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LCA FOR AGRICULTURE

Swiss Agricultural Life Cycle Assessment: A method to assess the emissions and environmental impacts of agricultural systems and products

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Abstract

Purpose Agricultural production, which dominates the environmental impacts of the food sector, has specific characteristics that need to be considered in life cycle assessment (LCA) studies. Agricultural systems are open, difficult to manage and control, strongly depend on natural resources and their impacts are highly variable and influenced by soil, climate and farm management. A specific framework, efficient methods and tools are thus needed to adequately assess the environmental impacts of agricultural systems.

Methods We present the Swiss Agricultural Life Cycle Assessment (SALCA) concept and method, developed for a detailed and specific analysis of agricultural systems. It comprises rules for the definition of system boundaries, functional unit and allocation, emission models, a life cycle inventory (LCI) database, calculation tools, impact assessment methods and concepts for analysis, interpretation and communication. This paper focuses on emission models for gaseous N, nitrate leaching, P emissions to water, soil erosion, pesticides, heavy metals, emissions from animal production and impact assessment methods for soil quality and biodiversity. The models are calculated at the crop, field, animal group and farm levels and are integrated in a consistent and harmonised framework, which is ensured by exchanging intermediate results between models.

Results and discussion The SALCA concept has been applied in numerous LCA studies for crops and crop products, cropping systems, animal husbandry systems and animal products, food and feed products, farms and product groups, the agrifood sector and food systems. The SALCA methodology has also been a backbone of the LCI databases ecoinvent, AGRIBALYSE and the World Food LCA database. The strengths of SALCA lie in its comprehensiveness, specificity to agriculture, harmonisation, broad applicability, consistency, comparability, flexibility and modularity. The extensive data demand and the high complexity, however, limit the application of SALCA to experts. The geographical scope is limited to Central and Western Europe, with a special focus on Switzerland. However, due to the modular and flexible design, an adaptation to other contexts is feasible with reasonable effort.

Conclusions SALCA enables answering a wide range of research questions related to environmental assessment and is applicable to various goals and scopes. A further development would be the inclusion of the social and economic dimensions to perform a full sustainability analysis in the SALCAsustain framework.

Keywords Life cycle assessment · Agriculture · Crop · Livestock · Farm · Farming system · Environmental impacts · Emissions

1 Introduction

Agricultural systems and food supply have a strong impact on the environment. For instance, one-third of greenhouse gas (GHG) emissions (Crippa et al. 2021), up to 80% of eutrophying emissions (Poore and Nemecek 2018) and 86% of the species at risk of extinction (Ritchie and Roser 2020) are caused by the agrifood sector. A detailed quantification of these impacts and mitigation options is therefore of crucial importance for farmers and policy makers. In this

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respect, life cycle assessment (LCA) has been successfully applied to agriculture and food systems for decades to perform a comprehensive assessment of environmental impacts and provide a basis for the design of mitigation measures (Dijkman et al. 2018; Thoma et al. 2022).

1.1 Relevant aspects of agriculture

We must consider the specific characteristics of agricultural systems. In contrast to typical industrial sectors dominated by relatively few companies with large, standardised production facilities, farms are comparatively small, numerous production units. The number of farms was estimated at ~ 570 million worldwide in 2016 (Lowder et al. 2016). The large variability between environmental impacts and the economic performance of different farms points to significant potential for improvement (Grassauer et al. 2022b; Pedolin et al. 2021, 2023). Agriculture relies heavily on natural resources; therefore, emissions do not depend only on management, but also to a large extent on climate, soil and topography, showing a high spatial and temporal variability (Lee et al. 2020). This means that large samples are required to obtain reliable estimates of the environmental impacts of agricultural production. Agricultural systems are open, in contrast to many industrial production systems that operate in closed environments. Emissions are thus difficult to control and measurements of emissions in real field situations or from grazing livestock are challenging. Therefore, agricultural LCA relies heavily on the use of models to take into account both management and site conditions.

1.2 Challenges of agricultural LCA

Efficient and reliable tools are required to quantify and mitigate the environmental impacts of agricultural production, to make informed choices in the supply chain, to enable consumers to select food with low environmental impacts and finally to adapt their diet accordingly. Methods and models for use in agricultural LCA should ideally have a number of characteristics. They should represent the main environmental mechanisms based on scientific evidence. It should be possible to consider the environmental impacts of different mitigation measures in order to assess their effectiveness. The main influencing factors, particularly different management options and their interactions with pedoclimatic conditions, need to be considered. Even if the pedoclimatic conditions cannot be significantly changed by the farmer, it is crucial that the main factors are properly reflected (Lee et al. 2020). This can help, for instance, to choose the best-suited regions for a particular production and to focus mitigation measures on the regions with the highest impact. Furthermore, we need to understand and represent the variability of the system and its environmental impacts. Often, the aim is to compare systems that are relatively similar. Examples are different types of fertilisers (Avadí 2020), various weed control techniques (Russo et al. 2015) or different feed rations in beef production (Hengen et al. 2016). The models should be detailed and accurate enough to discriminate between systems with relatively small changes, which makes screening LCA methods and rough estimates inadequate. Flexible models are needed for application in different situations to answer various research questions.

1.3 Methods and tools for agricultural LCA

Numerous methods, models and tools have been developed for LCA applications in agriculture and other economic sectors. Standard LCA software, such as SimaPro, OpenLCA and GaBi, are generic tools that can be used to model agricultural systems, but are not specifically adapted to the needs of agricultural LCA research and offer only limited options for calculating direct agricultural emissions and reflecting the complexity of agricultural systems.

Several software solutions have therefore been developed for specific use cases in the agricultural sector. At the farm level, the Cool Farm Tool (Hillier et al. 2011) is an example that enables analyses of farms in terms of GHG emissions, water and biodiversity. APS-Footprint (Braconi et al. 2021) measures and monitors the environmental impacts of animal production systems. In general, such tools have a specific focus on one or more environmental impacts and do not offer a full and comprehensive LCA.

In addition, there are different methods and tools for the creation of background life cycle inventories (LCI) of agri-food products, such as ecoinvent (Nemecek and Erzinger 2005; Wernet et al. 2016), AGRIBALYSE (Koch and Salou 2016, 2020), the World Food LCA database (WFLDB) (Nemecek et al. 2019) and Agri-Footprint (Tyszler et al. 2022). Furthermore, a methodology for calculating the product environmental footprint (PEF) has been developed in Europe, which is based on existing concepts and standards for life cycle assessment (Zampori and Pant 2019).

In recent years, several initiatives have been launched to standardise or combine the available environmental information and LCIs. Geofootprint combined available LCI from WFLDB and ecoinvent with globally available geo-referenced data to provide spatially explicit environmental footprints (Reinhard et al. 2021). The Harmonised Environmental Storage and Tracking of the Impacts of Agriculture (HESTIA) is an open-source and open-access platform that provides a standardised format and glossary of terms for the presentation of agri-environmental data (Henriksson et al. 2022; HESTIA 2023). The HESTIA platform allows users to upload and download agricultural life cycle inventory (LCI) and life cycle impact assessment (LCIA) data, which are validated and stored in a standardised format.

Approaches that aim to link nutritional aspects with environmental information are becoming increasingly important. Optimeal (Broekema et al. 2019) is an example of a software tool developed to optimise human nutrition, considering nutrients and sustainability aspects. Optimisation software for the food industry is being developed in the OptiSignFood project to support product development and optimise food quality, nutritional value and environmental impacts (Nemecek et al. 2022b).

Most of these methods and tools have a specific focus on either a predefined selection of environmental impacts such as GHG emissions, biodiversity or water footprint (Hillier et al. 2011) or a narrow focus on certain production systems and therefore do not offer a full and comprehensive LCA (Zah et al. 2009). Some are limited to generic background applications and are not suitable for a detailed assessment of the foreground system (Reinhard et al. 2021). The foreground system consists of processes which are under the control of the decision-maker (e.g. farmer), while the background system consists of processes, which can only indirectly be influenced (e.g. production of purchased inputs for production). Others are not specific to agriculture and therefore do not allow us to discriminate sufficiently between systems with only minor or moderate differences (PRé Consultants 2019). Certain tools were developed for other geographical contexts, limiting their application in Central Europe (e.g. Emhart et al. 2014).

1.4 Specific methods and tools needed for agricultural LCA

The combination of different models and tools offers a possible means to deal with these limitations, but can cause inconsistencies; therefore, harmonisation is needed to achieve a consistent assessment and to ensure results can be compared. A first attempt to harmonise methodology for agricultural LCA was put forth by Audsley et al. (1997). More recent initiatives have been taken by LEAP from the FAO (FAO 2023), the UN Life Cycle initiative (UNEP 2023) and the PEF of the EU (European Commission 2018). Case-by-case modelling of each individual situation is not feasible and bears the risk of errors and inconsistencies, since not all situations are handled equally. For broad applications and comparable assessments, we need flexible, parametrised and harmonised tools that are versatile enough to answer different research questions.

Mechanistic simulation models can help analyse specific processes; however, in the context of LCA, they often turn out to be too complex. They require a great deal of input data, which are often unavailable for a specific application. Moreover, combining several simulation models typically exceeds the size of a project and can lead to inconsistencies. Their predictive power is often lower than that of simple empirical models. Therefore, in agricultural LCA, models of medium complexity are expected to be the best compromise: they are detailed enough to evaluate mitigation options and to take into account the main influences of pedoclimatic conditions, but they are also applicable on a large scale with moderate data demand (Freiermuth Knuchel et al. 2009). For over two decades, Agroscope has worked to develop harmonised and standardised emission and impact assessment models for agriculture within the Swiss Agricultural Life Cycle Assessment (SALCA) framework and has applied them in numerous LCA studies.

1.5 Objective and structure of this paper

The objective of this paper is to introduce the overall concept of SALCA, to describe the emission models and impact assessment methods for soil quality (SQ) and biodiversity in the recently updated version 1.0 of SALCA within the SALCAfuture project (Lansche and Stüssi 2022), to show applications of SALCA, and to discuss the strengths, weaknesses and limitations of a harmonised methodology. Chapter 2 presents the SALCA concept, Chapter 3 gives an overview of the LCI emission models and Chapter 4 presents impact assessment methods for soil quality and biodiversity. Typical examples of the application of SALCA are presented in Chapter 5, followed by a discussion of strengths, weaknesses, limitations and needs for further development in Chapter 6.

