Deliverable 5.1
Methodological development report for carbon sequestration and GHG metrics in LCAs

Contents

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Executive Summary

The production of animal sourced food significantly contributes to climate change. Cattle, for example, produce methane while digesting feed, which is a strong climate gas. In addition, manure management and feed production largely contribute to the carbon footprint of meat and milk. To mitigate climate impacts innovations are needed on both the supply side, such as changes in the production, as well as on demand side, such as reducing meat consumption and food waste. In the Re-Livestock project, innovations on the production-side will be analysed and how they contribute to a reduced carbon footprint, using the method Life Cycle Assessment (LCA). However, to calculate the impact of livestock systems on climate change, several methodological challenges exist. In this report, three issues are addressed: 1) choice of functional units; 2) accounting for carbon stored in soils and biomass; 3) integrating effects of long-lived greenhouse gases (such as carbon) and short-lived greenhouse gases (such as methane). For each methodological challenge, the relevant literature, models, guidelines and recommendations are reviewed. The findings were then discussed among experts involved in the project to draw conclusions and recommendations.

Functional units

LCAs consists of four phases. In the first phase, the goal and scope of a study are defined, as well as a so-called functional unit or reference flow, which is used to relate impacts to. For a livestock product system, this can typically be 1 kg meat produced. The choice of functional units has been shown to have a strong impact on the outcome of an LCA. Depending on their selection, results are for example rather showing the environmental benefits of extensive or intensive livestock systems.

In this report, several recommendations on the choice of functional units have been reviewed, such as the ILCD guideline (International Reference Life Cycle Data System) or the guidelines published by FAO LEAP (Livestock Environmental Assessment and Performance Partnership) for different livestock systems. In addition, a selective literature review on studies applying and comparing different functional units, such as mass, area, nutritional value, monetary output, or ecosystem services was conducted. Finally, the choice of functional units for systems with diverse outputs, such as crop rotations or agroforestry systems, was reviewed.

The functional units must be selected reflecting the goal and scope of the LCA study. For LCAs comparing various products, it is crucial to reflect which functions the different products offer. For multifunctional processes, the recommendation of ILCD should be followed, to use one functional unit for each function. If large differences in the nutritional values of different products exist, the functional unit should ideally mirror this, i.e. a nutritional index should be used as functional unit. If livestock systems providing very different levels of provisioning ecosystem service (e.g. meat) and non-provisioning ecosystem services (e.g. habitat creation and maintenance) are compared, several functional units should be chosen to present LCA results. Within the Re-livestock project, comparisons could be done between a baseline and an innovation introduced to that baseline system, rather than comparing across different livestock systems. This will avoid comparing systems fulfilling very different functions.

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Accounting for carbon stored in soils and biomass

In the second phase of an LCA, the life cycle inventory phase, all inputs (e.g. material) and emissions (e.g. carbon dioxide) related to a product system are quantified. Here, the changes in carbon contained in soils and biomass are calculated. How and if biogenic carbon is considered within agricultural LCAs has a large influence on the results. However, no harmonized method exists so far on how to quantify changes in LCAs of agricultural systems. This is especially relevant for the management of grassland systems or silvo-pastoral systems (i.e. introducing trees in pasture land).

In this report, the general principles and recommendations on how biogenic carbon can be assessed in LCAs were reviewed. The IPCC methodology, developed for national greenhouse gas inventories, but often used for product LCAs, is shortly presented as well as the approaches recommended by FAO LEAP and ILCD. Several models that have been used to assess biogenic carbon in LCA are presented and their usefulness for LCA discussed. Also, models used for agroforestry systems were reviewed. Finally, exemplary LCA studies are summarized that compared different models to assess biogenic carbon or assessed biogenic carbon in livestock systems or agroforestry systems.

When accounting for biogenic carbon in LCA of agricultural systems, both changes in soil organic carbon (SOC) as well as in woody biomass can be relevant, while carbon in grass and annual crops can be neglected, because it is quickly released again. Generally, the reversibility or temporal nature of carbon accumulation in soil or biomass poses a major challenge when accounting for biogenic carbon in LCA, which typically has a rather long-time horizon. A suitable reference situation needs to be defined, to ensure that only carbon additionally added by human management is accounted for. This reference situation can either be the potential natural vegetation (e.g. forest), a “no use” scenario under the current land use (e.g. unused grassland) or the land management preceding the current management (e.g. degraded grassland). The choice of reference situation strongly influences results. Both the reference situation as well as the situation under study show temporal dynamics and assumptions on the dynamics should be made transparent, and, if possible, only carbon in more stable pools be accounted for (e.g. passive pool in IPCC Tier 2). Although the general recommendation of ILCD is to not account for carbon stored less than 100 years as a default, a shorter-term perspective (e.g. 20 years) could still be relevant to elaborate the option-space of farmers or policy-makers. Therefore, a concept is needed to separately show the effect of temporal carbon storage on global warming. In general, carbon stored in soils or biomass should be reported separately from fossil carbon and results need to be interpreted with caution. In addition, an uncertainty or sensitivity analysis should, to the extent possible, be performed on SOC changes.

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Assessing short and long-lived greenhouse gases

In the third phase of the LCA, the life cycle impact assessment, the environmental relevance of all inputs and emissions are calculated. In this phase, the effect of different greenhouse gases (such as carbon dioxide, methane or nitrous oxide) is summarized. Typically, a global warming potential (GWP) of all gases is calculated, reported as CO_2-eqivalents, where the warming potential of different gases is accounted for. However, different metrics have been proposed on how to integrate the effect of different greenhouse gases, which have a different radiative forcing while also showing very different atmospheric lifetimes. Methane, for example, a potent greenhouse gas, is degraded in the atmosphere within few decades, while carbon or nitrous oxide stay in the atmosphere for centuries. The choice of a metric is particularly relevant for product LCAs of ruminants, because the rather short-lived methane stems mainly from enteric fermentation, while emissions of long-lived GHG such as CO_2 and N_2O often result from feed production (i.e. application of fertilizer or energy and fuel use or land use change).

Here, a review was done on the different concepts and metrics to account for the effect of different greenhouse gases, within and outside LCA. A special focus was on GWP*, an alternative approach of aggregating greenhouse gases proposed a few years ago. The usefulness and critique on the concept are reviewed.

Generally, it is important to account for the different atmospheric lifetimes of GHGs in climate impact assessment of food products, especially for ruminant livestock products. It is strongly recommended to report greenhouse gas emissions separately by gas, or at least distinguishing between short-lived (SLCP) and long-lived climate pollutants (LLCP). There is no single metric that captures all the relevant differences in climate impact of different gases and therefore the application of multiple metrics (e.g. GWP and GTP) is recommended. To calculate the carbon footprint of a product, established metrics such as GWP and global temperature potential (GTP) are recommended. Using GWP* for climate footprints is discouraged for two reasons: (1) method details around baselines have proven difficult and confusing for many people; and (2) GWP* has no advantage in principle over a time-dependent application of GTP, which is a more established and widely understood metric. Using a dynamic approach, such as GWP* or other simple climate models, can be useful to highlight potential trade-offs between short-term and long-term warming effects. In a dynamic approach, climate impact is however not reduced to one number, but rather a time series of numbers indicating the climate impact, e.g., as global temperature change (GTP) or CO_2 warming-equivalents (GWP*).

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1. Introduction

Relevance of different greenhouse gases

Human-induced climate change due to elevated concentrations of greenhouse gases (GHG) in the atmosphere has drastic consequences on humans and ecosystems. According to the IPCC (2022a), more than 3 billion people "live in contexts that are highly vulnerable to climate change" and "a high proportion of species is vulnerable to climate change". Carbon dioxide (CO_2) is the most relevant greenhouse gas, which accounts for 75% of the anthropogenic GHG emissions, mainly emitted from combustion of fossils fuels, industry and land use change. Beside CO_2, other gases contribute to global warming, mostly showing a higher warming effect per kg gas emitted. To be comparable, emissions of all GHG are typically expressed in kg CO_2-equivalents (CO_2-eq), representing a similar impact on the climate as 1 kg of CO_2. Methane (CH_4) is the second most important GHG and contributes to 18% of anthropogenic GHG emissions (in CO_2-eq), while nitrous oxides (N_2O; 4%) and fluorinated gases (2%) have lower shares (IPCC, 2022b). Although the need to limit GHG emissions was internationally agreed on in the 1990s, the anthropogenic emission of all these GHGs has considerably increased since then (+67% of CO_2 emitted from fossil fuels and industry, CH_4: +29%, N_2O: +33%; fluorinated gases: +254%) (IPCC, 2022b). However, even if net emissions would be reduced to zero today, the anthropogenic CO_2 and N_2O emitted in the past will remain in the atmosphere for centuries and contribute to the greenhouse effect in the future, while methane will degrade within decades and its contribution to the greenhouse effect will decrease over time (Lynch, Cain, et al., 2020). To account for the different warming potentials as well as the long-term dynamics of short- and long-lived greenhouse gases, a new metric, GWP*, has been proposed by Allen et al. (2018). This led to a debate in science, but also in the livestock and farming sector, on how to quantify and interpret the effect different greenhouse gases have on climate change.

Contribution of livestock systems to climate change

Food systems account for about 21–37% of annual GHG emissions, with important contributions from agriculture (within farm-gate; 9-14%), land use and land use change (e.g. agricultural driven deforestation and peatland degradation; 5-14%) and beyond farm gate sources (e.g. fertilizer production, food processing, transport, retail, consumption; 5-10%) (Mbow et al., 2019). Within the global food system, the livestock sector shows significant contribution to GHG emissions, mainly due to non-CO_2 gas emissions from enteric fermentation, manure left on pasture, manure applied to soils and through manure management. About 30% of global methane emissions are related to enteric fermentation, mostly from cattle (77% of livestock methane emissions), buffalo (14%) and small ruminants (9%). For intensive livestock production systems relevant GHG-emissions also occur beyond farm gate: The production of feed and fertilizer, transport and refrigeration can account for about 24-32% of total emissions in intensive systems (Mbow et al., 2019). On cropland, methane emissions from rice, CO_2 emissions from peatland and N_2O emissions from fertilizer application are the main sources of GHG emissions.

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Mitigation options from the food and livestock sector

To limit climate change, reducing net GHG emissions is crucial. For agriculture, mitigation concerns the supply side (agricultural production) as well as the demand side (e.g. changing diets, reducing food waste) (Mbow et al., 2019). On the supply side, reductions can be achieved by lowering emissions (from soils, reducing land use change, and improving land and livestock management) or increasing carbon stocks on agricultural land. The latter has gained increasing attention in the last years, as global soils contain more than four times more carbon than terrestrial biomass. Carbon uptake of soils could be increased by appropriate soil management (Stockmann et al., 2013). Grasslands are particularly important, because they contain about 20% of the world's soil organic carbon (SOC) (FAO, 2023a). However, the C sequestration potential of restoring degraded grasslands is still not well quantified (L. Liu et al., 2023). Another mitigation option on agricultural land is increasing both the amount of carbon taken up in biomass and soils by establishing agroforestry systems (Feliciano et al., 2018). Kay et al (2019) estimated that between 1.4 and 43.4% of the total European agricultural greenhouse gas emissions could be offset by agroforestry systems on 8.9% of European agricultural land. However, soils and biomass can act as both sources and sinks of carbon, and a range of interacting factors such as climate, land management, soil pH, clay content and other soil characteristics influence the degree of carbon uptake and release (FAO, 2023a).

Assessing the carbon footprint of innovations in livestock production

Improving the environmental performance of livestock systems is highly needed. To quantify and compare the environmental impacts of livestock products and assess the potential of innovations, a life cycle assessment (LCA) is frequently used. LCAs assess impacts along the life cycle of products, from extraction of raw material to production, agricultural production, processing, transport and disposal of waste. A range of environmental impacts can be assessed using LCAs, including climate change, pollution and related effects on ecosystems and human health, depletion of resources etc. Impacts are expressed in relation to the function of a product system, typically 1 kg of milk or meat produced, the so-called functional unit.

LCAs consist of four stages (ISO, 2006). In stage 1 the goal and scope of the study are defined. In stage 2, the Life Cycle Inventory (LCI) is compiled, where the inputs and emissions of a product throughout its life cycle are quantified. The third stage is the life cycle impact assessment (LCIA), where the magnitude and significance of the environmental impacts of all emissions and inputs are evaluated. Finally, results are interpreted, with regard to the goal and scope, and recommendations given and limitations explained.

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To assess the carbon footprint of innovations in livestock production systems, several methodological challenges exist. First, results have been shown to be sensitive to the choice of functional units to relate impacts to. As an example, the results in a study exploring the environmental impacts of dairy intensification showed either support for or against intensification depending on the functional unit chosen (Salou et al., 2017). Second, during the life cycle inventory stage, no harmonized method exists on how to quantify changes in soil organic carbon of agricultural systems (Goglio et al., 2015). This is especially relevant for the management of grassland systems or silvo-pastoral systems (i.e. introducing trees in pasture land). Finally, during the life cycle impact assessment stage, it is questioned if the typically used metric Global Warming Potential (GWP) is appropriately accounting for climate effects of short- and long-lived greenhouse gases. This debate is particularly relevant for LCAs of ruminants because the rather short-lived methane stems mainly from enteric fermentation, while emissions of long-lived GHG such as CO_2 and N_2O often result from feed production (i.e. application of fertilizer or energy and fuel use or land use change).

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2. Objectives

The objective of Task 5.1 was to get an in-depth understanding of and provide potential solutions for three main methodological challenges of assessing the global warming potential of agricultural and especially livestock systems within life cycle assessment (LCA):

a) Choice of functional units
b) Accounting for carbon in soils and biomass in LCA
c) Integrating effects of long and short-lived GHG

A clear understanding of these methodological challenges and existing recommendations on how to handle them is crucial to select and design methods for assessing the environmental impacts and particularly climate impacts of different livestock systems in Task 5.2. It is also a prerequisite for finally interpreting climate impacts and for conducting the appropriate sensitivity analyses to capture the main methodological uncertainties.

Research questions

In this task, the following research questions were addressed:

- Which **functional units** (FU) have been proposed to assess livestock systems? How much does the choice of FU influence results? Which FU can be recommended for which types of questions and case studies?
- Which approaches and recommendations exist to model **biogenic carbon** in agricultural LCAs and how do they differ? Which models have been used to model carbon in livestock agroforestry systems and grasslands, and how could those models be implemented in LCAs?
- Can the results derived from the **GWP*-metric** be used to derive carbon footprints for single products and if so, how has this to be done? If not, how are the results to be used to communicate about the climate impact of single products and the related production processes?

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3. Methods

Because this deliverable covers three very diverse topics, experts involved in the task were working in three groups to get an in-depth understanding of the methodological challenges, summarize these findings and provide conclusions and recommendations on how to address these challenges.

For all topics, existing recommendations were reviewed, such as the guidelines of the FAO LEAP (Livestock Environmental Assessment and Performance Partnership, an initiative aiming to harmonize methods to assess environmental impact in the livestock sector), ILCD (International Reference Life Cycle Data System from the European Commission), or IPCC (methodologies from the Intergovernmental Panel on Climate Change). In addition, a selective literature review was conducted for each of the topics addressed. Finally, findings were discussed among experts involved in the project to draw conclusions and recommendations.

Additionally, to get a better understanding of the different approaches to assess biogenic carbon in LCA, respective models that have been used by researchers involved in the Re-Livestock project were described in detail.

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4. Choosing appropriate functional units

4.1. General recommendations of ILCD

According to the ILCD Handbook of the European Commission-Joint Research Center (2010), the functional units should be defined including (a) the function provided (what), (b) in which quantity (how much), (c) for what duration (how long), and (d) to what quality (in what way and how well is the function provided). If there are temporal changes in the function provided, these shall be explicitly considered and quantified and the use of parameterised data sets is recommended. For multifunctional processes, one functional unit or reference flow should be given for each function.

4.2. Recommendation of FAO LEAP regarding livestock

FAO LEAP provide separate guidance for environmental assessment of different animal supply chains (Large ruminants (FAO, 2016a); small ruminants (FAO, 2016c); poultry (FAO, 2016b) and pigs (FAO, 2018)) as well as for feed additives (FAO, 2020). An overview of proposed functional units is provided in Table 1.

For **milk** products, functional units correcting for fat, protein or lactose content are proposed. For large ruminants, the recommended functional unit is fat- and protein-corrected milk (FPCM), for small ruminants the energy-corrected milk (ECM), which corrects for fat, protein and lactose content, is proposed. They are calculated as follows:

kg FPCM = kg milk × (0.1226 × fat% × 0.0776 × protein% + 0.2534) (IDF, 2015)

kg ECM = kg milk × (0.1226 × fat% + 0.0776 × true-protein% + 0.0621 × lactose%) (FAO, 2016c)

Table 1. Functional units proposed by FAO LEAO for assessing different livestock products

Livestock category	Main product type	Cradle to farm gate	Cradle to primary processing gate
Large ruminants (FAO, 2016a)	Meat Draught Power Milk	Live weight (kg) MJ FPCM (kg)	Meat products (kg)[1] - Dairy product(s) with specific fat and protein content (kg)
Small ruminants (FAO, 2016c)	Meat Fibre Milk	Live weight (kg) Greasy weight (kg) ECM (kg)	Meat products (kg) - Milk product(s)
Pigs (FAO, 2018)	Meat Piglets Spent sows	Live weight (kg) Live weight (kg) Live weight (kg)	Carcass weight (kg)
Poultry (FAO, 2016b)	Meat Egg	Live weight (kg) Fresh, shelled weight (kg)	Carcass weight (kg) Liquid weight or dry (powder) weight (kg)

[1] Here, FAO LEAP recommends to specify the edible yield, moisture, fat and protein, which is packaged for secondary processing. It may contain small proportion of bones, which are then wasted at consumption stage.

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4.3. Review of publications comparing functional units

Many authors have analysed the effect of choosing different functional units for multi-functional and complex agricultural or food systems. Here, a selection of studies covering different aspects or functions of livestock systems are shortly presented. The section covers studies that compare or develop functional units based on mass, area, nutritional value, monetary output and ecosystem services.

4.3.1. Mass vs. area

One of the most commonly used functional units for agricultural LCAs is the mass of produced products, such as 1 kg of wheat or 1 kg of meat. Another often applied functional unit is an area of land, e.g. 1 ha. According to Basset-Mens and van der Werf (2005), the mass represents the function of producing market goods, the area reflects the function of producing non-market goods (e.g. environmental services). Many LCA studies therefore report both mass and area based functional units. Salou et al. (2017) analysed mass (kg milk) and area based functional units (both on- and off farm) for evaluating the environmental impacts of dairy system intensification. They show that the choice of functional unit is decisive for the conclusions drawn and that mass-based functional units tend to favour intensive systems, which on a local scale can have negative environmental impacts. The authors recommend to use both mass-based and area-based functional units in agricultural LCAs, especially when low-input and high-input systems are compared. Ross et al. (2017) proposed to use a functional unit combining productivity and land use, i.e. milk yield per hectare, to better capture trade-offs between land and production efficiencies for greenhouse gas assessments of dairy systems.

4.3.2. Nutritional value

In recent years, a stronger focus was made on the function of food to feed people and providing healthy diets, looking not only at the mass produced, but also at the nutritional value of food. Since nutrition is a complex issue, a variety of indices and functional units have been tested and suggested. In a comprehensive review on LCA studies of foods Poore & Nemecek (2018), present results per unit of primary nutritional benefit (e.g. **protein** for meat, sea food and legumes, **kcal** for grains, **litre** or kg for milk, oil, fruit, vegetable and sugar and **serving unit** for alcoholic drinks, coffee and chocolate). Using **human edible protein** as the functional unit for systems producing different foods, such as dual-purpose cattle producing milk and beef has the advantage, that allocation of environmental impacts can be avoided (Letelier et al., 2022). In a study comparing allocation methods and functional units in different cattle systems in Costa Rica, much more human edible protein was provided by milk than meat (Letelier et al., 2022). The authors highlight that accounting not only for the amount, but also the quality of proteins or other nutrients could help to better reflect the function of food. A **protein quality index** (based on digestibility and content of essential amino acids) was tested as the functional unit by Sonesson et al. (2017) in context of different dietary contexts (average Swedish diet, lacto-ovo vegetarian diet and low-meat diet). They compared the LCA results of six foods using a weighted protein quality index, as well as mass, protein and digestible protein. The authors

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conclude that using proteins, an important factor in human nutrition, global food security and health, as the functional unit of food has the advantage of having relatively accessible data. However, other nutrients important for human nutrition are not accounted for and this functional unit is not suitable for foods such as vegetables and fruit. In general they conclude that the nutritional value of food needs to be assessed in the context of diets, which is very data-demanding and implementation in LCA might therefore be difficult. Tessari et al. (2016) compared environmental impacts of different animal and vegetal foods using mass of edible parts, mass of essential amino acids (the building blocks for proteins), or the **recommended daily allowance** of essential amino acids as functional units. They conclude that taking the daily allowance of essential amino acids into account, the environmental benefits of crops vs. animal foods are less obvious then when using mass based assessments. McAuliffe et al. (2018) analysed the effect of different functional units for livestock production systems, including the nutritional value of meat. They assessed different functional units, i) based on different types of omega-3 poly-unsaturated fatty acids, which show various health benefits, and ii) based on different **nutrient indices**, considering different desirable nutrients (such as proteins, omega-3 poly-unsaturated fatty acids, calcium, iron, etc). Applying these different functional units to a range of livestock production systems, including cattle, sheep, pigs, and poultry, they conclude that results of LCA studies can be dramatically altered when considering the differences in nutrient content of meat. A methodology to assess multiple recommended nutrients was developed by Saarinen et al. (2017) to either assess different nutrients separately as functional units or as a combined nutrient index. They assessed 29 food products using these functional units. Looking at separate nutrients were deemed unsuitable for sustainability assessments of foods, but useful in situations of scarce nutrients or a deficiency of a certain nutrient. A nutrient index, including an index for beneficial nutrients as well as for nutrients where intake should be limited was proposed by the authors as a general method to take nutritional aspects into account in LCA of foods.

Increasingly, nutrient density is used as a concept in LCA, especially the **Nutrient Rich Food (NRF)** index family, which reflects dietary guidelines and is considered “robust, versatile and validated” (Bianchi et al., 2020). An NRF is flexible to include different desirable and non-desirable nutrients. It is calculated as the sum of all ratios of desirable nutrients content i in foods divided by their dietary reference intakes (DRI) minus the sum of all ratios of non-desirable nutrients j divided by the maximum recommended intake (MRI):

$$\mathrm{NRFx.y} = \sum_{i=1}^{x} \frac{Nutrient\ i}{DRI\ i} - \sum_{j=1}^{y} \frac{Nutrient\ j}{MRI\ j}$$

An NRF11.3, for example, is calculated based on eleven desirable (protein, fibre, vitamins A, C, E, D, filate, Ca, Fe, Mg, K) and three non-desirable nutrients (saturated fat, added sugar, Na). The choice of nutrients and dietary recommendations can be adapted depending on a populations dietary guidelines. However, this means that the index needs to be adapted to a population and data are needed for the nutrient content of food products. As nutrition and health are complex topics, the interpretation of LCA results based on nutrition indicators could require the involvement of nutrition scientists to understand their full implications (Bianchi et al., 2020)

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An overview on metrics and methods to jointly study environmental impact with **nutritional aspects** and/or health dimensions is provided by Green et al. (2020). Health and nutritional metrics are evaluated, which are based on nutrient quantity, nutrient diversity or nutrient quality. Typical food components considered in different indices are macronutrients (e.g. protein, fibre, unsaturated fat), vitamins, minerals (e.g. Calcium or Iron), or non-desired nutrients (e.g. saturated fat, added sugar, etc.). Most indices do not account for the importance of different nutrients, because there is no scientific agreement on how to weight nutrients (e.g. how important are proteins in comparison to carbohydrates). Criteria such as bioavailability, nutrient quality or local nutrient deficiencies have used been to weight different nutrients.

