinerated to replace fossil fuels, slightly negative if the waste wood was re-used in another construction and strongly positive if the wood was disposed of to landfill. When they assumed that all of the CO_2 produced from the calcination of $CaCO_3$ was re-absorbed during the lifetime of the building by carbonation, then the use of a concrete frame was equivalent to a timber frame if all of the demolition was disposed to landfill and none of the methane collected. However, if carbonation was not assumed then the net GHG emission was twice that of the timber frame alternative. GHG mitigation efficiency was higher if the excess wood waste and logging residues were used to replace fossil fuels for energy production. The excess forest left in the concrete frame scenario was used to replace fossil fuels, but if it was left standing to act as a carbon store, the mitigation potential was higher for the first rotation, but lower for the following rotations. Some of the data used for the analysis was uncertain, but it was found that the time perspective employed in the study affected the results markedly. The difference in operating energy between the concrete- and timber-framed buildings was not taken into consideration in this study, based upon the findings of Adalberth (1997, 2001) who calculated that the difference was of the order of only 1%. The analysis of Börjesson and Gustavsson (2000) was criticised by Lenzen and Treloar (2002) who found that their values of embodied energy and GHG burdens were underestimated by a factor of two. However, the main finding – that the environmental burdens of the concrete structure were higher than those of the timber structure – was verified.

Pingoud et al. (2001) investigated the wood substitution potential in new building construction in Finland. They determined the total amount of materials used in the construction of different buildings, as well as the commercial potential for increased wood use. They showed that the carbon pool in sawn wood and wood-based panels in the building sector in Finland had increased from 8.7 million tonnes in 1980 to 11.5 million tonnes in 1995, showing that this sector was a net absorber of atmospheric carbon dioxide. When taking into account building not subject to permission, the estimated stock of carbon in buildings was 16.5 million tonnes in 1995. The average lifetime of sawn wood in buildings was found to be less than 40 years. They found that the built environment was a net absorber of atmospheric carbon throughout the period 1860 to 1995, apart from a brief period in the 1920's and during the Second World War.

A study of residential construction in the Netherlands concluded that a potential reduction of 50% in GHG emissions was possible if timber was used to substitute for conventional construction materials (Goverse et al. 2001).

The use of various wood materials substituting for non-wood materials in Norway was analysed by Petersen and Solberg (2002, 2003, 2004, 2005). The emissions and costs that occur at different times during the material life cycles were analysed using a discounting method, and they were able to calculate an index for the cost-efficiency of the material substitution. This was also linked to an analysis of the carbon fixation dynamics of forests. They concluded that timber construction had consistently lower GHG emissions when compared with non-wood materials. The GHG savings depended upon waste material management and how forest carbon flows were considered. Scharai-Rad and Welling (2002) analysed central European single-family houses constructed in wood or brick.

In their analysis they considered the utilisation of processing and demolition residues to replace fossil fuels, and found that net GHG emission decreased as the volume of recovered wood increased.

Gustavsson et al. (2006b) compared the net carbon dioxide emissions associated with a timber-frame and concrete-framed building in Finland and Sweden. The results showed that the production of materials for the timber framed building required less energy and produced lower CO_2 emissions compared with the materials used for the concrete construction.

In an extension of this work, Gustavsson and Sathre (2006) explored a large number of possible scenarios when comparing two buildings constructed using a timber frame or a concrete frame. These included: clinker production efficiency, cement blending, aggregate crushing, steel recycling, lumber drying efficiency, material transportation distance, carbon intensity of fossil fuel, recovery of logging, sawmill, construction and demolition residues for biofuel, and growth and exploitation of surplus forest not required for wood material production. They found that the timber frame construction method exhibited superior environmental credentials in all explored scenarios except for one. The recovery of wood wastes and timber materials from demolition at end of life and their incineration with energy recovery in place of fossil fuels contributed very significantly to the lower energy consumption and carbon footprint associated with the timber frame construction. The carbon stock in the building and in the forest was assumed to remain unchanged during the 100 year analysis period. It was assumed that the volume of wood removed due to the harvesting of the timber would be replaced by regrowth during the 100 years. In the concrete reference case, it was assumed that the carbon stock in the (unharvested) forest would increase by 50% over the 100-year reference period. Only with the worst-case scenario assumed for wood and the best-case scenario assumed for concrete was the timber building found to have higher carbon emissions and energy consumption. The energy balance for the timber building was always negative, except when the wood processing residue and demolition wood was not recovered and used as fuel. The worst-case scenario for timber assumed low timber drying efficiency, no recovery of logging residue, no recovery of wood processing residue, no recovery of construction waste, no recovery of demolition waste and a low carbon intensity of fossil fuels.

Gerilla et al. (2007) studied a house in Japan with a reinforced concrete or wooden frame, concluding that the timber framed building had a lower environmental impact. Sathre and Gustavsson (2007) showed that only timber has a negative energy of production, compared to the manufacture of particleboard, steel, concrete, or plasterboard. This was because of the high-energy value obtained from the timber processing residues compared to the energy of production. In a later study of an eight-storey timber framed dwelling constructed in Sweden, Gustavsson et al. (2010) found that during the construction phase, more energy could be derived from the biomass residues obtained from processing the timber for the structure than was used to produce the building. Additional energy could also be obtained when wood-based demolition residues were recovered and used as a substitute for fossil fuel.

Asif et al. (2007) compared the environmental impact of building construction, concluding that the embodied energy of concrete was responsible for 65% of the embodied energy of the building, but 99% of the house building sector because of its high impact and the large quantities used.

In research supported by the Portland Cement Association, Marceau and VanGeem (2006) presented a very detailed analysis and comparison of an insulating concrete form house and a timber frame house. A large number of impact categories were used to measure the environmental impacts associated with the two types of structure. The timber frame house was found to have a slightly higher impact in the GWP and all other categories, when compared to the concrete equivalent. The most significant contribution to the environmental burdens was due to the use of electricity and natural gas by the occupants, rather than the materials. Both the heating and cooling requirements of the timber frame house were higher than that of the concrete equivalent. This was due to the thermal mass of the concrete house moderating internal temperature changes and peak energy loads. The two houses used in the comparison were therefore not functionally equivalent. Upton et al. (2008) reviewed several early studies of different construction systems and concluded that: ‘for systems with comparable heating and cooling-requirements, wood-based building systems generally contain lower embodied energy and CO_2 emissions than steel-, concrete-, and brick-based systems’.

In comparison studies of timber and concrete frame buildings, there have been criticisms that end of life carbonation of the concrete has not always been included and that this can affect the outcome. However, Dodoo et al. (2009) showed that even with the inclusion of end of life carbonation of concrete, timber frame buildings were still better performing in terms of environmental profile.

In a review of 48 studies of the environmental impact of the GHG impacts of using harvested wood products, Sathre and O’Connor (2008) investigated whether actively managing forests for wood production compared with leaving them in their natural state is better from the point of view of climate change. Several mechanisms were identified where product substitution affected GHG balances:

- The fossil energy used to manufacture the wood products compared to alternative materials;
- The avoidance of GHG emissions associated with industrial processes, such as cement manufacture;
- The physical storage of biogenic carbon in the wood products;
- The recovery of the inherent energy content in the wood at the end of the product life cycle.

They concluded that there were a range of GHG benefits arising from using wood in place of other construction materials and that there was a clear rationale for using wood to substitute for other materials from the point of view of climate change mitigation.

Salazar and Meil (2009) conducted an LCA of two Canadian residential buildings, one using a conventional construction (timber frame with brick cladding, uPVC windows, asphalt shingles and fibreglass insulation) and a wood frame house that maximised the use of timber. The wood-intensive house exhibited a complete offset of the manufacturing GHG emissions because of the credit given for the forest re-growth. By using the wood materials for energy production at the end of the life of the timber house, it was possible to obtain negative GHG emissions and a nearly energy-neutral construction. Allocating of the biomass energy generation and carbon sequestration in the forest on an economic rather than a mass basis significantly improved the life cycle balances of both types of house.

Sathre and Gustavsson (2009) compared the GHG emissions and embodied energy associated with the construction of a two-storey building made using a timber frame or reinforced concrete. They found that the timber framed building required 72% of the embodied energy and released 56% less CO_2 equivalent emissions over its lifetime. Kellenberger and Althaus (2009) investigated the effect of simplifications upon the outcome of LCAs of buildings. These showed that the transportation of components to the building site was a significant factor and that ancillary components could also have a major influence.

