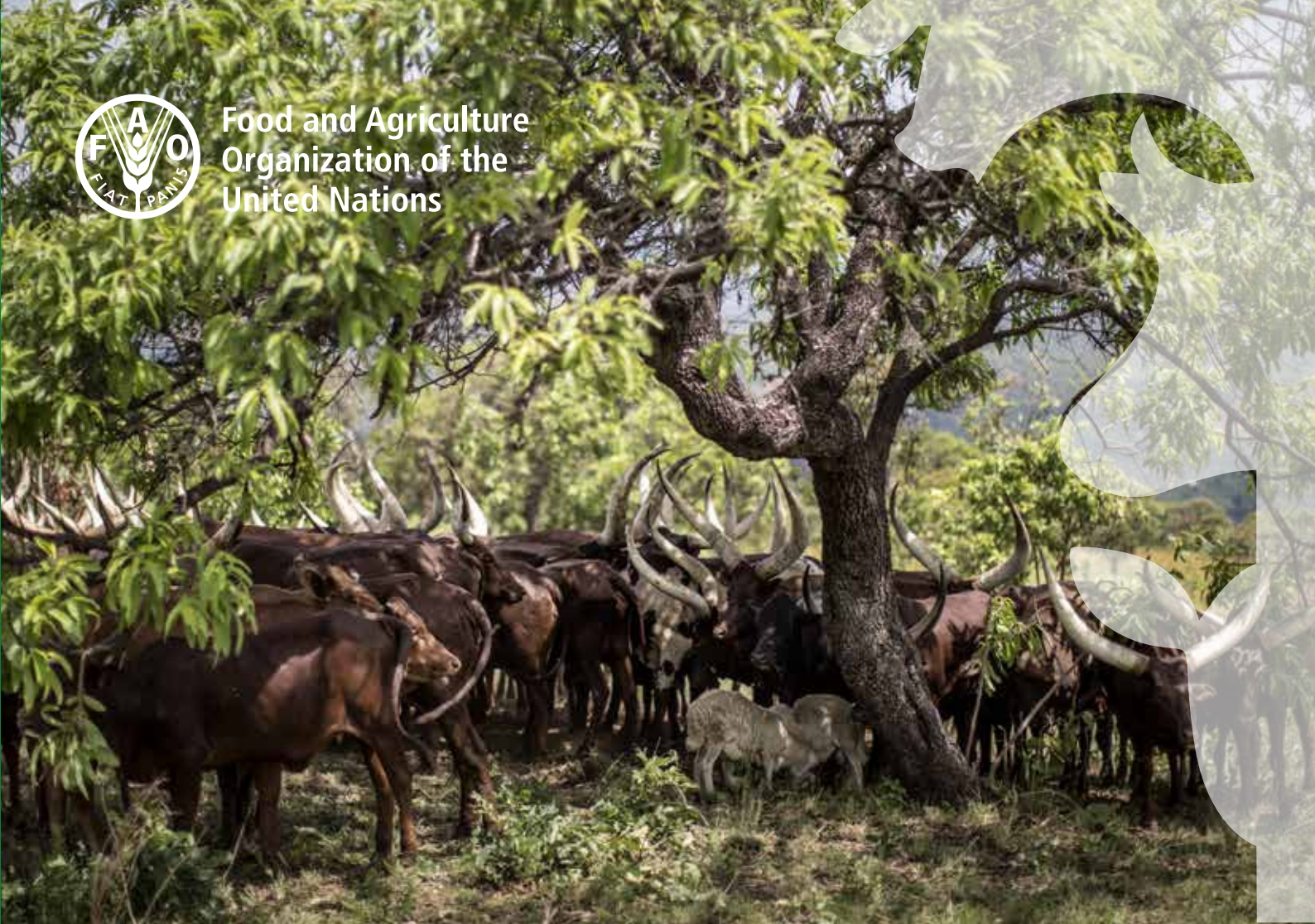




Food and Agriculture
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United Nations



DRAFT FOR PUBLIC REVIEW

Biodiversity and the livestock sector

Guidelines for quantitative assessment



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Guidelines for quantitative assessment

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1 Acknowledgements

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3 Abbreviations

BAP	Biodiversity Action Plan
BDP	Biodiversity Damage Potential
CBD	Convention on Biological Diversity
CF	Characterization Factor
EC	European Commission
EDP	Ecological Damage Potential
EEA	European Environment Agency
FAO	Food and Agriculture Organization of the United Nations
GAP	Good Agricultural Practices
GHG	Greenhouse Gas
GRI	Global Reporting Initiative
HCW	Hot Carcass Weight
HDPE	High-density Polyethylene
ISO	International Organization for Standardization
IUCN	International Union for Conservation of Nature
KPI	Key Performance Indicator
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LEAP	Livestock Environmental Assessment and Performance Partnership
LPI	Living Planet Index
LU	Land Use
LUC	Land Use Change
LULCIA	Land Use Life Cycle Impact Assessment
LULUC	Land Use and Land Use Change
LW	Live Weight
MEA	Millennium Ecosystem Assessment
NGO	Non-Governmental Organization
OECD	Organization for Economic Cooperation and Development
PDF	Potentially Disappeared Fraction (of species)
PNV	Potential Natural Vegetation
PSL	Potential Species Loss
PSR	Pressure-State-Response
SETAC	Society for Environmental Toxicology and Chemistry
SOM/SOC	Soil Organic Matter/Carbon
TAG	Technical Advisory Group
UN	United Nations
UNEP	United Nations Environment Programme