2 Swiss Agricultural Life Cycle Assessment (SALCA): Concept

The overall aim of SALCA is to provide a flexible and efficient framework for LCA studies in agriculture based on scientific evidence (Gaillard and Nemecek 2009; Nemecek et al. 2010). It supports the four phases of LCA and contains the following elements:

1. Goal and scope
    (a) Rules for the definition of system boundaries, functional unit and allocation for agricultural products, farms and the agrifood sector
2. LCI
    (a) Models to calculate direct field and farm emissions relevant for agriculture, such as N and P emissions, methane, carbon dioxide, heavy metals, pesticides and soil erosion
    (b) A database with LCI for inputs and processes for agrifood systems

(c) A software environment to collect input data, check their plausibility and calculate agricultural LCAs in a robust and efficient manner

3. LCIA

(a) Midpoint impact assessment methods specific to agricultural applications due to direct land use (biodiversity and SQ)
(b) Selection of impact assessment methods specific to agricultural systems

4. Interpretation

(a) A concept for analysing and interpreting LCA results
(b) A concept to communicate the results, conclusions and recommendations

The SALCA methodology comprises a series of harmonised and consistently parametrised models. The models follow a modular structure, with each having a clearly defined interface. This makes the model complexity manageable and allows for independent testing and updates of the models. Furthermore, a model can easily be exchanged for an alternative model if SALCA has to be applied in a different context. The methodology follows ISO 14040/44 standards (ISO 2006a, b) and considers the development of international standardisation bodies (European Commission 2018; FAO 2023; UNEP 2023) to the extent that the standards are considered relevant for the main purpose of SALCA. Studies applying the SALCA method have been critically reviewed three times according to ISO 14040/44 standards (Bystricky et al. 2020; Nemecek et al. 2005; Wolff et al. 2016).

This paper describes the modelling of field and farm emissions (2a, Chapters 3.1–3.8), as well as the midpoint impact assessment methods SALCAbiodiversity and SALCAsoilquality (3a, Chapters 4.1–4.2). Other aspects of the SALCA concept are partly described elsewhere: impact assessment (Bystricky et al. 2020), software tools (Lansche and Stüssi 2022), analysis and interpretation (Nemecek et al. 2011a), database (Nemecek and Schnetzer 2012) and interpretation and communication of LCA results (Bystricky et al. 2020; Herndl et al. 2016; Hersener et al. 2011; Nemecek et al. 2011a; Zumwald et al. 2018a).

Agricultural production is modelled at four levels (Fig. 1):

1. Field: a delimited piece of land that can have one or more crops, either in a sequence (one crop following another) or by splitting the field into several areas. Together, all fields form the agricultural area of a farming system or farm.

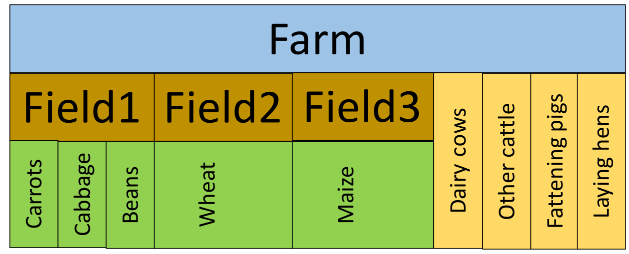

Fig. 1 Schematic representation of the four levels of organisation of SALCA (illustrative example). Green, crops; yellow, animal groups

2. Crop: the whole crop is managed in the same way and assigned to a specific field.
3. Animal group: a category of animals with similar properties and the same emission factors (EFs).
4. Farm: an entity comprising all fields, crops and animal groups. In addition, a farm includes general infrastructure, such as buildings and machinery, used in different types of production. SALCA includes only those parts of a farm that are dedicated to food and feed production (i.e. no forestry, flower production).

The calculation level of each emission model (Table 1) is chosen to allow the consideration of specific conditions on the one hand and to perform efficient calculations on the other hand. Therefore, the level of detail is simultaneously as low as needed and as high as possible.

In addition to the four calculation levels, product groups are distinguished that summarise similar products of a farm, e.g. cereals, potatoes, fruits, vegetables, milk, beef, pork and poultry. Pedolin et al. (2021) offered additional details on the concept of product groups. The latter can be defined individually for each study, depending on the goal and scope. The sum of all product groups represents the whole farm; therefore, the sum of the environmental impacts of all product groups equals the farm's total environmental impact. Different allocation

Table 1 Calculation levels for the different models

	Crop	Field	Animal group	Farm
SALCAfieldN		x		
SALCAfieldP	x			
SALCAfieldC		x		
SALCAnitrate		x		
SALCAerosion		x		
SALCAanimal			x	x
SALCAheavymetal	x		x	x
SALCApesticides	x			
SALCAsoilquality	x			x
SALCAbiodiversity	x	x		x

rules are applied to assign inputs, resources and emissions to one or more product groups. The allocation criteria are related to the area (e.g. agricultural area, arable land), to the livestock (e.g. livestock units) or to the financial return as the most generic criterion (Pedolin et al. 2021). The aim is to use specific allocation criteria where possible and to apply generic criteria, such as financial return, only if no other criterion is suitable. To provide consistent results, the models exchange data, as shown in Fig. 2.

The following descriptions of the models provide a general overview of the concepts, processes and relevant factors. A detailed description of the models can be found in the Supplementary Material, as referenced below.

3 SALCA life cycle inventory models

3.1 Gaseous N field emissions (NH_3, N_2O, NO_x, SALCAfieldN)

In this section, we present the calculations of N emissions in the air after the application of fertilisers or from crop residues. A detailed model description is given in Online Resource 1.

3.1.1 Ammonia (NH_3)

NH_3 emissions are important contributors to eutrophication and acidification impacts. They stem from fertiliser use and animal husbandry (see Chapter 3.7).

Mineral fertilisers The EF for mineral fertiliser stems from the EMEP guidelines 2019 (EEA 2019), since they distinguish between different types of mineral fertilisers and consider soil pH. EEA (2019) provides factors for pH values below and above 7, reflecting that ammonia emissions increase by ~ 70% on average from acid to basic soils. Such a stepwise change in the EF can cause a bias in some LCA studies, where a small change around a pH of 7 would lead to a sharp increase or decrease in ammonia emissions. Therefore, the EFs for low pH are applied to pH < 5.5, those for high pH to pH > 7.3 and linearly interpolated in between. The thresholds of 5.5 and 7.3 are derived from (Bouwman et al. 2002).

Farmyard manure To calculate ammonia emissions from farmyard manure, two models can be selected in SALCA:

1. EEA/EMEP model (EEA 2019): a Tier 2 model with a geographical scope of Europe
2. Agrammon (Agrammon Group 2022), a Tier 3 model adapted for Switzerland

Tier 1–3 denote different levels of methodological complexity: Tier 1 is the simplest method, Tier 2 intermediate and Tier 3 the most demanding in terms of complexity and data requirements (EEA 2019). The farmyard manure is calculated in SALCAanimal as a mix of all liquid manure types used on the analysed farm on the one hand and the solid manure types on the other. Therefore, the EFs for both liquid and solid manure are calculated as a weighted mean of the respective EFs.

A correction factor is applied for the climate conditions according to the month of application as a function of the saturation deficit, which in turn is calculated from relative humidity and temperature (monthly average values).

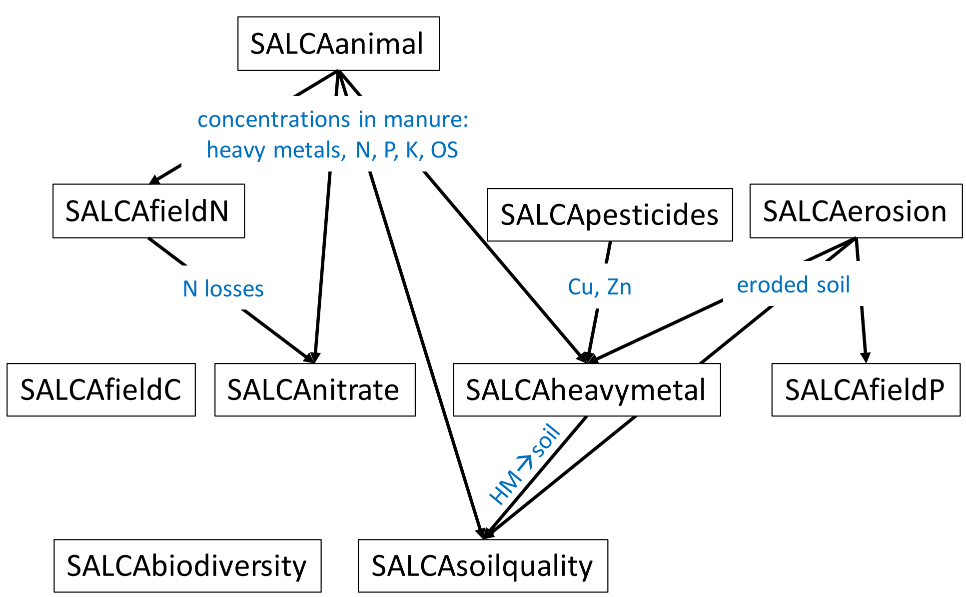

Fig. 2 Data flow among the SALCA models. HM, heavy metals; OS, organic substance

Correction factors from Agrammon are applied for the following:

- Liquid manure application technique: Broadcaster (1.0), trailing hoses (0.7), slurry drilling (0.3), deep injection (0.2), trailing shoes (0.5) and application in the evening (0.8). Example: a factor of 0.2 means that emissions are reduced by 80%.
- Incorporation of solid manure with reductions of up to 90% for incorporation within one hour.

Although NH_3 emissions can also stem from N-rich crop residues left on the soil surface (de Ruijter et al. 2010), the estimates are currently too uncertain and therefore not considered, in accordance with EEA (2019).