Other authors propose to include the health aspect of diets alongside the impact assessment of entire meals, rather than the functional units. Weidema & Stylianou (2020) propose to assess beneficial and detrimental nutritional components of food, comparing the food composition with thresholds provided by Global Burden of Disease (GBD) studies. A new impact indicator DALY Nutritional Index (DANI) is suggested, which calculates the health impacts (positive or negative, expressed as DALYs; disability-adjusted life years) per meal. Ridoutt (2021) questions if there are benefits of calculating health impacts of food consumption and if this should then, for consistency, also be done for other products serving human well-being such as electricity in hospitals.

There are several **challenges and critiques** of using the nutritional value of food as functional unit. Regarding nutritional indices for functional units, Ridoutt (2021) states that food is multifunctional and does not only provide nutrients, but also satiety and pleasure and has a role in social and cultural exchange. In addition, it is not easy to express nutrition in a single number. Therefore, Ridoutt (2021) expressed that in many LCA studies it might be more useful to just separately report nutritional composition of foods instead of calculating direct health impacts or express the impacts per nutritional value. In addition, the nutritional value of food can only be assessed in the context of diets, because single food products are not consumed in isolation. Therefore, the appropriateness for using nutritional values as functional units in product LCAs has been questioned. Assessing LCA of entire diets is however very data-demanding and implementation of nutritional values in LCA might therefore be difficult (Sonesson et al., 2017). In addition, results of nutrient indices tend to be arbitrary and difficult to compare, because of the selection of nutrients considered, the way indices are calculated and the mostly missing weighting of the importance of nutrients (Green et al., 2020). An additional conceptual challenge results from subtracting disqualifying nutrients in nutrient indices, which are actually not a function of food. As a result, an LCA of unhealthy food such as chocolate could result in a negative functional unit and thus also the related environmental impacts, which could be misinterpreted as beneficial outcomes (Saarinen et al., 2017). In addition, the non-linearity of the function of nutrition is a challenge, because exceeding the daily required amount of e.g. calcium is not adding additional value. Therefore, functional units can be “capped”, i.e. no further benefit is assigned to producing more calcium than is needed for a region (Green et al., 2020). However, on a micro-scale, e.g. farm level, this approach does not work, because supply is not given by one farm and is dependent on the production of other farms.

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4.3.3. Monetary output

For systems with very diverse outputs, such as agri-photovoltaic or agroforestry systems, a monetary functional unit has been proposed. (Leon & Ishihara, 2018) used the summed price of produced tomato and generated power as monetary-based functional unit to analyse agri-photovoltaic systems on tomato greenhouses.

A study by Grassauer et al. (2022) analysed three functions of dairy farms (food production, income generation and providing biodiversity), using a sample of 44 dairy farms. These three functions (i.e. outputs) of dairy farms were then used in a data envelopment analysis (DEA) to calculate the eco-efficiency, considering the cumulative exergy demand, global warming potential, eutrophication potential and aquatic ecotoxicity as “inputs” required to provide the functions. The eco-efficiency of each farm showed the possible reduction of excess inputs and the desired increase in deficient outputs. The authors conclude that their approach allows to identify improvement potential of individual dairy farms in various ways, by increasing one of the outputs (food, income, biodiversity) or decreasing one of the four inputs (i.e. environmental impacts). For all farms, the amount of purchased concentrate had a strong impact on outputs as well as environmental impacts.

4.3.4. Ecosystem services

Agroecosystems provide different ecosystem services: provisioning services (e.g. provision of food), regulating and maintenance services (e.g. regulating carbon and water flows) and cultural services (e.g. recreational value of landscapes). Using the mass or nutritional value of food as the functional unit, only the provisioning function of agroecosystems is accounted for. Boone et al. (2019) propose to consider all ecosystems services as functions provided by agroecosystems, and allocate the impacts among different services. This is the reverse approach of other authors, which propose a cause-effect chain in which damages to ecosystem services are either assessed as midpoint (e.g. erosion resistance potential [t/ha/yr]) or endpoint indicators (ecosystem quality; Koellner et al., 2013) or as an own area of protection (Hardaker et al., 2022) in the life cycle impact assessment.

Midpoint indicators focus on single environmental problems, for example climate change or acidification. Endpoint indicators show the environmental impact on three higher aggregation levels, being the 1) effect on human health, 2) biodiversity and 3) resource scarcity.

A selection of studies has applied the approach “Economic allocation including ecosystem services” (Ripoll-Bosch et al., 2013), by calculating the total economic value of different services (e.g. income generated by meat and milk and by direct payments for ecosystem services and allocating the impact to the different outputs (Bragaglio et al., 2020; Ripoll-Bosch et al., 2013). Because it is not always straightforward to link payment schemes with ecosystem services, von Greyerz et al. (2023) tested different allocation procedures. All studies found substantially lower GHGs associated with (extensive) livestock systems, when accounting for the non-provisioning ecosystem services.

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4.3.5. Agroforestry systems and crop rotations

Agroforestry systems are by definition multi-functional and provide different outputs. A review of LCA studies of agroforestry systems showed that the majority used mass of food products as functional units, but also area, economic units, energy or composite units were used (Quevedo-Cascante et al., 2023).

Goglio et al. (2018) discuss how to appropriately assess **entire cropping systems**, with multiple crops grown in sequence or simultaneously (intercropping or agroforestry), and where strong effects of previous crops on the performance of the subsequent crops occur. For system LCA (i.e. an assessment or comparison of production systems, focussing on better understanding the mechanisms behind environmental impacts) they recommend to apply a cropping system approach. For product LCA, they propose different approaches to attribute environmental burdens to different outputs of cropping systems. For management interventions, resources and emissions, a classification is proposed based on the causal relationship to the cultivation of single crops. Three classes are suggested for inputs or emissions (1) mainly attributed to one crop (e.g. seed inputs), (2) allocated to different crops using specific criteria, such as fertilizer demand, or (3) allocated using generic criteria, such as cereal unit or crop rotation. Cereal unit is the metabolizable energy content of each agricultural product, normalized by the respective content of the reference crop barley. The crop rotation approach calculates the environmental burdens of entire crop rotations in the inventory phase and then allocates impacts to single crops based on biophysical, cereal unit, mass, energy, or economic relationships. Alternatively, crops can be assessed separately from the crop rotation, which is actually most commonly done in LCAs. However, when assessed separately, the time variability across crops is not accounted for and there is a high risk of double counting effects such as GHG emissions, nutrient dynamic and biodiversity impacts.

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5. Accounting for biogenic carbon in LCA

5.1. Standard accounting methods and international recommendations

5.1.1. Biogenic carbon in LCA: General principles

In agricultural systems, carbon is stored in different pools, such as soils, dead organic matter (dead wood and litter) and biomass (above- and belowground). Management practices, such as reduced tillage, crop residue management, use of organic fertilizers, or planting of woody vegetation can affect the amount of carbon contained in these pools. When a new management practice is introduced, the carbon of the different pools changes at a high rate, particularly soil organic carbon (SOC), the main pool where carbon is stored in agricultural systems. The rate of change slows down over time until a new equilibrium is reached, where carbon content varies mainly depending on climate variability, and without showing a clear trend (Goglio et al., 2015). However, the dynamics to reach a steady-state are complex, especially for soil organic carbon. Baveye et al. (2023) highlight the difference between carbon storage (short-term retention of carbon in soils after application of organic matter) and carbon sequestration (long-term retention of carbon in soils). They illustrate that a substantial portion of the soil carbon input in one year is rapidly lost within the next 20 years. In addition, the storage capacity of carbon in soils is finite (Wang et al., 2023) and thus depends on how much carbon is already contained in soils.

If and how the changes in carbon stored in these pools should be accounted for in LCA thus strongly depends on temporal horizons considered. Following IPCC, typical time horizons used in LCA to account for climate impacts are 20, 100 and 500 years, with 100 years being most commonly used. Carbon absorbed in biomass (especially in crops and animals, but also trees) is typically considered as carbon-neutral, because it is re-released within a 100 year time frame. Exceptions exist, for example for wood used in buildings (IDF, 2015). How to account for the benefits of temporarily removing carbon from the atmosphere in woody vegetation of forests (Brandão et al., 2013) or agroforestry systems (Quevedo-Cascante et al., 2023) is being debated.

Recommendations from ILCD and environmental footprint

In LCA, a distinction is made between changes in biogenic carbon from land management and from land use change. While changes in biogenic carbon in biomass and soil due to land use change (e.g. deforestation) are typically accounted for, changes in soil organic carbon due to changes in land management are often not assessed (however with some exceptions e.g. Moberg et al. (2019) and Hammar et al. (2022)).

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According to the **ILCD** Handbook (European Commission-Joint Research Center, 2010), to account for changes in soil organic carbon due to **land use change**, the most recent IPCC CO_2 emission factors should be used, but if available, more accurate and specific data can be used. To **allocate** the impact of land use change to subsequent land uses (e.g. crops), ILCD separates between changes happening over longer periods (e.g. slower SOC changes) and changes within the first year after conversion (e.g. biomass burning). For slower changes, a new equilibrium should be assumed after 20 years as default, assuming that 90% of losses or gain happen within this time frame. Linear changes can be assumed for simplicity. Using a triangular allocation function, higher impacts are given to crops directly planted after land use change and lower impacts for the crop planted 19 years after the transformation. For carbon stock changes happening within a year after land transformation, the carbon emission is allocated evenly to all crops within the subsequent 20 years.

According to ILCD, the **temporary storage of carbon** (i.e. shorter than 100 years) shall not be considered per default. However, changes can be considered if this is explicitly required for meeting the goal of the study. In that case, they recommend to report temporary carbon storage or delayed emissions (i.e. within less than 100 years) in separate inventory flows called "Correction flow for delayed emission of biogenic carbon dioxide (within first 100 years)". These correction flows are calculated as the amount of C stored multiplied by the years of storage. These correction flows are then multiplied by -0.01. Storing 1 kg of C in a wood product for 20 years thus results in a carbon sequestration of -0.2 kg C, if the delayed emissions are accounted for in the LCA (assessing impacts using GWP100). Although not explicitly stated, this approach might also be applicable for establishing woody vegetation within agroecosystems (e.g. agroforestry) or temporarily increasing soil organic carbon due to land management.

ILCD states, that "only the net interventions related to human land management activities shall be inventoried". Emissions that would occur also without management ("unused site") shall not be inventoried. For most GHG emissions and carbon storage in soils, only changes due to human management should thus be assessed and not the total carbon stored in soils (when part of it would be there without interventions). As a **reference situation**, "no use" shall be assumed.

The guide on **environmental footprints** (Fazio et al., 2020) also requires that carbon emissions from land use and land use change are reported. However, SOC uptake due to changes in agricultural management shall not be included in the modelling, but can be reported separately.

Time horizons

Temporarily removing carbon from the atmosphere or delaying GHG emissions could contribute to lower the global temperature peak (Matthews et al., 2023). So far, no consensus exists on how to account for such **temporal storage**. In an expert workshop, Brandão et al. (2013) compared six approaches to account for temporal storage: one with a fixed GWP, used in conventional LCAs, which does not account for any benefits of temporal storage, and five methods that account for temporal storage in different ways (Moura-Costa method, Lashof

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method, PAS 2050 method, dynamic LCA method and ILCD method). The authors did not reach consensus on whether temporal storage of carbon should be accounted for in LCA and if so, with which method. They conclude that value choices are involved when choosing how to account for temporal carbon removal, and that choices should be made explicit and transparent. Instead of discounting for delayed emissions, Kendall (2012) proposed to directly calculate "time adjusted global warming potentials" (TAWPs). For one kg CO_2 emitted in 10 years in the future, a CO_2-equivalent of a current emission is calculated. For example, based on a 100 year time perspective, a 10 year delayed emission results in 0.93 kg CO_2-today, whereas with a 20 years perspective, a 10 year delayed emission corresponds to 0.58 CO_2-today. Other authors propose to use time-dependent modelling of SOC, where yearly fluxes of greenhouse gases are calculated and the temperature response over time is assessed to reflect the impact on climate change (e.g., Hammar et al., 2022).

LCA land use assessment framework and reference situations

For assessing the impacts of land use change on ecosystem quality, the land use assessment framework (Koellner et al., 2013; Milà i Canals et al., 2007) is usually applied in LCA (Figure 1). This framework can also be applied to assess changes in soil organic carbon (Müller-Wenk & Brandão, 2010). A reference situation is used to evaluate the impacts of land use and land use change. A distinction is made between effects of changing land use or management practices (referred to as land transformation impacts) and impacts of maintaining the current land use or management (referred to as land occupation impacts). Figure 1 shows two simplified situations of changes in SOC. The reference situation is at steady-state and positive (A) or negative (B) changes to SOC occur after the land management is changed, until it reaches a new steady-state under the new land management (occupation situation). As soon as the human activity stops, SOC levels slowly return to the reference situation. The impact of land occupation is given as the difference between the altered and reference situation, multiplied by time and area of the occupation. Transformation is calculated as the difference between SOC under land management and reference situation is calculated, multiplied by half the regeneration time (time used until SOC reaches the level of the reference situation, after land is not used anymore (dashed triangle).

Accounting for SOC under the land use assessment framework thus implies, that impacts are calculated as SOC change multiplied by time and area, and that a distinction is made between land transformation and occupation. In addition, steady-state assumptions are typically used for both reference and land management situation.

Following this approach, results are highly sensitive to the selection of the **reference situation**. For example, maintaining no-till cropland might show reduced SOC levels (C emissions) when using as a reference the potential natural vegetation in regions with natural forest cover, , while using tilled cropland as a reference might show increases in SOC.

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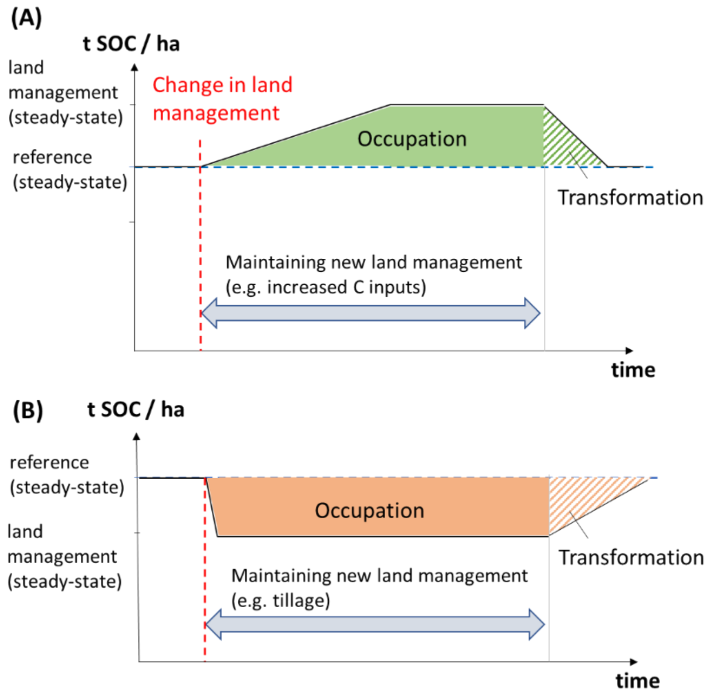

Figure 1. LCA land use assessment framework (Koellner et al., 2013) to assess changes in soil organic carbon stocks (SOC) due to land management. (A): reference situation is at a steady-state, the introduction of a new land management (e.g. increased yearly input of C) slowly increases SOC, until it reaches a new equilibrium. As soon as the management is changed again (e.g. no additional yearly input of C) the SOC is rather quickly returning to the reference situation. (B) introducing a new land management (e.g. ploughing the land) quickly releases SOC, until it reaches a new steady state. As soon as the land management is stopped (e.g. land is not ploughed any more), SOC slowly increases back to the reference situation. Filled areas: SOC changes due to land occupation (maintaining land management); dashed areas: SOC changes due to land management change. Green: increased SOC, orange: decreased SOC.

Conceptually, other situations are possible, where no steady-state is prevalentFigure 2. Figure 2 shows two alternative situations. In situation A, degrading SOC-levels are assumed (e.g. due to warming climate) and with a change in land management, the SOC can again be increased. In situation B, a new steady state is calculated with a time horizon of 100 years, but the new steady state is highly dependent on the assumptions on the starting point at time = 0 (Joensuu et al., 2021).

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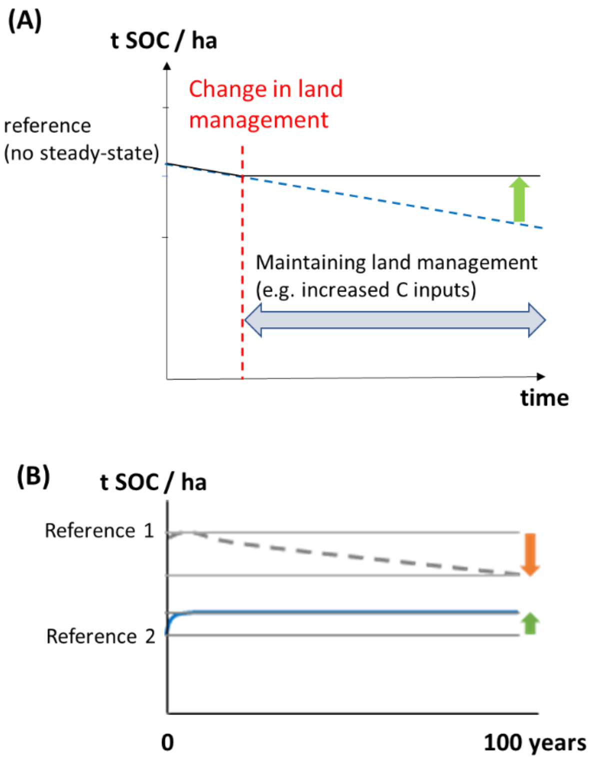

Figure 2. Changes of SOC over time of reference situation (A) or land management situation (B). (A): the reference situation is not at a steady-state, SOC is lost over time (e.g. due to warming climate). With the introduction of a new land management, the SOC is increased. (B) Assessment of changes in SOC comparing SOC at beginning of modelling period to SOC stocks after 100 years for two different reference situations (starting points) (adapted from Joensuu et al., 2021). Green: increased SOC, orange: decreased SOC.

Allocation of impacts

Within an LCA, SOC changes due to land use change, happening over a longer or shorter time period, finally need to be allocated to crop or animal production, disregarding the approach on how they are assessed. For land use change e.g. from forest to crop land, impacts are typically allocated to the products of this land for the first 20 years. The amount of carbon lost due to land use change is thereby distributed over the crops grown within the first 20 years, thereafter no impact is assigned to crops. A triangular distribution, giving more weight to crops grown in the first years after land use change, is proposed by ILCD, while IPCC uses a uniform distribution (Bessou et al., 2020).

A different approach is presented by Pendrill et al. (2019), which assessed drivers of deforestation and quantified the deforestation "embodied" in production and trade of agricultural and forestry commodities. They analysed the commodity groups per country associated with deforestation as well as the share of deforestation attributed to exported commodities.

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Review on accounting SOC changes in LCA

A review on how changes in soil organic carbon due to land management and land use change is assessed in LCA was performed by Goglio et al. (2015). They identified four different methods: emission factors, simple C models, dynamic crop-climate-soil models and observations. While emission factors (e.g. IPCC Tier I factors) are easy to apply in LCAs, they have a low level of certainty. Dynamic crop-climate-soil models, such as DAYCENT; DNDC, CERES-EGC, CropSyst, are process-based models that ensure maintaining the mass balance for C, N, and water in agro-ecosystems. They have a higher certainty but are not easy to apply. Simple C models (such as C-Tool, RothC, ICBM), consider soil C dynamics but do not model crop production and the input of carbon needs to be specified. These models are of intermediate applicability and certainty. While Goglio et al. (2015) identified LCA studies applying all three types of models for assessing soil C changes due to land management, only one study was identified that measured soil C. For land use change, the reviewed studies covered all four methods. In general, about 20% of the reviewed studies assessed both soil C changes due to land use change and land management change and around 40% each covered either only land use or only land management.

Goglio et al. (2015) conclude that a "compromise between accuracy and completeness in LCA methods is necessary" and that currently most LCAs considering land use change only do this in a coarse manner due to limited data and available methodology in specific countries.

They provide the following recommendations:

- Apply a method consistent with the objective of the study (e.g. apply a site-specific assessment if a site-specific case study is assessed).
- The choice of methods should depend on data availability and user expertise. Generally, they suggest an order of preference: measurements (only for small-scale site-specific assessments) > dynamic crop-climate-soil models > simple C models > IPCC Tier 2 methodology > IPCC Tier 1 emission factor.
- For large-scale assessments, a method should be selected which is dependent on soil and climate variability and the timing of the emission should be considered.
- A time horizon of at least 20 years should be considered, but a longer time horizon (e.g. 30-100 years) would be preferable for example in cool climates with slower dynamics.
- If indirect land use change could be relevant (e.g. increasing the use of existing croplands for producing animal feed or biofuel, leading to deforestation elsewhere to produce crops for direct human consumption), at least a sensitivity analysis should be performed on it.
- The biogenic carbon flows resulting from land use and land management change should be reported separately from fossil carbon flows.

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5.1.2. IPCC methodology

The intergovernmental panel on climate change (IPCC) has proposed guidance on how to calculate national greenhouse gas inventories (IPCC, 2006). The methodology has been revised and developed over the years, and the most recent methodological update has been published in 2019. IPCC methodology is a standard methodology applied in LCA.

IPCC (2019) distinguishes between **three carbon pools** (which are again split into sub-pools): biomass (above- and below-ground), dead organic matter (litter and dead wood) and soils (mineral or organic). Carbon fluxes for all managed land should be assessed (managed land principle), but not for unmanaged (natural) land. Generally, according to IPCC, emissions and removals of carbon should be reported separately for **six land use categories**: Forest Land, Cropland, Grassland, Wetlands, Settlements, and Other Land. For complex land uses, such as agroforestry systems, emissions and removals of carbon needs to be reported under one of these six categories. It depends on the type of agroforestry and the national definitions of forests, under which category agroforestry should be reported. Different methods apply, if land is converted from one category to another (land use change) or if land is remaining in the same category (and only the land management optionally changes, e.g. tillage practices). IPCC provides guidance on different aspects relevant for the livestock sector: CO_2 emissions and removals resulting from C stock changes in biomass, dead organic matter and mineral soils, GHG emissions from fire, N_2O emissions from all soils; CO_2 emissions associated with liming and urea application, CH_4 emissions from flooded land, C stock change associated with harvested wood products, as well as CH_4 emission from enteric fermentation and CH_4 and N_2O emissions from manure management systems.

IPCC methodology follows a tiered-structure:

- **Tier 1:** simplest methods to use, equations and default parameter values are provided directly by IPCC.
- **Tier 2** same methodological approach as Tier 1, but using country- or region-specific data, higher temporal and spatial resolution.
- **Tier 3**, higher order methods are used (e.g. process-based models), which provide estimates of greater certainty than lower tiers, but require high-resolution data or repeated measurements and may also include interannual variability. Here, often country-specific methodologies can be applied.

Generally, moving to higher tiers reduces uncertainty, but increases complexity and data requirements. A combination of tiers can be used for assessing different aspects, depending on the data availability. Within the same Tier, often different methodologies are proposed, depending on the data availability.