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2 Glossary

Biodiversity	Variability among living organisms from all sources including, <i>inter alia</i> , terrestrial, marine and other aquatic systems and the ecological complexes of which they are part, including diversity within species, between species and of ecosystems. [Article 2 of the CBD]
Biome	The world's major communities classified according to the predominant vegetation and characterized by adaptations of organisms to that particular environment. For instance, tropical rainforest, grassland, tundra. [Campbell 1996]
Characterization	Calculation of the magnitude of the contribution of each classified input/output to their respective impact categories, and aggregation of contributions within each category. This requires a linear multiplication of the inventory data with characterization factors for each substance and impact category of concern. For example, with respect to the impact category "climate change", CO ₂ is chosen as the reference substance and kg CO ₂ -equivalents as the reference unit. [Adapted from Product Environmental Footprint Guide, European Commission, 2013]
Characterization factor	Factor derived from a characterization model which is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator. [ISO 14040:2006, 3.37]
Conservation value (high)	Conservation value is a concept used to prioritize conservation efforts. Several factors can determine if a species (or ecosystem) is of high conservation value: the endangerment, risk or uniqueness; the functional contribution and potential to provide ecosystem services; the extrinsic value to local populations and stakeholders... Because the value of biodiversity is subject to value judgement, conservation value should be defined through stakeholder engagement. [LEAP biodiversity TAG and]
Data quality	Characteristics of data that relate to their compliance with stated requirements. [ISO 14040:2006, 3.19]
Ecoregion	Relatively large units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change [Olson et al. 2001]
Ecosystem	A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional entity [Article 2 of the CBD]

Ecosystem services	The benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual and recreational benefits; and supporting services such as nutrient cycling that maintain the conditions for life on Earth. [MEA 2005]
Elementary flow	Material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation. [ISO 14040:2006, 3.12]
Endemism	Association of a biological taxon with a unique and well-defined geographic area. [The Encyclopedia of Earth, http://www.eoearth.org]
Endpoint impact category	Attribute or aspect of natural environment, human health, or resources, identifying an environmental issue giving cause for concern [ISO 14040:2006, 3.36]
Environmental impact	Any change to the environment, whether adverse or beneficial, wholly or partially resulting from an organization's activities, products or services. [ISO/TR 14062:2002, 3.6]
Functional unit	Quantified performance of a product system for use as a reference unit. [ISO 14044:2006, 3.20] It is essential that the functional unit allows comparisons that are valid where the compared objects (or time series data on the same object, for benchmarking) are comparable.
Habitat	The place or type of site where an organism or population naturally occurs. [Article 2 of the CBD]
Hotspot analysis	Hotspot analysis aims to define areas of high occurrence versus areas of low occurrence of a feature of interest. Here, it refers to an assessment of the relative contribution of different elements (locations, steps of a supply chains, types of pressure), with the aim of identifying those that make the strongest contribution to biodiversity loss. [LEAP Biodiversity TAG]
Hotspot (of biodiversity)	Biogeographic region that is both a significant reservoir of biodiversity with high concentration of endemic species and is threatened with destruction. A more restrictive and quantitative definition is that a hotspot should have lost 70% or more of its primary vegetation and host at least 0.5% of the world's plant species as endemics [Meyers et al., 2000]
Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned. [ISO 14040:2006, 3.39]

Invasive alien species	An alien species whose introduction and/or spread threaten biological diversity [Convention on Biological Diversity]
Land occupation	Life cycle inventory flow related to use of a land area by activities such as agriculture, roads, housing, mining, etc. [Adapted from Product Environmental Footprint Guide, European Commission, 2013]
Land use change	Change in the purpose for which land is used by humans (e.g. between cropland, grassland, forestland, wetland, industrial land). [PAS 2050:2011, 3.27]
Life cycle	Consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal. [ISO 14040:2006, 3.1]
Life Cycle Assessment	Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle. [ISO 14040:2006, 3.2]
Life Cycle Impact Assessment (LCIA)	Phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product [ISO 14040: 2006, 3.4]
Life Cycle Inventory (LCI)	Phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle [ISO 14040: 2006, 3.3]
Livestock	Domesticated animals raised on a farm to produce labour or commodities (e.g., meat, milk, eggs, wool).
Midpoint impact category	Environmental impact category located between the life cycle inventory (human interventions) and the endpoints (final indicators, under areas of protection), for instance climate change or acidification.
Pressure, State and Response framework	The PSR indicator framework provides a way to structure indicators which facilitates interpretation and decision making. The PSR framework is based on causality. Indicators evaluate the pressures of human activities (e.g. pollution, habitat change, climate change) that lead to changes in the state of biodiversity (e.g. species abundance, richness or composition, ecosystem degradation), causing responses (decision and actions) from the stakeholders (political, socio-economic), aimed at reaching a more sustainable state.
Semi-Natural Habitats	Permanent woody or herbaceous areas in agricultural landscapes such as permanent grasslands, grassy field margins or ditch banks, tree or shrub hedgerows, woodlands.

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2 Grassland terminology [Allen et al., 2011]

Cultivated grassland	Forage is established with domesticated introduced or indigenous species that may receive periodic cultural treatment such as renovation, fertilization or weed control.
Grassland	The term 'grassland' is synonymous with pastureland when referring to an imposed grazing-land ecosystem. The vegetation of grassland in this context is broadly interpreted to include grasses, legumes and other forbs, and at times woody species may be present.
Meadow	A natural or semi-natural grassland often associated with the conservation of hay or silage.
Native or semi-natural grassland	Natural ecosystem dominated by indigenous or naturally occurring grasses and other herbaceous species used mainly for grazing by livestock and wildlife.
Pastureland	Land (and the vegetation growing on it) devoted to the production of introduced or indigenous forage for harvest by grazing, cutting, or both. Usually managed to arrest successional processes.
Rangeland	Land on which the indigenous vegetation (climax or sub-climax) is predominantly grasses, grass-like plants, forbs or shrubs that are grazed or have the potential to be grazed, and which is used as a natural ecosystem for the production of grazing livestock and wildlife.
Semi-natural grassland	Managed ecosystem dominated by indigenous or naturally occurring grasses and other herbaceous species.