3.1.2 Nitrogen oxides (NO_x, NO and NO_2)

Nitrogen oxides stem mainly from the nitrification process and contribute to eutrophication and photochemical ozone formation. The importance of NO_x emissions from N fertilisers and manure management is relatively small compared to combustion processes (Fowler et al. 2013). Therefore, simple EFs from (EEA 2019) Tier 1 are used. The emission is calculated after subtraction of the N volatilised as NH_3.

3.1.3 Nitrous oxide (N_2O)

Nitrous oxide (N_2O) is a very powerful GHG produced during the nitrification and denitrification processes. For nitrous oxide, we follow IPCC guidelines (IPCC 2019) Tier 1 for crop production. We use the default EF1 for mineral fertilisers, crop residues and soil organic matter (IPCC 2019). For organic fertilisers, however, we follow the approach of Nemecek and Ledgard (2016). An EF of 0.01 kg N_2O-N/kg N is applied to the total ammonia nitrogen (TAN) fraction of N in organic fertilisers, which is easily available for microbial processes. For the rest of the N, which is contained in organic compounds and is not easily available in the short term, we apply a lower EF of 0.025 kg N_2O-N/kg N. This gives values in the same range as IPCC (2019), but allows to take better into account differences between production systems. Thus, we use the same relationship as for urine and dung excreted on pasture by grazing animals. Higher rates of N application tend to result in higher emissions of N_2O (Bouwman et al. 2002). No additional emissions are counted for symbiotic N fixation by legumes (Rochette & Janzen 2005). Based on Shcherbak et al. (2014), we adjust the emissions after fertiliser application by a quadratic function of the N fertiliser rate, leading to slightly lower emissions below 146 kg N ha^{-1} $year^{-1}$ and higher emissions above this threshold. N_2O released during the decomposition of organic matter in the soil after land use change is a further source of emissions.

Following the default values of IPCC (2019), we calculate induced emissions from redeposition of NH_3 and NO_x (EF = 0.01 kg N_2O-N/kg N) and leaching of NO_3 (EF = 0.011 kg N_2O-N/kg N). These factors are applied to all sources of emissions, as calculated by SALCA.

3.2 Nitrate leaching (NO_3, SALCAnitrate)

Nitrate contributes to eutrophication of aquatic systems and is considered as a major driver of marine eutrophication. The SALCAnitrate model estimates the expected nitrate leaching from arable crops, grassland and horticulture, but no surface run-off. Nitrate leaching is given by the sum of the monthly values within the assessment period starting after the harvest of the previous crop and ending in the month in which the given crop is harvested. It considers N mineralisation in the soil and from crop residues, N uptake by the plants and N fertilisation (Fig. 3). This is an updated version of the model described by Richner et al. (2014) (see Online Resource 2 for details). Nitrate emissions from grazing animals are calculated in SALCAanimal.

N mineralisation depends on the following factors:

- Month of the year.
- Region: three regions are distinguished (valley, hills and mountains), representing different temperature curves.
- Humus content: N mineralisation of soil organic matter (SOM) increases with humus content.
- Clay content: N mineralisation decreases with higher clay content.
- Soil tillage: intensive soil tillage leads to mixing of soil particles and aeration of the soil and therefore increases N mineralisation.
- Type of land use: N mineralisation rates in grassland soils are lower than in arable soils. Within permanent grassland, we further distinguish between intensive, medium intensive and extensive management; the mineralisation decreases with lower management intensity.
- Organic manure supply: the N mineralisation from soil organic matter is further corrected for the average stocking rate. On farms with more livestock, a higher mineralisation rate results.
- Previous crop effects: N mineralisation further depends on the previous crop due to crop residues remaining in the field or being incorporated into the soil. The rate of N mineralisation is increased for a period of four to eight months. The rate and duration of the increase depend on the previous crop. We distinguish permanent grassland, temporary grassland, grain legumes, overwintering green manure and sugar beets with the incorporation of leaves into the soil.

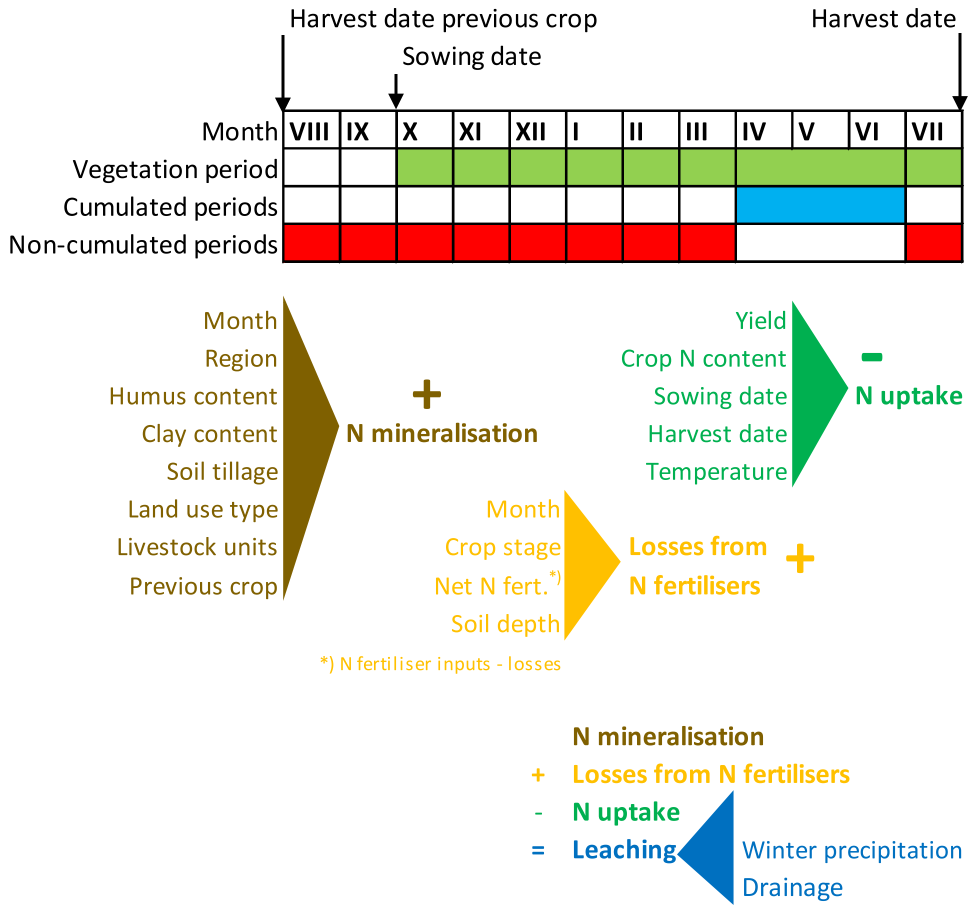

Fig. 3 Schematic representation of SALCAnitrate for the example of a winter wheat crop

The total N uptake is estimated from the effective yield, the standard yield and the standard N uptake according to Richner et al. (2017) and increased by 10% to account for additional N in roots and crop residues. For legumes, we assume that 60% of the N contained in the biomass stems from symbiotic N fixation and the remaining 40% is taken up from soil N. The temporal dynamics of N uptake by the crop were estimated based on the STICS model (Brisson et al. 2003). The monthly distribution of N uptake considers the date of sowing, the date of harvest, the crop-specific base temperature (0, 5, 7.5 and 10 °C) and the growing degree-days for the specific crop in the region's valley, hills and mountains.

Generally, no seepage occurs during the intensive vegetation period because the evapotranspiration is similar to or higher than the precipitation; therefore, no nitrate leaching is expected. To take this into account, cumulated periods are defined. The model accumulates the monthly values of N mineralisation, nitrate uptake by the plants and nitrate losses from fertilisation during this period, and the balance is calculated at the end of this period.

The amount of N in mineral fertilisers and the TAN in organic fertilisers is corrected by subtracting gaseous N losses, as calculated in SALCAfieldN. For non-cumulated periods, leaching is calculated separately for each month as the difference between N uptake and N mineralisation. Losses from fertiliser application are determined by monthly risk factors depending on the crop. The losses are increased on soils with a potential rooting depth < 100 cm. Leaching typically occurs during a cold period. To account for differences in climate, the total leaching is further corrected for precipitation during the cold period (October–March on the Northern hemisphere). Then, the amount is converted from N to NO_3. Finally, N leaching is counted as emission to groundwater, or—if drainage is present—to surface water.

The model was calibrated (Richner et al. 2014) and partly validated (Bystricky et al. 2018) for Switzerland and covers the three main Swiss production regions. It can be applied to regions with similar pedoclimatic conditions, e.g. in Central Europe. The model is flexible and can also be adapted to different conditions if needed, as, for example, was done for Austria (Bystricky and Nemecek 2015). For different pedoclimatic conditions, other models might be more suitable (e.g. de Willigen 2000).

3.3 CO_2 emissions from lime, urea and land use (SALCAfieldC)

Field applications of urea and lime lead to CO_2 emissions of fossil origin, which need to be accounted for. The EFs are taken from IPCC (2019) and are based on the C contents of urea and lime, assuming a complete decomposition in the soil and conversion to CO_2.

CO_2 emissions from the cultivation of organic soils are calculated according to Tiemeyer et al. (2020) by taking into account the depth of the water table. For well-drained soils (water tables below 0.6 m), the CO_2 emission amounts to 9.52 t CO_2-C ha^{-1} a^{-1}, following Leifeld et al. (2009).

Changes in soil organic carbon (SOC) in the six types of grassland are estimated according to Wüst-Galley et al. (2020). This study made estimates for different soil and climate types. The effects of land use change on C stored in the soil, living biomass and dead organic matter are calculated according to IPCC (2019). For details see Online Resource 3.