No land use change

Table 2 gives an overview on the different methods proposed for assessing biogenic carbon stock changes for **land remaining in the same land use**, following the different tiers. Some pools are considered as stable in Tier 1 approach without a change in land use and are therefore not considered in the calculations.

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IPCC proposes two main methods to account for changes in C stocks in different carbon pools: Gain-Loss method and Stock difference method. The **Gain-Loss method** accounts for all processes that bring about changes in a pool within a year. Gains (increase in biomass or transfer of carbon from another pool) are summed and losses (decay, burning or transfer to other pools) are subtracted. It is the default method for Tier 1, but can also be used for Tier 2 and 3 with refinements. The **Stock-Difference Method** calculates the annual average difference between estimates at two points in time.

Table 2 Overview on different biogenic C stocks and IPCC-assessment methods (IPCC, 2019) for different tiers for land remaining in the same land use category (simplified presentation, additional options might be available). Agroforestry needs to be assessed under one of these three categories, depending on national definitions. The equations in the table refer to the original IPCC report.

	Cropland	Grassland	Forest
Biomass	Annual crops: **not considered** (assumed stable)	Tier 1: **not considered** (assumed stable) Tier 2 and 3: Gain-Loss or Stock difference	Changes are assessed
Above-ground	Perennial woody crops: Tier 1: Gain-Loss Method Tier 2: Gain-Loss or Stock difference	See above	Tier 1-3: Gain-loss Tier 2-3: Stock difference
Below-ground	Perennial woody crops: Tier 1: **not considered** (assumed stable) Tier 2: measurement of root-to-shoot-ratio	See above	Tier 1 Gain-Loss: Below-ground biomass estimated with ratio (Equation 2.10) or with biomass conversion and expansion factors (BCEFI)
Dead organic matter (same methods for dead wood and litter proposed)	Tier 1: **not considered** (assumed stable) Tier 2: Gain-Loss or Stock difference calculated separately for dead wood and litter	Tier 1: **not considered** (assumed stable) Tier 2: Gain-Loss or Stock difference	Tier 1: **not considered** (assumed stable) Tier 2: Gain-Loss or Stock difference
Soil organic carbon			
Mineral soils	Tier 1: stock change factor (Eq. 2.25) Tier 2: Steady state method	Tier 1: stock change factor (Eq. 2.25) Tier 2: Eq. 2.25 with country specific stock change factors	Tier 1: **not considered** (assumed stable), Tier 2: Eq. 2.25 with country specific stock change factors
Organic soils	Tier 1-2: annual carbon loss from drained soils (Eq. 2.26)	Tier 1-2: annual carbon loss from drained soils (Eq. 2.26)	Tier 1-2: annual carbon loss from drained soils (Eq. 2.26)

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Biomass

Changes in carbon in **above-ground biomass** are only considered for woody vegetation and are not accounted for in arable crops and grasslands. The gain loss method can be applied for Tier 1 – 3 and the stock-difference method for Tier 2 and 3. For Tier 1, default estimates of biomass stocks, growth rates and losses are provided for major climatic regions and agricultural systems. For agroforestry in temperate climates, default values are given for hedgerows, silvo-arable and silvo-pastural systems, and for some systems different values for cool and warm temperate and different continents is provided by IPCC. In addition, default values are provided for temperate, perennial monocultures of olives, orchards (e.g. apple), vineyards and short rotation coppice.

In Tier 1, changes in **below-ground biomass** of cropland and grassland are not considered. For forests, the below-ground biomass is estimated with a below-ground to above-ground biomass ratio (root-shoot ratio). In Tier 2, default values derived from measurements are provided for woody cropland and grassland, but they are highly dependent on species and communities considered and show wide ranges and the use of region and species-specific values is recommended. For forests, country specific above-ground to below-ground ratios should be used in Tier 2.

Dead Organic Matter

For Tier 1, changes in dead organic matter are not considered. For Tier 2 and 3, both the Gain loss and the Stock-Difference method can be applied. To estimate the carbon content in the dead organic matter, a default value of 50% C (calculated from dry matter) for dead wood and 40% C for litter can be applied.

Soil organic carbon

For assessing changes in soil organic carbon, different approaches and data are recommended for mineral soil and organic soils. For **mineral soils** under forests, no change in SOC is assumed under Tier 1. For cropland and grassland, as well as for forest under Tier 2, a “stock change factor” method and “steady state” method are proposed. The **stock change factor** method calculates the change in SOC in the top 30 cm after a management change compared to a reference condition. It is based on the assumptions that a stable SOC content (equilibrium) is reached after 20 years; and that the transition to the new equilibrium SOC occurs linearly. The new equilibrium SOC is depending on soil, climate, land-use and management practices. So-called “stock change factors” are provided, depending on climate and soil type, which reflect the impact of land use, land management and C inputs to soils. The **steady state method** can be applied to cropland and distinguishes three C pools in the top 0-30cm layer of the soil with different turnover time: active (month – years), slow (decades) and passive (centuries) pools. It estimates C stock changes from combinations of tillage and C-input management activities under conditions defined by the soil texture and the weather. It is based on the CENTURY ecosystem model, which calculates steady-state solutions for the three SOC sub-pools (Ogle et al., 2012; Parton et al., 1987).

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For **organic soils**, an annual emission factor is assigned that estimates the carbon lost following a drainage.

Land use change

For land, which is **converted from one category to another**, the assessment of biogenic carbon changes is different than for land remaining in the same category. An overview of approaches is given in Table 3.

For the re-establishment of a new land use type after conversion (biomass, soil and litter), 20 years are assumed as a default, but other regeneration times can be used.

Table 3. Overview on different biogenic C-stocks and IPCC-assessment methods for different tiers for land which is converted to a new land use category. A conversion to agroforestry needs to be assessed under one of the listed land use categories, depending on national definitions If agroforestry (defined as cropland) is established on cropland, no land use change is assumed. The equations in the table refer to the original IPCC report.

	Land converted to		
	cropland	**grassland**	**forest**
Biomass		Is accounted for Tier 1: Eq. 2.15, 2.16; Tier 2: Gain-Loss or Stock Difference	Is accounted for Tier 1: gain-loss (Eq. 2.7), Tier 2: Eq. 2.15
Above-ground	Tier 1: Eq. 2.15, Tier 2: Gain-Loss or Stock Difference	See above	See above
Below-ground	**not** accounted for on cropland	See above	See above
Dead organic matter (same methods for dead wood and litter proposed)	Tier 1: Assumed zero after conversion, Tier 2: gain-loss or stock-difference	Tier 1: Assumed zero after conversion, Tier 2: gain-loss or stock-difference	Tier 1: linear increase assumed, Tier 2: gain-loss or stock-difference
Soil organic carbon			
Mineral soils	Tier 1 & 2: stock change factor (Eq. 2.25)	Tier 1 & 2: stock change factor (Eq. 2.25)	Tier 1 & 2: stock change factor (Eq. 2.25)
Organic soils	Tier 1 & 2: annual C loss after drainage (Eq. 2.26)	Tier 1 & 2: annual C loss after drainage (Eq. 2.26)	Tier 1 & 2: annual C loss after drainage (Eq. 2.26)

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Biomass

Land use can change slowly (e.g. reforestation of crop- or grassland by natural or artificial regeneration) or fast (e.g. deforestation and establishing crop- or grassland). For fast conversions, two-phases are considered: an initial abrupt change in biomass estimated for the year of conversion and a gradual loss or gain of biomass until a new steady-state is reached. For the initial change, biomass stocks before and after the conversion, as well as the carbon fraction of the biomass are considered. For annual changes thereafter, annual increases and decreases in carbon stocks in biomass are assessed.

In Tier 1, it is assumed that the land is cleared of all vegetation before another land use is established, thus carbon stocks in biomass is assumed zero after conversion, and all carbon of the previous vegetation is lost (and not remaining as dead organic matter). For grassland and cropland, the new vegetation is assumed to be fully established shortly after the conversion. For perennial woody vegetation, accumulations and losses in biomass are accounted for as in land remaining within the same category. For forests, growth rates of trees are considered and a distinction is made between intensively managed forests and naturally regenerating forests, different tree species and climatic regions.

In Tier 2, land use transitions can be taken into account, where not all vegetation is removed from the land at once. A disturbance matrix can be used to summarize the retention, transfers and release of carbon from one to another pool (above-ground, below-ground biomass, dead wood, litter, soil organic matter, harvested wood products, atmosphere). Generally, there is limited data on below-ground biomass in croplands, and calculations are mainly done for above-ground biomass.

Dead organic matter

It is assumed under Tier 1, that carbon stocks in dead wood and litter pools are negligible in non-forest land. A linear increase of dead organic matter and litter in mature forests is assumed over a default time period of 20 years. For crop- and grassland, a zero carbon stock is assumed after conversions, and all carbon in dead wood and litter is assumed to be removed from the previous land use.

In tier 2, a two phase approach is applied (fast initial change, slower change thereafter). The gain-loss or stock-difference method can be applied. A disturbance matrix should be used to summarize transfer of carbon between different pools.

Soil organic matter

For organic soils, the assessment is similar than for land remaining in its land use category. For mineral soils, the change in SOC stocks is estimated using Equation 2.25 (IPCC, 2019). The SOC stocks before and after the land conversion are determined from default references and default stock change factors, depending on the land use and management both pre- and post-conversion.

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5.1.3. FAO LEAP recommendations

Soil organic carbon in LCA

In 2019, in parallel to the updated IPCC guidelines, the Livestock Environmental Assessment and Performance (LEAP) Partnership (FAO LEAP) published a guideline for measuring and modelling soil carbon stocks and their changes for livestock systems (FAO, 2019).

FAO LEAP recommends to **account for SOC changes** in LCA to have a comprehensive assessment and to avoid burden-shifting (i.e. when the mitigation of an environmental problem worsen another). For livestock systems, grasslands supporting livestock production, as well as land use and direct or indirect land use change of feedstock production should be included in the evaluation. If possible, carbon sequestered in or emitted from soils should be included in the overall GHG balance and the ratio of grassland surface required to produce one functional unit shall be made explicit.

The **temporal dimension** is crucial, because after a management change, SOC levels can rapidly change until they reach a new steady state, and SOC stocks often are slow to build up but are lost fast after management changes ("slow in, fast out" pattern). The choice of the time perspective strongly impacts the results of the LCA, as shown by Petersen et al. (2013). Typical time perspectives are 20 years (IPCC, 2006), but often this is too short to reach a new equilibrium. FAO LEAP proposes to use 100 years, the same time perspective typically used for GWP (in line with Petersen et al., 2013). Benefits of carbon sequestration (or the burden of carbon emissions) can be linearly allocated throughout a fixed period (e.g. equally over 20 or 100 years of production).

FAO LEAP highlights the importance of the appropriate selection and transparent communication of the **system boundary**, in line with the goal and scope of the study. An incomplete system boundary could foster burden-shifting, i.e. if the benefits in focus (e.g. reduced methane emission due to changes in cattle feed) cause greater burdens elsewhere in the supply chain (e.g. deforestation for producing feed). Therefore, all GHG emissions from land use and land use change should be included and effects of soil carbon changes on all components (e.g. imported feedstock or manure application) need to be considered too (FAO, 2019). In a separate document, guidance on how to account for effects of feed additives (e.g. on enteric fermentation) in environmental assessments is provided (FAO, 2020).

Since accounting for SOC can have a strong impact on the overall GHG balance, and there is still no consensus on how SOC should be accounted for, a **critical review** should be made before reporting LCA results including SOC changes. In such a review, it should be assessed if the model and data chosen were appropriate for the study.

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Three level approach of FAO LEAP

FAO LEAP proposes a three level approach to estimate SOC stocks and SOC dynamics using simulation models. The level should be selected depending on the purpose of the study, the spatial scale, and data availability. An overview on the validity of the model for different modelling purposes is given in Table 4. For LCA, FAO LEAP considers level 2 and 3 models valid and common practice (especially for land management change). Level 1 methods are rated as having limited accuracy and acceptance, but are suitable for assessing land use change or to get a first estimate of the expected SOC change direction or amplitude, when specific data is not available.

Level 1: 'Empirical' Models provide a first indication to predict the magnitude or direction of carbon stock changes, e.g. using IPCC factors that can be adapted based on region-specific experiments. However, these simple models often have limited accuracy for a specific region or system. Examples are the stock change factor method (IPCC, 2006) or simple carbon balance equations based on a set of predictors. The latter considers first order kinetic decay (dependent on a coefficient of mineralisation) and humification rates (dependent on types of crop residues and plant carbon inputs). At level 1, different SOC pools, their stability or temporal changes in mineralisation and humification rates are however not accounted for.

Level 2 'Soil' Models simulate SOC dynamics over time, and use data on plant carbon inputs and environmental parameters affecting carbon sequestration and losses. Examples are YASSO (Liski et al., 2005), ICBM (Andrén & Kätterer, 1997), C-TOOL (Taghizadeh-Toosi et al., 2014), CANDY (Franko et al., 1997), Roth-C (Coleman et al., 1997). These models are often based on different conceptual C pools, with different decomposition rates and stabilization mechanisms, with carbon transfers from plant and animal biomass to microbial biomass and to different pools of soil organic carbon. However, models at level 2 do not include other complex dynamics such as plant growth or nutrient dynamics and may have limitations when applied to specific situations.

Level 3 'Ecosystem' Models integrate the feedbacks from multiple soil-plant- atmospheric processes on SOC dynamics and can be used to analyse impacts between agricultural management, crops and soils, as well as trade-offs between SOC change and other environmental indicators or crop-yields. They simulate above and belowground plant biomass growth and carbon inputs, soil water dynamics, nutrient dynamics and their interactions, based on different organic C pools (active, slow, passive, inert). Examples are EPIC (Williams et al., 1984), CENTURY (Parton, 1996), DNDC (Li, 1996), DAISY (Svendsen et al., 1995), SOCRATES (Grace et al., 2005), MEMS 2.0 (Zhang et al., 2021). Some level 2 models have been incorporated into ecosystem or farm models (e.g. ICBM in HOLOS (Kröbel et al., 2016), which was developed to estimate SOC changes and whole-farm greenhouse gas emissions). Some ecosystem models have SOC subroutines (DSSAT (Jones et al., 2003), APSIM (McCown et al., 1996)) or are specifically oriented to livestock systems, and also simulate SOC dynamics (e.g. ECOMOD Suit, including EcoMod, DairyMod, and SGS Pasture models (Johnson et al., 2008) or PaSIM (Riedo et al., 1998)).

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Table 4 Validity of simulation models of different complexity (Level 1, 2 and 3) for various purposes (source: FAO (2019)). Red: invalid, yellow: limited accuracy and acceptance, green: valid and common practice.

	Model complexity		
Purpose	**Level 1**	**Level 2**	**Level 3**
National accounting			
Comparing management practices			
Optimizing ecosystem services			
Climate change scenarios			
Benchmarking			
Life cycle assessments			
Cross compliance (including other GHGs)			
2-3D modelling (depth profile, lateral fluxes)			
Upscaling			
Commercial farm assessment			

5.2. Examples of carbon models in agricultural LCAs

5.2.1. FarmLCA

Short description of model

The FarmLCA model (Meier & Moakes, 2019; Schader et al., 2014) can be used to assess the environmental and economic performance of farm products, enterprises and systems. It consists of i) a farm system model, ii) an LCA part, and iii) an economic assessment module, and allows to calculate cradle-to-farm gate LCAs. The livestock and plant production enterprises and their interlinkages can be modelled in detail, as well as changes in carbon pools in soil, based on the land use history and management. The farm-specific transfer between livestock and plant production, such as on-farm feed and forage as well as diet-specific manure, can be considered. FarmLCA is specific to different livestock herd structure, their nutrient requirements at different life stages, the enteric fermentation and manure management, as well as plant nutrient requirements and N, P and carbon emissions. Direct emissions, e.g. of nitrous oxide (IPCC, 2019), ammonia (EMEP/EEA, 2019) and nitrate (Faist et al., 2009) are modelled. The FarmLCA version 4.1 also allows to assess agroforestry systems, by permitting to model multiple land uses on one field (e.g. crop production under trees and grazing of crop residues) with multiple outputs (e.g. fuelwood, grains and meet). Data from ecoinvent version 3.8 (Wernet et al., 2016) and AGRIBALYSE® (2023) are linked to the FarmLCA to calculate impacts of background processes (such as the production and transport of agricultural inputs which are not part of the main system analysed). A set of plausibility checks are implemented to allow validating data during data entry and a set of standard data is implemented to assist in filling potential data gaps. Finally, environmental impacts on-farm as well as off-farm (e.g. of purchased inputs) can be calculated based on the Impact World+ methodology (Bulle et al., 2019) for different functional units.

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Soil C-Sequestration

The FarmLCA model allows the user to choose how to model SOC changes: 1) not include changes; 2) calculate SOC changes based on IPCC (2019) Tier 1; or 3) based on IPCC (2019) Tier 2 approach. Results of SOC changes are always displayed separately from other GHG emissions. In the following, the IPCC Tier 2 approach is described in more detail. It is based on a steady state method calculated with a simplified CENTURY model, to quantify carbon sequestration (IPCC 2019). It is a dynamic model, thus it is designed to capture changes on soil carbon across time. At its core, carbon pools representing different stages of decomposition and stabilization of carbon in soil are modeled along with microbial and chemical processes, integral to the breakdown of organic matter (Figure 3). Varied decomposition rates across carbon pools influence the release of carbon dioxide into the atmosphere. To determine the status-quo of soil organic carbon, the dynamic model is initialized over 20 years, using monthly climatic data of the past, encompassing temperature and precipitation, which influences microbial activity and decomposition rates. The model incorporates feedback mechanisms, where alterations in carbon pools influence future decomposition rates and carbon cycling.

Farm management practices, such as tillage, crop residue management, and organic amendments like manure, are considered in carbon inputs, outputs, and overall soil carbon balance.

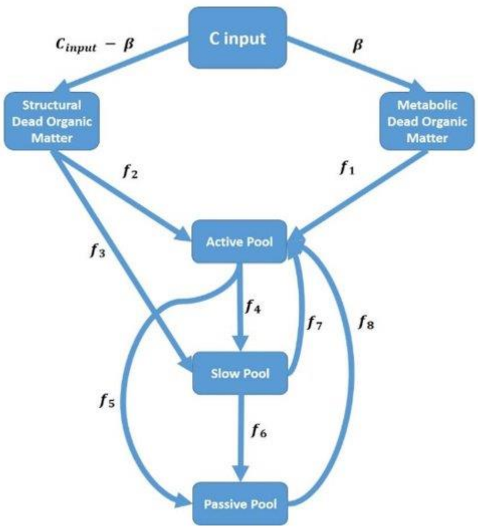

Figure 3 : Dynamics modelled by the CENTURY model with carbon pools and the varied decomposition rates across carbon pools that influence the release of carbon dioxide into the atmosphere.

The FarmLCA model employs an individual plot modeling approach for each year, accommodating several crops such as green manure or mixed crops within a single plot. For each plot, explicit definitions are required for tillage practices and carbon input. This carbon input is determined by calculating the residue left on the field, incorporating both crop residues, including green manures, and carbon from manure input. The model offers flexibility by

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allowing adjustments of manures to meet crop requirements while considering the availability of manure on the farm.

To initialize the CENTURY model embedded in the FarmLCA model, an assumption is made that the crop combination being modeled aligns with a long-term crop rotation. Additionally, it assumes that the practices modeled have been consistently implemented over the preceding 20 years. Subsequently, the model calculates the average soil carbon in the year before the assessment (t-1) for all fields. The soil carbon in the current year (t) for each field is then determined based on the specific carbon input of the plot in that year relative to the average soil carbon in the previous year (t-1). This process ensures a dynamic representation of soil carbon dynamics within the FarmLCA model.

The model configuration allows for the initiation of the CENTURY model using a specified baseline, as depicted in Figure 4. In the context of scenario runs, there is a feature that enables reliance on the outcomes of the baseline without the necessity for re-initialization. This design enhances efficiency and streamlines the modeling process, allowing for the seamless continuation of scenarios based on the established baseline conditions.

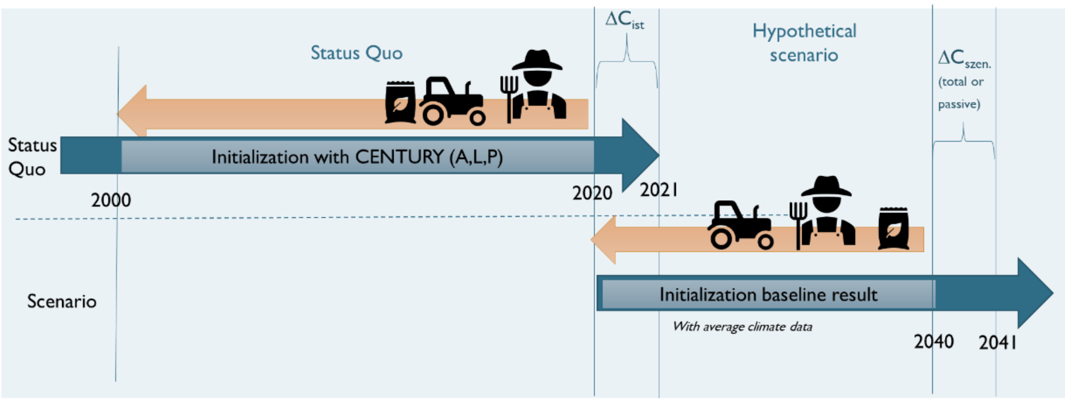

Figure 4: Initialisation of the CENTURY model from scratch or based on the baseline

The model provides flexibility in handling climatic data by allowing easy substitution with alternative sources. While the model is adaptable to various data inputs, the standard data available is sourced from CRU v4 CY (Harris et al., 2020). This dataset represents the average climate conditions at the country level. For future climate projections, the default setting relies on the average data from the last two decades.

The FarmLCA model introduces flexibility in accounting for soil carbon within the life cycle assessment (LCA) framework, accommodating different perspectives. Two primary approaches are offered to model users, each aligning with distinct viewpoints:

1) Yearly Perspective (IPCC Recommendation): This approach adheres to the IPCC recommendation, where soil carbon is computed as the summation of the active, slow, and passive pools. This yearly perspective provides a comprehensive overview of soil carbon dynamics over shorter time intervals, as suggested by IPCC guidelines for annual national GHG inventories.
2) LCA Perspective: In alignment with an LCA perspective, where the emphasis is on considering carbon storage over the long term, it becomes crucial to address the

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potential for short-term losses, especially when following the IPCC approach. Baveye et al. (2023) show that a substantial portion of the soil carbon input in the year of assessment might experience rapid losses within the following 20 years. This temporal vulnerability is particularly notable for the active and slow pools.

Given these considerations, the FarmLCA model introduces a nuanced approach to account for long-term carbon storage. In line with the findings of Baveye et al.(2023), the model offers the option to focus exclusively on the passive pool. This pool, representing carbon that has mineralized, tends to exhibit more extended persistence over the 20-year timeframe, as opposed to the active and slow pools.

The FarmLCA model, recognizing the non-linear accumulation of the passive pool over time, compute the total accumulation of the passive pool over 20 years as uses the yearly average to account for carbon sequestration in the specific year for which the life cycle assessment is conducted.

Biomass and dead organic matter in agroforestry systems

Similar to SOC changes, the FarmLCA model allows users to choose if the changes of carbon stored in biomass should be account for with IPCC (2019) Tier 1 or Tier 2 or not included in the analysis. In the Tier 1 approach, emission factors from Cardinael et al. (2018) were adopted. For Tier 2, the Gain-loss-Method was implemented, using standard data of the weight of different tree species for silver birch, beech, larch, Norway spruce, oak, scots pine, conifers and other broad-leaved trees.