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Summary of key messages and guidelines

Introduction

- Biodiversity is essential to agriculture and human well-being, but it is declining at an unprecedented rate.
- Depending on the ecological context and land use history, livestock is either among the most harmful threats to biodiversity or necessary to maintain high nature value farmland.
- Including biodiversity in environmental assessments is challenging, mainly due to its intrinsic complexity, scale issues and the significant difficulty associated with reducing biodiversity assessment to a single measure or conservation objective.
- Quantitative indicators and assessment methods are needed to assess biodiversity along with other environmental criteria, to meet international commitments on biodiversity and avoid the risk of burden shifting among environmental criteria.

Objective and scope of this document

- The objective of this document is to develop guidelines for quantitative assessment of the effects of livestock production on biodiversity, based on existing indicators and methods.
- Indicators and methods described in these guidelines are relevant to a range of assessment objectives, users, scales, geographical regions, livestock species and production systems.
- This document focuses on biodiversity at the species level and discuss links with the ecosystem level.

General information on livestock & biodiversity

General ecological principles

- Biodiversity describes the variability of life on earth and is often positively affected by intermediate levels of disturbance as this often opens up new niches (i.e. a variety of conditions and resources) for a greater diversity of species to become established.
- Extensively managed and low input livestock systems can maintain intermediate levels of disturbance, and be of high nature value, where past land use disrupted natural disturbance regimes, and replaced wild with domestic herbivores.
- Inappropriate management practices can occur in both low-input extensive (e.g., overgrazing, abandonment) and high-input intensive (e.g., off-farm feed produced in simplified landscapes, nutrient pollution due to animal concentration) systems and will determine impacts on biodiversity per unit of livestock product. In addition, several indicators can reflect negative impact on biodiversity from extensive systems, such as soil erosion, degraded soil and livestock density.

Ecosystem services

- Ecosystem services (ES) are the outcomes from ecosystems that lead to benefits valued by people. Agroecosystems are both providers (e.g. food production but also soil fertility, water regulation, soil

carbon storage) and beneficiaries (e.g. vegetation productivity, pollination, pest control, water availability) of ES.

- Biodiversity plays a key role in ES provision as a regulator of ecosystem processes, an ecosystem service, and a good or benefit.
- In livestock production systems, there can be synergies and trade-offs between biodiversity and the different types of ES, for instance intensification is often associated with higher food production (a provisioning service), but lower biodiversity or regulating services (water quality, soil carbon).
- Temporal trade-offs also exist – highly productive systems can have an important impact on biodiversity and regulating ecosystem services, potentially damaging the natural processes that are essential for food production, leading to collapse of the system over the long term.
- To date considerable effort has been devoted to quantifying the life cycle impacts of products on ES (this section provides a number of methods), but key challenges remain unsolved.

Genetic biodiversity

- The assessment of livestock genetic resources is beyond the scope of this document but Section 3.2 provides background information and data sources on this aspect of biodiversity.
- More than 8800 livestock breeds have been recorded globally, representing a valuable resource and a high biodiversity at the genetic level.
- Animal genetic resources remain at risk and many drivers of loss are common with those of wild biodiversity, such as demand, intensification, degradation of natural resources, and climate change.
- The Domestic Animal Diversity Information System (DAD-IS) collects information on animal genetic resources from 182 countries and provides a searchable database of information related to livestock breeds.

Definition of the assessment goal and method

- Goal definition is the first step of the assessment and all further steps (scope, data, methods, results and conclusions) should align with the defined goal
- This document frames potential assessments within three major scales – global, regional and local – and using two main methods – Life Cycle Assessment (LCA) and Pressure, State, Response (PSR) indicators. Selection of the method depends on the overarching goal of the assessment, its scale and its constraints.

LCA regional and global assessments

Impact pathway (Cause-effect chain)

- This section describes the impact pathway used in the document, i.e. the conceptual cause-effect chain that links inventory flows associated with livestock production (e.g., land use, nutrient inputs, water use) to resulting impacts on biodiversity.
- Life cycle impact assessment (LCIA) models translate inventory flows into specific biodiversity indicators; the impact pathway points to the main LCIA model recommended by these guidelines (Chaudhary and Brooks 2018) and to alternative models.

- Not all inventory flows detailed in the impact pathway are necessary to the different LCIA models, but it is recommended that as much information as possible on inventory flows is collected and reported.

Functional unit

- In LCA, impacts on biodiversity will always be expressed in relation to a functional unit – e.g., per litre of milk or kg of carcass or protein – to ensure system definition and comparability.
- For biodiversity assessments using LCA, the recommended functional units for different livestock supply chains (i.e. species, commodities) are the ones indicated in the LEAP sectoral guidelines (FAO 2015, 2016); for instance at farm gate, kg fat and protein corrected milk or kg of live weight.

System boundaries

- While carrying out an LCA study, a flow diagram of all assessed processes in the livestock production system should be drawn, indicating system boundaries.
- The recommended system boundaries are those indicated in the LEAP sectoral guidelines (FAO 2015, 2016) and typically encompass all stages of production, from raw material extraction to the primary processor gate.

Scale of assessment

- LCA is well suited to consider complex supply chains encompassing geographically distributed locations, that are common to most livestock sectors at both the regional and global scale.
- Currently, LCA is not well suited for assessing local biodiversity effects as global data lacks site-specific resolution.
- This document recommends several LCIA models for regional to global assessment but the PSR approach is more suitable for local assessments – complementarities between the two approaches are discussed.

Description of biodiversity indicators in LCIA

- Different LCIA models can address different biodiversity indicators such as species richness, abundance or functional diversity.
- The main LCIA model recommended in this document (Chaudhary and Brooks 2018) uses the potentially disappeared fraction (i.e., impact on species richness) as an indicator.
- Other models can be used to address different biodiversity components, if justification and discussion is provided.

Life cycle inventory

- Land use (in $\text{m}^2 \times \text{years}$) and land use change (in m^2 transformed from one land use class to another) are the main inventory flows that should be collected in the context of a LCA assessment using these guidelines.