According to Tsunetsugu and Tonosaki (2010), the substitution of wood products for functionally-equivalent steel results in savings of 38% in terms of GHG emissions for buildings up to 3 stories high in Japan.

Bolin and Smith (2011a) undertook a comparative cradle to grave LCA study of a timber deck (treated with alkaline quaternary copper preservative) compared with a wood plastic composite (WPC) equivalent. The timber deck had a considerably reduced environmental impact compared to the WPC equivalent in all impact categories studied except for eutrophication, where the impact was similar.

In a study of the use of different materials in a model dwelling, it was found that the use of modern methods of construction (MMC) (with the assembly of off-site manufactured timber frame panel units) resulted in a 34% reduction in embodied energy compared to the same building constructed using conventional methods (Monahan and Powell, 2011). The materials and energy used in construction was responsible for the emission of 34.6 tonnes CO_2 eq. for a 3-bedroom semi-detached house, which was 405 kg CO_2 eq. per m^2 of useable floor area. Based upon the figures supplied in their paper, a carbon footprint of approximately 381 kg CO_2 per m^2 internal floor area (for all building materials) can be offset by 186 kg CO_2 per m^2 because of the stored carbon in the construction timber in the MMC structure. The importance of maximising the timber content of a building to act as a carbon store can be readily appreciated. Although timber was the predominant construction material in the low energy house, it was found that concrete was the most significant material in terms of embodied carbon dioxide emissions, being responsible for 36% of the materials-related CO_2 eq. emissions.

Guardigli et al. (2011) studied comparable timber and concrete structures. They used a variety of databases and reported the outcome of the study as endpoint impact categories. They came to the conclusion that the timber building had a lower impact overall.

Nässén et al. (2012) considered that the case for favouring timber over concrete as a construction material was more ambiguous. The methodology that they chose involved the evaluation of net present costs and carbon balances over the whole life of building structures with different material compositions. They modelled the same building type that had already been used in the studies of Adalberth (2000), Gustavsson et al. (2006b) and Dodoo et al. (2009). They noted that although some studies had used material substitution factors of 1:40 (1 tonne of timber replaces 40 tonnes of concrete), they considered this to be too high and chose a factor of 1:10. The energy costs for erecting and operating the buildings were not included since they were assumed to be independent of building type. In the study of Dodoo et al. (2009), recycling rates for timber and steel were assumed to be 90%, which was considered optimistic by Nässén et al. (2012) who assumed 80%. Building lifetimes were assumed to be 100 years and two scenarios were considered with construction years of 2010 and 2050. They used three energy supply system scenarios in their modelling, one of which was a baseline scenario, one a climate constrained scenario with renewable energy dominating and one climate constrained and using carbon capture and storage (CCS) from the year 2020. It was assumed that the timber was incinerated with heat recovery at the end of use. The carbon emissions and uptake for forest biomass (growing during the lifetime of the building) and carbonation in the concrete were equal in the scenarios. For the 2010 scenario, it was found that the concrete used in construction was responsible for twice the CO_2 emissions compared to the timber frame. Wood frames had a better carbon balance in most scenarios, but this advantage was strongly affected by the different land use scenarios in the models. Where wood was felled for construction, the land was used for new growth and the carbon sequestered was counted in the model. Where concrete was used, land was assumed to be left unharvested and the amount of sequestration was assumed to be much less since the forest was mature. Alternatively, the forest was felled and the wood was used for pulp, with the land now being used for bioenergy production and carbon credits for substitution of fossil energy were included. The residues from wood processing (0.5 tonne for every tonne of wood) were used as an energy source. No consideration was given to the use of small roundwood thinnings or harvesting residues for energy production, both of which would improve the situation in favour of wood. No carbon credits were allocated for the carbon sequestered in the timber in the building structure. The energy used for the materials in the timber frame building was 1.68 GJ/m^2 and 2.26 GJ/m^2 for the concrete building.

Aktas and Bilec (2012) pointed out the lifetime of a building had a very important influence on the results obtained from LCA when comparing different products. They found that US residential buildings had a mean lifetime of 61 years and that interior renovation energy consumption had a mean of 220 GJ over the life cycle of the model and represented about one third of the initial embodied energy of the materials for a conventional home. They concluded that choosing an arbitrary lifetime for a building and ignoring interior renovation impacts introduced a major error into LCA calculations for interior buildings.

A cradle to construction gate LCA comparison of a North American mid-rise office building made using cast-in-place concrete and a functionally equivalent building constructed from cross-laminated timber (CLT) was undertaken by Robertson et al. (2012). The LCA showed that the CLT building had a lower environmental footprint in all impact categories (except fossil fuel depletion), significantly so in the case of GWP (30% of the impact of the concrete building). The energy stored in the timber structure (embedded energy) available for fuel use at end of life was also significantly higher (71.3 TJ compared with 13.3 TJ for the concrete structure). This end of life phase with energy recovery was not included in the cradle-to-construction gate analysis, where it would significantly reduce the fossil fuel depletion impact, if incineration with energy recovery was used. The paper also pointed out that uses must be found for the large quantities of timber available due to the devastation caused by the mountain pine beetle (estimated to be one billion cubic metres of timber before the epidemic subsides in British Columbia). This epidemic has resulted in Canadian forests becoming net emitters of carbon dioxide because of the huge volumes of timber presently decaying in the forests. Such unpredictable events cannot be modelled.

Gong et al. (2012) studied three residential buildings constructed using frames constructed of concrete, steel or timber, finding that the timber frame performed better overall in terms of energy consumption and CO_2 eq. emissions. The embodied energy of the concrete structure was similar to that of the steel frame and 30% higher than that of the timber frame. The associated GHG emissions (CO_2 equivalents) of the concrete structure were 49% higher than those attributable to the wooden frame equivalent.

Nässén et al. (2012) compared the whole life carbon emissions associated with buildings constructed using concrete or timber frames. The model assumed that the land used for providing timber for the timber framed house was replaced with new long-rotation forest for the future production of wood materials. For the concrete framed house it was assumed that this land area of forest was harvested and used for some other purpose and that future production was used for bioenergy production. The scenario analysis showed that the timber framed house had a better carbon balance in most cases, but that the outcome was particularly sensitive to the land-use assumption. The environmental impacts associated with the construction materials were quite small compared to the whole life cycle and the two types of construction were almost identical in terms of environmental impact in nearly all scenarios. The energy demand for producing the materials for the timber framed building was 1.68 GJ/m^2 and that for the concrete framed building was 2.26 GJ/m^2. This paper refers to work by Gustavsson and Sathre (2006) and to Adalberth (2000) which shows that the energy difference in the use phase is less than 1%. However, the authors note that over the lifetime of a building that this difference is not insignificant when compared to the embodied energy of the materials. The difference in the use-phase energy was about 0.34 GJ/m^2, which was almost 60% of the difference in embodied energy of the materials (0.58 GJ/m^2). Their study concluded that the difference in impacts between the concrete and timber framed houses and the advantage of one system over another was not immediately obvious.

Ximenes and Grant (2013) examined the GHG impacts associated with the construction and use of a domestic house (5-year lifespan) with a conventional construction and a timber maximised design. It was found that the timber-maximised design had half the GHG emissions compared with the conventional structure.

McDevitt and Allison (2013) conducted life cycle assessments of four building types: comparing concrete and timber flooring, steel and timber framing and fibre cement and timber cladding. The substitution of steel framing by timber resulted in a decrease in global warming emissions, eutrophication and photochemical oxidation, but an increase in land and water use. Substituting a concrete floor with timber gave similar results.

Although LCA has the potential to be used to help with materials selection in the built environment, it is important not to make choices based upon over-simplified assumptions or data analysis. Buyle et al.