For dead organic matter of woody biomass, the annual change is calculated according to the Tier 2 gain-loss-method.

In its current form, the model allows to account for a set of tree species, but it has the flexibility to add new species that have been parameterized beforehand.

5.2.2. Introductory Carbon Balance Model (ICBM)

Short description of model

ICBM is a simple model to simulate soil C dynamics in agricultural soil (Andrén & Kätterer, 1997; Bolinder et al., 2018). It is a dynamic two-compartment model with first-order kinetics. Multiple pools of “young” materials are represented as independent pools with first-order decay, with a time constant of roughly 1 year. Part of the C leaving the young pools enter a pool of “old” materials, which also has first-order decay, however with a much longer time constant of roughly 200 years.

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A recent calibration of ICBM based on data from long-term field experiments at two sites in Sweden included separate young pools for above-ground plant residues, below-ground plant residues, manure, sewage sludge, sawdust, peat, and compost (Bolinder et al., 2018).

ICBM has been used in research on soil C dynamics(e.g., Kätterer et al., 2011; Poeplau, Aronsson, et al., 2015; Poeplau, Kätterer, et al., 2015) and in applications including national inventory reporting (Sweden) and LCA studies (e.g., Hammar et al., 2022).

Expressed in equations, the model has the following parts.

The young pools j = {manure, above-ground, below-ground, ...} have (possibly time-dependent) C inputs I_j and exponential decay with a common rate constant $r\ k_Y$:

$dY_j/dt = I_j(t) - r\ k_Y$

A pool-specific fraction h_j of the C leaving each young pool is transferred to the old pool ("humified"); and the old pool has exponential decay with rate constant $r\ k_O$:

$dO/dt = \Sigma_j\ h_j\ r\ k_Y\ Y_j(t) - r\ k_O\ O(t)$

The exponential decay parameters have a common factor r, which represents soil and climate conditions. The idea is that the factors k_Y and k_O represent what is intrinsic to the C-containing substrates while r represents site-specific (possibly time-dependent) conditions.

Main features of model

ICBM predicts SOC response to different crops and management practices. It is limited to mineral soils. As the model is time-continuous, results are easily calculated for any chosen time frame. The model is not spatially explicit, but in principle it can be adapted to any spatial resolution subject to calibration data availability.

Required input data

Each of the young pools is associated with different parameters for "humification", corresponding to substrate quality (above-ground crop residues, below-ground crop residues, manure and other organic amendments). Each pool has a time constant for the first-order dynamics. A combined soil-climate factor is calibrated using daily weather data, pedotransfer functions for soil/air temperature, and response functions for the influence of soil water and temperature on biological activity.

After this calibration, the inputs needed are simply quantities (e.g., annual) of inputs of above-ground and below-ground crop residues and organic amendments.

Usefulness for LCA

ICBM can be used to estimate SOC balance in mineral soils under different crops and crop management. See, for example, Hammar et al. (2022).

5.2.3. MEMS framework

Short description of model

The MEMS model is a complete ecosystem model including N cycling, plant growth, microbial processes and carbon pools dynamics (Zhang et al., 2021). The pools included in the model are physically defined and can be directly measured, as opposed to traditional models (e.g., RothC, Century or DayCENT) that represent conceptual pools based on turnover time (e.g. active or passive) but that cannot be directly measured. The MEMS model belongs to a new generation of models based on recent evidence and advancements on conceptual frameworks about the processes of SOM formation and persistence, where measurable soil C pools and fluxes play a central role (Zhang et al., 2021). Among these new generation of models, MEMS is a SOC model that includes physically measured pools that accords with recent frameworks on SOM formation such as the microbial efficiency–matrix stabilization (MEMS) hypothesis (Cotrufo et al., 2013). Under this framework, there is a paradigm shift where labile plant residues would lead to stable SOM, due to their higher usage by microbes and affinity to the mineral soil matrix.

Main feature of model

The main feature of the model is the consideration of functionally defined and measurable SOM pools that results from known biogeochemical processes. This characteristic has the potential to overcome the limitations of traditional SOM models, that have shown contrasting results under similar environmental and land management scenarios. A key point of the MEMS model is the coupling of C and N cycles. This allows the incorporation of feedbacks between both elements, such as the effect of N limitation on decomposition or the presence of constrains based on known stoichiometric relationships.

Required input data

A simplified version that only simulates litter and soil components for a single layer and a site requires data about three major categories: daily weather data (maximum temperature, minimum temperature and precipitation), soil characteristics such as (pH, soil bulk density, sand content and rock fraction) and land use. Some input variables such as site net primary production (NPP) or C inputs are need at a daily basis.

Limitations of the Model

The MEMS model does not incorporate common agricultural practices. Land use is defined in broad categories and related to main vegetation types. It is a complex model which requires knowledge and time to master. The calibration of the model is related to the fractionation of SOM, which makes its usage by non-experts difficult. It is in the forefront of SOM dynamic research.

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Usefulness for LCA

The model can provide detailed information on the mechanisms operating for SOM dynamics and turnover. It can help to understand the potential of C accrual in the different soil fractions and may help to interpret LCA results. The model can estimate key ecosystem variables, including SOC changes, in response to land use change and management. However, the model might not be suitable to directly be applied in an LCA because of the lack of a thorough validation of the accuracy of SOC estimations following land use change (Zhang et al., 2023).

5.2.4. Agrecalc

Short description of model

Agrecalc is a user-friendly farm-scale carbon calculator that includes agricultural soil C balances, based on the IPCC tier 1 model (Topp et al., 2017). Parameters fit to data that is easily accessible to farm owners and managers, with assumptions relating to soil type using national soil data and farm coordinates. It has been used widely for farm-specific carbon calculation as a farm management aid, as well as various LCA studies (Barnes et al., 2022; De Vries et al., 2022; Sukhoveeva, 2021; Topp, 2017; Topp et al., 2017).

Main feature of model

The primary feature of the model is that it can estimate baseline carbon stocks based on geographic location as well as national soil data and can predict SOC response to land use change, management and application practices. The model can be applied on a farm scale, for various farm types, sizes and locations (Sukhoveeva, 2021). The model assesses soil C-carbon impacts for a chosen year, based on the average expected annual changes over the course of a 20-year timeframe (it is assumed that soil-C impacts will remain steady for 20-years and level off/normalise after 20-years) and is calibrated for arable and pasture mineral soils.

Required input data

Agrecalc requires details of the farm location and enterprise details such as which livestock and crops are grown on-farm. It also requires information on farm inputs such as imported feed, livestock bedding, fertilisers and pesticides, as well as exports of livestock products, crops, and other products (Topp et al., 2017). Additionally, information is needed on land use change practices (such as conversion from grassland to cropland), land use management practices (such as livestock grazing intensity and tillage), as well as input use and application (such as chemical and organic fertiliser application as well as cover cropping)(Sukhoveeva, 2021; Topp et al., 2017).

Usefulness for LCA

The model can be used to provide quick, easy and accessible estimates of SOC balance under different livestock and arable farming systems (Sukhoveeva, 2021; Topp et al., 2017).

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Limitations of the model

The Agrecalc model was developed as an easy to use model to aid decision making in farms, which means accessibility to farmers is prioritized ahead of flexibility and reducing assumptions. It is not possible to adjust the internal parameters and embedded questions according to farm type or specific research aims, as the tool is designed to capture all farms with minimal input from the end-user (Sukhoveeva, 2021). Again, due to the tool being designed specifically for farmers to assess farms on a given year, it is not possible to assess different spatial scales (specific parts of a farm, or regional farm systems), or to make assessments over a time-scale more, or less than a year.

5.2.5. Implementing IPCC models in LCAs

Short description of models

IPCC has 3 tiers of models based on different sets of equations that utilise the most advanced datasets and experimental findings to estimate agricultural soil C balance. The Tier 1 model is the most generalised and requires the least data, mostly using default figures with high levels of assumptions that are provided by IPCC (Hergoualc'h et al., 2021; Mathivanan et al., 2021). If more detailed (rigorously documented country-specific) emission factors are available then the Tier 2 model may be applied. This requires more data, but uses less assumptions and is country-specific, thus providing more accurate results (Thiagarajan et al., 2022). The Tier 3 model is the most accurate model but requires additional data components and modelling beyond those provided by IPCC. For all tiers, parameters can fit to long or short-term field experiments and/or data can be provided by IPCC for different data components (Rodrigues et al., 2021).

Main feature of models

The main feature of the IPCC models (applied in LCAs) is to estimate the implications of different crops and management practices to SOC. While the models are not spatially explicit, they are commonly used at a national scale. Moreover, the models can be adapted for any chosen time frame and can be applied to arable and pasture mineral soils (Rodrigues et al., 2021), but also organic soils.

Required input data

There are a range of different data inputs relating to crops and management practices that are required for various equations associated with estimating SOC changes. These data inputs can come from IPCC default values, from the literature or from experimental work. Equations are used to determine various components associated with carbon inputs and outputs from the system, such as the carbon content of manure at the point of application (based on livestock type and diet, in addition to manure storage type and duration), as well as the fraction of this carbon that is absorbed by the soil (Thiagarajan et al., 2022).

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Usefulness for LCA

IPCC are flexible models that can require limited data and expertise (Tier 1 model), or additional data and expertise (Tier 2 and 3) (Hughes et al., 2023). They can be used to estimate SOC balance under different crops and livestock management systems yet they do not reflect local conditions (especially Tier 1) (Mathivanan et al., 2021).

5.3. Carbon models used for agroforestry-systems

In agroforestry systems, woody vegetation is deliberately integrated into crop or animal systems. These systems are highly diverse and change in non-linear ways, as the different components (trees, crops and livestock) show substantially different growth periods and diverse interactions (Burgess et al., 2019). Getting experimental data about agroforestry systems is a challenge, because the long time periods involved in woody vegetation growth would require long-term data collection and research financing.

An alternative solution is the development of models to predict tree, crop, and livestock growth. More specifically quantifying the carbon stocks and flows in the system. There are two major carbon stocks considered in agroforestry systems, above ground and below ground (Nair et al., 2010). Above ground considers tree and crop growth, whereas below ground takes into account the carbon from leaf and branch fall, livestock or other fertiliser, and root decomposition.

To better understand the different methods involved in carbon modeling, three models are summarised below. Each model excels in certain areas but may make assumptions or exclude other areas.

5.3.1. Agroforestry Carbon Code

Short Description of Model

This model was developed as part of a feasibility study to identify if a woodland carbon code-style model could be useful (Soil Association, Woodland Trust et al., n.d.). The user creates a customized tree growth model using input measurements from their land/surrounding landscape. The model then predicts how biomass and thus carbon changes over time. Tree biomass is estimated using the Bunce Equation (Bunce, 1968) which relates the tree Diameter at Breast Height (DBH) to the dry weight and thus carbon of the tree systems. This is only relevant for above ground biomass. Belowground biomass estimates are made based on woodland carbon assessment protocol and their relevant equations relating DBH to root biomass (Jenkins, 2018).

Main Features of the Model

The primary outputs of the model include values for aboveground and belowground biomass in open-grown trees for a 30-year period. In addition, carbon emissions through planting and management activities are also produced, the results of which are subtracted from the biomass to calculate net carbon sequestration for the agroforestry system.

Required Input Data

The tree growth models require DBH data for species of interest at varying ages, ideally from samples on or local to the site being modelled. In order to produce reliable models, at least three ages/development stages ranging from three to 150 years are required. This improves the applicability of the results to that particular site. Tree numbers and density are also important factors required to calculate total biomass/carbon sequestration over an area (normally per hectare). Finally, the emissions from planting and management need data on soil preparation, staking methods, fencing materials, tree guards, mulching, restocking practices, understory management, canopy and root management, and product harvesting.

Limitations of the model

The Bunce equation was developed on broadleaved species in Scotland and thus may offer limited evaluation for species and locations that are significantly different. The Bunce equation was also specifically used for its relevancy to open-grown tree systems. As such denser systems including hedgerows and woodlands are not well represented by this model. The reliance on in-field measurements for model parameterisation can mean getting a suitable range of ages difficult, thus limiting the outputs of the model.

This model was an initial attempt to evaluate the potential of an independent agroforestry carbon code model. As such it lacks the time and detail required of a fully fledged predictive model. Further evaluation and refinement of this model is ongoing.

Usefulness to LCA

Alone this model offers a low-input way to calculate carbon sequestration for trees in an agroforestry system. It lacks the holistic approach of more detailed models, however this detail is not always appropriate depending on the goals and scale of the LCA. It would be best combined with other simple models, such as a typical whole-farm carbon calculator which takes into account carbon dynamics other than trees.

5.3.2. Integrating RothC (soil C) into YieldSAFE (C in biomass)

Short Description of Model

Yield-SAFE, is a parameter sparse model to estimate aboveground biomass in agroforestry systems (van der Werf et al., 2007). It may be considered a “single-leaf” type of model where all the resource competition is estimated at single area unit. Originally developed for silvo-arable systems, it has been improved over the years to consider grassland/pastures, improvement of provision of pan-european climate datasets, estimation of carrying capacity, carrying capacity effect on pasture yields, or effects on microclimate and thus animal welfare

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incorporating animal stress indexes (e.g. temperature humidity index: THI), amongst other (see Palma et al., 2017). Within these improvements, there was the integration of the widely used model RothC to simulate underground carbon dynamics. The Yield-SAFE has been improved also to interact with RothC on the estimation of input plant material into soil (i.e. leaf fall and root mortality) while maintaining the original aspiration for a simple conceptualization of agroforestry modelling (Palma et al., 2018).

Main Features of the Model

Yield-SAFE can simulate silvo-arable, silvo-pasture, and treeless systems (just crop, pasture, or forest). Within the silvo-arable/silvo-pasture systems, resource competition (water and light) between crops and trees are modelled in a daily time step. This is done via simple equations and parameter values, many of which are available in the literature (e.g. radiation use efficiency, water use efficiency). This makes it relatively easy to parameterise new species into the model.

The model only has a single soil layer. This soil layer uses models of soil tension with the van Genuchten equation (1980) and commonly occurring parameters e.g. Wösten et al. (1999). Soil carbon is calculated through the integrated RothC model. This estimates the incorporation of tree leaf fall, tree root mortality (using the theory of the whole-plant economics spectrum), and crop residues (above and below-ground) after harvest as organic matter inputs to RothC.

The model operates on a daily time step providing a detailed output. The interface is currently excel-based but work is ongoing on a python version for self-development and applications. For a list of trees and crops available in the model, see Palma et al. (2017).

Required Input Data

Daily weather data is required for the model calculations. This can be retrieved automatically via clipick (Palma, 2017)[2]. This tool can be used for retrieving climate format for Yield-SAFE and Hi-sAFe model (section 5.3.3). Soil data is also necessary, specifically depth and texture (choice of five FAO classes which determine parameters for the van Genuchten equation, as defined by Wösten et al. (1999). Dates of planting and pruning for trees, and sowing and harvesting for crops need to be entered. For soil carbon the inputs for RothC are required; soil organic matter, bulk density, soil depth until carbon dynamics are present, and rate of decomposition. Finally, if new trees/crops are added into the model then they require additional parameters (~10).

[2] Code also available via github: https://github.com/euraf/clipick

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Limitations of the model

The Yield-SAFE model does not model nutrient dynamics and is not 3D nor spatially explicit. The preset assumes a single rotation of trees (i.e. full harvest for timber) and has not been developed with coppice in mind, although it may be possible to include a coppice-style rotation. The model is also limited in that it doesn't deal with mixed tree systems (i.e. multiple species of trees) or perennial undercover (shrubs). Livestock interactions are also limited to provision/grazing and nitrogen or carbon inputs from livestock are not included.

There are also technical limitations. If the model is set to run with many years (e.g. rotations of slow growing trees), and if multiple climatic scenarios are stored in the MSExcel file, then the file size can become large and may struggle to open/run on some computers. The ongoing python version can help with these technical limitations but requires some programming knowledge so, currently, is less user friendly.

Usefulness to LCA

Yield-SAFE has been used for LCA assessments before using a "carbon balance" method (Crous-Duran et al., 2019). However, the model requires some experience, time and data to calibrate new species, which can limit its usage in LCA analysis.

5.3.3. Hi-sAFe

Short description of model

Hi-sAFe is a 3D model that simulates tree and crop growth and management interventions, soil water and nitrogen fluxes, and soil organic matter processes (Dupraz et al., 2019). The model combines in a single platform the crop model STICS and the tree model sAFe-Tree. The model simulates a rectangular scene divided into cells. Each cell is then divided into various soil layers, as decided by the user. In each cell one crop or tree can be planted depending on the agroforestry design. Tree and crop above- and below-ground biomass is estimated daily. Tree biomass is calculated based on allometric equations. The integration between trees and crops is based on equations that calculated the interaction between vegetation layers for light, water and nitrogen. The model simulates several variables related to these three components. Hi-sAFe can model soil N_2O emissions but not CH_4. In addition, it does not include livestock components.

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Main features of the model

Hi-sAFe considers the three-dimensional tree-crop interactions; light, water, and nitrogen. The soil compartment is divided into layers and voxels, each one having a high temporal resolution of resource dynamics. Within the model there are several tree and crop species already available. Main crop species are included, as well as some tree species of interest. The distribution of vegetation types in the scene is flexible and different vegetation layers can be modelled. Multiple types of trees and crops management are also considered, e.g. pruning, harvesting, mulching, fertilization, tillage, etc. Crop rotations can be also modelled. Similar to Yield-SAFE, the time-stop unit is daily. The trees and crop's dynamic response to changing climatic conditions and resource availability is also modeled.

Hi-sAFe relies on the STICS model to simulate SOC dynamics. In the STICS model the involvement of residues, microbial biomass, and active pools are considered in the mineralization process, while the stable pool is presumed to remain inert over the course of a century. The simulation encompasses the decomposition of organic residues such as plant residues or organic amendments. The organic matter within residues is assigned to a singular pool undergoing decomposition through first-order kinetics, governed by a decomposition constant. The resulting decomposed organic matter undergoes either mineralization (yielding CO_2) or assimilation by the microbial biomass.

Required input data

The model requires multiple parameters for trees and crops related to allometry, physiology or phenology. The parameterization of a new species is a daunting task, being mostly reliant on the available species included in the model. Tree parameterization requires less parameters than crops. The model needs detailed information on soil properties and layers, regarding soil texture, organic matter, stone content, initial soil water and N content or pH among others. Daily weather data (precipitation, temperature, relative humidity, wind, and radiation) should be also provided. Management interventions need to have dates and intensity described. For instance, for fertilizers quantity, quality, date of application and associated labour must be supplied. All the information about trees, crops, soil and weather should be supplied in dedicated data files.

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Limitations of the model

The model was created to simulate silvo-arable systems. It has a great flexibility to be adapted to other systems, but it requires the measurement of several parameters in the field. The learning-curve is steep and error prone as the model contains more than 400 parameters that can be individually changed. It is a complex model because it simulates in detail various specific mechanisms (e.g. soil water, nitrogen and carbon dynamics, tree and crop production and phenology) and the interaction among them.

Usefulness to LCA

At its core, Hi-sAFe relies on the STICS model to simulate soil and crop dynamics. STICS has been extensively used to model agricultural systems. It is regarded as a complex model (sensu Avadí et al., 2022), thus it can overcome some limitations of simple models to estimate direct field emissions or SOC dynamics. On the other hand, this model requires high expertise, time and data which can limit its usage in LCA.

5.4. Examples of agricultural LCA studies including biogenic carbon

5.4.1. Comparison of measurements and different models for SOC changes in crops

Goglio et al. (2018) compared several methods to estimate CO_2 and N_2O emissions from soils in agricultural LCAs:

(a) measurements from a seven-year crop rotation field experiment in Manitoba, Canada

, (b) Tier 1 and (c) Tier 2 IPCC methodology,

(d) ICBM for CO_2 combined with Tier I for N_2O emissions, and

(e) the DNDC agroecosystem model.

Comparisons were made for perennial and annual crops, legumes, cereals and whole cropping systems for a time series with 28 time intervals (four per year over seven years). They did not find statistically different results between measured soil CO_2 emissions and any of the four modelled results for any of the crops or the cropping system as a whole.

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Although this suggests that modelled results were in agreement with measured data, the absolute values and the direction of predictions (net emission or sequestration) deviated strongly for single time intervals. For only about half of the time intervals, the measured CO_2 emissions were within the range of the modelled results, for the other half, the measured CO_2 emissions or sequestration was considerably larger than any of the models. The measured CO_2 emissions from or to soil were highly variable across the 28 reported measurements times and ranged from net positive emissions to negative (SOC accumulation, see Figure 5), while modelled results showed lower variation between time intervals. For about half of the 28 assessed time intervals, the predictions of the four models ranges between net positive and negative CO_2 emissions, while for the other half the predictions ranged between no emissions to net positive or no emissions to net negative emissions. The authors also stated that results from measuring CO_2 flux and soil carbon can be highly variable and inaccuracies in quantifying net CO_2 can occur in site-specific assessments. The study by Goglio et al. (2018) shows, that a model validation is difficult to perform in the context of LCAs, where the long-term changes on SOC are of interest, while in reality dynamics are highly seasonal as well as inter-annual. In addition, it shows how different modelling approaches that have been applied in LCA contexts even disagree in the short-term direction of SOC changes (positive or negative). This agrees with the difficulties of traditional models to assess SOC dynamics (see section 5.2.3).

The strong variability on SOC is particularly relevant for arable crops in crop rotations. However, also for permanent grassland (non-pasture), recently published data from a 16 year experiment showed large inter-annual variations: in nine years, the grassland accumulated carbon, in seven years it acted as a carbon source (Feigenwinter et al., 2023). The authors conclude that under the ongoing climate change, it will be difficult to maintain even a small grassland C sink in the future and will require continuous organic C imports (in this case study from slurry).

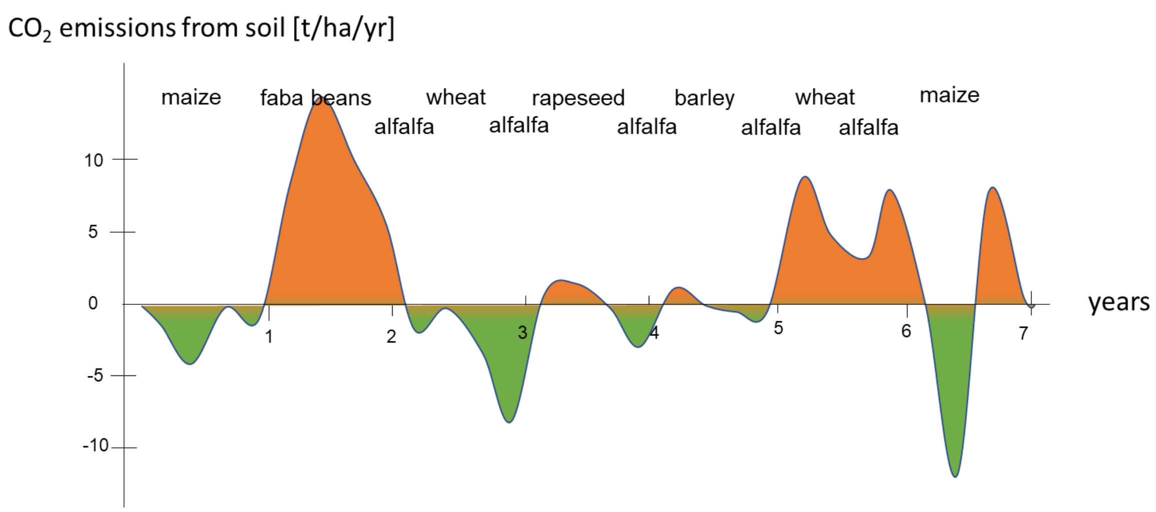

Figure 5. Illustration of dynamics of CO_2 emissions from arable soils over time, based on measurements presented in (Goglio et al., 2015). Orange: CO_2 emission; green: CO_2 accumulation.