- Inventory flows should be spatially differentiated – i.e. the location where they occur should be known with as high of precision as possible.
- These guidelines detail several levels of differentiation between land use categories; the highest level of differentiation possible should be used.
- For regional/global assessments, the main LCIA model recommended by these guidelines differentiates between 15 land use classes and spatially between more than 800 ecoregions.

Life cycle impact assessment models

- Chaudhary and Brooks (2018) model provides characterisation factors (CFs) reflecting regional or global species extinctions for different taxa; these guideline recommend using the global taxa-aggregated CFs.
- Recommendations in these guidelines are consistent with those of the UNEP-SETAC.
- One important limitation of the recommended LCIA model is its limited ability to reflect beneficial impacts on biodiversity – this should be discussed as part of results interpretation and the use of complementary PSR indicators is recommended in an attempt to overcome this limitation.

Reference state

- In LCIA, impacts on biodiversity are always expressed compared to a reference state.
- The potential natural vegetation (PNV, which describes the mature state of vegetation in the absence of human intervention) is often used as a reference state, but a historic reference or the current mix of land use can also be chosen.
- The CFs recommended by this document (Chaudhary and Brooks 2018) use the PNV as a reference state.
- The reference state decision has important implications for the results. Both the reference state and these implications should be reported and discussed, especially when using different methods.

Local assessments using PSR indicators

The framework for local assessments

- When conducting a local biodiversity assessment using the PSR framework, 5 major steps can be considered – (1) definition of goal, (2) definition of scope, (3) indicator identification, (4) data collection and analysis and (5) interpretation and communication – along with stakeholder engagement that should occur iteratively along the 5 steps.

Within the 5 major steps, key principles to consider are the following:

- The objectives of a biodiversity assessment and the objectives of any related initiatives shall be clearly stated and appropriate indicators and methodologies chosen to reflect these objectives. The intended use of the results shall also be specified.
- A scoping and a hotspot analysis shall be conducted. The scoping analysis consists of a preliminary assessment of the scope and dimension of the study, in order to map key concepts and issues, identify gaps and challenges related to biodiversity and livestock production. The hotspot analysis aims to provide a qualitative evaluation of the relative contribution of the livestock system to different biodiversity issues, and to identify the most prominent positive and negative impacts.

- The boundaries of the assessment shall be clearly defined. Processes such as feed production, in particular off-farm feed production, shall be included in the system boundaries of livestock systems. This is due to feed's substantial contribution to the overall impact on biodiversity.
- Under the PSR model, it is necessary to select specific pressure, state and/or response indicators to respectively describe the pressures from human activities on the environment, the resulting changes in environmental conditions, and the societal response to environmental concerns, either to mitigate negative effects, to reverse damages, or to conserve habitats and biodiversity.
- Given the context-dependency of biodiversity conservation, engagement with multiple stakeholders (i.e., anyone who may be impacted by, or have an impact on, an issue) can improve several facets of the assessment, including goal definition, scoping/hotspot indicators, indicator selection, data collection/analysis and interpretation of results.
- Indicators are identified and prioritized for the biodiversity assessment, based on expert and stakeholder input and relevant resources.
- Relevant information shall be identified and a plan for data collection developed to be able to compute the selected indicators.
- The impacts on biodiversity can be identified through analysis of data collected for the chosen indicators and presentation of results. Data analysis and presentation of results shall be undertaken by personnel with appropriate expertise.
- Interpretation shall be aligned with the goal of the assessment, identify issues, guide decision making for improving biodiversity performance, and discuss limitations. Interpretation shall be undertaken by personnel with appropriate expertise and consideration for data quality and results.
- Communication of the assessment results shall ensure transparency and be adapted to the target audience

Recommended list of biodiversity indicators for local assessments

- This section provides a list of recommended pressure, state and response indicators addressing key thematic issues that were identified: habitat protection, habitat degradation, wildlife conservation, invasive species, aquatic biodiversity, off-farm impacts and landscape scale conservation.
- The indicators in the list are recommendations and not requirements. Users shall consider each of the indicators in turn, and provide a short justification of why an indicator is selected or not, or why an alternative indicator is used.

As a good practice, the selected indicators shall include:

- All indicators related to 'procedural checks';
- At least one indicator from each category (i.e., pressure, state and response) to show if actions do have an effect on decreasing pressure and improving the state of biodiversity;
- At least one indicator for each of the thematic issues identified as relevant during the scoping and hotspot analyses;
- Indicators reflecting both positive and negative impacts on biodiversity;
- Indicators covering off-farm impacts when relevant.

Interpretation and communication

Interpretation

- The interpretation stage makes use of available evidence to evaluate, draw conclusions, and inform specific decision and policy making contexts.
- Interpretation shall be aligned with the goal and scope of the assessment.
- The limitations to robustness, uncertainty and applicability of the assessment results also shall be explicitly acknowledged and discussed.

Communication

- A major success factor in maintaining and improving sustainability (including biodiversity) is an effective knowledge transfer strategy, and the achievement of cultural awareness and appreciation of biodiversity
- Information provided shall be transparent about the aims and methods of an assessment
- For transparent communication, the limitations of an assessment shall be clearly described and discussed

Policy implications

- LCA has arisen as a structured, comprehensive, internationally standardized tool that is capable of offering objective data for use as an environmental decision support; however, there is a risk for a decision-maker to assume that LCA generates simple answers to complex environmental questions, like biodiversity impacts for which describing the complexity with models remains a challenge.
- It is critically important to model impacts at adequate spatial and temporal scales, particularly by using more accurate local and regional data, and to use appropriate indicators to address policy- and decision-making processes, bearing in mind that a specific indicator for one biodiversity level or dimension such as species composition are not fully adequate to depict linkages between ecosystem function, biodiversity and ecosystem services.

Data and data quality

Introduction

- Biodiversity data shall be aligned with the scale at which the analysis will be conducted, when relevant, and/or be scalable to enable cross-scale analyses.
- When using data at a large geographical scale the risk of simplification, lack of specificity and not considering all aspects and interactions shall be minimized.
- When using data at a small geographical scale, the risk of lacking representativeness and over generalization shall be minimized.