(2013) reviewed the literature examining LCAs of the building sector and noted that timber structures generally had more favourable LCAs, citing lower embodied energy associated with wood and its carbon neutrality, citing Cole and Kernan (1996), Gerilla et al. (2007) and Mithraratne and Vale (2004) in support. However, they also noted that the research of Marceau and VanGeem (2006) which concluded that a concrete structure was preferable to timber. The most significant environmental impacts according to Marceau and VanGeem (2006) were due to the use of electricity and natural gas by the occupants of the houses and not due to the production of the construction materials. The occupational energy use represented 96% of the total energy use of the concrete framed house and 97% of the total energy use of the timber framed house. The functional unit was a single-family house of 228 m^2 living space with a lifetime of 100 years, including maintenance and replacement of components. The non-equivalent houses were modelled for different climate zones in the US and with different orientations. In all modelled scenarios, the timber frame house had higher energy use compared with the insulated concrete form house and this resulted in higher environmental impacts in nearly all impact categories. The modelled concrete house performed better than the timber framed house because it had better insulation and also because of the thermal mass of the concrete. For the materials impacts, cement with 25% fly ash was used for the concrete. Energy recovery from the timber at end of life and from by-products was not included in the calculation, but given the very low contribution of the materials to the overall impact, it is unlikely that this would have made a significant difference to the final result.

Basbagill et al. (2013) undertook a study to understand which building material design choices had the greatest environmental impact, for a mid-rise multi-residential building. They found that the largest change in impact came from choices regarding the building cladding and the second greatest change from choices regarding the substructure. Choices regarding column and beam were sixth in the ranking of materials' choices, but CLT was not included in the assessment. They obtained their data from SimaPro and the Athena EcoCalculator. Sequestered carbon was not included in the assessment.

Cabeza et al. (2013, 2014) conducted a review of published LCA studies of building materials (cement and concrete, bricks, timber, rammed earth, sandstone) focussing on embodied energy and CO_2 eq. emissions. This review showed that material substitution using low embodied energy materials could make an important contribution to reducing the environmental impacts associated with the building sector.

Dodoo et al. (2014a) compared the primary energy consumption of a multi-storey timber structure made from cross laminated timber (CLT), a beam and column system using glulam and laminated veneer lumber (LVL) and pre-fabricated modules using light frame volume elements. The highest consumer of primary energy was the timber in the construction in all cases, reflecting the high volume of timber used. When compared on a weight for weight basis, the timber had a much higher embodied energy than concrete, as was also reported by Hill and Dibdiakova (2016). However, the energy recovered at the end of lifecycle for the timber was considerable. They also found that the amount of carbon (as CO_2 equivalents) stored in the timber in the buildings, exceeded that emitted due to the total material production and construction, illustrating the benefits of using timber in construction as a means of offsetting GHG emissions (Dodoo et al. 2014b).

Takano et al. (2014) used a five dwelling three storey reference building (gross floor area 1 243 m^2, heated floor area 986 m^2) planned for Helsinki, to make a comparison of the environmental impacts associated with the use of different materials. Windows and doors were excluded from the analysis, since they were the same in all cases. The following indicators were used: Resource use, embodied energy (EE), energy content (EC), embodied greenhouse gas emissions (GWP), carbon storage (CS), material cost. The embodied energy was divided into renewable and non-renewable components and reported as primary energy. Six frame materials were analysed: light weight timber panel (LWT), cross laminated timber (CLT), reinforced concrete panel (RCP), autoclaved aerated concrete (AAC), brick (BRK), light gauge steel (LGS). The LCA analysis was performed using the Ecoinvent database, with energy content reported as lower heating value. The results for embodied energy, global warming potential, energy content and carbon storage per m² of heated floor area are shown below in Table 24. Note that all of the buildings have some recoverable energy because they all have some timber content.

Table 24 Environmental impact data per m^2 heated floor area for different frame building materials from Takano et al. (2014).

	EE (MJ)	EC (MJ)	GWP (kg CO_2 eq.)	CS (kg CO_2 eq.)
LWT (light weight timber panel)	2738	-2275	123	-205
CLT (cross laminated timber)	4082	-3896	186	-348
RCP (reinforced concrete panel)	2418	-712	221	-53
AAC (autoclaved aerated concrete)	3066	-696	256	-53
BRK (brick)	2946	-712	263	-53
LGS (light gauge steel)	2466	-729	159	-52

EE: embodied energy, EC: energy content, GWP: global warming potential, CS: carbon storage

The storage of carbon in the buildings was accounted as a negative flow, as was the energy recovered from the materials. All of the options also had a certain amount of timber, which accounts for the carbon storage occurring with all of the building types. Other materials, such as bitumen and polymers which had a calorific content were also included in the inventories. The CLT option had the highest embodied energy per sq. metre, but the inventory did not state how much CLT was used. The timber options also had the highest value of recoverable energy and the highest levels of carbon storage, both yielding negative GWP impacts for the building fabric. There is no indication that carbonation of the concrete was included in the calculations.

Grant et al. (2014) compared three different wall systems: brick, timber cladding and aluminium. Five different service life models were used and the analysis included material manufacturing, construction, operation and maintenance. The building models were manipulated so that they all had the same operational energy requirements. The analysis found that the life cycle impact was dependent upon modelled frequency of the maintenance cycles and material replacement intervals. End of life scenarios were not considered in the analysis. For the GWP impact category, the brick wall was the best material option of those studied.

Pajchrowski et al. (2014a,b) conducted an LCA upon four building types in a Polish context:

- A traditional masonry building;
- Passive masonry building;
- Traditional wooden building;
- Passive wooden building.

All of the buildings were of the same functional unit, of 98.04 m² floor area and with an assumed lifespan of 100 years. Maintenance cycles were included and replacement of various elements, such as windows and doors was also included. The final disposal of waste was also included in the analysis, for the timber this was incineration with energy recovery. The climate benefit of CO_2 storage of the timber in the structure was also accounted for. The results were reported on the basis of ecopoint impacts using IMPACT 2002+ (a mixture of endpoint and midpoint impacts). Both the timber buildings were found to have a lower impact compared with the masonry equivalents.

In a study of the air pollutant emissions and water footprints of buildings in China, Chang et al. (2016) concluded that public buildings had higher impacts than residential buildings because their heavier structural designs relied upon the use of large quantities of steel and cement. In a study of the embodied energy used in the construction of agricultural buildings in Norway, it was found that the one of the largest contributors to EE was the amount of concrete used in the building (Koesling et al. 2015).

Huang and Bohne (2012) conduced an input and output analysis of embodied atmospheric emissions in the Norwegian construction sector and showed that their contribution to total national emissions inceased between 2003 and 2007. Nine types of main air pollutants were considered in the study: greenhouse gasses (CO_2, N_2O and CH_4), acidification precursors (NO_x, SO_x, NH_3), ozone precursors (NMVOC, NO_x, CO and CH_4) as well as PM10. The intensities of GHG emissions, however, are expected to slowly decrease by 2020. They found that the largest potential for the reduction of embodied energy emissions lies in the use of low embodied energy materials in the building sector.

Fouquet et al. (2015) used a dynamic LCA approach (Levasseur et al. 2010) to determine the GWP impact of three houses having the same floor area, same geometry and same level of insulation but built with different materials. The houses were all built to a Passivhaus standard and either manufactured from timber, cast concrete, or concrete blocks. Two scenarios were considered for the timber house: landfilling, or incineration with energy recovery. Biogenic carbon was included in the study, with the timber assumed to come from a sustainably managed pine forest and CO_2 uptake by trees counted as a negative emission. For the landfill scenario, the wood was considered to be poorly degraded during the lifetime of the project, with 1.3% of the mass being converted to CO_2 and 1.9% as CH_4 and 75% of the remainder was recovered and burnt (with release of CO_2) but with no energy recovery. For the concrete houses, the carbonation of cement was included. The operating energy and recurring embodied energy were all included in the model. A 100 year time span was used for the building and replacement of materials within this time span was included where appropriate. This analysis showed that the cast concrete house has the greatest GWP impact followed by the concrete block house. Both the instantaneous and cumulative radiative forcing were considered. The timber house had a much lower impact (35% for a 100-year time horizon and 45% for a 150 year time horizon).

Rebane and Reihan (2016) evaluated a building made with concrete or timber and reported that the timber alternative had a lower embodied energy, even when using the Bath ICE database for the embodied energy data. Nepal et al. (2016) studied the implications of increased wood use in low-rise non-residential construction in the US. Increased use of wood was found to lead to an overall emissions reduction of 870 million tonnes of CO_2 eq., or 2.03 tonnes CO_2 eq. per tonne of wood used.