3.4 Soil erosion (SALCAerosion)

Soil erosion is relevant for soil quality and emissions of P and heavy metals to water. The theoretical background of SALCAerosion follows the revised universal soil loss equation (RUSLE, Renard 1997). This equation considers the following six factors: rainfall erosivity (*R*), soil erodibility (*K*), slope length (*L*), slope steepness (*S*), cover and management (*C*) and support practice (*P*). The RUSLE model can be expressed by Eq. 1 for annual soil loss e_a:

$$e_a = R \cdot K \cdot L \cdot S \cdot C \cdot P \tag{1}$$

where e_a is the average annual soil loss [t ha^{-1}], *R* is a rainfall erosivity factor [MJ mm ha^{-1} h^{-1}], *K* is a soil erodibility factor [t h MJ^{-1} mm^{-1}], *L* and *S* are slope length and slope steepness factors, respectively [-], *C* is a land management factor [-] and *P* is a conservation practice factor [-]. RUSLE predicts the long-term average and annual rate of erosion based on rainfall pattern, soil type, topography, crop system and management practices. The product of the factors *R*, *K*, *L* and *S* describes potential soil erosion loss, while the factors *C* and *P* take into account the effects of soil cover and land management on erosion. Factor *P* in Eq. 1 characterises soil loss changes related to specific support practices, such as contour farming or stone walls (Renard 1997).

The parameterisation of the relevant RUSLE factors in SALCAerosion follows different approaches. The computation is split into the parameterisation of the site-dependent factors (*R*, *K*, *L* and *S*) and factor *C*, which depends on the farmer's management practices and share of crops in a rotation. The site-dependent factors (*K*, *L* and *S*) are estimated using tabulated values, depending on the soil type (largely depending on granularity), the steepness of the plot and the distance from the next open body of water or canalisation. In addition, tabulated values are differentiated between five Swiss regions, which are characterised by similar erosive effects of precipitation, grain size composition of the most frequent soil types and humus contents (see Online Resource 4 for details). Factors *P* and *R* are not considered in the computation and are thus set to 1. The computation of crop composition factor *C* is based on tabulated standard values, supplemented by a number of correction factors to consider additional effects from crop rotations and soil cultivation. The most important influencing parameters are the fraction of certain crop groups (e.g. winter grain, rapeseed and maize) and catch crops, as well as the frequency of reduced tillage technique application, such as strip sowing, no-till and mulch sowing.

3.5 Field P emissions (SALCAfieldP)

P emissions to water are the main driver of freshwater eutrophication. SALCAfieldP predicts both dissolved and soil-bound phosphorus (P) emissions to water bodies at the plot level (Prasuhn 2006). Four discharge paths are taken into account: (i) soil erosion, (ii) surface runoff, (iii) drainage losses to surface waters and (iv) leaching to groundwater. The discharge path via erosion leads to P emissions bound to the soil particles, while the latter three lead to emissions of soluble phosphate. The calculations are based on mean climatic conditions. Thus, P emissions due to extreme events are not simulated by the model. The amount of eroded soil is estimated in the SALCAerosion model (Chapter 3.3, Fig. 2).

For each discharge pathway, default values are assumed, which are then adjusted to the respective conditions by means of correction factors. These correction factors depend on various parameters: distance to the nearest water body, topography (slope steepness, slope shape and length), soil type (granularity), P content of the soil, soil P status class, P amount in applied organic and mineral fertilisers and the presence of drainage. Plant-available P in the topsoil, characterised by the soil's P status class, is crucial, as it largely determines the amount of P leaching in permeable soils or via macropore flow (see Online Resource 5 for details).

Default values for P leaching into ground water and surface water bodies are provided for different land use types, such as permanent grassland, arable land, vineyards and orchards (Prasuhn 2006). P leaching by drainage is parameterised like P leaching to groundwater, but increased by a factor of six to consider the preferential flow through drainage pipes. The model was developed for Swiss and similar climate and soil conditions.

3.6 Pesticide emissions (SALCApesticides)

Pesticides are important contributors to toxicity impacts and can have strong effects on biodiversity. The distribution of an active ingredient to the different emission compartments uses the PestLCI consensus model and follows the methodology described in Nemecek et al. (2022a). We use default values per crop and target group (herbicides, fungicides, insecticides, etc.) to calculate the initial or primary distribution to the following compartments:

- Air, low population density: a fixed fraction of the active ingredient is calculated as airborne, depending on the application method and drift reduction.
- Off-field surfaces: the fraction deposited outside the field depends on the application technique, the drift reduction and the width of the buffer zone. This fraction is further subdivided into the compartments agricultural soil, natural soil and surface water by taking the shares of the respective areas in the considered region.
- The remaining substance is deposited in the field and split between the agricultural soil and the crop surface according to the intercepted fraction, which in turn depends on the crop and the development stage at the time of application.

Currently, there is no emission compartment to consider pesticides deposited on crop surfaces. Therefore, following the recommendation of Nemecek et al. (2022a), this amount is added to the emissions to agricultural soil. This is a worst-case assumption that most likely overestimates the effective toxicity impacts. However, at the time being, this is considered the best approach, as issued by the OLCA-Pest consensus project (Nemecek et al. 2022a). We distinguish the target groups of herbicides (pre- or post-emergence), insecticides, fungicides, plant growth regulators, acaricides/miticides and applications with soil incorporation and 17 crop groups: berries, bulbs, citrus fruit trees, fruit trees temperate, fruit trees tropical, grapes/vines, grass, oil-bearing crops, oil-bearing trees, other permanent crops, paddy rice, panicoideae (maize, sorghum, etc.), pooideae (cereals), pulses, roots and tubers, vegetables fruit and vegetables leafy. For Cu- and Zn-containing pesticides, the initial distribution is calculated by the method described here. The emissions to water and soil are then estimated using the SALCAheavymetal model (see Chapter 3.8).

The amounts of the active ingredients applied is provided by the user. All EFs sum up to 100% of the quantity applied to the field to maintain the conservation of mass. The different environmental compartments are assigned corresponding characterisation factors in the LCIA phase.

3.7 Modelling of animal husbandry-related emissions (SALCAanimal)

Emissions from animal production are dominating many environmental impacts, related to the food system. The model calculates the excretion of N, P, K, heavy metals (HMs) and organic substance as well as emissions of NH_3, N_2O, NO_x, NO_3 and CH_4 for 15 animal groups: dairy cows in loose housing respectively tied housing, other cattle, fattening pigs, sows, sheep, goats, other ruminants, horses (including mules and asses), laying hens, broilers, turkeys, ducks, geese and other nonruminants (Fig. 4, see Online Resource 6 for a detailed description).

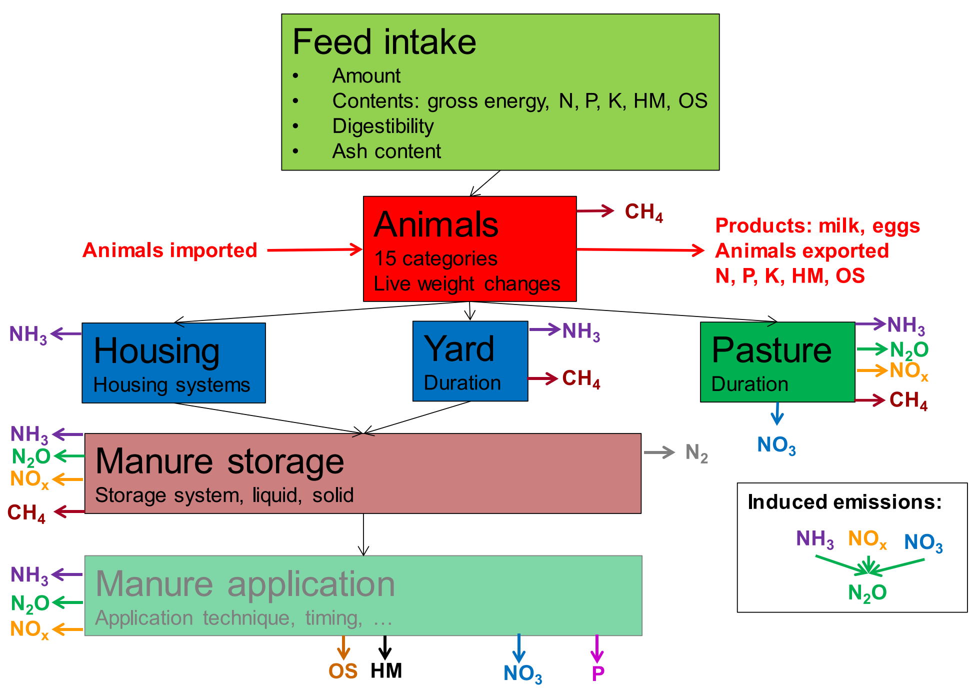

Fig. 4 Concept of balances and emission calculations for animal husbandry. HM, heavy metals; OS, organic substance. Emissions from manure application are calculated in other modules (see text)

Inputs, outputs and husbandry parameters are recorded, and nutrient balances are calculated within each animal group separately. These groups were chosen according to the following criteria: (1) animal groups relevant to the environmental impacts and (2) different EFs provided for the emissions of NH_3, N_2O, NO_x, NO_3 or CH_4.

The balances, excretion and emissions are calculated as follows:

1. The feed ration is provided by the user, and the feed consumed is assigned to different animal categories.
2. From the feed ration and the tables with nutrient and HM contents of the different feedstuffs, the nutrient/HM intake is calculated for each animal group. Nutrient contents and digestibility are taken from feedbase.ch and HM contents from sources reported in Online Resource 7.
3. Methane from enteric fermentation is calculated by the feed intake per animal category, according to IPCC (2019) Tier 2 and for dairy cows optionally, according to Kirchgessner (2004). Methane emissions from manure management and grazing animals follow the guidelines of IPCC (2019) Tier 2. Along with methane emissions, a balance of organic substance and organic carbon is calculated, which is subsequently used in SALCAsoilquality.
4. The balances of nutrients, HMs and organic carbon per animal group are calculated from feed intake, purchase and sale of animals, change in live weight (if relevant) and the products of the animals. The excretion of N, P, K and HM is given by the net balance.
5. The nutrient and HM excretion is categorised into housing, yard and pasture by using the time fractions spent in the respective areas.
6. The excretion is further subdivided into solid and liquid manure depending on the N concentration in the diet (Nemecek and Ledgard 2016), since different EFs apply to these types of manure.
7. Emissions of NH_3, N_2O, NO_x and CH_4 occur in the housing, yard and pasture. The amount of N emitted is subtracted from the total N and the TAN to ensure a correct mass balance.
8. The same procedure applies to manure storage. Here, the different manure management systems are distinguished. N mineralisation in liquid manure and N immobilisation in solid manure during manure storage are calculated. N_2 emissions are subtracted to ensure the mass balance.
9. NH_3, N_2O and NO_3 emissions from grazing animals are estimated using EFs differentiated by season.
10. Induced (or indirect) N_2O emissions are calculated from all NH_3 and NO_x emissions to the air and from NO_3 leaching.
11. At the farm level, the average nutrient/HM concentration of liquid and solid manure is calculated. Here, the average from all animal groups is computed and manure imports and exports are taken into account. These concentrations are subsequently used in SALCAfieldN, SALCAfieldP, SALCAnitrate and SALCAheavymetal for the calculation of emissions and balances. Organic substance (OS) is used in SALCAsoilquality to calculate humus balance (Fig. 2).