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5.4.2. Comparison of biogenic carbon models in LCA for crops and livestock

Bessou et al. (2020) analysed SOC for five crop and two livestock products, subjected to land use change and land management change. They applied the three methods: i) IPCC Tier 1-2, ii) Müller-Wenk and Brandaõ (2010) and iii) Levasseur et al. (2012) and used a total of 22 scenarios of land use and land management change. For the IPCC approach, both SOC and carbon in biomass were assessed for changing the land use from situation A to B, and allocating impacts over the first 20 years. The method of Müller-Wenk and Brandaõ (2010) connects to the land use assessment framework (see Figure 1) (Koellner et al., 2013; Milà i Canals et al., 2007) and calculates both occupation and transformation impacts for SOC and biomass for a reference situation and the land use under study. The method accounts for the reversibility of biogenic carbon emissions, for example after deforestation. A residence time of biogenic carbon in the atmosphere is calculated, based on half of the time an ecosystem needs to regenerate after e.g. deforestation. For fossil carbon, the residence time in the atmosphere is typically larger, here assumed to be 157 years according to the Bern carbon cycle model. The third approach of Lavasseur et al. (2012) is a dynamic LCA approach, where a dynamic global warming potential (GWP_{dyn}) of GHG emissions/sequestration is assessed based on the timing of GHG emission/sequestration. Impacts are assessed over a 100-year horizon and all GWP_{dyn} of GHG fluxes are integrated over this period.

Brandaõ (2010) found that accounting for SOC was highly relevant compared to the overall GHG emissions of the assessed products. Impacts from land use change ranged from − 23 to + 1702% (IPCC method) and from − 5 to + 336% (Müller-Wenk and Brandaõ, 2010). For land management change, impacts ranged from − 130 to +54% (IPCC) and from − 31 to + 11% (Müller-Wenk & Brandaõ, 2010), respectively. Dynamic LCA could only be calculated for a subset of scenarios, because of high data requirements and were mostly lower than the other approaches.

Bessou et al. (2020) conclude that IPCC was easiest to implement from a practical point of view. Müller-Wenk and Brandaõ (2010) would need more differentiation of land use archetypes and regeneration times to be fully applicable. Dynamic LCA was most difficult to apply, because of the high data requirements to model carbon stock dynamics.

5.4.3. Examples of SOC models applied to LCAs of livestock systems

Two LCA case studies, calculating SOC changes, are mentioned in FAO LEAP (2019). Henry et al. (2015) calculated net greenhouse gas emissions per kg greasy wool at the farm-gate for Australian pastoral lands, including changes in SOC and vegetation. (Little et al., 2017) applied the ICBM in Holos-FarmModel to assess the effect of forage source on the GHG intensity of milk production (corn silage vs. alfalfa), including effects on enteric CH_4 emissions and SOC changes. They highlight that SOC will change over time, and therefore report effects separately.

Hammar et al. (2022) assessed the dynamic climate impact of beef production, calculating yearly fluxes of different GHGs, including from SOC changes, and assessing their effect on temperature response over time. For SOC changes from land use, the ICBM model was applied, assuming fallow land as a reference situation. The authors conclude that about 15–

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22% of GHG emissions arising from beef production could be compensated by carbon sequestration. However, when choosing potential natural vegetation as a reference (in this case forest), then the land would most likely result in net carbon emissions rather than a net sequestration (compared to fallow land).

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5.4.4. Examples of biogenic carbon assessments within LCAs of agroforestry systems

A recent review of 32 LCA studies on agroforestry systems showed that 17 did not account for carbon sequestration in biomass or soils, four studies included both carbon sequestration in soil and biomass, four included carbon stored in trees only and seven studies considered only carbon stored in soils (Quevedo-Cascante et al., 2023). In addition, the approaches and time horizons considered varied across studies, from soil carbon being at an equilibrium, to assuming an indefinite rate of change to accounting impact over 10-years (Quevedo-Cascante et al., 2023).

The carbon footprint study of Reyes-Palomo et al., (2022) assessed biogenic carbon in soil and biomass comparing historical and current measurements on six organic and nine conventional cattle farms in the Spanish Dehesa agroforestry system. They calculated a carbon sequestration rate for soils and biomass, assuming a linear yearly change from the carbon stored at measurement time 1 to carbon stored at measurement time 2, which were on average 22 years apart for soils and 10 years for biomass. A historical reference was thus used, which was 10 to about 20 years in the past. Tree above and belowground biomass stocks were estimated using data from the national forest inventory, based on the parameters "volume over bark" (m3 ha−1), biomass expansion factor, and ratio of aboveground to belowground biomass (as dry matter). To calculate changes in carbon stocks, the carbon fraction contained in each tree species were used when available. The study found an average sequestration rate of carbon in soils of 0.91 t C·ha−1·year−1 and of 0.07 t C··ha−1·year−1 in trees. Including carbon sequestration reduced the net GHG emissions of cattle (in terms of CO_2-eq) by an average of 68% (95% in organic and 54% in conventional farms). In some cases, C sequestration surpassed the GHG emissions of cattle and resulted in a "negative carbon footprint" of calf meet. The total C sequestration strongly varied between farms, from 1.36 to 5.09 t CO_2eq ha−1·year−1. Because C sequestration per hectare was finally linked to meat production, a lower livestock density resulted in a higher sequestration per functional unit and thus in a lower carbon footprint. The authors did not assess the reversibility of carbon storage or non-linear dynamics of carbon sequestration (e.g. a levelling-off of carbon sequestration over time).

In the study of Crous-Duran et al. (2019) a new indicator "carbon balance" is proposed to compare the carbon emissions during food production with the carbon sequestered during food production on a specific area and time. The method is tested for wheat production in Portugal in monoculture and agroforestry systems. They applied the Yield-SAFE model combined with RothC model (Palma et al., 2018) to estimate carbon stored in biomass of trees and crop and in soils. Changes in SOC and carbon in biomass is modelled over 80 years, starting with the establishment of the agroforestry or monoculture system. A reference situation is not directly mentioned. Reversibility of carbon storage was not addressed in the article. The study showed that the production of wheat in the newly established agroforestry system showed a negative carbon balance (1 kg CO_2eq / kg wheat) while after 50 years of tree growth, the balance became positive, i.e. more carbon was stored in the biomass and soil compared to GHG emitted for food production.

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6. Assessing effects of short and long-lived greenhouse gases

6.1. Introduction to greenhouse gases

Substantial parts of the greenhouse gas emissions associated with livestock systems are in the form of methane (CH_4) and nitrous oxide (N_2O). Methane has a substantially higher heat-trapping ability than carbon dioxide (CO_2). However, the effect of methane diminishes over time as it breaks down to carbon dioxide in the atmosphere (after approximately 10 years). Hence, methane is classified as a short-lived climate pollutant (SLCP). Nitrous oxide has an even greater heat-trapping ability (radiative efficiency) than methane and also stays in the atmosphere much longer (around 110 years). Despite being less prevalent in the atmosphere than carbon dioxide or methane, its potency as a greenhouse gas makes it a substantial contributor to global warming.

In LCA and climate reporting it is usual to aggregate the climate impact of different gases into one common unit using one of several so-called climate *metrics*. Two common metrics are GWP (Global Warming Potential), which compares greenhouse gases in terms of the cumulative warming (radiative forcing) they cause over a given time period (for example 100 years), and GTP (Global Temperature change Potential), which compares greenhouse gases in terms of the temperature change they cause after a given time period (for example 100 years). This chapter is dedicated to exploring the implications of the use of different climate metrics in LCA of livestock systems.

In a first subsection, a closer look is taken at methane, its role in nature, and recent changes in its emission levels are discussed. Subsequently, attention is directed to various methodologies for measuring greenhouse gases with differing lifespans, including a relatively recently developed metric known as GWP*. The third section summarizes some of the literature on GWP*. The organization of this review is based on different perspectives regarding the GWP* metric, accompanied by the presentation of various case studies that have implemented this metric. The concluding section synthesises the presented knowledge, shows implications for LCA and proposes some recommendations on the use of climate metrics in LCA.

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6.2. A closer look at methane

6.2.1. What is methane?

In the realm of short-lived greenhouse gases, methane is most significantly relevant to agriculture. Methane is characterized by an atmospheric lifespan of approximately 9 to 15 years which is notably shorter than that of carbon dioxide. Also, it has a considerably higher potency, methane exhibits a global warming potential around 84-87 times greater than CO_2 over a 20-year span. The sources of methane are multifaceted, originating predominantly from anaerobic situations where organic matter undergoes decomposition in the absence of oxygen. These sources encompass both anthropogenic and natural origins. Anthropogenic sources include activities such as livestock husbandry, rice cultivation, waste management, and fossil fuel extraction. Meanwhile, natural sources comprise wetlands, wildfires, and geological processes.

Once released into the atmosphere, methane undergoes natural degradation through various processes. One primary mechanism involves oxidation by hydroxyl radicals (OH) present in the atmosphere. Hydroxyl radicals act as a natural "cleaner" by reacting with methane molecules, initiating a sequence of chemical transformations. This oxidation process results in the formation of water vapor and carbon dioxide, ultimately rendering methane less potent as a greenhouse gas over a century.

Recent studies show that the world is releasing more methane than the atmosphere can effectively break down (Saunois et al., 2020). Notably, the total amount of methane in the atmosphere is on the rise, and studies emphasize that the gap between emitted methane and its degradation is growing exponentially (Nisbet et al., 2019). Because of the higher short term warming potential, higher methane concentrations in the atmosphere accelerate global warming.

6.2.2. Methane from livestock

Methane from livestock is considered as anthropogenic, as livestock are kept by humans for food. Yet, methane is the result of a natural process related to microbial activities during the decomposition of feed in the rumen, known as enteric fermentation. The fiber content of the feed plays a crucial role in influencing methane emissions per cow. High-fiber diets, often associated with forage-rich feeds like grass, can lead to increased methane production in the rumen due to the metabolism of the involved microorganisms. The digestion of fibre-rich material results in acetate as a co-product, as well as hydrogen which is then converted to methane. A digestion of diets richer in starch rather results in propionate and less acetate and hydrogen and therefore less methane is produced.

from the fact that fibrous materials are more challenging for microbes to break down, necessitating prolonged fermentation processes in the rumen.

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In addition to enteric fermentation, methane emissions in livestock production arise from manure management practices, particularly when manure is stored with little or no oxygen. This occurs in anaerobic manure management systems, such as lagoons and pits, where organic matter undergoes decomposition in the absence of oxygen. These systems create favorable conditions for methanogenic microorganisms to thrive, leading to the production and release of methane as a by-product of the anaerobic digestion process.

Figure 6 : Methane emission trends from livestock over last 60 years by continent split by source (source : FAOstat)

Over the last 60-year, methane from livestock has been increasing across all continents except Europe, where livestock numbers have decreased and rations have become less fibre rich (Figure 6).

6.3. Accounting for greenhouse gases with different lifespan

6.3.1. The de facto standard for aggregating greenhouse gases

To date, the Global Warming Potential over 100 years (GWP100) is the most common metric in LCA. The GWP is a measure of how much heat a greenhouse gas traps in the atmosphere over a specific timeframe (commonly 100 years) compared to CO_2. Technically it is defined as the cumulative radiative forcing of the gas over that time horizon. The radiative forcing quantifies the change in energy balance in the Earth's atmosphere due to a particular gas. GWP thus takes into account both the heat trapping ability (radiative forcing) of the gas and its persistence in the atmosphere.

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An alternative metric is the Global Temperature change Potential (GTP). GTP measures the impact of gases based on how they affect global temperatures at a specific point in time (e.g. in 100 years in case of GTP100). Hence, GWP and GTP measure different aspects of climate change; GWP measures the trapped heat during a specific time period, while the GTP measures the temperature change at a certain future point in time (IPCC, 2022). The choice of metric can heavily influence results. Table 5 below shows the variation in characterization factors across metrics and timeframes (typically 20, 100, 500 years). These characterization factors are used to weight the different gases, before adding them together as aggregated carbon dioxide-equivalents (CO_2e).

Table 5. GWP and GTP characterization factors (IPCC, 2021b)

Gas	GWP-20	GWP-100	GWP-500	GTP-50	GTP-100
Carbon dioxide	1	1	1	1	1
Methane, fossil	82.5	29.8	10	13.2	7.5
Methane, non-fossil	79.7	27.0	7.2	10.4	4.7
Nitrous oxide	273	273	130	290	233

6.3.2. An alternative approach - GWP*

A few years back, the Global Warming Potential star (GWP*), was introduced as an alternative approach to account for gases with different lifespans. While GWP and GTP indicate the marginal global warming caused by emitting versus not emitting an emission pulse of GHGs (i.e. an emission of a certain amount of GHG at a given point of time), GWP* indicates the future warming or cooling of changes in GHG emission rates, compared to the present emission rate. An emission rate thereby refers to a certain amount of GHG emitted over a certain time in a certain region.

As an indicator of future warming or cooling in an emission scenario (i.e. a scenario of how GHG emissions rates will change over time), GWP* captures the dynamics of short-lived GHGs such as methane better than GWP. Roughly speaking, GWP* quantifies the climate impact compared to a hypothetical scenario where emission rates of SLCPs like methane would be held constant at the rates seen recently (20 years ago, is the timespan proposed by the creators of GWP*). That is, the calculation underpinning GWP* equates a *change* in the emission rate of methane as roughly equivalent to a large *pulse* emission or removal of CO_2 (Lynch, Garnett, et al., 2020). In the GWP* framework, a constant emission rate of an SLCP has roughly zero climate effect; a decreasing emission rate has a net cooling effect (shown as a negative GWP* value); and an increasing emission rate has a warming effect (shown as a positive GWP* value).

For GWP and GTP on the other hand, results always take positive values, reflecting the warming impact that an emission of a GHG gas always has, compared to if the emission had never occurred.

Several evolutions of the GWP* have been proposed, shown in

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Table 6. The initial alternative metrics has been proposed by Allen et al. (2018) with subsequent refinements from Cain et al. (2019) and Smith et al. (2021).

The initial GWP* approach proposed by Allen et al. (2018) relate changes in emission rates of SLCP to pulse emissions of carbon dioxide. As such it approximates the temperature implications of emission time series, providing insights into the warming potential of different pollutants over time.

A modification was proposed by Cain et al. (2019) to also account for the delayed warming response from SLCP. It incorporates the terms "r" and "s" to better capture this delayed effect and introduces the concept of 'warming equivalents'. *r* represents the weighting given to the impacts of changing the rate of SLCP emissions, and *s* the weighting given to the impacts of the current emissions rate. Here, methane is treated as a combination of 75% stock pollutant (s) and 25% flow pollutant (r). Lynch et al. (2020) proposed a simplified variant of this calculation, valid under most emission scenarios except near-constant emissions. These improvements aim to streamline calculations while retaining key features related to SLCP and their impact on global temperatures.

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Table 6. Metrics to report GHG and its improvements. E: emission of GHG, H: time horizon considered (in years); GWP: global warming potential; SLCP: short lived climate pollutant; eGWP*: equity criteria applied to GWP*

Metric / approach	**Equation**	**Description and main changes**
GWP100	$E_{CO2} = E_{SLCP} \times GWP_H$	Aggregates GHG emissions in relation to CO_2
GWP* (Allen et al., 2018)	$E_{CO2} = \frac{\Delta E_{SLCP}}{\Delta t} \times GWP_H \times H$	Introduction of GWP*. Relates emission rate changes of SLCP to emissions of CO_2. Approximates the temperature implications of emission time series
GWP* (Cain et al., 2019)	$E_{CO_2we} = \left[r \times \frac{\Delta E_{SLCP}}{\Delta t} \times H + (s \times E_{SLCP}) \right] \times GWP_H$	Incorporates the terms “r” and “s” to represent better the delayed warming response from SLCP, where r represents the weighting given to the impacts of changing the rate of SLCP emissions, and s the weighting given to the impacts of the current emissions rate. Treats methane as 75% stock (s) pollutant and 25% flow pollutant (r).
GWP* (Lynch, Cain, et al., 2020)	$E_{CO_2we\,(SLCP)} = [4 \times E_{SLCP(t)} - 3.75 \times (E_{SLCP\,(t-20)})] \times GWP_{100}$	Simplification of Cain et al. 2019 under most scenarios except near-constant emissions.
GWP* (Smith et al., 2021)	$E*(t) = \left[g \left(\frac{1-s)\, H \Delta E\,(t)}{\Delta t} \right) + gsE(t) \right]$	Incorporates the constant “g” for further consistency with linear models in metric calculations.
eGWP* (Rogelj & Schleussner, 2019)	See equations in reference paper: • eGWP*-CE: “Constant emissions” per capita • eGWP*-CW: “Constant warming” per capita • eGWP*-MW: “Minimal methane induced warming” per capita • eGWP*-ZR: « Zero Reference » per capita	Four equity concepts with different grades of “fairness”. Note: “While the choice of the accounting metric leads to differences in absolute terms, the general patterns are similar between all alternative eGWP* metrics”.

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Smith et al. (2021) proposed a modified version that incorporates the constant "g" to enhance consistency with linear models in metric calculations. This addition contributes to the reliability and coherence of the GWP*, ensuring a more accurate representation of the warming potential associated with different pollutants.

Additionally, modified versions of the metrics have been proposed to address certain limitations, as will be explored in the following section. Among these alternatives, the eGWP* has gained prominence, specifically addressing equity concerns associated with the application of GWP* when not used at global scale (Rogelj & Schleussner, 2019).

6.4. Critiques of the GWP*-approach

The GWP* approach has received considerable attention and has been especially welcomed by livestock organisations and businesses. The fact that with GWP* constant methane emissions only contributes little to future *additional* warming has been used by some to argue that emission reductions from livestock in stable or declining herds (which is the case in many European countries, see Figure 6) should not be prioritised. This reasoning however neglects the fact that constant methane emissions contribute to *maintained* warming. There has also been numerous discussions and debates in the scientific literature of the consequences of using GWP* for different applications. We shortly summarize these here.

Equity and fairness concerns

Rogelj & Schleussner (2019) argue that when GWP* is used for individual emitters (as opposed to at the global level), e.g., at the country level, countries with high past SLCP emissions are rewarded because reducing emissions from these high levels would give them the right to continue emitting similar amounts in the future or to receive credits for other GHGs. This raises concerns about equity and fairness, as developing countries with low historical SLCP emissions would be punished for increasing their emissions. Because GWP* calculations are often based on a country's past emissions levels, this is comparable to adopting the "grandfathering" principle. As an alternative, the authors propose approaches with varying degrees of fairness to allocating future emissions contributions. For instance, by assigning future emissions on a per-capita basis. Using these approaches substantially alters the outcomes compared to the GWP* metric calculated at the national level.

Perceived negative warming contributions

With GWP*, negative results are possible, e.g. if a country reduces methane emission from high levels, this would be reported as negative emissions in GWP*. This could be perceived as a "cooling" which is usually the case for negative emissions, e.g. by removing carbon dioxide from the atmosphere. However, in the case of methane is it rather the case that these negative emissions imply "less warming" compared to the current emissions rate (Cain et al., 2019; Rogelj & Schleussner, 2019). However, it is important to consider that an emission of methane will always result in warming compared to if the emission would not have occurred at all.

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Meinshausen & Nicholls (2022) claim that capturing such marginal effects of emissions should actually be one (of several) requirements of a 'metric':

"…a metric should…approximate the marginal climate effect of an emission action (climate effect of emitting one additional ton of a greenhouse gas, compared to a world in which that emission did not happen), so that a policy framework can appropriately reflect that externality."

Collins et al. (2020) on the other hand, claim that a metric, although traditionally applied to pulse emissions, can also be applied to step changes in emissions rate. Hence, there seems to be some conflicting views in the literature on what constitutes a 'metric'.

Additional concerns

Rojeli & Schleussner (2019) point out that the choice of a particular time horizon for measuring emissions has significant implications for the calculation of GWP*, and the arbitrary nature of this choice (i.e. 20 years) is a disadvantage for the application of these metrics (Rogelj & Schleussner, 2019).

Meinshausen & Nicholls (2022) highlight that when current emissions are accounted for in relation to historic emissions, as in some applications of GWP*, this metric fails to capture the contribution to warming as compared to a scenario without those emissions. That is, GWP* does not reflect the marginal warming of an emission. Rather, it considers the temperature contribution from past emissions (which decreases over time) along with the effect of future emissions, which is inappropriate for assessing the climate impact of a particular year or a specified period of time (Meinshausen & Nicholls, 2022).

Moreover, the compatibility with existing climate policy is contested. There are concerns that the new metric might be inconsistent with existing policy frameworks, and that this would lead to the need of setting new targets: "*GWP* would ask countries to start from scratch in terms of their political target setting processes: a bold ask to policy makers*" (Meinshausen & Nicholls, 2022). In contrast, the authors who proposed GWP* argue that in the case of SLCP, climate warming targets are compatible with stable emission rates because the atmospheric concentration does not accumulate if methane emission would be kept within earth's capacitiy to break down methane. For stock gases such as CO_2 and N_2O, stable emission rates in contrast do lead to increased atmospheric concentrations (Lynch et al., 2021).

6.4.1. Review of case studies

In this section, we present a short summary of case studies that have used GWP* to assess climate impacts of livestock systems (Table 7). In all the analysed papers, the data comes from countries or regions where the number of livestock has been reducing in the past years, that is Europe, USA and New Zeeland. Therefore, all demonstrate a reduced warming/no additional warming. Values calculated with GWP* are lower (sometimes below zero) compared to GWP100. Results would be opposite in countries where methane emissions from livestock have increased over the last decennia.

The improvement of farm efficiency is also mentioned as a factor that has led to reduced methane emissions and a potential lever to achieve climate neutrality when it is combined with

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decreases in livestock populations. In some of the papers, the “cooling”/ negative warming contribution narrative is present. However, it is not acknowledged that this cooling effect is only present because it builds on the reduction of warming that was already created before the study period. In the case of the carbon footprinting of beef and sheep done for New Zealand, the calculated climate effect from methane emissions using GWP* was negative, but the authors chose to represent it as zero. They highlight that there is not yet international agreement on how to report this relatively new approach.