Representativeness

- Data used in biodiversity assessment shall be representative regarding three main aspects: time, space and taxa.
- Representativeness shall be considered when designing the sampling procedure for data collection.

Data quality assessment

- Data quality shall be assessed, reported and discussed.
- Data quality assessment shall include several key criteria – precision, error, completeness, consistency, reproducibility and uncertainty.
- Databases supporting biodiversity assessment in livestock should ideally be made open-access.

Existing data sources

- This section provides a number of sources of global and regional data; other sources can also be used if sufficient information is provided to assess their representativeness and quality.
- Key aspects of global and regional datasets are their spatial/temporal extent and resolution; there are frequently trade-offs among these dimensions which shall be considered and justified when selecting data matching the assessment goals.
- With local data, accessibility is an important issue and engagement of data owners as stakeholders in study design, including data handling provisions, is likely to aid data access.

Preparation process

The LEAP Partnership

LEAP Partnership is a multi-stakeholder initiative launched in July 2012 with the goal of improving the environmental performance of livestock supply chains. Hosted by the Food and Agriculture Organization of the United Nations, LEAP brings together the private sector, governments, civil society representatives and leading experts who have a direct interest in the development of science-based, transparent and pragmatic guidance to measure and improve the environmental performance of livestock products. The first phase of LEAP Partnership (2012-2015) focused mainly on the development of guidelines to quantify the greenhouses gas emissions, energy use and land occupation from feed and animal supply chains as well as the principles for biodiversity assessment. The second phase (2016-2018), known as LEAP+, broadened the scope and is focusing on water footprinting, nutrient flows and impact assessment, soil carbon stock changes, quantification of the impact of livestock on biodiversity and assessment the effect of feed additives on GHG emissions.

In the context of environmental challenges such as climate change and increasing competition for natural resources, the projected growth of the livestock sector in coming decades places significant pressure on livestock stakeholders to adopt sustainable development practices. In addition, the identification and promotion of the contributions that the sector can make towards more efficient use of resources and better environmental outcomes is also important.

The LEAP Partnership addresses the urgent need for a coordinated approach to developing clear guidelines for environmental performance assessment based on international best practices. The scope of LEAP is not to propose new standards, but to produce detailed guidelines that are specifically relevant to the livestock sector, and refine guidance as to existing standards. LEAP is a multi-stakeholder partnership bringing together the private sector, governments and civil society. These three groups have an equal say in deciding work plans and approving outputs from LEAP, thus ensuring that the guidelines produced are relevant to all stakeholders, widely accepted and supported by scientific evidence.

The work of LEAP is challenging but vitally important to the livestock sector. The diversity and complexity of livestock farming systems, products, stakeholders and environmental impacts can only be matched by the willingness of the sector's practitioners to work together to improve performance. LEAP provides the essential backbone of robust measurement methods to enable assessment, understanding and improvement in practice. More background information on the LEAP Partnership can be found at www.fao.org/partnerships/leap/en/.

TAG on biodiversity

The TAG on biodiversity was formed in June 2017. The core group included 25 international experts in ecology, biodiversity indicators, agronomy, life cycle assessment, livestock production systems, and environmental science. Their backgrounds, complementary between systems and regions, allowed them to understand and address different perspectives. The TAG was led by Tim McAllister (Agriculture and Agri-Food Canada and the University of Alberta), assisted by Félix Teillard, technical secretary of the TAG.

1 The TAG met in two workshops. The first was held on 18-20 September 2017 at FAO in Rome, Italy and
2 the second was held on 22-26 January 2017 at ILRI, Nairobi, Kenya. Between and after the workshops, the
3 TAG worked via online communications and teleconferences.

5 **Period of validity**

6 It is intended that these guidelines will be periodically reviewed to ensure the validity of the information
7 and methodologies on which they rely. At the time of development, no mechanism is in place to ensure
8 such review. The user is invited to visit the LEAP website (www.fao.org/partnerships/leap) to obtain the
9 latest version.

Part 1. About this document

1 Introduction

Key messages

- Biodiversity is essential to agriculture and human well-being, but it is declining at an unprecedented rate.
- Depending on the ecological context and land use history, livestock is either among the most harmful threats to biodiversity or necessary to maintain high nature value farmland.
- Including biodiversity in environmental assessments is challenging, mainly due to its intrinsic complexity, scale issues and the significant difficulty associated with reducing biodiversity assessment to a single measure or conservation objective.
- Quantitative indicators and assessment methods are needed to assess biodiversity along with other environmental criteria, to meet international commitments on biodiversity and avoid the risk of burden shifting among environmental criteria.

1.1 Background

Global biodiversity is essential for ecosystem functioning, service provision and human well-being, but is declining at an unprecedented rate of over 100 times the normal rate prevailing between previous mass extinctions (Pimm et al. 2014). This decline is primarily due to habitat loss driven by human conversion of natural ecosystems to other land uses, mainly for producing commodities for consumption, providing transportation corridors and urbanization. To date, international agreements have slowed down, but have not completely halted this loss (Tittensor et al. 2014). It is increasingly clear that to mitigate this crisis, traditional interventions such as establishing protected areas (Watson, et al. 2014) and addressing the direct drivers (e.g., habitat loss, pollution), need to be complemented by policies addressing the underlying (indirect) drivers (MEA 2005; CBD 2014; Gibbs et al. 2015; Lenzen et al. 2012). A first step in this direction is environmental footprinting, i.e., quantifying the impact of individual commodity production activity on biodiversity and informing the producers, consumers and other stakeholders of the impact to promote the adoption of more sustainable management practices.