Emissions from pasture are included in the SALCAanimal model, while emissions from manure application are calculated in the SALCAfieldN and SALCAfieldP models.

3.8 Heavy metal emissions (SALCAheavymetal)

Heavy metals (HM) can have strong impacts on toxicity. The SALCAheavymetal model calculates HM balances at the crop level and emissions to surface water, ground water and agricultural soil (details are provided in Online Resource 7). The model includes cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), nickel (Ni), lead (Pb) and zinc (Zn). These seven elements are considered the HM emissions from agriculture with the highest environmental relevance. The calculation is performed in five steps (Fig. 5):

1. The balance of HMs of the animal herd is calculated in SALCAanimal to determine the contents and concentrations of HMs in farmyard manure (see Chapter 3.7).
2. The HM inputs through mineral and organic fertilisers, pesticides and seeds into a specific field are calculated.
3. An allocation factor is calculated, accounting for the share of the HM inputs caused by agricultural management in the total inputs (including HMs from deposition). Deposition is caused by the industry, households and transports and is therefore largely of non-agricultural origin. This allocation factor is subsequently applied to the emissions by erosion and leaching and to the exports by harvested goods (in Steps 4 and 5).
4. The emissions from erosion and leaching, as well as the HMs exported in the harvested goods, are determined. Emissions by erosion are determined using the mass of eroded soil from SALCAerosion, the HM concentration in the soil, an accumulation factor for the topsoil and the allocation factor calculated in Step 3. The emissions by leaching are estimated by constant leaching rates for each HM per area and year. The HMs exported in the harvested main products and co-products are determined by the harvested dry mass and the HM concentrations of the respective types of products or co-products. HMs from pesticides leaving the field with harvested products are also included.

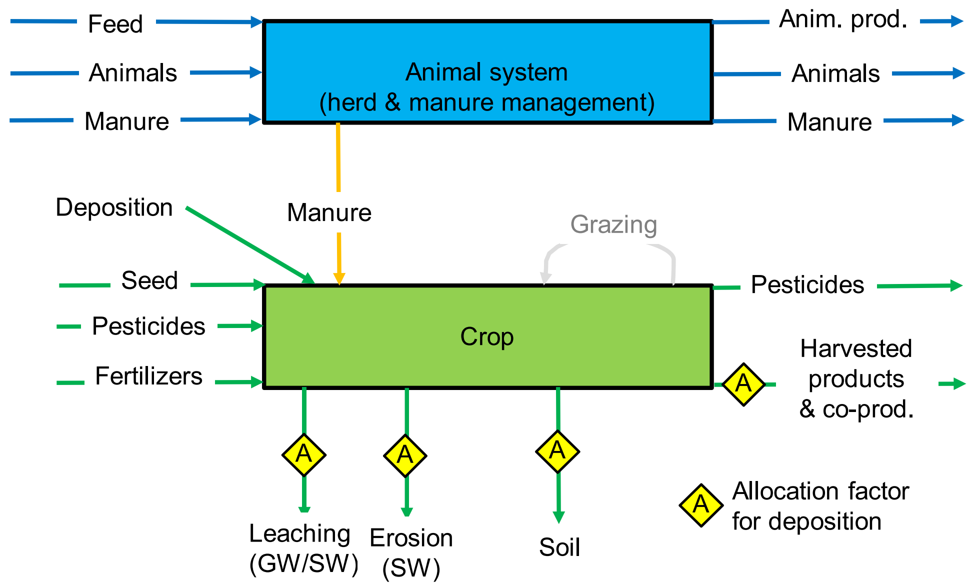

Fig. 5 Heavy metal flows considered in the SALCAheavymetal model. GW, ground water; SW, surface water

5. The emissions to the agricultural soil are calculated as the difference between the inputs and the outputs (erosion, leaching and harvest).

4 SALCA life impact assessment models

4.1 Soil quality (SALCAsoilquality)

SALCAsoilquality aims to assess the impact of agricultural management practices on SQ. A former version was presented by (Oberholzer et al. 2012). Here, we present an updated method (see Online Resource 8 for a detailed description).

The model estimates on-farm SQ with a set of nine measurable soil properties (hereinafter called SQ indicators [SQI]), three each in the areas of soil physics, soil chemistry and soil biology (Table 2). These indicators typically do not change in the short term but rather in the medium term (crop rotation). SALCAsoilquality estimates expect relative changes due to soil and crop management practices, using empirical modelling based on expert knowledge and supported by available literature. These relative changes are approximated based on impact classes, which summarise the effects of farm management activities on soil processes. Examples are the risk of soil compaction, humus dynamics or toxic impacts of slurry application (see Online Resource

Table 2 Description of soil quality indicators (SQIs) as used in the SALCAsoilquality model. P, physical SQI; C, chemical SQI;B, biological SQI.

Soil quality indicators	Description
P: Potential plant rooting depth	Potential depth at which plant roots can take up the maximum amount of plant-available water. This can be affected by soil erosion, which is calculated using SALCAerosion
P: Macropore volume	Volume of macropores (> 50 m diameter) defined as continuous soil pores that are significantly larger than the intergranular or interaggregate soil pores.
P: Aggregate stability	Stability of soil aggregates (a group of primary soil particles that cohere to each other more strongly than to other surrounding particles)
C: Organic carbon	Balance (gains minus losses) of SOM
C: Heavy metals	Amount of HMs (Cu, Cd, Zn, Pb, Ni, Cr and Hg) emitted into the soil from SALCAheavymetal
C: Organic pollutants	Amount of organic chemicals emitted into the soil: dioxins, furans, polychlorinated biphenils (PCDD/PCDF and PCBs) and polycyclic aromatic hydrocarbons (PAH)
B: Earthworm biomass	Mass of earthworms per m^2 soil surface
B: Microbial biomass	Total mass of microorganisms (sum of the mass of bacteria, fungi and protozoa). This sum is an indication of the number of microbes
B: Microbial activity	Soil microbial activity reflects microbiological processes providing important soil functions (e.g., mineralisation of organic matter)

8, Table 2). SALCAsoilquality requires data on site characteristics, fertilisation, pesticide applications, soil tillage, crop rotation, crop residues, machinery usage and grazing animals. The final SQ score is computed separately for arable land and grassland and then added together on an area-weighted basis (see also Fig. 6):

1. Collect input data (LCI).
2. Assign all management practices with similar effects to the corresponding impact classes (classification, e.g. risk of soil erosion, risk of soil compaction by wheeling). Impact classes are generally influenced by more than one management practice.
3. Quantify the effects of each management practice on the impact class.
4. Categorise the calculated values of the impact classes on a five-step scale: highly unfavourable, unfavourable, neutral, favourable and highly favourable.
5. Assign each impact class to the SQIs affected (first-level impact category). The effects of contributing impact classes are weighted according to their relevance and then summed.
6. Categorise SQI values on a five-step scale: highly unfavourable, unfavourable, neutral, favourable and highly favourable.
7. Aggregate the effects on all nine SQIs to a final single SQ score ('second level' midpoint indicator), using a nonlinear aggregation scheme. Since each SQI can limit SQ, a threat to SQ is assumed as soon as an individual SQI is adversely affected. The overall SQ is rated again on a five-step scale.

The SALCAsoilquality method was verified on 14 farms by comparing model results with farmer's own assessments (Marbot 2012). It can be applied to Swiss and similar pedoclimatic conditions (Central/Western Europe).

4.2 Biodiversity (SALCAbiodiversity)

The SALCAbiodiversity model (Jeanneret et al. 2014) was developed as an expert system for including biodiversity (i.e. organismal diversity) in agricultural LCA. It provides a detailed assessment of the impact of agricultural management on eleven indicator species groups (ISG): vascular plants (grassland and crop flora), birds (*Aves*), small mammals (*Mammalia*), amphibians (*Amphibia*), snails (*Gastropoda*), spiders (*Araneae*), carabid beetles (*Carabidae*), butterflies (*Rhopalocera*), wild bees (*Apoidea*) and grasshoppers (*Orthoptera*). The ISGs cover different trophic levels, aboveground ecological niche widths, i.e. seminatural habitats (SNH, e.g. conservation crop margins, hedgerows, groves and litter meadows) and cultivated fields (crops and grassland) and their known response to agricultural management.