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Table 7. Overview on case studies using GWP* as a metric to assess climate impacts

Source	Methodology	Conclusions
(Correddu et al., 2023) Aim: Recalculating the global warming impact of Italian livestock methane emissions with new metrics	**Metrics**: GWP100 compared to GWP* (Smith et al 2021) of CH_4 emissions CH_4 **emissions**: enteric fermentation + anaerobic digestion of manure. **Scope**: Livestock supply chains **Time period**: 2010 to 2020 **Region**: Italy	**Emission rates**: Decreasing CH_4 emission rates (for total livestock), with an average annual variation of -0.5%. **Climate impact:** The total cumulative contribution of Italian livestock production to global warming over the past 10 years, including the nitrous oxide (N_2O) emissions, has been negative (-48,759 kt of CO_2we) calculated with GWP* compared to the data calculated using the GWP100 method (+206,091 kt of CO_2e).
(Hörtenhuber et al., 2022) Aim: Calculate the effect of switching from a GWP100 to a GWP* metric for the product carbon footprints of milk, cattle and pig carcasses.	**Metrics**: GWP100 (IPCC 2014, CH_4+$N_2$0) vs GWP* (Smith et al 2021) **Emissions data:** CH_4 (direct, w/o upstream processes), $N_2$0 **Scope**: Livestock (dairy cattle, beef cattle and pig) and products (including manure mgmt., feed production, electricity use). **Time period**: 1990 to 2019 **Region**: Austria	**Emission rates:** Large reduction in emissions in dairy cattle and pigs, but not in other species. **Climate impact:** Reduction by 32% using GWP100 from 1990 to 2019 from the dairy sector, but low reduction from other animals and non-dairy cows. Climate impacts with GWP* have lower values than with GWP100: the total CO_2-we is 16% lower for the past 20 y.
(S. Liu et al., 2021) Aim: evaluate the actual effects of the CH_4 emission from U.S. dairy and beef production on temperature and initiate a rethinking of CH_4 associated with animal agriculture.	**Metrics**: GWP100 vs GWP* (Lynch et al 2020) of CH_4 **Emissions data:** USA: FAOSTAT, California: own calculations. **Scope**: US cattle (dairy and beef industry), California dairy cows **Time period**: 1981–2017 for USA, 1951-2017 California **Region**: Country (USA), region (California).	**Emission rates:** Decreasing herd population from 1961-2017 reflected in decreasing annual emissions, which peaked in 1975. **Climate impact:** Since 1990, the annual GWP* for US cattle has been decreasing. GWP100 shows an increasingly cumulative impact in which the reduction of herd number is not reflected. Conversely, GWP* shows decreasing (and negative) CO_2-we since 1990. *“Using GWP*, the projected climate impacts show that* CH_4 *emissions from the U.S. cattle industry have not contributed additional warming since 1986. Calculations show that the California dairy industry will approach climate neutrality in the next ten years if* CH_4 *emissions can be reduced by 1% per year, with the possibility to induce cooling if there are further reductions of emissions.”*

(del Prado et al., 2021) Aim: Investigate how European small ruminant dairy systems have contributed to global temperatures changes in an integrated manner and illustrate the contrasting climate impacts resulting from emissions of CH_4,CO_2 and N_2O using GWP and GWP* methodologies	**Metrics**: GWP100 (IPCC 2013) vs GWP* of N_2O, CO_2, CH_4 **Emissions data:** LCA-based **Scope**: Dairy sheep and goats **Time period**: 1990 to 2018 **Region**: Europe	**Emission rates:** Larger expansion of goat dairy systems compared to sheep. Reduction in total GHG emissions and a change in GHG emissions profile (data obtained by extrapolation): Apparent large reduction in emissions intensity per kg of milk from CH_4 and $N_2$0, but increase in CO_2 -derived from fossil fuels and land use change). Increased production efficiency. **Climate impact:** Warming estimations reflect that the CO_2-we metric results in lower climate impact than using CO_2-e. Using GWP*, European sheep and goat dairy sectors have not contributed to additional warming in the period 1990–2018 *("dairy goat production has led to some level of additional warming into the atmosphere, but these have been compensated by larger emission reductions in the dairy sheep sector")*.
(Pressman et al., 2023) Aim: to compare GWP and GWP* based climate impact of dairy systems in California.	**Metrics**: GWP100 (IPCC, 2007) vs GWP* (Lynch et al 2020) of CH_4 **Emissions data:** US EPA Greenhouse Gas Inventory and California Air Resources Board. **Scope**: US cattle (dairy industry), California dairy cows **Time period**: 1950-2017 **Region**: Country (USA), region (California).	**Emission rates:** Increasing methane emissions from livestock between 1950 and 2008, but decreasing methane emissions between 2008 and 2017. **Climate impact:** GWP* estimated higher CO_2 warming equivalent emissions than GWP during the time period where methane emissions were rising, but lower warming equivalent emissions during the period when methane emissions were decreasing.
(Mazzetto et al., 2023) Aim: Calculate the cradle-to-grave carbon footprint of beef and sheep meat in NZ. Compare different GWP metrics	**Metrics**: different versions of GWP100, GTP, GWP* **Emissions data:** **Scope**: NZ sheep and beef meat **Time period**: 2017-2018 **Region**: New Zealand	***Emission rates:** significant decrease from the methane component due to decreased animal numbers, particularly for sheep. For beef, reduction in traditional livestock numbers but increase in dairy cows. ***Climate impact:** -Sheep: With GWP*, very low contribution due to "zero contribution" from CH_4. -Beef: Larger carbon footprint than sheep with all metrics. Values from GWP* were smaller than those calculated with GWP100 but not with GTP100.

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6.5. When is GWP* useful and when not?

6.5.1. GWP* may be useful as a simple alternative to a dynamic climate model

The central argument behind the original GWP* proposal was that GWP does not adequately capture the very different roles of long-lived and short-lived GHGs in ambitious mitigation scenarios (Allen et al., 2018). Specifically, GWP* was designed to capture the fact that emissions of long-lived GHGs need to reach zero for long-term climate stabilization, whereas short-lived GHGs do not. Emissions of short-lived GHGs rather affect the available emission budget for long-lived GHGs under a given long-term temperature target. This original argument for GWP* has wide support in the research community.

GWP* is, especially after subsequent improvements (Lynch, Cain, et al., 2020) established as a simple but useful alternative to a dynamic climate model. For example, Meinshausen & Nicholls (2022) noted that "*when taken in aggregate and considered as a complete timeseries, GWP* emissions are a better predictor of global-mean temperature changes than GWP*". As a closing statement, they explicitly positioned GWP* as an alternative to more sophisticated climate models:

"Let us take GWP for what it is: A new class of 'micro climate models' (MCMs) that should be welcomed in the hierarchy of climate models. There are now GWP* and the combined global temperature change potential (CGTP) formulas* (Collins et al., 2020), *which open the door for educational tools and various applications, if quick temperature projections are required from time series of emissions."* (Meinshausen & Nicholls, 2022)

As a "micro climate model", GWP* can be used to evaluate the climate impact of different prospective emission scenarios. When considering emission pathways of different GHGs, GWP* will more accurately than GWP capture multi-decadal temperature change as it accounts for the difference between the cumulative warming of carbon dioxide and the non-cumulative warming of methane. For example, Clark et al. (2020) used GWP* to relate GHG emissions from the global food system under different mitigation scenarios to GHG emission budgets compatible with different temperature targets (1.5 and 2 degrees). In the study, the GWP* results were compared to what would be obtained using GWP, and this comparison demonstrates that results in methane-intensive scenarios can differ very substantially between GWP* and GWP.

However, GWP* is not a universal replacement for other climate models. It does not capture all climatic details that more sophisticated models do. Therefore, whether to use GWP* in place of a more complex model needs to be decided depending on the application (del Prado et al., 2023).

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6.5.2. Using GWP* to calculate the climate impact per unit product ("carbon footprint") requires zero-emission baseline

For some applications, it is necessary to estimate the climate impact per unit product (the "carbon footprint"), as an estimate of the climate impact caused by each unit of consumption. Examples of such applications include comparisons of the climate impact of different foods for purposes such as environmental labelling, environmental taxation (Moberg et al., 2021), or in other types of consumer communications (Karlsson Potter & Röös, 2021). This is commonly done using LCA methodology in which different GHGs from different life cycle stages (primary production, processing, transports etc.) are aggregated using a metric, typically GWP, into one carbon footprint.

A carbon footprint is typically intended as an estimate of the marginal climate impact of production/consumption of a product. That is, it is intended to answer the question: what is the climate impact of producing this additional unit of product (hence causing some GHG emissions) compared to not producing this additional unit (hence causing no GHG emissions)?

The GWP* metric, if used with a baseline of historical emissions (e.g., emissions from previous production from the same farm or production system) is not suitable for this application. A simple example clearly demonstrating this is that GWP* values can become negative if short-lived GHGs like methane have been declining in recent time. Clearly, this negative GWP* value would not mean that additional marginal consumption would have a cooling effect on the climate. It merely means that the decline in short-lived GHG emissions leads to cooling compared to if the short-lived GHG emissions had remained constant.

More generally, as Meinshausen & Nicholls (2022) put it, "*GWP* ... is not a 'neutral' metric as it weighs emissions differently depending on what the emission history of the country, project or facility has been.*" One of the results of using GWP* as an indicator per unit product would be that a product from the same production system, e.g., meat from a livestock farm, would be assigned different footprint values depending on whether the production system is growing or shrinking. As an example, if there were two identical farms, one recently growing and another one recently shrinking, the meat from the recently growing farm would have higher product footprint, even though the two farms, and their annual emissions and production volumes, were identical.

One way of potentially using GWP* for product-level assessments is to use it with a zero-emission rate as the reference scenario, based on the idea that zero emissions is what would result from non-consumption of the product. This type of application was explored by McAuliffe et al. (2023). They used GWP* to calculate the time-dependent CO_2 warming-equivalent impact per unit live weight production in a beef production system. The results presented, notably, are not one "footprint" per unit product, but rather a time-dependent "footprint" from 1 to 100 years after the production emissions take place. This is very roughly equivalent to calculating the time-dependent temperature change, i.e., calculating a curve consisting of GTP values with time horizons from 1 to 100 years. Indeed, McAuliffe et al. (2023) explicitly position their approach as an alternative to using multiple GTP values with different time horizons:

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"*GWP* provides a shorthand approximation of how these change over time, rather than having to calculate a full temporal evolution of GTP for every year* x." (McAuliffe et al., 2023, p. 9)

Such a calculation in terms of GTP, rather than GWP*, would be made using a climate model (e.g. Persson et al., 2015; Persson & Johansson, 2022).

The key observation made by McAuliffe et al. (2023) and others is that **no single metric number can fully account for the time-dependent differences in climate impact between products, and therefore a multidimensional "metric" is needed for this purpose**. The time-dependent CO_2 warming-equivalent impact as calculated using GWP* is one among several methods that can be used to this end.

6.5.3. What can GWP* bring that could not be delivered by accounting for the different GHGs separately?

Not all applications require the aggregation of different GHGs. Accounting for gases separately in emission pathways is the most transparent and completely avoids the use of metrics.

Much of current climate policy is based on the aggregating GHG emissions using GWP, but, e.g., GHG reduction targets can also be set for individual gases. When the IPCC talks about what is required to limit climate change, it differentiates between the different gases:

"From a physical science perspective, limiting human-induced global warming to a specific level requires limiting cumulative CO_2 *emissions, reaching at least net zero* CO_2 *emissions, along with strong reductions in other greenhouse gas emissions."* (IPCC, 2021a).

That is, the IPCC highlights the need for CO_2 emissions to reach net zero, while for other GHGs it will suffice with "strong reductions". Hence, an alternative to aggregating GHG with a metric, studies like Clark et al. (2020), or even product level LCA studies, could assess emissions of different gases separately and relate them to their individual targets. However, this could add complexity to the interpretation and communication of study results.

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6.6. Recommendations for evaluating climate impacts of GHGs in product-based LCA

6.6.1. Recommendation from other authors

Report gases separately

Several authors encourage to report climate impacts of separate GHG, at minimally separating them by short-term climate change and for long-term climate change (Allen et al., 2022; Jolliet et al., 2018; McAuliffe et al., 2023).

In the UNEP-SETAC Life Cycle Initiative global guidance (Jolliet et al., 2018), it is argued that no single measure can fully capture the contributions of different climate forcing agents to both shorter-term and the long-term temperature changes. For short-term climate change (over the next decades) GWP100 is suggested, while for long term climate change impacts GTP is. Furthermore, a sensitivity analysis (including GWP20) is strongly advised. In line with these suggestions, Allen et al., 2022 point out that "the separate specification of individual gases minimises ambiguity in determining the climate impact of past emissions".

A recent carbon footprint study (McAuliffe et al., 2023) concluded that the selection of impact assessment methods has major effects on the interpretation. As major recommendations from the study, they encourage LCA practitioners to perform sensitivity analysis by using different metrics and reporting SLCP and LLCP separately.

Use multiple metrics to capture differences between SLCPs and LLCPs

In addition to using the de facto standard metric GWP100, it is advisable to use climate models or metrics capturing the differences between SLCPs and LLCPs, or at least perform sensitivity analyses through other metrics and different time frames. This helps to test the robustness of results obtained using different metrics. This is especially relevant for agri-food systems, where SLCPs are a substantial part of the GHGs (Jolliet et al., 2018; Manzano et al., 2023; McAuliffe et al., 2023).

Several authors have demonstrated the value of using climate models, or tools derived from climate models, to assess time-dependent climate impacts. This can be done using a simple climate model (e.g., Persson et al., 2015; Persson & Johansson, 2022) or a tool such as GWP*.

FAO LEAP guidelines (FAO, 2023b) also recognize the difference between pulse-emission metrics (e.g. GWP100, GWP20, GTP100, GTP20) and Step-pulse metrics (e.g. GWP*, CGTP). Pulse-emission metrics provide information about future climate impacts of emission units, as opposed to the absence of those emissions, which are called the "marginal" impacts. Step-pulse metrics provide information about "additional" impacts relative to a specified date. The guideline suggests using either pulse-emission metrics or step-pulse depending on the question posed. They also suggest using multiple metrics to test whether results are consistent across different timescales or with respect to different impacts.

Report different metrics aligned to existing policy contexts

FAO LEAP guidelines also highlight that since metrics are used as tools by policy makers, it is important to consider them within the wider context of the Paris Agreement, definitions of climate neutrality, sustainable agriculture and equity considerations.

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6.6.2. General consideration on GWP

In formulating recommendations regarding metrics, it is imperative to consider nuanced perspectives on GWP. Firstly, it is incorrect to assert that GWP fails to capture dynamics; rather, it adeptly captures aggregated warming impacts over a given timeframe (commonly 100 years). It is crucial to communicate this accurately to avoid misconceptions. Secondly, GWP has a couple of clear advantages as a metric: its simplicity makes it easily understandable for a broad audience, rendering it an effective tool for conveying complex climate concepts; and the wide adoption of GWP100, always using the same reference, enables comparison across many scientific studies.

6.6.3. Recommendations for Re-livestock LCA assessments

No single metric can capture all the relevant differences in climate impact of different gases. Moreover, the science and the policy context around climate impacts is continuously evolving. Therefore, it is strongly recommended to report greenhouse gas emissions separately by gas to the extent possible, since this allows readers to assess the climate impact using any metric of choice and/or reassess in the future given new scientific advances (e.g., changed characterisation factors).

For some purposes, a useful simplification of the multi-gas reporting is the so called 2-basket approach, distinguishing SLCPs from LLCPs, as proposed by Allen et al. (2022).

To calculate the carbon footprint of a product, i.e., a one-dimensional indicator of the climate impact of marginal consumption/production per unit product, a metric is needed to summarize the contribution of different gases. Established metrics such as GWP and GTP with different time scales serve distinct purposes, yielding different results. It is therefore strongly recommended to use more than one metric, as a sensitivity analysis and to indicate the inherently time-dependent nature of climate impacts. This aligns with the perspective advocated by Jolliet et al. (2018) as well as del Prado et al. (2023), emphasizing the importance of using different metrics to showcase various effects. While GWP100 serves as a de facto standard metric, there is a compelling case for incorporating other metrics too. In essence, there is no argument against using GWP100, but a strong rationale exists for complementing it with alternative metrics, as endorsed by expert recommendations.

Using GWP* for climate footprints (as an indicator of marginal climate impact per unit product, etc.) is discouraged for two principal reasons. The first reason is that the choice of baseline has a large effect on results, and that this method detail has proven difficult and confusing for many people. This is most clearly demonstrated by examples of GWP* based “footprints” showing negative values, which may lead to the mistaken idea that marginal consumption would have a cooling effect on the climate. The second reason is that the only appropriate way to use GWP* for carbon footprints, namely with a zero-emission baseline (McAuliffe et al., 2023), has no advantage in principle to a time-dependent application of GTP, which is a more established and widely understood metric. By using GTP with a selection of fixed time horizons (e.g., the commonly applied GTP20, GTP50, and GTP100), the time-dependent climate impacts of different gases are well summarized in the well understood language of temperature

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units. In this context, the only clear advantage of GWP* (with a zero-emission baseline) is the relatively simple calculations involved.

For some use cases (e.g. when trade-offs between reducing SLCP or LLCP are expected), an explicitly dynamic approach is useful as an alternative or a complement to one-dimensional carbon footprints. A dynamic approach can help to clearly communicate the difference between SLCPs and LLCPs, possibly including trade-offs between short-term and long-term warming. In a dynamic approach, climate impact is not reduced to one number, but a time series of numbers indicating the climate impact, e.g., as global temperature change potential (GTP) or CO_2 warming-equivalents (GWP*). For such approaches, it is possible to use a simple climate model (e.g., Persson et al., 2015; Persson & Johansson, 2022) or a "micro climate model" (Meinshausen & Nicholls, 2022) with GWP* as a good example.

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7. Conclusions and recommendations

7.1. Functional units

The choice of the functional unit can have a strong impact on results and is thus highly sensitive. The functional units should be selected depending on the goal and scope of the study. In cases where the focus of an LCA is on comparing various products rather than on identifying environmental hot-spots within the value chain of a single product, it becomes imperative to reflect if the products offer only a single function or if the production systems provide different functions to different degrees.

If large differences in the nutritional values of different products exist (e.g. comparing different food products), this should either be reflected in the functional unit or otherwise captured in the analysis. Nutritional indices should encompass both qualifying (i.e. beneficial) as well as disqualifying (i.e. unhealthy) nutrients. Sensitivity in the results due to how a nutrient index is designed should be evaluated. When evaluating similar livestock products, the recommended functional units outlined by FAO LEAP (see Table 1), accounting for fat or protein content, are advised.

For multifunctional processes, the recommendation of ILCD should be followed, to give one functional unit or reference flow for each function. For example, if very different livestock systems (e.g. low and high intensity) are compared in an LCA study, which typically show very different levels of provisioning (e.g. meat) and non-provisioning ecosystem services (e.g. habitat creation and maintenance), several functional units should be chosen to present LCA results. In the simplest case, a functional unit per kg milk or meat should be complemented with a functional unit per ha. Expressing impacts per ha provides additional information on local environmental impacts, especially if primary impacts are concentrated within a specific location, such as the farm. This is most relevant for freshwater and terrestrial eutrophication, acidification, or ecotoxicity. This dual perspective provides additional insights into the potential adverse effects of intensification on the ecosystems surrounding the farm. If feasible, non-provisioning ecosystem services provided by the systems under comparison could be quantified and impacts calculated per unit of ecosystem services (e.g. by calculating the economic value of provisioning and non-provisioning services). Complementing LCA studies with other assessments, such as for example farm-level or territorial sustainability assessments, can provide additional information to decision-makers on potential trade-offs between different policy goals.

Within the Re-livestock project, comparisons could be done between a baseline and an innovation introduced to that baseline system, rather than comparing across different livestock systems. This will illustrate the environmental benefits of an innovation in a particular system and avoids comparing high and low intensity systems, which offer very different levels of provisioning and non-provisioning ecosystem services.

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7.2. Biogenic carbon in LCA

When accounting for biogenic carbon in LCA of agricultural systems, both changes in soil organic carbon (SOC) as well as in woody biomass can be relevant. Assessing carbon contained in the biomass of grass or annual crops is less relevant (or just reported as “biogenic carbon” without climate effect), because it is quickly released again.

Impacts related to land use change (e.g. from forest to crop land), are typically assessed in LCA, e.g. using the IPCC approach. Land use change is often relevant in background processes (e.g. for imported feed, when assessing impacts of milk production). For deforestation, the challenge is on how to allocate the ongoing deforestation in some world regions to agricultural production and single inventories of feed or food. Approaches on how to do this allocation are provided by ILCD or IPCC, typically using a 20 year time horizon. For traded commodities, Pendrill et al. (2019) provide data on how to allocate deforestation to crops.

For land management change (e.g. avoid ploughing or planting single trees on cropland), ILCD recommends to not account for changes in soil carbon or biomass as a default, but if specifically required, it can be assessed. Land management change is more often in the foreground system of LCA studies. Assessing changes in SOC and biomass due to land management change is challenging, because land management can change rather quickly, impacts can be very site dependent, and uncertainties in the assessments can strongly affect results in the foreground system.

Existing studies show that LCA results are highly sensitive to the inclusion of SOC. Many approaches have been proposed on how to assess SOC changes, but no consensus has been reached so far, on which approach should be chosen.

Several challenges exist to model changes in SOC or biomass in LCAs. First, a reference situation or baseline needs to be defined to make sure only carbon additionally added to systems by human management is accounted for. When assessing biogenic carbon in LCAs, a reference situation should be used, to avoid accounting for carbon which is “naturally” stored in systems, and would be stored in soils or biomass even without human intervention. This is especially relevant when carbon stock and not changes in stocks are assessed. This reference situation can either be the potential natural vegetation (e.g. a forest that would establish on unused grassland), a “no use” scenario under the current land use (e.g. unused grassland) or the land management preceding the current management (e.g. degraded grassland). The choice of reference situation strongly influences results. *The selection of the reference situation should be in line with the goal and scope of the study and should be explicitly stated.* Both the reference situation as well as the situation under study show temporal dynamics. *The assumptions on the dynamics of SOC (e.g. steady state assumption) for both the reference as well as the system under study should be made transparent and be in line with the goal and scope of the study.* For assessing SOC or carbon in biomass in LCA, the land use assessment framework (Koellner et al., 2013) can be useful to conceptualize temporal changes of SOC or carbon in biomass due to land use or management change (land transformation) as well as for maintaining a certain land use or management (land occupation).

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Storage of SOC shows strong temporal dynamics and changing stocks of carbon in soils as well as in biomass are mostly reversible. Long-term measurements have shown the strong temporal dynamics of soil carbon within and across years. If C inputs are temporarily reduced and climate extremes occur, parts of the C stored in soils can be quickly released and only parts of the carbon are stored more stable. As a conservative approach, only carbon stored over longer periods could be assessed. For example, if the *IPCC Tier 2 approach for assessing SOC is applied to LCA, only the "passive pool" should be accounted as long-term storage. Slow and active pools should to the extent possible be reported separately, as they are often released within short time.* This deviates from IPCC method for annual national GHG inventories, because of the different perspective compared to LCA. While the IPCC methodology is intended to provide transparency on the yearly contributions of countries and sectors to GHG emissions and their development, LCA takes a product and long-term perspective.

Although the general recommendation of ILCD is to not account for carbon stored less than 100 years as a default, a shorter-term perspective (e.g. 20 years) could still be relevant to elaborate the option-space of farmers or policy-makers. Therefore, a concept is needed to separately show the effect of temporal carbon storage on global warming. One option is to further test the approach of delayed emissions proposed by ILCD or to directly calculate the warming effect over a certain time period. Results of temporally stored carbon should be reported separately and the temporal dynamics assumed for the modelling (when is which gas emitted/stored) should be made explicit. For carbon in biomass stored for more than 100 years (e.g. agroforestry-systems with Spanish oak trees), carbon storage can be fully accounted for.

Measuring and modelling SOC changes is often uncertain, because it is highly dependent on the soil, climate, weather, management, land use history etc. Different models provide quite different results. In addition, future climates will most likely strongly influence the amount of carbon stored in soils. Therefore, carbon stored in soils or biomass should be reported separately from fossil carbon and results interpreted with caution. In addition, an uncertainty or sensitivity analysis should, to the extent possible, be performed on SOC changes.

In crop rotations of arable land or temporary grassland, the long-term storage or release of carbon needs to be allocated across different crops or even grazing livestock. How this allocation is performed (e.g. linearly, temporal, physical, economic allocation) should be explicitly mentioned.

Carbon stored in biomass should be reported separately from carbon in soils and from other carbon emissions (e.g. fossil fuels). The methods, time perspectives and uncertainties are different and to make results transparent, a separate reporting is highly recommended. When carbon storage in woody biomass is accounted for, then the carbon released during the use of products (e.g. burning of fuel wood) needs to be considered as well.

For the innovations assessed within the Re-Livestock project, which affect biogenic carbon in soil or woody biomass, changes in carbon stocks should be quantified following the recommendations above.