Røyne et al. (2016) used a range of different methods to calculate the GWP of a timber building and an equivalent concrete building. The studies included no credit given for carbon storage in the timber, as well as discounting of emissions at 1% per year, the GWP_{bio} methodology developed by Cherubini et al. (2011b,c) and further extended by Guest et al. (2013), ILCD methodology and dynamic LCA (Levasseur et al. 2010). Their study included potential climate effects arising from soil disturbances or albedo changes, as well as emissions of GHG. Although there were variations in the results obtained, depending on the methodology employed, the data consistently showed that the timber building had a lower GWP impact compared to the concrete alternative. The main influence on the results was the operational phase of the life cycle. During the operational phase, the concrete building was found to have a greater capacity to store heat compared to the timber building. The concrete building had a larger GWP impact compared to the timber building in the production phase in all scenarios considered. This work included a review of the literature where they found that none of the 101 papers included a credit in the LCA for avoided radiative forcing due to storage of atmospheric carbon in long-life timber products.

Mitterpach and Štefko (2016) calculated the environmental impacts associated with the construction of a timber and a brick house. The analysis showed that the timber house had a lower environmental impact compare with the brick equivalent. The two houses were identical apart from the materials used in construction. The GWP impact of the wooden house was 48 tonnes CO_2 eq. and that of the brick house was 103 tonnes CO_2 eq.

There are very few LCAs of tall buildings at the present time (Trabucco 2016). Skullestad et al. (2016) undertook a study where they compared the climate change impact of buildings from 3 to 21 storeys made from reinforced concrete or from cross laminated timber. They found that there was a climate change mitigation benefit from using timber. Different scenarios were modelled (best case, medium and worst case), which included carbonation of the concrete during the lifetime of the building and at end of life after crushing of the concrete. Reinforcing steel was modelled with different recycled contents. It was assumed that the timber was derived from a sustainably managed forest, where the biogenic carbon pool was constant. The LCI data was obtained from information gathered from Norwegian forestry and timber producers in the MIKADO project (Wærp et al. 2009). This used average data obtained for spruce and pine for production chains and sawmills, which showed that most of the emissions in the supply chain came from fossil fuel use by harvesters and for transport. Electricity use was the main source of impacts in the rest of the production chain. Several different allocation procedures were applied to deal with the multi-output processes in the timber production chain. Where incineration of the timber residues and wastes was modelled, this was assumed to replace natural gas for heating. Storage of atmospheric carbon in timber for the lifetime of the building was dealt with using the method described by Guest et al. (2013).

12.3 Building components

Engelbertsson (1997) compared the environmental impacts of the production of steel and glulam beams used for support in roof construction (functional unit 1 m^2 of roof), finding that the steel beams had impacts 2-8 times higher than the timber alternatives. Jarnehammar (1998) studied roof and floor constructions (FU = 1 m^2, lifetime 100 years) made from crosslam, lightweight stud, or concrete construction. It was found that the timber structures had approximately 60% lower GWP impact compared with the concrete construction. Jarnehammar (1998) also studied a timber wall compared with a plaster façade, finding that the timber system had a 5-40% better performance in terms of GWP impact. Bergman et al. (2014) compared wood-based products with a range of non-wood alternatives (framing, doors, decking, cladding) and reported that in all cases the use of wood as a material resulted in lower net carbon emissions. Bolin and Smith (2011b) compared a house frame constructed using borate-treated timber, compared to a functionally-equivalent structure made of galvanised steel. The impact categories reported were: GHG emissions, fossil fuel use, water use, acidification, ecological diversity, smog forming potential and eutrophication. The cradle to grave impacts of the timber frame structure were approximately 4x lower for fossil fuel use, 1.8x less for GHG emissions, 83x less for water use, 3.5x lower for acidification, 2.5x lower for ecological impact, 2.8x less for smog formation and 3.3x less for eutrophication, when compared to the functionally-equivalent steel frame.

In a comparison of different flooring materials (carpets made of wool, polyamide, vinyl flooring, linoleum and solid oak) it was found that the wood flooring had the lowest GHG emissions, although linoleum had very similar performance in this respect. The assumptions that had the greatest influence on the results were the choice of discount rate, carbon fixation in the forest and waste handling of the materials (Petersen and Solberg 2004).

Citherlet et al. (2000) compared window frames made from different materials (wood, plywood, aluminium, PVC, wood-aluminium, plywood-aluminium), with an LCA over the whole life cycle. The highest impact in the impact categories Embodied energy, GWP, Acidification Potential, was aluminium followed by PVC, the highest impact in the category Photochemical Ozone Creation Potential was PVC followed by aluminium. Salazar and Sowlati (2008) undertook a literature survey of published LCAs of windows, finding that in all the published studies the timber framed windows had the lowest embodied energy. However, the use-phase dominated the impacts in the studies and this was very heavily influenced by the maintenance requirements of the windows. The reviewed studies (with one exception) did not conduct sensitivity analysis, ignored uncertainties and provided results based upon what they considered to be the most likely scenarios. Because of this unwillingness to explore the uncertainties, the usefulness of the LCAs was limited. An LCA study of wooden windows was undertaken by Tarantini et al. (2011). This showed that the use-phase of the life cycle contributed most to the GWP impact category, but that the manufacturing phase was responsible for the greatest impact in the photo-oxidant formation category. A maintenance cycle of 5 years was assumed for the windows, with repainting using a water-based paint. At the end of their technical life of 30 years, it was assumed that the windows were recycled (30%) or disposed of to controlled landfill, with the further assumption that 25% of the methane derived from the decomposition of the wood in the landfill was recovered and burnt, with the remainder released to the atmosphere, thereby contributing to the greenhouse effect.

A comparison of the beams used for the new Gardermoen airport terminal showed that the use of glulam for the construction resulted in a reduced embodied energy of 2-3x and a reduction of 6-12x in the use of fossil fuels compared to the use of steel for the equivalent purpose (Petersen and Solberg 2002). They also compared the use of a natural stone floor with a wooden equivalent, finding that the stone floor had a lower embodied energy, but resulted in higher overall GHG emissions (Petersen and Solberg 2003). It was necessary to use the wooden floor as a biofuel at the end of the lifecycle to ensure that the GHG footprint was more favourable.

The use of timber windows and doors and surrounding frames was compared with PVC coated steel core equivalents, with the analysis including maintenance, building services, operational energy and replacements (Blom et al. 2010). The biggest impact was found to arise from the use of older glazing units, since the environmental impact from energy use for heating was found to exceed that from maintenance and replacement activities by a factor of 40. The environmental impact of using PVC components was found to be 1.5-2 times larger for every impact category, except that for ozone layer depletion, where it was the same. However, the main impact from the PVC components actually came from the steel-core of the frame.

Broun and Menzies (2011) undertook an LCA analysis of a partition wall system made from brick, hollow concrete block, or timber frame. Per square metre of wall, the GWP (CO_2 eq.) was 25.5, 10.4, 3.1 and the embodied energy 191.2, 93.8, 38.3, for the manufacture of these functional units from hollow concrete block, brick and timber, respectively. Timber also performed well in other parts of the life cycle of the wall.

The effect of different lifetimes on the GHG emissions impact for a wall made of different materials (concrete, brick, stone, wood, aerated concrete) was studied by Mequignon et al. (2013). It was found that the lifespan of the product had an important influence on the outcome of the study. Extending the lifespan of the wall was found to have a major influence and of greater importance compared with choosing the materials with the lowest impact at the build stage of the process. Stone was found to have the lowest overall impact due to its long life (500-1000 years), but a material lifetime does not reflect the lifetime of the structure which may be replaced for reasons other than the failure of the material. However, the wood wall was found to be the most effective solution, whatever the lifetime of the structure because of the long-term storage of atmospheric carbon dioxide in the material.

The effect of end of life fates of construction materials upon the outcomes of LCA studies was investigated by Sandin et al. (2014), who studied two roof support systems: glulam beams and steel framing. In all comparable scenarios, it was found that the glulam beams had clear environmental advantages over the steel frames, except in a scenario where the steel frames were recycled (with current steel production technology), where the environmental impacts were similar. The importance of considering the end of life phase was shown in this study, along with the use of a consequential or attributional approach to the LCA. When considering the end of life fates of construction materials, the credits that may accrue through such practices as recycling, re-use, disposal, or incineration, depend upon whether current or potential future technologies for production are considered.