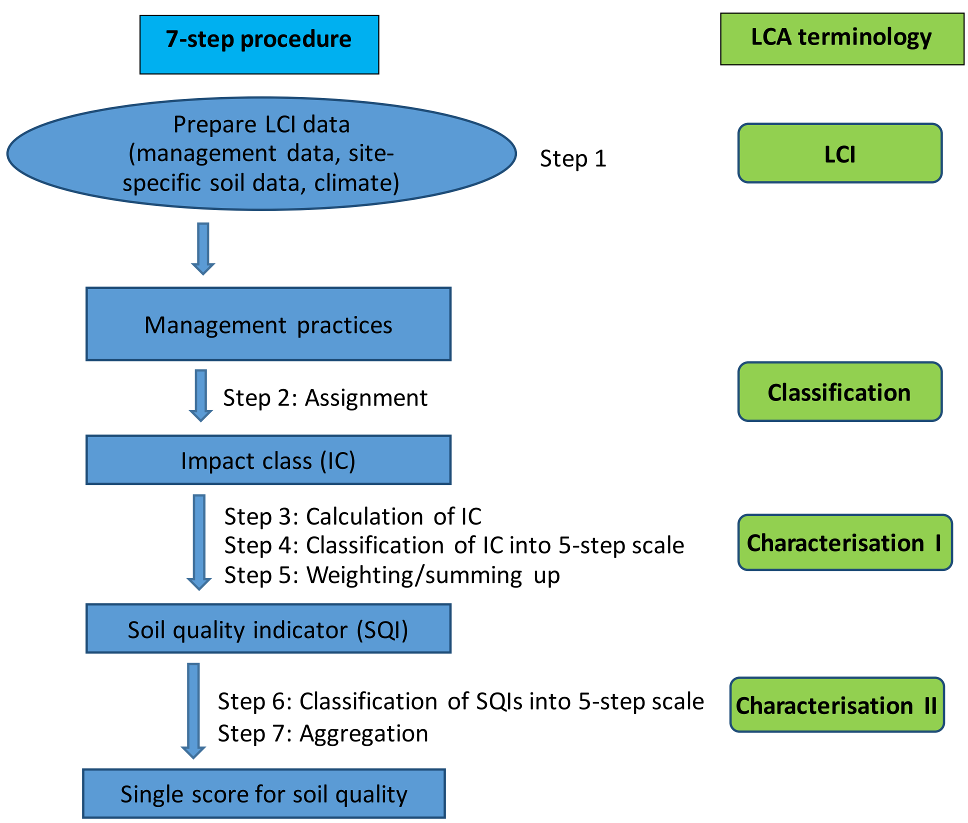

Fig. 6 Overview of the 7-step procedure to calculate the final SQ score. IC, impact class

Two characteristics from the ISG are considered: (1) the overall species diversity within each ISG and (2) ecologically demanding species for amphibians, snails, spiders, carabid beetles, butterflies and grasshoppers (stenotopic species, red list species), resulting in a total of 17 indicators.

The potential effect of agricultural management on ISGs is assessed using a scoring system based on an extensive literature survey and expert knowledge (Jeanneret et al. 2014; Pépin et al. 2023; van Der Meer et al. 2020). A low score indicates a habitat or management option that is unfavourable for biodiversity, and a high score indicates a favourable impact. For each ISG, the score S combines the suitability of the SNHs and the cultivated fields as habitats *per se* (habitat coefficient $C_{habitat}$, from 1 to 10, indicating increasing suitability, e.g. $C_{habitat[\mathrm{potatoes}]}$ for spiders = 4), the importance of the management category (management coefficient $C_{management}$ from 1 to 10, $C_{management[\mathrm{insecticide}]}$ for wild bees = 3) and the response of each ISG to the management options (rating R from 1 to 5; 1 is damaging, 5 is favourable, e.g. two insecticide applications = 3 for carabids in potato fields) occurring in those fields and habitats (Fig. 7):

$$S = R\frac{C_{management} + C_{habitat}}{2} \tag{2}$$

The scores of the eleven ISGs are aggregated to a total biodiversity score by weighting each ISG score on the basis of the trophic relationships between the ISGs and on the species richness of each ISG. Crop protection, fertilisation, crop rotation, soil cultivation, harvesting, grass cutting and grazing are considered management options.

SALCAbiodiversity was validated by comparing the scores with data from field surveys. Both were significantly correlated for vascular plants and grasshoppers (Jeanneret et al. 2014), for spiders and wild bees (Lüscher et al. 2017), although sometimes with high variability.

SALCAbiodiversity is applicable to grasslands, arable crops and SNHs of the farming landscape and enables the estimation of the impact of management systems on biodiversity (Jeanneret et al. 2014). Van Der Meer et al. (2020) adapted the method to fruit orchards, which was applied in Mathis et al. (2022), Pépin et al. (2023) for vegetables and Neel (2021, unpublished master's thesis) for vineyards. With these recent developments, the model covers the main types of agricultural land use, and production systems can be compared to make recommendations for good practice.

The evaluation of the impacts on biodiversity related to products (e.g. functional unit = 1 kg of apples) requires a reference state to be defined and the land use area to be taken into account. Mathis et al. (2022) exemplified this process for apple orchards by setting the average agricultural land use as a reference. An application to dairy products is presented by (Zumwald et al. 2018a, b).

The assessment includes an estimation of the impacts on biodiversity of land under agricultural use (UAA) and does not encompass other land uses, such as forests, urban areas and traffic areas. This method can be applied to Central and Western Europe. For other geographical contexts, a revision of the scoring system might be needed. Moreover, other ISGs and management options might be relevant.

5 Application examples for SALCA

5.1 Scope of application and target audience

The main application of the SALCA methodology is the detailed analysis of the environmental impacts of agriculture. As such, the target audience is manifold: researchers, decision-makers from companies, public authorities, producer organisations,

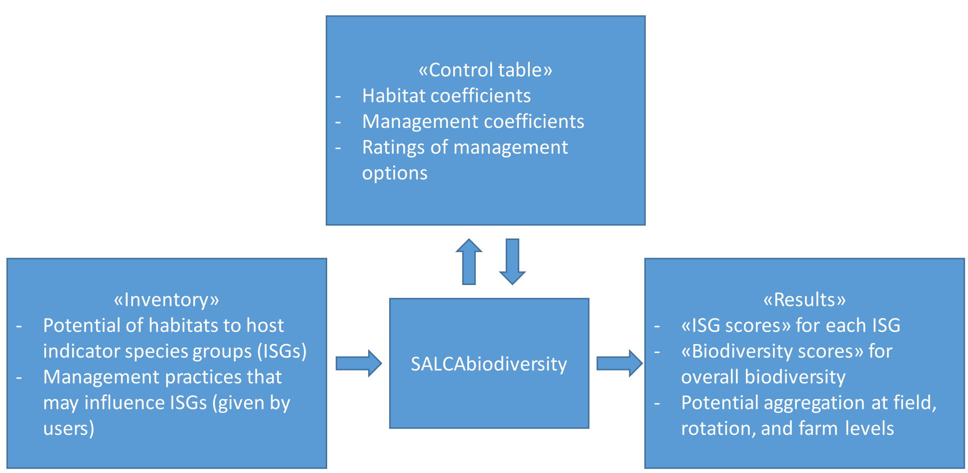

Fig. 7 The SALCAbiodiversity framework. Users provide the inventory file and import the control table into the expert system. The expert system assigns ratings and coefficients from the control table to the inventory data to calculate the final scores (adapted from Pépin et al. 2023). ISG, indicator species group

NGOs, advisors and specialists in agricultural production and environmental assessment. SALCA is not intended for direct use by farmers but can serve as a basis for extension tools. Consumers are not a user group either, but profit from insights gained by research projects that use SALCA.

The following subchapters provide examples of typical applications of SALCA. They comprise several levels of complexity: single crops and their products, crop rotations and cropping systems, animal products, product groups of a farm, LCA of complete farms and the entire agrifood sector. SALCA can be applied for a detailed and specific analysis of foreground systems as well as for the generation of background LCI datasets used as inputs into LCI databases. Although the cited examples stem partly from studies using earlier versions of the model, the insights are still valid.

5.2 LCA of crops or crop products

The simplest level of application of SALCA is the analysis of a single crop and its products. Nemecek et al. (2011b) analysed typical arable crops and grassland types (temporary and permanent grassland) at different intensities in Switzerland. They found that extensive products have lower environmental impacts per area unit, but that impacts can be higher, similar or lower per product unit. A medium intensity was revealed to be optimal for arable crops, while a combination of intensive and extensive management in grassland can reconcile conflicting goals of high productivity and quality with the preservation of biodiversity. Kägi et al. (2007) analysed the suitability of different crops and farming systems for biofuel production (bioethanol, biodiesel and biogas). Organic maize and soybeans performed better than integrated production for bioethanol and biodiesel production, respectively. For cereals and rapeseed, organic farming had no clear advantage. The lowest environmental impacts for biofuel were achieved with biogas production from extensive grassland.

5.3 LCA of cropping systems

A cropping system is highly multifunctional since it includes different crops and their products. Numerous interactions between crops take place in a cropping system. A typical example is the nutrients remaining in the soil or left in harvest residues. The advantage of analysing cropping systems is that the multiple relationships between different crops can be properly reflected, avoiding the need to model nutrient carryover from the previous crop to the following crop (Goglio et al. 2018). Another example is the application of lime for soil improvement or weed control, or the addition of biochar, which is carried out for a certain crop but serves the entire cropping system. Consequently, disaggregation or allocation is needed to analyse the environmental impacts of individual products.

SALCA has been applied in a series of LCA studies of cropping systems. Nemecek et al. (2008) showed that the introduction of grain legumes in European crop rotations reduces climate change, energy demand, eutrophication and acidification thanks to savings of mineral N fertilisers. These results were also confirmed for cropping systems in three regions of France (Nemecek et al. 2015). The environmental impacts of high N fertiliser application rates can best be reduced through the introduction of legumes or reduced N fertilisation; the highest reductions were achieved by combining both options. Nemecek et al. (2011a) compared organic farming to integrated production based on two long-term Swiss experiments. In general, organic farming had similar or lower impacts compared to integrated production, but for some organic products, higher impacts were found per unit of organic product, particularly for impacts related to nutrient losses. To improve eco-efficiency, the optimisation of organic farming should be mainly output-driven (improving yields), while the optimisation of integrated production was found to be input-driven (improving the efficiency of input use). Biodiversity impacts varied more between different grassland management intensities than between arable cropping systems. Soil quality was more influenced by input of organic fertilisers than by the farming system. Lütke-Börding (2016) investigated the environmental impacts of a conventional crop rotation, a crop rotation with reduced tillage and the use of biogas digestate and a diversified crop rotation with the use of biogas digestate in Germany. The environmental performance of the three systems strongly depended on the site conditions. The use of digestate led to lower use of non-renewable resources, but also to higher nutrient emissions. Prechsl et al. (2017) analysed conventional cropping systems vs. organic farming, with intensive tillage, reduced or no tillage and different cover crops. The strongest effect was observed for the farming system, followed by the tillage system and the cover crop. Biodiversity scores were higher for the organic systems, mainly due to the absence of pesticides. Integrated weed management strategies for French arable cropping systems were evaluated by Deytieux et al. (2012). Alternative strategies (except for mechanical weeding) could reduce environmental impacts per area, but these advantages were largely offset by a lower value of the harvested good or a lower income. Both pesticide application and mechanical weeding can have detrimental effects on biodiversity. The largest differences in soil quality indicators were found for the earthworm biomass.