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7.3. Short and long-lived GHG

It is important to account for the different atmospheric lifetimes of GHGs in climate impact assessment of food systems and food products. This is more relevant for food products than many other products because some food products, most importantly ruminant livestock products, cause considerable emissions of methane, while many other food products mostly cause emissions of nitrous oxide and carbon dioxide.

It is strongly recommended to report greenhouse gas emissions separately by gas to the extent possible, since this allows readers to assess the climate impact using any metric of choice and/or reassess in the future given new scientific advances (e.g., changed characterisation factors). There is no single metric that captures all the relevant differences in climate impact of different gases. Moreover, the science and the policy context around climate impacts is continuously evolving. For some purposes, a useful simplification of the multi-gas reporting is the 2-basket approach, distinguishing SLCPs from LLCPs, as proposed by Allen et al. (2022).

To calculate the carbon footprint of a product, i.e., a one-dimensional indicator of the climate impact of marginal consumption/production per unit product, established metrics such as GWP and GTP are recommended. To capture relevant differences between long-term and short-term climate impacts, it is strongly recommended to use more than one metric (e.g., GWP and GTP) and/or metrics with different time horizons (e.g., GTP20, GTP50, and GTP100).

Using GWP* for climate footprints (per unit product, etc.) is discouraged for two reasons: (1) method details around baselines have proven difficult and confusing for many people; and (2) GWP* has no advantage in principle over a time-dependent application of GTP, which is a more established and widely understood metric. By using GTP with a selection of fixed time horizons (e.g., the commonly applied GTP20, GTP50, and GTP100), the time-dependent climate impacts of different gases are well summarized in the well understood language of temperature units.

For some use cases, an explicitly dynamic approach is useful as an alternative or a complement to one-dimensional carbon footprints. A dynamic approach can help to clearly communicate the difference between SLCPs and LLCPs, possibly including trade-offs between short-term and long-term warming. In a dynamic approach, climate impact is not reduced to one number, but a time series of numbers indicating the climate impact, e.g., as global temperature change (GTP) or CO_2 warming-equivalents (GWP*). For such approaches, it is possible to use a simple climate model (e.g., Persson et al., 2015; Persson & Johansson, 2022) or a "micro climate model" (Meinshausen & Nicholls, 2022) with GWP* as a good example.

Within the Re-Livestock project, climate impacts should, to the extent possible, be assessed with GWP and GTP and the different GHGs should in addition be reported separately.

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8. References

AGRIBALYSE. (2023). *AGRIBALYSE Database*. https://doc.agribalyse.fr/documentation-en/

Allen, M. R., Peters, G. P., Shine, K. P., Azar, C., Balcombe, P., Boucher, O., Cain, M., Ciais, P., Collins, W., Forster, P. M., Frame, D. J., Friedlingstein, P., Fyson, C., Gasser, T., Hare, B., Jenkins, S., Hamburg, S. P., Johansson, D. J. A., Lynch, J., … Tanaka, K. (2022). Indicate separate contributions of long-lived and short-lived greenhouse gases in emission targets. *Npj Climate and Atmospheric Science*, *5*(1), 5. https://doi.org/10.1038/s41612-021-00226-2

Allen, M. R., Shine, K. P., Fuglestvedt, J. S., Millar, R. J., Cain, M., Frame, D. J., & Macey, A. H. (2018). A solution to the misrepresentations of CO2-equivalent emissions of short-lived climate pollutants under ambitious mitigation. *Npj Climate and Atmospheric Science*, *1*(1), 1–8. https://doi.org/10.1038/s41612-018-0026-8

Andrén, O., & Kätterer, T. (1997). ICBM: The introductory carbon balance model for exploration of soil carbon balances. *Ecological Applications*, *7*(4), 1226–1236. https://doi.org/10.1890/1051-0761(1997)007[1226:ITICBM]2.0.CO;2

Avadí, A., Galland, V., Versini, A., & Bockstaller, C. (2022). Suitability of operational N direct field emissions models to represent contrasting agricultural situations in agricultural LCA: Review and prospectus. *Science of the Total Environment*, *802*, 149960. https://doi.org/10.1016/j.scitotenv.2021.149960

Barnes, A. , Bevan, K. , Moxey, A. , Grierson, S., & and Toma, L. ,. (2022). Greenhouse gas emissions from Scottish farming: an exploratory analysis of the Scottish Farm Business Survey and Agrecalc. *Scotland's Rural College*.

Basset-Mens, C., & Van Der Werf, H. M. G. (2005). Scenario-based environmental assessment of farming systems: The case of pig production in France. *Agriculture, Ecosystems and Environment*, *105*(1–2), 127–144. https://doi.org/10.1016/j.agee.2004.05.007

Baveye, P. C., Berthelin, J., Tessier, D., & Lemaire, G. (2023). Storage of soil carbon is not sequestration: Straightforward graphical visualization of their basic differences. *European Journal of Soil Science*, *74*(3), 1–8. https://doi.org/10.1111/ejss.13380

Bessou, C., Tailleur, A., Godard, C., Gac, A., de la Cour, J. L., Boissy, J., Mischler, P., Caldeira-Pires, A., & Benoist, A. (2020). Accounting for soil organic carbon role in land use contribution to climate change in agricultural LCA: which methods? Which impacts? *International Journal of Life Cycle Assessment*, *25*(7), 1217–1230. https://doi.org/10.1007/s11367-019-01713-8

Bianchi, M., Strid, A., Winkvist, A., Lindroos, A. K., Sonesson, U., & Hallström, E. (2020). Systematic evaluation of nutrition indicators for use within food LCA studies. *Sustainability (Switzerland)*, *12*(21), 1–18. https://doi.org/10.3390/su12218992

Bolinder, M. A., Menichetti, L., Meurer, K., Lundblad, M., & Kätterer, T. (2018). *New calibration of the ICBM model & analysis of soil organic carbon concentration from Swedish soil monitoring programs*. *20*, 1–23. www.smed.se

Boone, L., Roldán-Ruiz, I., Van linden, V., Muylle, H., & Dewulf, J. (2019). Environmental sustainability of conventional and organic farming: Accounting for ecosystem services in life cycle assessment. *Science of the Total Environment*, *695*, 133841. https://doi.org/10.1016/j.scitotenv.2019.133841

Bragaglio, A., Braghieri, A., Pacelli, C., & Napolitano, F. (2020). Environmental impacts of beef as corrected for the provision of ecosystem services. *Sustainability (Switzerland)*, *12*(9), 1–15. https://doi.org/10.3390/su12093828

Brandão, M., Levasseur, A., Kirschbaum, M. U. F., Weidema, B. P., Cowie, A. L., Jørgensen, S. V., Hauschild, M. Z., Pennington, D. W., & Chomkhamsri, K. (2013). Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. *International Journal of Life Cycle Assessment*, *18*(1), 230–240. https://doi.org/10.1007/s11367-012-0451-6

Bulle, C., Margni, M., Patouillard, L., Boulay, A. M., Bourgault, G., De Bruille, V., Cao, V., Hauschild, M., Henderson, A., Humbert, S., Kashef-Haghighi, S., Kounina, A., Laurent, A., Levasseur, A., Liard, G., Rosenbaum, R. K., Roy, P. O., Shaked, S., Fantke, P., & Jolliet, O. (2019). IMPACT World+: a globally regionalized life cycle impact assessment method. *International Journal of Life Cycle Assessment*, *24*(9), 1653–1674. https://doi.org/10.1007/s11367-019-01583-0

Bunce, R. G. H. (1968). Biomass and Production of Trees in a Mixed Deciduous Woodland: I. Girth and Height as Parameters for the Estimation of Tree Dry Weight. *The Journal of Ecology*, *56*(3), 759. https://doi.org/10.2307/2258105

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Funded by the European Union. Views and opinions expressed are however those of the author(s) only and do not necessarily reflect those of the European Union or European Commission. Neither the European Union nor the European Commission can be held responsible for them.

Burgess, P., Graves, A., de Jalón, S. G., Palma, J., Dupraz, C., & van Noordwijk, M. (2019). Modelling agroforestry systems. In M. R. Mosquera-Losada & R. Prabhu (Eds.), *Agroforestry for Sustainable Agriculture* (pp. 209–238). Burleigh Dodds Science Publishing. https://doi.org/10.19103/as.2018.0041.13
Cain, M., Lynch, J., Allen, M. R., Fuglestvedt, J. S., Frame, D. J., & Macey, A. H. (2019). Improved calculation of warming-equivalent emissions for short-lived climate pollutants. *Npj Climate and Atmospheric Science*, *2*(1), 29. https://doi.org/10.1038/s41612-019-0086-4
Cardinael, R., Umulisa, V., Toudert, A., Olivier, A., Bockel, L., & Bernoux, M. (2018). Revisiting IPCC Tier 1 coefficients for soil organic and biomass carbon storage in agroforestry systems. *Environmental Research Letters*, *13*(12). https://doi.org/10.1088/1748-9326/aaeb5f
Clark, M. A., Domingo, N. G. G., Colgan, K., Thakrar, S. K., Tilman, D., Lynch, J., Azevedo, I. L., & Hill, J. D. (2020). Global food system emissions could preclude achieving the 1.5° and 2°C climate change targets. *Science*, *370*(6517), 705–708. https://doi.org/10.1126/science.aba7357
Coleman, K., Jenkinson, D. S., Crocker, G. J., Grace, P. R., Klír, J., Körschens, M., Poulton, P. R., & Richter, D. D. (1997). Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. *Geoderma*, *81*(1–2), 29–44. https://doi.org/10.1016/S0016-7061(97)00079-7
Collins, W. J., Frame, D. J., Fuglestvedt, J. S., & Shine, K. P. (2020). Stable climate metrics for emissions of short and long-lived species-combining steps and pulses. *Environmental Research Letters*, *15*(2). https://doi.org/10.1088/1748-9326/ab6039
Correddu, F., Lunesu, M. F., Caratzu, M. F., & Pulina, G. (2023). Recalculating the global warming impact of italian livestock methane emissions with new metrics. *Italian Journal of Animal Science*, *22*(1), 125–135. https://doi.org/10.1080/1828051X.2023.2167616
Cotrufo, M. F., Wallenstein, M. D., Boot, C. M., Denef, K., & Paul, E. (2013). The Microbial Efficiency-Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: Do labile plant inputs form stable soil organic matter? *Global Change Biology*, *19*(4), 988–995. https://doi.org/10.1111/gcb.12113
Crous-Duran, J., Graves, A. R., Garcia-De-Jalón, S., Paulo, J. A., Tomé, M., & Palma, J. H. N. (2019). Assessing food sustainable intensification potential of agroforestry using a carbon balance method. *IForest*, *12*(1), 85–91. https://doi.org/10.3832/ifor2578-011
del Prado, A., Lynch, J., Liu, S., Ridoutt, B., Pardo, G., & Mitloehner, F. (2023). Animal board invited review: Opportunities and challenges in using GWP* to report the impact of ruminant livestock on global temperature change. *Animal*, *17*(5), 100790. https://doi.org/10.1016/j.animal.2023.100790
del Prado, A., Manzano, P., & Pardo, G. (2021). The role of the European small ruminant dairy sector in stabilising global temperatures: lessons from GWP* warming-equivalent emission metrics. *Journal of Dairy Research*, *88*(1), 8–15. https://doi.org/10.1017/S0022029921000157
De Vries, M. , Haan, M. H. , Hargreaves, P. R. , Verge, X. , Kadžienė, G. , Cieslak, A. , Galama, P., & and Vellinga, T. V. ,. (2022). Comparing outcomes of three GHG emission calculation tools applied on dairy production systems. *European Association for Animal Production Book of Abstracts*, *28*.
Dupraz, C., Wolz, K. J., Lecomte, I., Talbot, G., Vincent, G., Mulia, R., Bussière, F., Ozier-Lafontaine, H., Andrianarisoa, S., Jackson, N., Lawson, G., Dones, N., Sinoquet, H., Lusiana, B., Harja, D., Domenicano, S., Reyes, F., Gosme, M., & Van Noordwijk, M. (2019). Hi-sAFe: A 3D agroforestry model for integrating dynamic tree-crop interactions. *Sustainability (Switzerland)*, *11*(8). https://doi.org/10.3390/su11082293
EMEP/EEA. (2019). Crop production and agricultural soils. In *EMEP/EEA air pollutant emission inventory guidebook 2019*. https://doi.org/https://doi.org/10.2800/293657
European Commission-Joint Research Center. (2010). ILCD Handbook. General guide for Life Cycle Assessment - Detailed guidance. In *European Commission, JRC*. https://doi.org/10.2788/38479
Faist, M., Zah, R., & Reinhard, J. (2009). Sustainable Quick Check for Biofuels (SQCB): A Web-based tool for streamlined biofuels' LCA. *Environmental Informatics and Industrial Environmental Protection: Concepts, Methods and Tools*, *1*(Tc383), 297–303.
FAO. (2016a). *Environmental Performance of Large Ruminant Supply Chains. Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. FAO.
FAO. (2016b). *Greenhouse gas emissions and fossil energy use from poultry supply chains. Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. FAO.

Funded by the European Union

Funded by the European Union. Views and opinions expressed are however those of the author(s) only and do not necessarily reflect those of the European Union or European Commission. Neither the European Union nor the European Commission can be held responsible for them.

FAO. (2016c). *Greenhouse gas emissions and fossil energy use from small ruminant supply chains: Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. FAO.

FAO. (2018). *Environmental performance of pig supply chains. Guidelines for assessment*. Livestock Environmental Assessment and Performance Partnership. FAO. http://www.fao.org/3/a-bl094e.pdf

FAO. (2019). *Measuring and modelling soil carbon stocks and stock changes in livestock production systems Guidelines for assessment* (Version1 ed.).

FAO. (2020). *Environmental performance of feed additives in livestock supply chains*. Livestock Environmental Assessment and Performance Partnership. FAO.

FAO. (2023a). *Global assessment of soil carbon in grasslands. From current stock estimates to sequestration potential*. https://doi.org/10.4060/cc3981en

FAO. (2023b). *Methane emissions in livestock and rice systems*. Livestock Environmental Assessment and Performance Partnership. FAO.

Fazio, S., Zampori, L., De Schryver, A., Kusche, O., Diaconu, E., & Thellier, L. (2020). *Guide for EF compliant data sets. Joint Research Centre, European Commission*. https://doi.org/10.2760/537292

Feigenwinter, I., Hörtnagl, L., Zeeman, M. J., Eugster, W., Fuchs, K., Merbold, L., & Buchmann, N. (2023). Large inter-annual variation in carbon sink strength of a permanent grassland over 16 years: Impacts of management practices and climate. *Agricultural and Forest Meteorology*, *340*(March 2022). https://doi.org/10.1016/j.agrformet.2023.109613

Feliciano, D., Ledo, A., Hillier, J., & Nayak, D. R. (2018). Which agroforestry options give the greatest soil and above ground carbon benefits in different world regions? *Agriculture, Ecosystems & Environment*, *254*, 117–129. https://doi.org/10.1016/J.AGEE.2017.11.032

Franko, U., Crocker, G. J., Grace, P. R., Klír, J., Körschens, M., Poulton, P. R., & Richter, D. D. (1997). Simulating trends in soil organic carbon in long-term experiments using the CANDY model. *Geoderma*, *81*(1–2), 109–120. https://doi.org/10.1016/S0016-7061(97)00084-0

Goglio, P., Brankatschk, G., Knudsen, M. T., Williams, A. G., & Nemecek, T. (2018). Addressing crop interactions within cropping systems in LCA. *International Journal of Life Cycle Assessment*, *23*(9), 1735–1743. https://doi.org/10.1007/s11367-017-1393-9

Goglio, P., Smith, W. N., Grant, B. B., Desjardins, R. L., Gao, X., Hanis, K., Tenuta, M., Campbell, C. A., McConkey, B. G., Nemecek, T., Burgess, P. J., & Williams, A. G. (2018). A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. *Journal of Cleaner Production*, *172*, 4010–4017. https://doi.org/10.1016/J.JCLEPRO.2017.03.133

Goglio, P., Smith, W. N., Grant, B. B., Desjardins, R. L., McConkey, B. G., Campbell, C. A., & Nemecek, T. (2015). Accounting for soil carbon changes in agricultural life cycle assessment (LCA): A review. *Journal of Cleaner Production*, *104*, 23–39. https://doi.org/10.1016/j.jclepro.2015.05.040

Grace, P. R., Ladd, J. N., Robertson, G. P., Gage, S. H., & Kellogg, W. K. (2005). *SOCRATES-A simple model for predicting long-term changes in soil organic carbon in terrestrial ecosystems*. https://doi.org/10.1016/j.soilbio.2005.09.013

Grassauer, F., Herndl, M., Nemecek, T., Fritz, C., Guggenberger, T., Steinwidder, A., & Zollitsch, W. (2022). Assessing and improving eco-efficiency of multifunctional dairy farming: The need to address farms' diversity. *Journal of Cleaner Production*, *338*, 130627. https://doi.org/10.1016/J.JCLEPRO.2022.130627

Green, A., Nemecek, T., Chaudhary, A., & Mathys, A. (2020). Assessing nutritional, health, and environmental sustainability dimensions of agri-food production. *Global Food Security*, *26*(January), 100406. https://doi.org/10.1016/j.gfs.2020.100406

Hammar, T., Hansson, P. A., & Röös, E. (2022). Time-dependent climate impact of beef production – can carbon sequestration in soil offset enteric methane emissions? *Journal of Cleaner Production*, *331*. https://doi.org/10.1016/J.JCLEPRO.2021.129948

Hardaker, A., Styles, D., Williams, P., Chadwick, D., & Dandy, N. (2022). A framework for integrating ecosystem services as endpoint impacts in life cycle assessment. *Journal of Cleaner Production*, *370*(January), 133450. https://doi.org/10.1016/j.jclepro.2022.133450

Harris, I., Osborn, T. J., Jones, P., & Lister, D. (2020). Version 4 of the CRU TS monthly high-resolution gridded multivariate climate dataset. *Scientific Data*, *7*(1), 1–18. https://doi.org/10.1038/s41597-020-0453-3

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Funded by the European Union. Views and opinions expressed are however those of the author(s) only and do not necessarily reflect those of the European Union or European Commission. Neither the European Union nor the European Commission can be held responsible for them.

Henry, B. K., Butler, D., Wiedemann, S. G., Henry, B. K., Butler, D., & Wiedemann, S. G. (2015). Quantifying carbon sequestration on sheep grazing land in Australia for life cycle assessment studies. *The Rangeland Journal*, *37*(4), 379–388. https://doi.org/10.1071/RJ14109

Hergoualc'h, K. , Mueller, N. , Bernoux, M. , Kasimir, Ä. , Van Der Weerden, T. J., & and Ogle, S. M. ,. (2021). Improved accuracy and reduced uncertainty in greenhouse gas inventories by refining the IPCC emission factor for direct N2O emissions from nitrogen inputs to managed soils. *Global Change Biology*, *27*(24), 6536–6550.

Hörtenhuber, S. J., Seiringer, M., Theurl, M. C., Größbacher, V., Piringer, G., Kral, I., & Zollitsch, W. J. (2022). Implementing an appropriate metric for the assessment of greenhouse gas emissions from livestock production: A national case study. *Animal*, *16*(10), 100638. https://doi.org/10.1016/j.animal.2022.100638

Hughes, H. M. , McClelland, S. C. , Schipanski, M. E., & and Hillier, J. ,. (2023). Modelling the soil C impacts of cover crops in temperate regions. *Agricultural Systems*, *209*.

IDF. (2015). A common carbon footprint approach for the dairy sector. *Bulletin of the International Dairy Federation*, *479*, 70. https://doi.org/10.1016/s0958-6946(97)88755-9

IPCC. (2006). 2006 IPCC Guideline for National Greenhouse Gas Inventories. In S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.), *2006 IPCC Guideline for National Greenhouse Gas Inventories* (Vol. 4, Issue 1, pp. 1–14).

IPCC. (2019). 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. In E. Calvo Buendia, K. Tanabe, A. Kranjc, J. Baasansuren, M. Fukuda, S. Ngarize, A. Osako, Y. Pyrozhenko, P. Shermanau, & S. Federici (Eds.), *IPCC Guidelines for National Greenhouse Gas Inventories* (Vol. 4, p. 194). http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_04_Ch4_Forest_Land.pdf

IPCC. (2021a). Summary for Policymakers. In *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1–30). https://doi.org/10.1017/9781009157896.001.3

IPCC. (2021b). The Earth's Energy Budget, Climate Feedbacks and Climate Sensitivity. In *Climate Change 2021 – The Physical Science Basis* (pp. 923–1054). Cambridge University Press. https://doi.org/10.1017/9781009157896.009

IPCC. (2022a). Summary for Policymakers. In *Climate Change 2022: Impacts, Adaptation and Vulnerability*.

IPCC. (2022b). Summary for Policymakers. In *Climate Change 2022: Mitigation of Climate Change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (Issue 1). https://www.ipcc.ch/report/ar6/wg2/

ISO. (2006). *ISO 14040: Environmental management - Life cycle assessment - Principles and framework*. ISO.

Jenkins, T. A. R. et al. (2018). FC Woodland Carbon Code: Carbon Assessment Protocol (v2.0). *Forestry Commission*. https://www.woodlandcarboncode.org.uk/images/PDFs/WCC_CarbonAssessmentProtocol_V2.0_March2018.pdf

Joensuu, K., Rimhanen, K., Heusala, H., Saarinen, M., Usva, K., Leinonen, I., & Palosuo, T. (2021). Challenges in using soil carbon modelling in LCA of agricultural products—the devil is in the detail. *International Journal of Life Cycle Assessment*, *26*(9), 1764–1778. https://doi.org/10.1007/S11367-021-01967-1/FIGURES/7

Johnson, I. R., Chapman, D. F., Snow, V. O., Eckard, R. J., Parsons, A. J., Lambert, M. G., Cullen, B. R., Johnson, I. R., Chapman, D. F., Snow, V. O., Eckard, R. J., Parsons, A. J., Lambert, M. G., & Cullen, B. R. (2008). DairyMod and EcoMod: biophysical pasture-simulation models for Australia and New Zealand. *Australian Journal of Experimental Agriculture*, *48*(5), 621–631. https://doi.org/10.1071/EA07133

Jolliet, O., Antón, A., Boulay, A.-M., Cherubini, F., Fantke, P., Levasseur, A., McKone, T. E., Michelsen, O., Milà i Canals, L., Motoshita, M., Pfister, S., Verones, F., Vigon, B., & Frischknecht, R. (2018). Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use. *The International Journal of Life Cycle Assessment*, *23*(11), 2189–2207. https://doi.org/10.1007/s11367-018-1443-y

Jones, J. W., Hoogenboom, G., Porter, C. H., Boote, K. J., Batchelor, W. D., Hunt, L. A., Wilkens, P. W., Singh, U., Gijsman, A. J., & Ritchie, J. T. (2003). The DSSAT cropping system model.