Iribarren et al. (2015) conducted an LCA comparative study of a functional unit comprising 1 m² of external wall, involving 175 different wall systems with an assumed life span greater than 50 years and insulation of U = 0.15 W/(m²K). The impacts associated with the production phase were included, but those with the use phase and end of life were not. Based on this analysis, the lowest impact constructions were timber frame or solid wood structures and two massive block (lightweight concrete with pumice, adobe) structures. The most suitable insulation materials were cellulose, hemp, cotton, or mineral wool. Kahhat et al. (2009) compared exterior wall systems composed of concrete block, poured in place concrete, wooden frame and steel stud framing. In the cradle to gate part of the life cycle, the timber frame walls had the lowest environmental impact of the wall systems studied. During the in-service phase, the insulated concrete walls had the lowest impact because they required less operational energy and over the 50-year assumed life span they consumed 5% less overall energy (operational and embodied) than the timber frame systems. The use phase accounted for over 90% of the total energy in all of the systems studied. The study highlighted the importance of thermal mass of the walls for improving the performance of the buildings in a hot climate zone. This study clearly shows how using different functional units can affect the outcome.

Chau et al. (2015) quoted GHG emissions (in kg CO_2 eq. per kg) and embodied energy intensities for a range of building materials, including plywood, obtaining these from the Bath ICE database, Basbagill et al. (2013) and Jeong et al. (2012) (Table 25).

Table 25 Embodied energy and GWP data for building materials (per kg) taken from Chau et al. (2015).

Building material	Embodied energy (MJ)	GWP (kg CO_2 eq.)
Aluminium	155.0-227.0	8.24-11.40
Bricks, blocks	0.9-4.6	0.20-1.13
Concrete	0.5-1.6	0.050-5.15
Galvanised steel	35.8-39.0	2.82
Glass	15.0-18.0	1.06-1.50
Stone, gravel, aggregate	0.3-1.0	0.016-0.056
Paint	20.0-81.5	2.95-3.56
Plaster, render, screed	1.4-1.8	0.12-0.16
Plastic, rubber, polymer	67.5-116.0	2.20-16.20
Plywood	8.5-15.0	0.75-1.35
Precast concrete element	2.0	0.22
Reinforcing bar and structural steel	9.9-35.0	1.03-3.51
Stainless steel	51.5-56.7	3.38-6.15
Thermal and acoustic insulation	3.0-45.0	0.15-0.86
Ceramic and tile	0.8-11.1	0.43-0.65

In a study of window frames made from different materials, the option with the lowest impact was an aluminium framed window according to Carlisle and Friedlander (2016) in a study funded by the International Aluminium Foundation. The four materials options studied were aluminium, wood clad aluminium, wood and uPVC framing. Each window frame option was considered against a full building lifetime, which was considered to be 80 years, with the data normalised to include the amount of material required to make 1 m² of glazing. The LCA included end of life scenarios and a range of in-service scenarios, with different maintenance regimes and different replacement lifetimes. These different scenarios had a significant effect upon the environmental impacts and it was recommended that this should be the subject of further research. The work showed that the maintenance of the wooden window was a much more significant factor compared to the manufacture or disposal phases of the lifecycle. The aluminium frame window scored better due to reduced maintenance compared to wood and also received higher credits for recycling of the aluminium at the end of life.

In a comparison of a CLT wall panel and a functionally equivalent wall made of hollow bricks, it was found that the brick wall had twice the GWP impact of the CLT equivalent (85.9 kg CO_2 eq. compared with 35.2 kg CO_2 eq./m^2 of wall) (Santi et al. 2016). The 1 m^2 of CLT wall that formed the functional unit contained 97.1 kg of oven-dry wood, which stores 178 kg of atmospheric carbon dioxide. According to this paper, a 100 m^2 house would require 40 m^3 of CLT. The displacement factor calculated for the CLT wall compared to the brick wall was 0.52 tonnes CO_2 eq. per tonne of wood, based on the emissions only. This is much lower than the average displacement factor of 3.9 tonnes CO_2 eq. per tonne of oven-dry wood calculated by Sathre and O'Connor (2010) in their meta-analysis. But as noted previously, they found a wide range of displacement factors in their study, with post-use of the wood being a very important factor affecting the magnitude of the GHG emissions of the wood product life cycle. The use of wood as a biofuel at the end of life cycle is a very important contributor to reducing GHG emissions, but the magnitude of the savings is strongly influenced by the type of fossil fuel that is being substituted.

12.4 Bridges

An LCA was conducted on three Norwegian bridges (a concrete box girder bridge, a steel box girder bridge and a wooden arch bridge) by Hammervold et al. (2013). Although the bridges were similar, they were not exact comparisons (e.g., the wood bridge had a deck area of 229 m^2 and that of the concrete bridge was 417 m^2). The environmental impacts were reported for 1 m^2 of deck area. The study included all phases of the life cycle, including end of life, with the steel being recycled, the concrete used as filling material. The wood was assumed to be incinerated and it does not appear that the assumption included energy recovery, hence no benefits accrued from the substitution of fossil fuel. The benefits associated with re-use, or incineration, were considered to arise due to diversion from landfilling. The wooden bridge had the lowest GWP per m^2 of the three bridges studied, but came mid-way between the other two bridges (with concrete the lowest) when the weighted environmental impact categories were aggregated. As noted previously, the use of weighted impact categories and their aggregation carries with it large uncertainties and the use of value judgments.

13 Conclusions

Most of the studies reviewed at the building level indicate that there is a significant environmental benefit from using timber, rather than concrete or steel in construction. The comparison needs to be made with reference to an appropriate functional unit. For the purposes of a study of this nature, the most appropriate functional unit would be at the building level.

Timber use has nine main advantages over the use of other construction materials in Norway:

- Timber is a renewable resource available in perpetuity.
- Increased harvesting levels of Norwegian forests will maintain a younger age profile, resulting in higher levels of carbon sequestration than would be the case if the forests were allowed to mature. Managed forests are less susceptible to major losses of biomass due to fire.
- Harvesting and processing activities take place in rural areas, securing jobs and creating economic activity in these regions.
- Norwegian timber is indigenous and increased production and use of harvested wood products is good for the Norwegian economy.
- Increased Norwegian expertise in CLT production will allow for higher value-added products to be exported to major markets, such as the UK.
- Timber processing residues can be used as an energy source, substituting for fossil energy sources.
- Timber is a low embodied energy material when compared to other building materials on a functional unit basis. It also has a lower GWP impact compared to other building materials on a functional unit basis.
- At the end of life, or multiple lives, timber can be incinerated with energy recovery, resulting in low levels of demolition waste. The greatest benefit would be achieved by replacing fossil fuels in Norwegian cement production. The ash left over from the incineration process could be used as a clinker substitute in cement production. This would be most easily achieved by using waste timber as an energy source in the cement kiln, which would reduce fossil carbon emissions.
- Timber contains sequestered atmospheric carbon dioxide and there is a climate change mitigation benefit to be gained from using timber in long-life products in the construction sector. This benefit continues during the time of storage and increasing the use of timber in construction will ensure that the harvested wood products pool acts as sink during this critical time in human history. Ensuring that these structures have long lifetimes, extends the carbon storage benefit.

Life cycle assessment is a complex analytical tool that can potentially be used to inform choices regarding the selection of materials for the built environment. However, due to the complexity of the method there are considerable difficulties associated with the use of LCA as a decision-making tool. In order that LCA can be used to make comparative assertions it is necessary that these requirements are met:

- Only consequential LCA can be used when comparing different materials for use in construction.
- The functional unit should be the same. If two buildings are compared in a scenario they should have the same energy performance in terms of heating and cooling requirements. System expansion is necessary to include the building plus the generation of energy in order to take account of the recovered energy from waste wood.
- It is necessary to consider the whole life cycle of the materials using realistic assumptions regarding maintenance cycles and end of life scenarios.