5.4 LCA of animal husbandry systems and animal products

Zumwald et al. (2018a, b) analysed dairy production in different production systems: intensive production with higher amounts of concentrates, less intensive production with low concentrate inputs and full grazing systems. The intensive system with higher inputs of concentrate feed led to higher use of mineral resources, higher deforestation and ecotoxicity impacts. By contrast, more favourable results were found for global warming potential, ozone formation and landscape aesthetics than for the full-grazing system. Marton et al. (2016) investigated the environmental performance of mixed and specialised dairy farming systems. The authors showed that milk produced in collaboration between valley and mountain farms had lower environmental impacts than non-collaborative production.

5.5 LCA of food and feed products

SALCA models have also been applied in studies to calculate the environmental impacts of food supply chains beyond the farm gate. Bystricky et al. (2014) compared wheat bread, feed barley, potatoes, cheese and beef from Switzerland with imports from the most important countries of origin. To ensure consistent results, emissions were recalculated with the SALCA models instead of using the values from the original studies. The results showed that Swiss production has advantages for cheese (due to favourable conditions for grassland-based dairy production) and potatoes (due to shorter transport distances). For the other products, Swiss production had both strengths and weaknesses compared to imports. Bread from diversified low-input systems and distribution systems as alternatives to supermarkets was compared with conventional products (Kulak et al. 2015, 2016). Alternative systems were highly variable; some outperformed the conventional products, while others were clearly less eco-efficient. Alig et al. (2012) showed that meat from animal-friendly and organic production suffers from lower efficiency; this means that more feed has to be used per kg of meat, which increases the impact. This applies mainly to beef and chicken and less to pork. On the other hand, feeding cattle grass-based rations has environmental advantages. An optimised feed ration can effectively reduce the environmental impact of meat production (Wolff et al. 2016).

5.6 LCA of farms and product groups

Analysing the environmental impacts of farms considerably increases the complexity of an LCA study, due to various crops on different fields, several animal groups and interactions between the production branches: feedstuffs produced at the farm are used for animals, animal manure is used as fertiliser in crop production and there are interactions between crops, as explained above. In addition, inputs such as fertilisers, fuels, pesticides, feedstuffs and animals are brought to the farm, and various outputs, such as crop and animal products, manure and animals are exported. Detailed production data for up to 110 farms were collected for three years and analysed by SALCA (Hersener et al. 2011). Environmental impacts were evaluated for three functional units at the farm level: hectare and year, MJ of human digestible energy produced and gross return. In addition, the analysis was carried out for different product groups, where the dry matter of exported products was used as a functional unit for crop production, the amount of milk sold as a functional unit for dairy production and the live weight of animals for other animal product groups. The results revealed great variability in environmental impacts and clearly showed that the dominant product groups largely determined the environmental impact of the farms.

Pedolin et al. (2021, 2023) re-analysed Hersener et al. (2011) dataset by completely updating the LCA calculation and relating it to the economic performance of the farms to investigate economic eco-efficiency using data envelopment analysis. The authors found high variability in both the environmental impacts and the economic performance, which was only partly explained by the farming system (integrated or organic) or the production region.

Repar et al. (2017) analysed the global environmental impacts of Swiss dairy farms per unit of human digestible energy using SALCA. In addition, they calculated local impacts by considering only on-farm impacts and relating these impacts to the agricultural area of the farm to take the local carrying capacity of ecosystems into account. The authors found synergies between global environmental and economic performance and mostly no significant relationship between local environmental and economic performance. Frequent trade-offs were found between global and local environmental performance.

SALCA was integrated into an advisory tool for farms in Austria (HBLFA 2023). The SALCA methodology was adapted to conditions in Austria and tested on 51 farms (Herndl et al. 2016). The FarmLife tool combines LCA results with figures such as calorie production for human consumption or economic outcomes to calculate the eco-efficiency of farms and is used for scientific, advisory and educational purposes. Grassauer et al. (2022a, b) analysed the environmental and economic performance of Austrian dairy farms using the FarmLife tool, with integrated SALCA methodology. Environmental impacts were combined with data envelopment analysis to address the multifunctionality of agriculture. This allowed to identify strategies for improvement of individual farms. Purchased concentrate feed was identified as a main driver of environmental impacts in Austrian dairy production.

5.7 LCA of the agrifood sector and food systems

SALCA models were also used to analyse scenarios at the level of the Swiss agrifood sector. Four strategies to reduce diffuse nutrient losses from agriculture to water bodies were evaluated by a combination of the SWISSland (Möhring et al. 2016), MODIFFUS (Hürdler et al. 2015) and SALCA (Bystricky et al. 2017; Prasuhn et al. 2017) models. All scenarios led to extensification in agricultural production and lower N and P emissions to water. However, domestic production quantities would decrease for many products; consequently, import volumes would rise, leading to a shift in environmental impact to other countries.

Bystricky et al. (2020) analysed the environmental impacts of scenarios in which the use of pesticides and animal numbers would be reduced through direct payment measures. Such measures would lead to a reduction of environmental impacts in Switzerland; however, the total impacts of the agrifood sector would be similar or higher, with the exception of freshwater ecotoxicity, because domestic production quantities would decrease and would have to be replaced by imports.

Combining environmental impacts, calculated by SALCA, with a food system model, von Ow et al. (2020) estimated the potential to reduce environmental impacts by changing food consumption behaviour and optimising the food supply. Nutritional behaviour would need to be changed by substantially reducing the consumption of meat, oils, fats and alcohol on the one hand and increasing the consumption of cereals, potatoes, pulses, fruit and vegetables on the other hand. By changing food consumption combined with an optimisation of the food sector, the environmental impacts could be reduced by more than 50%.

Schader et al. (2013) combined environmental impacts from SALCA with the sector-representative FARMIS model to calculate the cost effectiveness of organic farming support in achieving environmental policy targets. Slightly higher direct payments were needed to reduce environmental impacts with organic farming compared to a combination of three single agri-environmental measures.

5.8 Life cycle inventory databases

SALCA models were also used in leading LCI databases to model agricultural products and production processes, e.g. for ecoinvent (Nemecek and Erzinger 2005; Wernet et al. 2016), AGRIBALYSE (Koch and Salou 2016, 2020) and the WFLDB (Nemecek et al. 2019). The ecoinvent database covers a variety of economic sectors. Various SALCA models were used to model agricultural LCIs in ecoinvent. The SALCA method was applied for the creation of the AGRIBALYSE database, an LCI database focusing on French agricultural production and food consumption (Koch and Salou 2016, 2020). The methodology for crop LCIs was largely based on SALCA in the first version. The SALCA methodology is the methodological backbone of the WFLDB, a global LCI database that represents the production of agricultural primary products and food products (Nemecek et al. 2019). To make the methodology applicable to other climatic regions, adaptations were needed, such as the use of the model de Willigen (2000) for nitrate leaching or the use of EFs for methane from manure storage for different climatic conditions.

6 Discussion

6.1 Strengths and weaknesses of a standardised and harmonised methodology

The examples presented in Chapter 5 illustrate the flexible application options of the SALCA models to answer numerous research questions at the level of a crop, a cropping system, an animal production system, a product group, a farm and the agrifood sector.

The SALCA concept has the following strengths:

- *Comprehensiveness*: SALCA strives to address all the relevant environmental impacts of agriculture. Thus, the method allows a comprehensive assessment of environmental impacts, the identification of hotspots and possible trade-offs between environmental impacts or along the value chain. In addition, a broad application of the most important areas of agricultural production is made possible, as shown in the examples above.
- *Specificity to agriculture*: the models were designed to take specific characteristics of agriculture into account and be applicable in the analysis of crop and livestock production, as well as farms at the level of detail necessary to improve agricultural production systems. Thus, SALCA can also show relatively small differences between similar systems, which supports the development of mitigation measures.
- *Harmonisation*: following a harmonised methodology ensures that the results comply with international standards (European Commission 2018; FAO 2023; ISO 2006a, b; UNEP 2023), which was also demonstrated by three critical reviews (see Chapter 2).
- *Broad applicability:* the same models and data can be used for applications along the value chain (cradle-to-grave approach), as well as on a territory unit, such as a set of farms.
- *Consistency*: application of the methodology ensures the consistency of results within a study but also across studies. Compared to case-by-case modelling, where the risk of inconsistency from individual decisions exists, this also makes documentation easier.
- *Comparability*: consistency also ensures that the results are fully comparable between different product systems, which is a requirement of ISO standards 14,040/44 (ISO 2006a, b).

- *Flexibility*: parametrised models enable the consideration of specific situations. This offers the potential to answer a wide range of research questions in different contexts, as demonstrated in Chapter 5.
- *Modularity*: the modular structure enables the management of the high complexity of agricultural LCA. Existing models can be adapted, and thanks to clearly defined interfaces, alternative models can be used if required by the goal and scope of the study.
- *Testing*: through numerous applications, the tools and methods are tested constantly, and possible methodological and implementation errors can be discovered and eliminated. The modular structure allows independent testing of individual modules.

However, the concept also revealed some weaknesses:

- *High data demand*: a detailed and specific assessment requires a great deal of input data. This challenge can be overcome by using default values for missing parameters, albeit at the expense of accuracy. Sensitivity analyses can help determine influential parameters for which detailed data need to be collected.
- *Complexity* of the models compared to simple screening of LCA models (Zah et al. 2009) and complexity of the whole modelling system.
- *Application limited to experts*: the high complexity and the relatively high data demand makes SALCA too demanding to be used directly to support the decision-making of individual farmers or to be used by consumers. To this end, simplified tools for specific purposes need to be created that could be based on the more complex models presented in this paper. An example of this is FarmLife (Herndl et al. 2016).
- The specificity of the models *limits* their *geographical scope* of application, as discussed in Chapter 6.3.