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European Journal of Agronomy, *18*(3–4), 235–265. https://doi.org/10.1016/S1161-0301(02)00107-7
Karlsson Potter, H., & Röös, E. (2021). Multi-criteria evaluation of plant-based foods –use of environmental footprint and LCA data for consumer guidance. *Journal of Cleaner Production*, *280*. https://doi.org/10.1016/j.jclepro.2020.124721
Kätterer, T., Bolinder, M. A., Andrén, O., Kirchmann, H., & Menichetti, L. (2011). Roots contribute more to refractory soil organic matter than above-ground crop residues, as revealed by a long-term field experiment. *Agriculture, Ecosystems and Environment*, *141*(1–2), 184–192. https://doi.org/10.1016/j.agee.2011.02.029
Kay, S., Rega, C., Moreno, G., den Herder, M., Palma, J. H. N., Borek, R., Crous-Duran, J., Freese, D., Giannitsopoulos, M., Graves, A., Jäger, M., Lamersdorf, N., Memedemin, D., Mosquera-Losada, R., Pantera, A., Paracchini, M. L., Paris, P., Roces-Díaz, J. V., Rolo, V., … Herzog, F. (2019). Agroforestry creates carbon sinks whilst enhancing the environment in agricultural landscapes in Europe. *Land Use Policy*, *83*, 581–593. https://doi.org/10.1016/J.LANDUSEPOL.2019.02.025
Kendall, A. (2012). Time-adjusted global warming potentials for LCA and carbon footprints. *International Journal of Life Cycle Assessment*, *17*(8), 1042–1049. https://doi.org/10.1007/s11367-012-0436-5
Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., Canals, L. M., Saad, R., Souza, D. M., & Müller-Wenk, R. (2013). UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *International Journal of Life Cycle Assessment*, *18*(6), 1188–1202. https://doi.org/10.1007/s11367-013-0579-z
Kröbel, R., Bolinder, M. A., Janzen, H. H., Little, S. M., Vandenbygaart, A. J., & Kätterer, T. (2016). Canadian farm-level soil carbon change assessment by merging the greenhouse gas model Holos with the Introductory Carbon Balance Model (ICBM). *Agricultural Systems*, *143*, 76–85. https://doi.org/10.1016/J.AGSY.2015.12.010
Leon, A., & Ishihara, K. N. (2018). Assessment of new functional units for agrivoltaic systems. *Journal of Environmental Management*, *226*(June), 493–498. https://doi.org/10.1016/j.jenvman.2018.08.013
Letelier, P., Aguirre-Villegas, H. A., Navarro, M. C., & Wattiaux, M. A. (2022). Milk, meat, and human edible protein from dual-purpose cattle in Costa Rica: Impact of functional unit and co-product handling methods on predicted enteric methane allocation. *Livestock Science*, *263*(May). https://doi.org/10.1016/j.livsci.2022.105013
Levasseur, A., Lesage, P., Margni, M., Brandão, M., & Samson, R. (2012). Assessing temporary carbon sequestration and storage projects through land use, land-use change and forestry: Comparison of dynamic life cycle assessment with ton-year approaches. *Climatic Change*, *115*(3–4), 759–776. https://doi.org/10.1007/S10584-012-0473-X/TABLES/2
Li, C. (1996). The DNDC Model. In D. S. Powlson, P. Smith, & J. U. Smith (Eds.), *Evaluation of Soil Organic Matter Models* (pp. 263–267). Springer-Verlag.
Liski, J., Palosuo, T., Peltoniemi, M., & Sievänen, R. (2005). Carbon and decomposition model Yasso for forest soils. *Ecological Modelling*, *189*(1–2), 168–182. https://doi.org/10.1016/j.ecolmodel.2005.03.005
Little, S. M., Benchaar, C., Janzen, H. H., Kröbel, R., McGeough, E. J., & Beauchemin, K. A. (2017). Demonstrating the effect of forage source on the carbon footprint of a Canadian dairy farm using whole-systems analysis and the holos model: Alfalfa silage vs. corn silage. *Climate*, *5*(4). https://doi.org/10.3390/cli5040087
Liu, L., Sayer, E. J., Deng, M., Li, P., Liu, W., Wang, X., Yang, S., Huang, J., Luo, J., Su, Y., Grünzweig, J. M., Jiang, L., Hu, S., & Piao, S. (2023). The grassland carbon cycle: Mechanisms, responses to global changes, and potential contribution to carbon neutrality. *Fundamental Research*, *3*(2), 209–218. https://doi.org/10.1016/j.fmre.2022.09.028
Liu, S., Proudman, J., & Mitloehner, F. M. (2021). Rethinking methane from animal agriculture. *CABI Agriculture and Bioscience*, *2*(1), 22. https://doi.org/10.1186/s43170-021-00041-y
Lynch, J., Cain, M., Frame, D., & Pierrehumbert, R. (2021). Agriculture's Contribution to Climate Change and Role in Mitigation Is Distinct From Predominantly Fossil CO2-Emitting Sectors. *Frontiers in Sustainable Food Systems*, *4*. https://doi.org/10.3389/fsufs.2020.518039
Lynch, J., Cain, M., Pierrehumbert, R., & Allen, M. (2020). Demonstrating GWP*: a means of reporting warming-equivalent emissions that captures the contrasting impacts of short- and long-lived

Funded by the European Union

climate pollutants. *Environmental Research Letters*, *15*(4), 044023. https://doi.org/10.1088/1748-9326/ab6d7e
Lynch, J., Garnett, T., Persson, M., Röös, E., & Reisinger, A. (2020). Methane and the sustainability of ruminant livestock. *Food Climate Research Network*, 20. https://tabledebates.org/building-blocks/methane-and-sustainability-ruminant-livestock#MSBB2
Manzano, P., Rowntree, J., Thompson, L., del Prado, A., Ederer, P., Windisch, W., & Lee, M. (2023). Challenges for the balanced attribution of livestock's environmental impacts: the art of conveying simple messages around complex realities. *Animal Frontiers*. https://doi.org/10.1093/af/vfac096
Mathivanan, G. P., Eysholdt, M., Zinnbauer, M., Rösemann, C., & Fuß, R. (2021). New N2O emission factors for crop residues and fertiliser inputs to agricultural soils in Germany. *Agriculture, Ecosystems and Environment*, *322*. https://doi.org/10.1016/j.agee.2021.107640
Matthews, H. D., Zickfeld, K., Koch, A., & Luers, A. (2023). Accounting for the climate benefit of temporary carbon storage in nature. *Nature Communications*, *14*(1), 1–10. https://doi.org/10.1038/s41467-023-41242-5
Mazzetto, A. M., Falconer, S., & Ledgard, S. (2023). Carbon footprint of New Zealand beef and sheep meat exported to different markets. *Environmental Impact Assessment Review*, *98*, 106946. https://doi.org/10.1016/j.eiar.2022.106946
Mbow, C., Rosenzweig, C., Barioni, L. G., Benton, T. G., Herrero, M., Krishnapillai, M., Liwenga, E., Pradhan, P., Rivera-Ferre, M. G., Sapkota, T., Tubiello, F. N., & Xu, Y. (2019). Food security. In P. R. Shukla, J. Skea, E. C. Buendia, V. Masson-Delmotte, H.-O. Pörtner, D. C. Roberts, P. Zhai, R. Slade, S. Connors, R. van Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. P. Pereira, P. Vyas, . Huntley, … J. Malley (Eds.), *Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems*. http://dx.doi.org/10.1038/544S5a
McAuliffe, G. A., Lynch, J., Cain, M., Buckingham, S., Rees, R. M., Collins, A. L., Allen, M., Pierrehumbert, R., Lee, M. R. F., & Takahashi, T. (2023). Are single global warming potential impact assessments adequate for carbon footprints of agri-food systems? *Environmental Research Letters*, *18*(8), 084014. https://doi.org/10.1088/1748-9326/ace204
McCown, R. L., Hammer, G. L., Hargreaves, J. N. G., Holzworth, D. P., & Freebairn, D. M. (1996). APSIM: a novel software system for model development, model testing and simulation in agricultural systems research. *Agricultural Systems*, *50*(3), 255–271. https://doi.org/10.1016/0308-521X(94)00055-V
Meier, M. S., & Moakes, S. (2019). *Swiss animal production adapted to local ecosystem boundaries: Production potential and eco-efficiency within different bio-geographic regions in Switzerland.*
Meinshausen, M., Homayounfar, Z., Turner, A., & Nicholls, Z. (2022). *GWP * is a model , not a metric OPEN ACCESS*.
Meinshausen, M., & Nicholls, Z. (2022). GWP * is a model , not a metric. *Environmental Research Letters*, *17*(041002), 1–6.
Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R. F., Gaillard, G., Michelsen, O., Müller-Wenk, R., & Rydgren, B. (2007). Key elements in a framework for land use impact assessment within LCA. *International Journal of Life Cycle Assessment*, *12*(1), 2–4. https://doi.org/10.1065/lca2006.12.296
Moberg, E., Säll, S., Hansson, P. A., & Röös, E. (2021). Taxing food consumption to reduce environmental impacts – Identification of synergies and goal conflicts. *Food Policy*, *101*(July 2020). https://doi.org/10.1016/j.foodpol.2021.102090
Moberg, E., Walker Andersson, M., Säll, S., Hansson, P.-A., & Röös, E. (2019). Determining the climate impact of food for use in a climate tax—design of a consistent and transparent model. *The International Journal of Life Cycle Assessment*, *24*(9), 1715–1728. https://doi.org/10.1007/s11367-019-01597-8
Müller-Wenk, R., & Brandão, M. (2010). Climatic impact of land use in LCA-carbon transfers between vegetation/soil and air. *International Journal of Life Cycle Assessment*, *15*(2), 172–182. https://doi.org/10.1007/S11367-009-0144-Y/TABLES/9
Nair, R. P. K., Nair, V. D., Kumar, B. M., & Showalter, J. M. (2010). Carbon Sequestration in Agroforestry Systems. In *Agroforestry Systems. Advances in Agronomy.* (Vol. 108, pp. 237–307). https://doi.org/10.1016/S0065-2113(10)08005-3
Nisbet, E. G., Manning, M. R., Dlugokencky, E. J., Fisher, R. E., Lowry, D., Michel, S. E., Myhre, C. L., Platt, S. M., Allen, G., Bousquet, P., Brownlow, R., Cain, M., France, J. L., Hermansen, O.,

Funded by the European Union

Hossaini, R., Jones, A. E., Levin, I., Manning, A. C., Myhre, G., … White, J. W. C. (2019). Very Strong Atmospheric Methane Growth in the 4 Years 2014–2017: Implications for the Paris Agreement. *Global Biogeochemical Cycles*, *33*(3), 318–342. https://doi.org/10.1029/2018GB006009

Ogle, S. M., Swan, A., & Paustian, K. (2012). No-till management impacts on crop productivity, carbon input and soil carbon sequestration. *Agriculture, Ecosystems & Environment*, *149*, 37–49.

Palma, J. H. N. (2017). Clipick – climate change web picker. A tool bridging daily climate needs in process based modelling in forestry and agriculture. *Forest Systems*, *26*(1), 1–4. https://doi.org/10.5424/fs/2017261-10251

Palma, J. H. N., Crous-Duran, J., Graves, A. R., de Jalon, S. G., Upson, M., Oliveira, T. S., Paulo, J. A., Ferreiro-Domínguez, N., Moreno, G., & Burgess, P. J. (2018). Integrating belowground carbon dynamics into Yield-SAFE, a parameter sparse agroforestry model. *Agroforestry Systems*, *92*(4), 1047–1057. https://doi.org/10.1007/s10457-017-0123-4

Palma, J. H. N., Oliveira, T., Crous-Duran, J., Graves, A. R., Garcia de Jalon, S., Upson, M., Giannitsopoulos, M., Burgess, P. J., Tomé, M., Ferreiro-Dominguez, N., Mosquera-Losada, M. R., Gonzalez-Hernandez, P., Kay, S., Mirk, J., Smith, J., Moreno, G., Pantera, A., Mantovani, D., Rosati, A., … Hermansen, J. (2017). *Modelled agroforestry outputs at field and farm scale to support biophysical and environmental assessments*. https://www.repository.utl.pt/handle/10400.5/14799

Parton, W. J. (1996). The CENTURY model. In J. U. Powlson, D.S., Smith, P. & Smith (Ed.), *Evaluation of soil organic matter models, NATO ASI Series (Series I: Global Environmental Change)* (pp. 283–291).

Parton, W. J., Schimel, D. S., Cole, C. V., & Ojima, D. S. (1987). Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal*, *51*(5), 1173–1179.

Pendrill, F., Persson, U. M., Godar, J., & Kastner, T. (2019). Deforestation displaced: Trade in forest-risk commodities and the prospects for a global forest transition. *Environmental Research Letters*, *14*(5). https://doi.org/10.1088/1748-9326/ab0d41

Persson, U. M., & Johansson, D. J. A. (2022). *Simple climate model*. https://zenodo.org/records/5957222

Persson, U. M., Johansson, D. J. A., Cederberg, C., Hedenus, F., & Bryngelsson, D. (2015). Climate metrics and the carbon footprint of livestock products: Where's the beef? *Environmental Research Letters*, *10*(3). https://doi.org/10.1088/1748-9326/10/3/034005

Petersen, B. M., Knudsen, M. T., Hermansen, J. E., & Halberg, N. (2013). An approach to include soil carbon changes in life cycle assessments. *Journal of Cleaner Production*, *52*, 217–224. https://doi.org/10.1016/J.JCLEPRO.2013.03.007

Poeplau, C., Aronsson, H., Myrbeck, Å., & Kätterer, T. (2015). Effect of perennial ryegrass cover crop on soil organic carbon stocks in southern Sweden. *Geoderma Regional*, *4*, 126–133. https://doi.org/10.1016/j.geodrs.2015.01.004

Poeplau, C., Kätterer, T., Bolinder, M. A., Börjesson, G., Berti, A., & Lugato, E. (2015). Low stabilization of aboveground crop residue carbon in sandy soils of Swedish long-term experiments. *Geoderma*, *237*, 246–255. https://doi.org/10.1016/j.geoderma.2014.09.010

Poore, J., & Nemecek, T. (2018). Reducing food's environmental impacts through producers and consumers. *Science*, *360*, 987–992. https://doi.org/10.1126/science.aaq0216

Pressman, E. M., Liu, S., & Mitloehner, F. M. (2023). Methane emissions from California dairies estimated using novel climate metric Global Warming Potential Star show improved agreement with modeled warming dynamics. *Frontiers in Sustainable Food Systems*, *6*. https://doi.org/10.3389/fsufs.2022.1072805

Quevedo-Cascante, M., Mogensen, L., Kongsted, A. G., & Knudsen, M. T. (2023). How does Life Cycle Assessment capture the environmental impacts of agroforestry? A systematic review. *Science of the Total Environment*, *890*(March). https://doi.org/10.1016/j.scitotenv.2023.164094

Reyes-Palomo, C., Aguilera, E., Llorente, M., Díaz-gaona, C., Moreno, G., & Rodríguez-est, V. (2022). Carbon sequestration offsets a large share of GHG emissions in dehesa cattle production. *Journal of Cleaner Production*, *358*(131918). https://doi.org/10.1016/j.jclepro.2022.131918

Ridoutt, B. (2021). Bringing nutrition and life cycle assessment together (nutritional LCA): opportunities and risks. *International Journal of Life Cycle Assessment*, *26*(10), 1932–1936. https://doi.org/10.1007/s11367-021-01982-2

Funded by the European Union

Riedo, M., Grub, A., Rosset, M., & Fuhrer, J. (1998). A pasture simulation model for dry matter production, and fluxes of carbon, nitrogen, water and energy. *Ecological Modelling*, *105*(2–3), 141–183. https://doi.org/10.1016/S0304-3800(97)00110-5

Ripoll-Bosch, R., de Boer, I. J. M., Bernués, A., & Vellinga, T. V. (2013). Accounting for multi-functionality of sheep farming in the carbon footprint of lamb: A comparison of three contrasting Mediterranean systems. *Agricultural Systems*, *116*, 60–68. https://doi.org/10.1016/j.agsy.2012.11.002

Rodrigues, L. , Hardy, B. , Huyghebeart, B. , Fohrafellner, J. , Fornara, D. , Barančíková, G. , Bárcena, T. G. , De Boever, M. , Di Bene, C. , Feizienė, D., & and Kaetterer, T. ,. (2021). Achievable agricultural soil carbon sequestration across Europe from country-specific estimates. *Global Change Biology*, *27*(24), 6363–6380.

Rogelj, J., & Schleussner, C.-F. (2019). Unintentional unfairness when applying new greenhouse gas emissions metrics at country level. *Environmental Research Letters*, *14*(11), 114039. https://doi.org/10.1088/1748-9326/ab4928

Ross, S. A., Topp, C. F. E., Ennos, R. A., & Chagunda, M. G. G. (2017). Relative emissions intensity of dairy production systems: Employing different functional units in life-cycle assessment. *Animal*, *11*(8), 1381–1388. https://doi.org/10.1017/S1751731117000052

Saarinen, M., Fogelholm, M., Tahvonen, R., & Kurppa, S. (2017). Taking nutrition into account within the life cycle assessment of food products. *Journal of Cleaner Production*, *149*, 828–844. https://doi.org/10.1016/j.jclepro.2017.02.062

Salou, T., Le Mouël, C., & van der Werf, H. M. G. (2017). Environmental impacts of dairy system intensification: the functional unit matters! *Journal of Cleaner Production*, *140*, 445–454. https://doi.org/10.1016/J.JCLEPRO.2016.05.019

Saunois, M., Stavert, A. R., Poulter, B., Bousquet, P., Canadell, J. G., Jackson, R. B., Raymond, P. A., Dlugokencky, E. J., Houweling, S., Patra, P. K., Ciais, P., Arora, V. K., Bastviken, D., Bergamaschi, P., Blake, D. R., Al, E., & Zhuang, Q. (2020). The global methane budget 2000-2007. *Earth Systems Science Data*, *12*(3), 1561–1623.

Schader, C., Jud, K., Meier, M. S., Kuhn, T., Oehen, B., & Gattinger, A. (2014). Quantification of the effectiveness of greenhouse gas mitigation measures in Swiss organic milk production using a life cycle assessment approach. *Journal of Cleaner Production*, *73*, 227–235. https://doi.org/10.1016/j.jclepro.2013.11.077

Smith, M. A., Cain, M., & Allen, M. R. (2021). Further improvement of warming-equivalent emissions calculation. *Npj Climate and Atmospheric Science*, *4*(1), 19. https://doi.org/10.1038/s41612-021-00169-8

Soil Association, Woodland Trust, F. E. F., Natural Environment Investment Readiness, & Forestry, S. (n.d.). *Investigating the feasibility for an agroforestry carbon code – NEIRF Phase 2 – Final Report and recommendations*.

Sonesson, U., Davis, J., Flysjö, A., Gustavsson, J., & Witthöft, C. (2017). Protein quality as functional unit – A methodological framework for inclusion in life cycle assessment of food. *Journal of Cleaner Production*, *140*, 470–478. https://doi.org/10.1016/J.JCLEPRO.2016.06.115

Stockmann, U., Adams, M. A., Crawford, J. W., Field, D. J., Henakaarchchi, N., Jenkins, M., Minasny, B., McBratney, A. B., Courcelles, V. de R. de, Singh, K., Wheeler, I., Abbott, L., Angers, D. A., Baldock, J., Bird, M., Brookes, P. C., Chenu, C., Jastrow, J. D., Lal, R., … Zimmermann, M. (2013). The knowns, known unknowns and unknowns of sequestration of soil organic carbon. *Agriculture, Ecosystems and Environment*, *164*(2013), 80–99. https://doi.org/10.1016/j.agee.2012.10.001

Sukhoveeva, O. E. (2021). Carbon calculators as a tool for assessing greenhouse gas emissions from livestock. *Doklady Earth Sciences (, Pp.). Pleiades Publishing.* , *Vol. 497*, 266–271.

Svendsen, H., Hansen, S., & Jensen, H. E. (1995). Simulation of crop production, water and nitrogen balances in two German agro-ecosystems using the DAISY model. *Ecological Modelling*, *81*(1–3), 197–212. https://doi.org/10.1016/0304-3800(94)00171-D

Taghizadeh-Toosi, A., Christensen, B. T., Hutchings, N. J., Vejlin, J., Kätterer, T., Glendining, M., & Olesen, J. E. (2014). C-TOOL: A simple model for simulating whole-profile carbon storage in temperate agricultural soils. *Ecological Modelling*, *292*, 11–25. https://doi.org/10.1016/j.ecolmodel.2014.08.016

Tessari, P., Lante, A., & Mosca, G. (2016). Essential amino acids: Master regulators of nutrition and environmental footprint? *Scientific Reports*, *6*(December 2015), 1–13. https://doi.org/10.1038/srep26074

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Thiagarajan, A. , Liang, C. , MacDonald, J. D. , Smith, W. , VandenBygaart, A. J. , Grant, B. , K. R. , Janzen, H. , Zhang, T. , McConkey, B., & and Ma, B. ,. (2022). Prospects and challenges in the use of models to estimate the influence of crop residue input on soil organic carbon in long-term experiments in Canada. *Geoderma Regional*, *30*.

Topp, C. F. E. (2017). Tools to support farmer decision–making in arable cropping systems. *FACCE MACSUR Reports*, *10*.

Topp, C. F. E., Reckling, M., Hanegraaf, M., Bachinger, J., Walker, R. L., Buckingham, S., Sykes, A. J., & Watson, A. (2017). Farmer friendly tools–how can they help support decision making under a changing climate. *Asp. Appl. Biol, 136*, 1–6.

van der Werf, W., Keesman, K., Burgess, P., Graves, A., Pilbeam, D., Incoll, L. D., Metselaar, K., Mayus, M., Stappers, R., van Keulen, H., Palma, J., & Dupraz, C. (2007). Yield-SAFE: A parameter-sparse, process-based dynamic model for predicting resource capture, growth, and production in agroforestry systems. *Ecological Engineering*, *29*(4), 419–433. https://doi.org/10.1016/j.ecoleng.2006.09.017

van Genuchten, M. Th. (1980). A Closed-form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils. *Soil Science Society of America Journal*, *44*(5), 892–898. https://doi.org/10.2136/sssaj1980.03615995004400050002x

von Greyerz, K., Tidåker, P., Karlsson, J. O., & Röös, E. (2023). A large share of climate impacts of beef and dairy can be attributed to ecosystem services other than food production. *Journal of Environmental Management*, *325*(October 2022). https://doi.org/10.1016/j.jenvman.2022.116400

Wang, Y., de Boer, I. J. M., Persson, U. M., Ripoll-Bosch, R., Cederberg, C., Gerber, P. J., Smith, P., & van Middelaar, C. E. (2023). Risk to rely on soil carbon sequestration to offset global ruminant emissions. *Nature Communications*, *14*(1), 1–9. https://doi.org/10.1038/s41467-023-43452-3

Weidema, B. P., & Stylianou, K. S. (2020). *SUSTAINABLE FOOD PRODUCTION AND CONSUMPTION Nutrition in the life cycle assessment of foods-function or impact?* 1210–1216. https://doi.org/10.1007/s11367-019-01658-y

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): overview and methodology. *International Journal of Life Cycle Assessment*, *21*(9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Williams, J. R., Jones, C. A., & Dyke, P. T. (1984). A modeling approach to determining the relationship between erosion and soil productivity. *Transactions of the ASAE*, *27*(1), 129–144.

Wösten, J. H. M., Lilly, A., Nemes, A., & Le Bas, C. (1999). Development and use of a database of hydraulic properties of European soils. *Geoderma*, *90*(3–4), 169–185. https://doi.org/10.1016/S0016-7061(98)00132-3

Zhang, Y., King, A. E., Hamilton, E., & Cotrufo, M. F. (2023). Representing cropping systems with the MEMS 2.0 ecosystem model. *AgriRxiv*. https://doi.org/10.31220/agriRxiv.2023.00206

Zhang, Y., Lavallee, J. M., Robertson, A. D., Even, R., Ogle, S. M., Paustian, K., & Cotrufo, M. F. (2021). Simulating measurable ecosystem carbon and nitrogen dynamics with the mechanistically defined MEMS 2.0 model. *Biogeosciences*, *18*(10), 3147–3171. https://doi.org/10.5194/bg-18-3147-2021

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