- Uncertainties need to be studied in greater detail using sensitivity analysis.
- The LCAs should be transparent and employ appropriate sensitivity analyses to show the effect that different assumptions have on the outcome. Very few studies meet these requirements, often for reasons of commercial confidentiality.
- An appropriate set of physical and temporal system boundaries needs to be chosen. The whole life cycle of the building needs to be considered. There may be arguments for extending the LCA beyond the life of the material, to consider recycling, or incineration with energy recovery. The comparison of timber with other building materials requires consideration of the energy recovery that is possible from timber thinnings, processing residues and also timber at end of life of the structure. It is also necessary to make appropriate allocations of environmental burdens, or use system expansion correctly. This is particularly important when the energy from timber residues is taken into account and compared with materials which have no inherent energy content. There is also the issue of concrete carbonation to be taken into account, although this is not as significant at end of life as is sometimes stated. Concrete carbonation in real-life situations is not fully understood and assumptions regarding carbonation at end of life require justification. These uncertainties need to be explored.
- Appropriate allocations need to be made with respect to timber co-products. This can be very complicated for timber products since the system boundary also includes the forest and forest operations from planting to harvest and may also include the next rotation with energy credits arising from thinnings and carbon sequestration by the growing biomass. This is a complex issue and can have a significant influence on the results. Allocations can be made on the basis of mass, economics or energy content; each usually giving different results. A sensitivity analysis should be employed to illustrate this.
- The assumptions that are made regarding the whole life cycle of the building and materials can have a very profound influence upon the associated environmental impacts. There needs to be an open transparent presentation of the assumptions made and in order to back up the decisions made regarding those assumptions, it is necessary to perform a sensitivity analysis.
- The deciding criteria as to what constitutes a lower overall environmental impact has to depend upon a value judgement. Different environmental impacts can have greater importance, depending upon the temporal and spatial scale considered. For this report Global Warming Potential and embodied energy were deemed to be of the greatest importance. They are also two impact indicators which are considered to be the most reliable.
- Most LCAs produced in Europe either use the GaBi or Ecoinvent databases and for the embodied energy and GWP impact, these should give similar answers. Where other databases are used, it cannot be assumed that this will be the case. Where comparisons are to be made the same impact assessment methods must be used.
- GWP and sequestered atmospheric carbon dioxide must be dealt with separately in the LCA and reported separately.

The introduction of environmental product declarations (EPDs) which use common product category rules (PCRs) has gone some way to meeting these requirements and to some extent allow comparative assertions to be made at least regarding the cradle to factory gate stage of the life cycle. However, the huge variety of scenarios regarding the other phases of the life cycle make comparisons extremely difficult. This situation has not yet been resolved satisfactorily.

Where timber waste or residues are used as an alternative energy source to fossil fuels, the magnitude of the climate change mitigation benefit depends upon:

- The fossil fuel substituted: Greater benefits are obtained if coal is replaced, followed by oil and finally natural gas. This is because proportionally more energy is obtained from the hydrogen atoms as one moves from coal to natural gas. In Norway, these benefits are more likely to arise if biomass is used for space heating because this is where fossil fuel is more likely to be used. Replacing the coal used in the kiln of the cement clinker production process would have the greatest benefit in a Norwegian context.
- If the biomass energy is used to generate electricity, then the benefit is different if the marginal or average grid primary energy mix is considered. Hydro-power is responsible for 96.1% of the grid mix, with 2.5% thermal power and 1.4% wind power (Statistics Norway). An LCA of wood use for energy in Norway would not produce necessarily yield GWP credits for electricity production. This would depend upon whether a marginal or an average approach to the replacement of primary energy for electricty production is used.The use of wood to generate electricity has to be viewed as part of a plan to increase bionergy electricity production in the country. The choice of whether to increase the use of biomass, wind of hydropower for energy production is a political decision, which is influenced by more factors than just climate change mitigation. Whether decisions of this nature can be based solely upon a single 'total sustainability impact assessment tool' is, at best, highly questionable.

The choice of where and when to use wood residues and wastes is therefore a strategic one, which depends upon whether the Norwegian Government views an increase in bioenergy production as an important element in a climate change mitigation strategy. The greatest benefits would arise when using combined heat and power facilities in the cement kilning process with biomass residues as the energy source.

Maintaining the harvesting of managed forests is important from a climate change mitigation point of view. Sustained, or even intensified harvesting will ensure that the age structure of the forests will have more vigorous young trees, thereby continuing to maintain high levels of carbon sequestration. Management will minimise risks dues to fires and storm events. Any potential decrease in soil carbon is likely to be outweighed by the substitution benefits of material substitution in buildings and storage of atmospheric carbon dioxide in the long-life HWPs used in construction.

The relative importance of the embodied energy associated with building materials increases as the operating energy of the building is reduced. But is must also be taken into consideration that the embodied energy will assume a greater importance in a growing building sector with a growing population.

The large increases in the number of published EPDs is making the task of comparing the environmental burdens of building materials easier, but there still remains errors in the data. EPDs are necessarily brief and do not contain all of the information that is required for interpretation. Although subject to a verification procedure, this is an inexact process that does not always pick up errors. Verification has to be a relatively superficial process in order to avoid excessive expense. EPDs are costly to perform and not always an option for small companies that may be at the cutting edge of innovation in low environmental impact building technologies. The assumptions that are used in different scenarios in EPDs may not be realistic and it is often difficult or impossible to obtain the original data from the reported information. As an example, it is necessary to report biogenic carbon emissions and fossil carbon emissions separately, the carbon sequestered in biogenic materials and assumed carbon absorbed in cement products due to carbonation should be reported separately. The advantage of EPDs is that data is separated into different life cycle stages, which does help with understanding of where the main impacts occur. It has to be accepted that LCAs and EPDs will be influenced by commercial considerations and that this requires careful monitoring. EPDs are often viewed as marketing tools used to claim advanatges over rival products.

This review has shown that there is need to improve the data quality on the inventories of materials used in the Norwegian construction sector. There is no up to date information on materials flows through the sector, the lifetimes of materials in the buildings and fate of materials at the end of life.

Impact assessment tools for buildings have some limited scope for raising the awareness of environmental issues in the construction industry, but have no place in policy decision making. This is also the case for impact assessment methods that aggregate all environmental impacts into one overarching category, or measurement. Only LCA has the necessary robustness to serve as a tool to inform policy, but it is an imperfect tool and the consequences of the uncertainties and scenario assumptions need to be thoroughly explored.

14 Appendix: Summaries of previous reports

Kittang, D., Narvestad, R. and Nyryd, A.Q. (2011) Tre i by – en kunnskaps-oversikt; SINTEF Prosjektrapport 74

English title: Wood in cities – an overview about what we know

Besides a review on the technical properties of wood as a building material and wood research institutions that work with wood in Norway, the Kittang et al. (2011) report studies different levels of decision making for material choices. Different types of builders in the building process, how the building will be used later and who is responsible for maintenance and use phase is important for the choice of material. Those builders being responsible for maintenance and use phase put more effort into the choice of material. The architects have a larger influence on the exterior than the actual load bearing construction, in contrast to the engineers who have a special influence on the load bearing construction. Due to the variation in wood, and sometimes too little knowledge and routines, the use of wood in large constructions was ignored. Missing, wrong and too little information was also the reason why wood often was not the first choice for contractors.

Statsbygg (2013) Tre for bygg og bygg i tre, Kunnskapsgrunnlag for økt bruk av tre i offentlige bygg, Analysedokument fra Strategi- og Utviklingsavdelingen

English title: Wood in buildings and building in wood, knowledgebase for increased use of wood in public buildings

The report (Statsbygg 2013) studies the situation on the building market in 2013 in Norway and potentials for increased use of wood in the public sector, especially the potential role of the public procurement as drivers for this development. It was concluded that there is a large potential for increased use of wood in the urban and public building sector. Despite long traditions in building with wood, the material is mostly used in single family and small house dwellings and not in the urban building sector. In 1998 building regulations in Norway opened for building in wood with more than three storeys and even though there is good knowledge on the material available, prefabricated and standardised solutions for building with wood in taller buildings have not yet been developed. This puts wood at a disadvantage in competition with other building materials. Public procurement can be a driver for developing structures and solutions that promote and facilitate increased use of wood.

Denizou, K., Hveem, S., Time, B. (2007) Tre i by – Hvilke mekanismer styrer materialvalget for større urbane byggverk? KMB-forprosjekt (NFR) «Fellessatsing Tre», SINTEF Byggforsk Prosjektrapport 409 – 2007

English title: Wood in cities – Which mechanisms control material choices for larger urban buildings?

The SINTEF report (Denizou et al. 2007) studies the mechanisms controlling material choices in larger urban buildings in 2007. The report shows that there are good possibilities for directing material choices by municipalities and development plans, even though these means were at the time of writing the report not yet so much in use. Technical guidelines in 2007 did not formally hinder the use of wood in load bearing constructions, but asked for special documentation and special solutions for fire protection. The special care necessary in construction, sound insulation, fire protection and documentation is a knowledge for which routines often are missing and wooden solutions were a financial risk for builders, due to little practically tested solutions. The report also shows examples of municipality strategies and projects on modern design and architecture with wood.