6.2 Comparison with other LCA tools for agriculture

The SALCA concept was developed as a generic tool to answer research questions in the context of (mainly) foreground-agricultural LCA modelling at the levels of field, crop, animal group and farm (see also Chapter 2), e.g. to assess changes in the management of a cropping system or different feed rations for animals. By using default values and national or regional averages, SALCA can be used to generate datasets for background LCI databases (see Chapter 5.8). This ensures the consistency of foreground and background datasets; this is a requirement of ISO 14040/44, which is difficult to fulfil with generic databases. Tools developed to generate LCI datasets (Auberger et al. 2018; Reinhard et al. 2021) are not specific or detailed enough for the assessment of foreground systems.

Many tools for environmental assessment in agriculture focus on one or more environmental impacts (Hillier et al. 2011; Kayatz et al. 2019; O'Brien et al. 2020). To achieve a comprehensive assessment, several tools need to be combined. These tools can differ in scope, system boundaries, reference flows, methods or data sources used, which creates the risk of inconsistencies. The SALCA concept integrates the different models in a consistent, comprehensive and harmonised framework. To the best of our knowledge, no other LCA method for agriculture has been developed that can model processes at the field, crop, animal group and farm levels within the same system. In addition, it is possible to apply it at the sectoral level and to take upstream and downstream areas of agricultural production into account.

The SALCA models follow the life cycle approach so that all upstream processes until the farm gate are included in the assessment. This is partly done in the models themselves and partly by using background LCI datasets generated using the same or similar methodology.

A trade-off exists between ensuring the global applicability of the models and being sufficiently specific and detailed to capture minor or medium differences between the analysed systems. SALCA has taken the approach of developing models for Western and Central Europe (with a focus on Switzerland); at the same time, the modular structure and the flexible design of the tool also make an adaptation to other geographical contexts feasible, as shown in the next section.

6.3 Geographical coverage

Originally developed for the context of Swiss agriculture, the models can also be applied with minor alterations in regions with similar pedoclimatic conditions, mainly in Central and Western Europe. Generally, the models are suitable for a cool temperate and wet climate (according to definitions of IPCC 2019) and users wishing to apply the models in different contexts should thoroughly check their suitability. To use them in a different geographical or climatic context, adaptations in site-dependent model parameters and equations are required. Some parts of the model are of a general nature and are not site-dependent, such as CO_2 emissions from urea and lime, calculated purely on stoichiometric relationships; these are universally applicable. In other cases, this may require changing the model equations or even changing the entire model. Changing parameters is supported by the modular structure of SALCA and the separate storage of parameters and data tables. Furthermore, the modular structure allows easy replacement of models. In different conditions, considering other indicators or processes might be required, e.g. the inclusion of wind erosion and salinisation in arid regions or different species groups in tropical zones.

Although the main application focus of SALCA is in the context of Swiss agriculture, the model has also been successfully adapted to and applied in other geographical contexts, e.g. in France (Deytieux et al. 2012; Nemecek et al. 2015), Austria (Grassauer et al. 2022a, b; Herndl et al. 2016), Germany (Lütke-Börding 2016) and to cropping systems Nemecek et al. (2008) and animal production systems (Baumgartner et al. 2008) in different European regions.

6.4 Limitations and the need for further development

The SALCA models described in this paper cover cradle-to-farm gate assessments of environmental impacts. To cover downstream processes, such as processing, transports, storage, packaging, retail or consumption, additional modelling steps are required (examples are given in Chapters 5.5 and 5.7). Furthermore, the modelling is limited to the agricultural sector. In studies where agricultural systems must be compared to non-agricultural systems (e.g. biofuels), the methodology needs to be extended. This applies particularly to the assessment of biodiversity and SQ. Currently, the respective SALCA impact assessment methods account only for the foreground system, i.e. the agricultural land directly used by a farm or crop. The impacts of upstream processes, such as feedstuff production or land used for infrastructure, are not taken into account. We propose combining foreground assessment with SALCAbiodiversity with a generic method, such as that of Chaudhary and Brooks (2018) for background systems, and foreground assessment with SALCAsoilquality with Bos et al. (2020) for background systems. This approach will allow for combining the strength of a specific and detailed foreground model with a more generic but less discriminative model for background systems.

The validation of model results is an important yet challenging step. SALCA models are based on experimental data, combined with modelling and expert know-how. Partial validation of selected emission results or impact assessment outcomes has been performed for some of the models, e.g. SALCAnitrate, SALCAbiodiversity and SALCAsoilquality, as explained in this paper. Furthermore, the results of SALCA were discussed with numerous experts in the LCA studies mentioned above and checked for their plausibility. Nevertheless, the robustness of the models would benefit from extended validation in contrasting situations.

Extensive sensitivity analyses would help to concentrate data collection efforts on the most sensitive parameters. Nemecek et al. (2022a) provide an example of the modelling of pesticide emissions. This is, however, a challenging task, as the sensitivity of emissions and impacts is highly dependent on the goal, scope and context of the study. It is generally recommended that a sensitivity analysis be run before data collection starts to ensure that the resources are invested in the most influential parameters. Contribution analyses of the environmental impacts of various agricultural systems help to set priorities for further development. Examples are shown in Chapter 5 (Bystricky et al. 2015; Nemecek et al. 2011a; Zumwald et al. 2018a). Uncertainty analyses were carried out in several of the studies (Alig et al. 2012; Bystricky et al. 2020; Nemecek et al. 2005); however, there is a need for further development of tools and databases to provide a reliable assessment of model and data uncertainty. Finally, there are a number of situations outside the models' range of validity. Examples are nitrate leaching on organic soils with > 15% humus content or the assessment of SQ in horticulture (vegetables, fruit orchards and vineyards).

The models presented in this paper take a number of processes and parameters into account; however, other potentially relevant aspects are currently not explicitly included. An example is the lack of explicit inclusion of the soil–water balance in SALCAnitrate. It is indirectly addressed by precipitation during the cold season relative to the Swiss average; however, the model could be improved by explicitly modelling this process, thus allowing for robust applications in different pedoclimatic contexts. Another example is the modelling of N_2O emissions from soil. More detailed models have been developed that consider the effects of climate, SOC, soil pH, drainage, soil texture, crop type, etc. (Albanito et al. 2017; Li et al. 2022; Rochette et al. 2018; Stehfest and Bouwman 2006), which can be used to refine the modelling. However, the uncertainty of these effects is high and not all of them have been shown to be significant (IPCC 2019). The integration of further site-specific parameters can help to develop optimised systems for different regions and thus to contribute to the mitigation of environmental impacts.

6.5 Extension to a comprehensive sustainability analysis

The SALCA models calculate field and farm emissions and enable the assessment of environmental impacts. To achieve a comprehensive assessment of sustainability, we also need to take the economic and social dimensions into account. The SALCAsustain methodology (Roesch et al. 2017) was developed to enable a sustainability assessment of farms. It was applied to a set of Swiss farms, and the relationships between different sustainability indicators were analysed (Roesch et al. 2021). Other model sustainability assessment methods developed in Switzerland include SMART (Schader et al. 2016; 2019) and RISE (Grenz 2017). They are applied globally to provide advice on sustainability aspects at the farm level. They work mainly with qualitative indicators, while SALCAsustain provides quantitative results (Roesch et al. 2018). Furthermore, SALCAsustain follows the life cycle approach for the environmental dimension, while the other tools do not include upstream processes, with the exception of some social indicators. Different system boundaries and reference flows are used in RISE and SMART, depending on the indicator chosen.

7 Conclusions

The proposed SALCA concept comprises models to calculate direct field and farm emissions, criteria for their applicability, impact assessment methods specific to agricultural applications, a database with life cycle inventories, a software environment for data collection and LCA calculation, a concept to analyse and interpret LCA results and a concept to communicate the results, conclusions and recommendations. It enables answering a wide range of research questions and is applicable to different goals and scopes: for the assessment of crops and their products, of cropping systems, of animal production and resulting products, for food production, for farms and for the agrifood sector as a whole. The SALCA models, which are the focus of the current paper, comprise models to calculate emissions of gaseous N, nitrate leaching, P emissions to water, soil erosion, pesticides, heavy metals and emissions from animal production, as well as SQ and biodiversity impact assessment methods in a harmonised system.

The strengths of SALCA are the comprehensive assessment of environmental impacts, taking into account specific aspects of agriculture, a harmonised and consistent framework, a flexible and modular modelling system, which allows managing the complexity of agricultural LCA along the value chain and through different territorial units, testing of independent modules and easy adaptation to other contexts of application. A weakness is the high demand for data and expert know-how. As a research concept, the use of SALCA is limited to user groups with sufficient expertise and makes the method unsuitable for direct use by farmers or consumers. The geographical scope is limited to Central and Western Europe, with a special focus on Switzerland. However, due to the modular and flexible design, an adaptation to other contexts is feasible with reasonable effort, as shown by various examples of applications throughout Europe. For applications to different conditions, e.g. in tropical regions, the concept provides a valuable basis, but an appropriate parametrisation is required, or the emission models can be completely replaced by alternative models. The need for further research and development includes the modelling of downstream processes in the food sector, such as processing, transport, storage, packaging, retail and consumption. A specific challenge is the assessment of land use-related impacts on biodiversity and SQ. Here, we propose combining the SALCA models for the foreground system with generic models having global applicability for the background systems. For certain research questions, an extension beyond the agricultural sector is needed. Furthermore, model validation and extensive sensitivity or uncertainty analysis would increase the robustness and accuracy of the environmental assessment results. Finally, extending the environmental assessment to a full life cycle sustainability assessment by including the economic and social dimensions, as proposed by Roesch et al. (2017), would allow for revealing trade-offs between the dimensions of sustainability, thus making SALCA even more useful for decision-makers.

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Declarations

Competing interests The authors declare no competing interests.

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