Livestock is among the sectors with the highest impact on biodiversity. Around 22% of ice-free land on Earth is used for pastures (18%) and feed crops (4%) (Mottet et al. 2017), that result in habitat modification and biodiversity change. Livestock also contribute to climate change – the second most important driver of global biodiversity loss (MEA 2005, CBD 2014) – by releasing about 14.5% of global anthropogenic greenhouse gases (GHG) (Gerber et al. 2013, using 100-year IPCC global warming potential to convert CH₄ and N₂O into CO₂ equivalents). However, an important specificity of the livestock sector is that its impacts on biodiversity can also be positive. For instance, extensive livestock grazing can be the only way to maintain semi-natural habitats hosting a unique pool of wild species and providing key ecosystem services (e.g., in temperate, Pogue et al. 2018; tropical, Overbeck et al. 2007; and semi-arid, Milchunas et al. 1989 grasslands).

Despite the strong relationship between livestock production and biodiversity, many assessments and initiatives on the environmental performance of the livestock sector have had a strong focus on GHG emissions (Roma et al. 2015) and biodiversity assessment has been largely ignored. This is mainly due to the intrinsic complexity of biodiversity, scale issues (e.g., context-dependency) and the significant

1 challenges associated with reducing biodiversity assessment to a single measure or conservation
2 objective.

3 The inclusion of biodiversity in environmental assessment is an emerging but increasingly important area
4 of work. Several recent initiatives have attempted to address the relationship between biodiversity and
5 livestock production. In the UNEP-SETAC life cycle initiative, a specific task force worked on the inclusion
6 of land use impacts on biodiversity in Life Cycle Assessment (LCA). Other initiatives on biodiversity
7 assessment exist at the global (CBD biodiversity indicators for commodity production, FAO-UNEP 10 Year
8 Framework Programme core initiative on biodiversity), regional (European Union Product Environmental
9 Footprint) and sectoral (Cool Farm Tool, Sustainable Agriculture Initiative Platform, International Dairy
10 Federation) levels. There is a need to ensure that the livestock sector and its specific relationship with
11 biodiversity (e.g., positive impacts) are not left behind in recent developments on biodiversity assessment.

12 During LEAP 1 (2012-2014), a first step was taken to tackle the challenge of biodiversity assessment in the
13 livestock sector, with the formation of a dedicated TAG on biodiversity and the development of *Principles
14 for the assessment of livestock impacts on biodiversity*. The present document builds on this previous work
15 and progresses it by moving from qualitative principles to guidelines for the quantitative assessment of
16 livestock impacts on biodiversity.

18 1.2 The need for quantitative indicators

19 The LEAP Partnership has enabled a high level of methodological consensus on how to quantify GHG
20 emissions and other environmental impacts (including nutrient cycles and water) from livestock supply
21 chains. It has allowed for a number of quantitative assessments and for technical and policy options to be
22 proposed in order to mitigate the livestock contribution to climate change. In particular, increasing the
23 efficiency and intensity of livestock production has been suggested as a mitigation option because more
24 intensive mixed production systems with a part of crop by-products feeding have lower GHG or nutrient
25 emissions per unit of product compared to grassland-based systems (Gerber et al. 2010; 2014). However,
26 changing to high input and intensively managed systems intensification could result in higher impacts on
27 biodiversity because of the associated habitat changes (e.g. natural to improved pastures, grassland to
28 feed crops) and negative effects of water withdrawal, pesticides or inorganic fertilizers. On the contrary,
29 extensively managed grassland-based systems can provide crucial biodiversity habitats, but with higher
30 GHG emissions per unit of product compared to intensively managed systems because these 'units of
31 product' usually focus on food or proteins and do not take into account other social and ecosystem
32 services.

33 To integrate biodiversity along with other environmental criteria, there is a need to move from principles
34 to quantitative and operational biodiversity assessments. In the absence of more holistic approaches, the
35 risk of pollution swapping is real, and unrecognized trade-offs among different dimensions of agri-
36 environmental sustainability may occur. Quantitative biodiversity assessments could help integrate
37 environmental criteria because biodiversity is at the endpoint of the environmental cause-effect chain
38 and is impacted by, for example, climate change, nutrient pollution, water withdrawals, and soil. Hence,
39 the other LEAP guidelines provide valuable information for conducting a biodiversity assessment
40 (Appendix 1).

Quantitative biodiversity assessments are needed to support international agreements that recognize the importance of biodiversity conservation, such as the 2020 Aichi Targets set by the Convention on Biological Diversity (CBD) and the 2030 UN Sustainable Development Goals number 14 and 15 on protecting, restoring and promoting sustainable use of terrestrial ecosystems. Furthermore, after the COP23 decision to address agriculture in the negotiation process, there is a potential for integration and synergies between biodiversity, climate change mitigation and nutrient management (e.g. UNFCCC and CBD, SDG 13, 14 and 15) in the transition towards sustainable livestock production (FAO 2018a).

2 Objective and scope of this document

Key messages

- The objective of this document is to develop guidelines for quantitative assessment of the impacts of livestock production on biodiversity, based on existing indicators and methods.
- Indicators and methods described in these guidelines are relevant to a range of assessment objectives, users, scales, geographical regions, livestock species and production systems.
- This document focuses on biodiversity at the species and ecosystem level.

2.1 How to use this document?

Figure 1 presents an overview of the different steps of assessment and points to the section of the document providing the corresponding guidelines. The first step is to select an assessment framework between LCA and PSR, or a combination of both depending on the overarching goal of the assessment, its scale and its constraints. Specific procedural guidelines on how to implement the LCA or PSR framework are then provided in Part 2 of the document.