Rambøll (2012) Analyse av dagens offentlige bygg i Norge

English title: Analysis of public buildings in Norway today

The Rambøll report (2012) was a request from Statsbygg to analyse the use of wood in public buildings. The analysis was based on a report review and interviews with key actors on the building market and focused on the impediments for use of wood as well as possible steps from authorities to support and facilitate the use of wood. Since the forest industry as such is scattered and often driven by small companies, increased profitability of the wood value chain in Norway was shown to be important, by for example encouraging collaborations horizontally and vertically throughout the industry, from the forest to products. The report pointed out that if the use of wood in the construction is wanted, this has to be communicated right from the beginning of the project phase, otherwise more common concrete and steel solutions are often likely to be chosen. Impediments to the use of wood in loadbearing constructions were more related to financial considerations, since building with wood is, due to little prefabrication and industrialised solutions is more expensive as compared to steel or concrete. With more buildings and more solutions for large buildings in wood, construction costs were assumed to decrease and builders would become more accustomed to build with wood. While quality and environmental aspects are usually not limiting, deficiencies in expertise among the decision makers limit the use of wood. There are, however, efforts to increase the technical and practical knowledge on building with wood throughout the industry.

Rambøll (2012) Analyse av bruk av tre I Sverige, Finland, Østerrike, Sveits and Sør-Tyskland

English title: Analysis of the use of wood in Sweden, Finland, Austria, Switzerland and Southern Germany

The Rambøll report (2012) was requested by Statsbygg to analyse the use of wood in some other countries and markets. Therefore, authorities, architects and industry associations were contacted and information was collected on the actual extent of use of wood as well as the political strategies and instruments in the respective countries. The report shows, that in all countries considered, wood is used as a construction material in a growing market. The development of the market however, was different for the different countries. All countries though have in common, that increased use of wood was enshrined in national politics. Political means resources were directed to increase research and development, financing of demonstration projects, establishing of networks and adjustment of the building requirements. According to the report, the industrialisation of the industry, as well as an active wood industry with innovative companies was shown to be important. In areas with good collaboration between the different actors of the industry and local political interest, the use of wood was at its largest.

Tellnes, L.G.F. (2012) Bruk av tre i offentlige bygg – Miljø og klimaeffekter. Treteknisk rapport

English title: Use of wood in public buildings – Environmental and Climate effects

The report from Tellnes (2012) was an order from Statsbygg to analyse environmental and climate effects with use of wood in larger public buildings.

The report shows that low emissions during production as well as energy recovery of waste are advantages using wood and increasing the use of wood in larger buildings. Increased need for maintenance, however, can reduce the climate benefits, and it is important to already consider these aspects in the project phase. The report points out environmental aspects to be most important when using wood. These are the sustainable resource with a good growth in Norway, the low energy production of building materials from wood, the high coverage of the needed energy by bioenergy, increasing prefabrication and industrialised solutions will lead to lucrative projects in wood, reduced climate gas emissions due to changed and increased use of wood in buildings and the energy recovery of wood products at the end of life.

Alfredsen, G., Asbjørnsen, B.R., Flæte, P.O. and Larnøy, E. (2008) Miljøeffekter ved bruk av tre, Sammenstilling av kunnskap om tre og treprodukter, Oppdragsrapport fra Skog og landskap 03/2008

English title: Environmental effects due to use of wood, compilation of knowledge on wood and wood products

The report from Alfredsen et al. (2008) was request from the Ministry for Agriculture and Food, to give a summary on the available knowledge on environmental effects with the use of wood, especially on LCA, wood protection and service life. Ensuing from the energy in wood, only three percent are needed to supply it to the industry, with half of it allocated to wood transportation. GHG emissions and energy recovery after use are dependent on material treatment after use. The potential energy in recovered wood materials is as least as large as its embodied energy. In most comparisons with other building materials, wood is considered to be as good or better (60%) in terms of its environmental performance. The positive environmental effect of wood will be utilised when the material has a long lifetime and carbon can be stored for a long time. Substitution of energy intensive materials by wood and simultaneously utilising the carbon storage in wood can have good effects. Intensive forestry could be very beneficial and lead to the largest reduction in CO_2 emissions to the atmosphere.

Bache-Andreassen, L. (2009) Harvested wood products in the context of climate change - A comparison of different models and approaches for the Norwegian greenhouse gas inventory, Statistics Norway report 12/2009

This report covers different accounting approaches and methods for estimating the annual change of emissions/removals of CO_2 due to HWP. The report distinguishes between approach (how are emissions allocated to countries) and methods/models (how are emissions and stocks estimated from national data). In order to avoid double accounting of emissions/stocks of HWP in the commitment period from 2012 under UNFCCC, common approach systems had to be chosen. Stock change approach (SCA, all HWP within national boundaries are accounted for, also imported), atmospheric approach (AFA, atmospheric carbon flux estimations for HWP within national boundaries, also import), production approach (PA, all nationally harvested wood is considered, including export), simple decay approach (SDA, flux estimates from domestic harvested wood) and stock change approach for HWP of domestic origin (SCDA, domestically harvested wood, no export nor import included) were discussed. For estimating the emissions/removals of CO_2 due to harvested wood products, the IPCC HWP model and a revised model were applied, the standard IPCC HWP model was also applied with country specific data. The revised model is a country specific inventory based on the Norwegian building stock.

The revised model is only applicable to the SCA and AFA, and gives according to the author the most accurate results since estimations are based on direct inventories. Comparisons of the revised and the IPCC HWP model, SCA and AFA show higher estimated removals, the revised model gives values of higher accuracy. Similar results might be due to a random occurrence where the half lives in the revised model work in different directions. The author however, points out that this is not a reason to favour the IPCC HWP model.

Skullestad, J.L. (2016) Bygging av høyhus i tre som klimatiltak – En sammenliknende LCA av bæresystemer i tre og betong med varierende antall etasjer. Masteroppgave, NTNU

English title: Building high-rise-buildings in wood as a climate efforts

The study of Skullestad (2016) aims on investigating the potential for reducing the climate change impact of the building sector by constructing highly populated cities with high-rise timber buildings. Therefore, LCA was performed, and the climate change impact of a reinforced concrete benchmark structure was compared to a timber structure for buildings with three to 21 storeys. The timber structure and the benchmark structure met the same load bearing criteria, the system boundaries included the structural system as well as the foundation and GHG emissions were calculated from cradle to gate. In a consequential approach it was also accounted for net benefits from recycling and reuse. In taller buildings, the climate change impact per m² gross floor area increases, since more structural materials are needed. This CO_2 premium for building height occurs for concrete structures from 12 storeys, for timber buildings, however, it is evident from 3-7 storeys. For the timber buildings, the premium is lower from 12 to 21 storeys as compared to concrete. After attributional LCA, the timber structures caused a significantly lower climate change impact for all building heights and methodological assumptions than the reinforced concrete benchmark structures. Taking into account also the net benefits of recycling and reuse, timber constructions could save GHG emissions.

Rønning, A., Vold, M. (2008) Miljøvurdering av nytt hovedkontor for SpareBank 1 SMN - Sammenligning av to alternative løsninger. Østlandsforskning Rapport OR 10.08

English title: Environmental evaluation of the new headquarters for SpareBank 1 SMN

When a new headquarters for SpareBank 1 SMN was planned to be built in Trondheim, Rønning and Vold (2008) estimated the environmental impact of two possibilities, rehabilitation of the old building or establishment of a new building at the same site. Comparisons were based on climate emissions related to raw material extraction, production of building materials, demolishing, new building, rehabilitation, energy consumption during use and conversion of the two cases as well as functionality energy consumption during use and area utilisation. According to the calculations made in the study, climate gas emissions for a new building are lower already after 14 years of use. Over the life time of the new building (60 years) savings of approximately 50 tonnes CO_2 eq. / workspace were estimated.