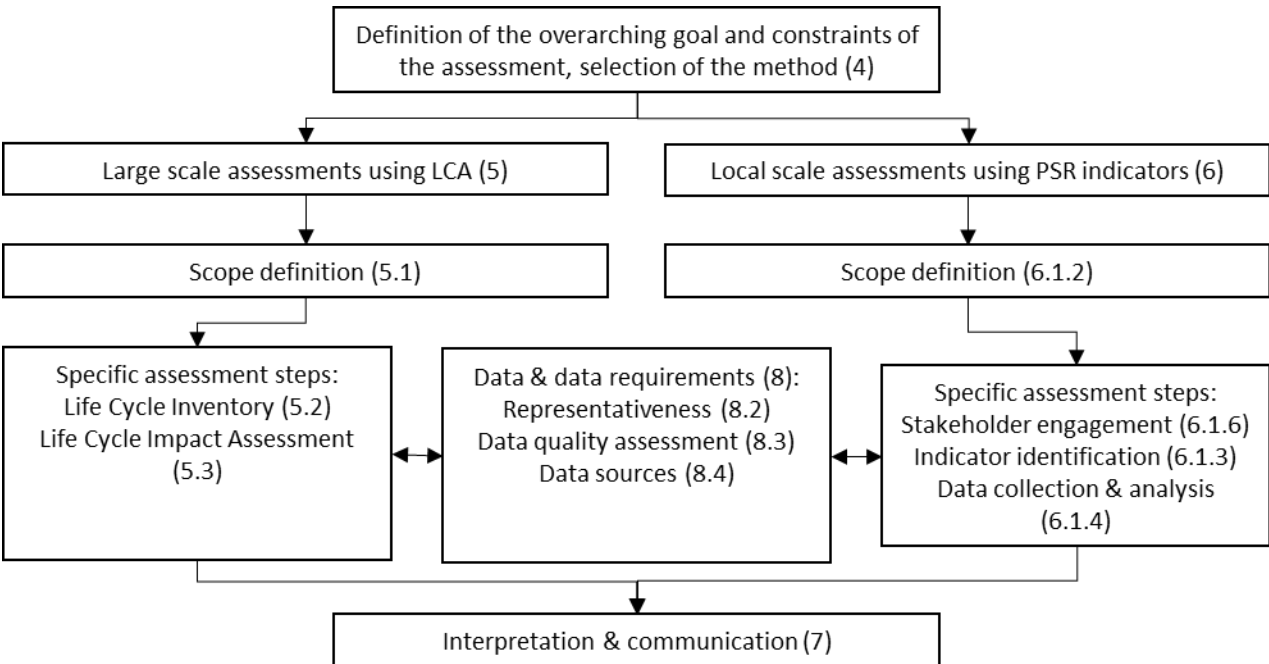


Figure 1. Overview of the main assessment steps and corresponding document sections (in parentheses).

2.2 Objective and intended users

The objective of this document is to develop guidelines for quantitative assessment of the impacts of livestock production on biodiversity, based on existing indicators and methods.

It is recognized that key biodiversity issues and conservation priorities vary among geographical regions and livestock production systems. Indicators and methods described in these guidelines are relevant to a range of assessment objectives, users, scales, geographical regions, livestock species and production systems.

In developing the guidelines, it was assumed that the primary users will be individuals or organizations with a good working knowledge of environmental assessment of livestock systems (including feed production). The guidance is relevant to a wide array of stakeholders in livestock supply chains including:

- Livestock producers who wish to know the environmental performance of their production units assessed or to adopt biodiversity friendly practices;
- Supply chain partners such as feed processors, livestock farming organizations, processors of animal products as well as retailers seeking a better understanding of the environmental performance of their production processes;
- Policy makers interested in developing biodiversity assessment and reporting specifications for livestock supply chains;
- Environmental organizations or land managers conducting biodiversity assessments for conservation objectives.

2.3 Biodiversity levels and components

These guidelines cover the range of positive and negative links between livestock production and biodiversity, adopt a life cycle perspective and include multiple possible and spatially dispersed impacts along livestock supply chains, and address biodiversity at both the species and ecosystem levels. The assessment of livestock genetic resources is beyond the scope of this document but Section 3.2 provides background information and data sources on this aspect of biodiversity. Section 3.3 describes the linkages between livestock, biodiversity and ecosystem services and highlights overlaps between assessment frameworks and methods. In the absence of additional precisions, the term “biodiversity” in this document refers to diversity at the species level.

2.4 Livestock species

These guidelines focus on the six main livestock species (i.e., cattle, sheep, goats, pigs, poultry and buffalos) although the recommended methods, procedure and certain indicators may be relevant to other types of animals (e.g., insects, aquaculture, ducks, reptilians, amphibians).

3 General information on the relationship between livestock, biodiversity and ecosystem services

3.1 General ecological principles

Key messages

- Biodiversity describes the variability of life on earth and is often positively affected by intermediate levels of disturbance as this often opens up new niches (i.e. a variety of conditions and resources) for a greater diversity of species to become established.
- Extensively managed and low input livestock systems can maintain intermediate levels of disturbance, and be of high nature value, where past land use disrupted natural disturbance regimes, and replaced wild with domestic herbivores.
- Inappropriate management practices can occur in both low-input extensive (e.g., overgrazing, abandonment) and high-input intensive (e.g., off-farm feed produced in simplified landscapes, nutrient pollution due to animal concentration) systems and will determine impacts on biodiversity per unit of livestock product.

Biodiversity is a concept that describes the variability of life on earth, and can refer to the variability of genes (genetic diversity), of species (species diversity), or of ecosystems. Species diversity is a typical measure of biodiversity and is usually calculated as the number of species and their relative abundance at a specific place and time. Species diversity varies as a result of a wide range of factors including ecosystem productivity, abundance of resources, predation intensity, spatial heterogeneity, climatic variability, and ecosystem age. In general, species diversity is higher within areas that contain a wide range of environmental conditions that provide the necessary resources (e.g., food, habitat) and conditions (e.g., climate, soil pH, humidity, disturbance regimes) required for the survival of different organisms. Heterogeneity is produced as a result of different habitat types or different vegetation structures (i.e., structural heterogeneity), and species diversity tends to be higher if these regions are subject to medium levels of disturbance. Depending on the situation, disturbances can arise as a result of natural factors (e.g., exposure, flooding, burning, grazing by wild herbivores) or linked to human-influenced management factors (e.g., deforestation, cropping, grazing by livestock, ploughing, breeding management, nutrient and pesticide input). In the latter case, they often occur in ways or at levels much higher than as a result of natural processes. High levels of disturbance can result in a reduction in biodiversity.