Kommune Trondheim (2015) Trebyen – en rapport om innovative bruk av tre i byutviklingen Trondheim 2005 – 2015

English title: Wooden city_ a report on innovative use of wood in the city development of Trondheim

The wooden city Trondheim was a city development project established in 2005 which was meant to further develop timber building projects in the area, highlight wood as an environmentally friendly and modern building material with good building characteristics as well as facilitate sustainable architecture and city development with high standards of energy efficiency and CO_2 emissions. During the project duration, a large number of public and large multi-storey buildings were constructed, increasing the knowledge on building and constructing with wood in large structures.

Fossdal, S. and Edvardsen, K.I. (1999) Bygningers energiforbruk og miljøpåvirkning – en studie av tradisjonelle og moderne trebygninger, Norges byggforskningsinstitutt, Prosjektrapport 262 – 1999

English title: Energy consumption and environmental impact of buildings.

The study (Fossdal and Edvardsen 1999) compares environmental impact of the two housing types (single family houses constructed as timber framing and log house) with same size throughout the entire service life (50 years). The log house has a smaller environmental impact as the timber frame example, although the log house has higher emissions in photo oxidations due to transportation and the higher weight of the building components. The energy consumption during lifetime is 20% higher for the log house. The building components in the log house, however are barely processed and produce therefore nearly no waste.

Sand, R. and Stene, M. (2016) Verdiskapingsmuligheter i Trøndelag ved økt bruk av tre i byggemarkedet. Trøndelag Forskning og Utvikling, arbeidsnotat 2016:101

English title: Possibilities for added value with increased use of wood in the building market in Trøndelag

The Sand and Stenes (2016) report studies the possibilities for added value with increasing use of wood on the building sector, especially in Trøndelag. They conclude that the demand for wood as a building material can increase considerably in Trøndelag (+ 0.8 BN NOK) and entire Norway (+8 BN

NOK) until 2030. This gives large potentials for added value for the market and production of construction timber, wood and prefabricated building components as well as the generally increased demand of wood from domestic sustainable forests. This gives also possibilities for Trøndelag to increase and expand its regional manufacturing of products for the building market, especially within the market of prefabrication of building components.

NAL (2004) Reduksjon av klimabelastninger fra byggesektoren – ved økt bruk av tre I Hedmark, Norske Arkitekters Landsforbund

English title: Reduction of environmental impact from the building market with increased use of wood from Hedmark

The report (NAL 2004) creates four scenarios for saving climate gas emissions (production phase, storage effects) in the building sector by increasing the use of wood (all public buildings are built in CLT; all future dwelling are built in CLT; all future buildings built of CLT; all future buildings built in light timber frame construction). Climate gas savings during production as well as storage effects were studied. Especially for the scenarios with CLT constructions, the carbon storage effect was important, and only the production process could give considerable reductions in CO_2 emissions (30 – 50%) as compared to todays' building and construction customs. Especially the carbon storage potential of CLT constructions could, according to NAL's (2004) calculations, store more climate gasses than would be emitted during production.

Flugsrud, K., Økstad, E., Kvissel, O.K., Backer, E.B., Søgaard, G., Granhus, A., Terum, T., Bøe, L.V., Rosland, A., Astrup, R., Grønningsæter, G. (2016) Vern eller bruk av skog som klimatiltak, Miljødirektoratet, Report M-519

English title: Protection or use of forests as climate mitigation

The report (Flugsrud et al. 2016) discusses the climate benefit of forest protection in terms of permanent forest protection against harvest; one of the key assumptions is a sustainable forest use in Norway. The climate effect of forest protection was compared to the climate effect of sustainable forestry where biomass from forests is used as substitute for energy and building materials which today would generate fossil carbon emissions. The IPCC points out that the climate problem is caused by introducing fossil carbon from the slow cycle to the atmosphere and the fast carbon cycle. A central point is to reduce the use of fossil carbon in a long term and management of the short term cycle. An increased substitution of fossil carbon and emission intense products by biomass is according to the IPCC necessary to reaching the goal of keeping the temperature increase by to degrees until 2100. On a shorter timescale, the effect of increased carbon storage could dominate substitution, but in the long term the substituting effect is more important. Therefore, Flugsrud et al. (2016) state that sustainable forest use, rather than not harvesting is a better strategy in terms of climate mitigation. It is however important how the forest biomass is used. The effect is largest with use of wood and secondary wood products in products with a long service life, hence long carbon storage.

Sørgaard, G., Granhus, A., Gizachew, B., Clarke, N., Andreassen, K. and Eriksen, R. (2015) En vurdering av utvalgte skogtiltak – innspill på veien mot Lavutslippssamfunn 2050. Skog og Landskap, Oppdragrapport 02/2015

English title: An evaluation of different forestry methods – contributions on the way towards the Low Emissions Society 2050.

The report (Sørgaard et al. 2015) describes different climate efforts in the forest in Norway, especially with regard to carbon storage and uptake. The average annual deforestation per year between 1990 and 2012 was about 70 km² and mostly due to other land use (forest roads, living area). Approximately one third of the harvested forest is harvested before the forest is ready for harvesting. The methods used in Norway to prepare the land for forestation are quite gentle and do not have an effect on the carbon storage in the soil. Denser forestation results in large productions of volumes. Today, approximately one third of the forestation area has a lower plantation density than recommended by sustainability regulations. Continued practice until 2100 would result in 83 million tonnes lower CO_2 uptake than in an area with the recommended planting density. There is potential for increased thinning without reducing the production. Thinning could be beneficial for two reasons, since it on the one hand there are increases in the volume of industry wood early on, and secondly can increase the volume of timber for sawn wood in the stand. Harvest residues are a raw material for substituting fossil energy with bioenergy, requiring, however, sufficient fertilization. Volume production in a spruce forest could be increased by planting birch in the under-storey.

Alfredsen, G., Sandland, K.M., Søgaard, G. (2017) Norges klimagassregnskap for treprodukter og trebruk i fleretasjes bygg – en analyse av trender. NIBIO Rapport; 3(35).

English title: Norwegian climate gas accounting for wood products and use of wood in multi-storey buildings – an analysis of trends

The report (Alfredsen et al. 2017) was prepared after a request from the ministry for Agriculture and Food in order to increase the knowledge on the environmental properties of wood as a building material and the climate effects of increased use of wood and bioenergy. The report contains values for HWP and their changing carbon stock due to national harvest, included are HWP use and export, import of HWPs is not included. HWP are three main product categories; construction timber, wood based panels as well as paper and cardboard products. The estimated storage of carbon in HWP use after 1960 is at 22 613 492 tonnes C in 2014, for export the storage is estimated to 6 929 580 tonnes C for the same time. Until 2009, the carbon storage in Norway increased, but has had some reductions since. The main factors leading to a reduction in carbon storage is a decreased export of paper and cardboard products, due to reductions in the production of the paper industry in Norway, especially after 2007. Also a reduction of use of construction timber from 2009 makes for a reduction in carbon storage, since importation is not included. Total numbers, however show an increased use of construction timber, mostly due to increased imports. The main contributor to an increase in carbon storage was construction timber (approximately 400 000 tonnes C/yr) while wood based panels as well as paper and cardboard only made for 100 000 – 200 000 tonnes C/yr. For export, paper was historically the most important contributor to carbon storage in Norway (2001: 787 826 tonnes C; 2014: 358 596 tonnes C), and was in 2014 still larger than construction timber and wood based panels together (174 590 tonnes C).

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Norsk institutt for bioøkonomi (NIBIO) ble opprettet 1. juli 2015 som en fusjon av Bioforsk, Norsk institutt for landbruksøkonomisk forskning (NILF) og Norsk institutt for skog og landskap.

Bioøkonomi baserer seg på utnyttelse og forvaltning av biologiske ressurser fra jord og hav, fremfor en fossil økonomi som er basert på kull, olje og gass. NIBIO skal være nasjonalt ledende for utvikling av kunnskap om bioøkonomi.

Gjennom forskning og kunnskapsproduksjon skal instituttet bidra til matsikkerhet, bærekraftig ressursforvaltning, innovasjon og verdiskaping innenfor verdikjedene for mat, skog og andre biobaserte næringer. Instituttet skal levere forskning, forvaltningsstøtte og kunnskap til anvendelse i nasjonal beredskap, forvaltning, næringsliv og samfunnet for øvrig.

NIBIO er eid av Landbruks- og matdepartementet som et forvaltningsorgan med særskilte fullmakter og eget styre. Hovedkontoret er på Ås. Instituttet har flere regionale enheter og et avdelingskontor i Oslo.

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