Taking plant species richness as an example, situations subject to high disturbance either in terms of severity or frequency of the disturbance events will produce conditions in which only a limited number of species can adapt, and hence the overall species richness in such situations will be relatively low. At the other end of the scale, situations subject to little or no disturbance will often result in the ecosystem being dominated by a limited number of plant species that out compete other species. Both extremes result in relatively homogenous vegetation structures, which limit biodiversity by restricting the growth and colonisation of other species.

However, medium disturbances promote increased plant biodiversity by providing more opportunity for other plant species to become established. In forest and, to a large extent in rangelands, biodiversity is much more driven by 'naturalness' and so can be influenced by a combination of natural disturbance events and anthropogenic management activities such as grazing. Conversely, farmed habitats that can

1 impact biodiversity are substantially more influenced by management and policy decisions. Hence, in
2 livestock production systems, biodiversity is primarily driven by the nature and intensity of the
3 management to which those habitats are subjected.

4 Biodiversity is also generally higher not only where there are a variety of habitats/structures in a landscape
5 but also where these occur at a large enough scale to allow species to survive and maintain viable
6 populations through avoidance of habitat fragmentation. The species area relationship is a consistent
7 pattern in ecology – where larger areas (e.g., regions, islands, patches of habitats) host more species than
8 smaller areas (Connor & McCoy, 1979). Having a sufficient amount of similar habitats in close proximity
9 to each other allows species to seek out and occupy suitable habitats, increasing their chances of
10 exchanging genes through migration and thereby supporting the maintenance of viable populations
11 (MacArthur, 1967; Levins, 1969).

12 Biodiversity at species or ecosystem level can be measured at different scales. Local or α (Whittaker 1972)
13 diversity refers to species diversity within a particular habitat or ecosystem. Certain species may be
14 endemic and common locally but dependant on a specific type of ecosystem and therefore rare or
15 endangered at a global scale. Looking not only at local diversity but also at regional and global diversity is
16 therefore an important component of an assessment.

17 Adverse impacts on biodiversity can be associated with both extensive and intensive systems (FAO 2016).
18 In extensive systems, unsuitable grazing management can lead to overgrazing which causes soil
19 degradation and reduces plant diversity as well as productivity. Abandonment (i.e., inadequate or no
20 grazing) can also lead to land degradation and biodiversity loss through high dominance of few species or
21 shrub encroachment, if wild herbivores are scarce or no longer present (Laiolo 2004; Koch et al. 2016). In
22 intensive or confined livestock production systems, a large proportion of the feed used is usually produced
23 off-farm in intensive farmland with simplified landscapes. Intensive livestock systems can also concentrate
24 manure at the site of production, which if improperly managed, can adversely impact soil and water
25 quality.

26 Reconciling food production and biodiversity conservation objectives could be achieved in contrasting
27 ways in extensive vs. intensive livestock production systems (Green et al. 2005). Intensive systems
28 typically require a lower area of land for feed production, per unit of livestock product produced and could
29 therefore theoretically spare land for natural and semi-natural areas. By increasing the efficiency and
30 integration of such systems in the circular bioeconomy, they maximize productivity while minimizing
31 resource use and externalities (Godfray et al. 2010).

32 Conversely, extensive systems use a higher area of land and actually exert less pressure on habitats. They
33 can therefore be considered of high biodiversity value, even after some amount of transformation of the
34 more natural habitats at a landscape level happened through disruption of natural disturbance regimes
35 by wild replaced by domestic herbivores (Appendix 2). Grazing influences and promotes biodiversity in
36 grassland ecosystems (Watkinson & Ormerod 2001) and several models have been proposed to describe
37 the effect of grazing on vegetation (Cingolani et al. 2005). Under adequate management and after a long
38 history of livestock grazing, domestic animals can perform the ecological role of wild herbivores in
39 maintaining this unique biodiversity in production systems (Eriksson et al., 2002; Bond & Parr 2010). This
40 is particularly the case where large herbivore guilds have undergone Pleistocene extinctions (Corlett,
41 2016). The biodiversity value of extensive livestock systems relates both to their spatial and temporal
42 heterogeneity and the ability to sustain high levels of habitat and species diversity (Pogue et al. 2018).

Furthermore, the ecological diversity within these extensive ecosystems often provides favourable conditions for plants and animals (especially invertebrates) to find habitats suitable for the completion of their life cycles (Bignal & McCracken 2000).

For more information on the link between livestock production and biodiversity, please refer to the LEAP review of indicators and methods to assess biodiversity (FAO 2015).

3.2 Ecosystem services

Key messages

- Ecosystem services (ES) are the outcomes from ecosystems that lead to benefits valued by people. Agroecosystems are both providers (e.g. food production but also soil fertility, water regulation, soil carbon storage) and beneficiaries (e.g. vegetation productivity, pollination, pest control, water availability) of ES.
- Biodiversity plays a key role in ES provision as a regulator of ecosystem processes, an ecosystem service, and a good or benefit.
- In livestock production systems, there can be synergies and trade-offs between biodiversity and the different types of ES, for instance intensification is often associated with higher food production (a provisioning service), but lower biodiversity or regulating services (water quality, soil carbon).
- To date considerable effort has been devoted to quantifying the life cycle impacts of products on ES (this section provides a number of methods), but key challenges remain unsolved.

Ecosystem services (ES) are “Nature Contribution to People” (Diaz et al. 2018), i.e. the outcomes from ecosystems that lead to benefits valued by people. The difference categories of ES include provisioning services such as water, wood, genetic resources, crop and livestock products; supporting services or ecosystem processes such as water cycling and soil formation; regulating services such as soil, air and water quality or climate regulation; and cultural services including cultural identity, recreation and tourism. In complex social-ecological systems such as livestock production, natural factors (e.g., geography, climatic conditions, wild and domestic biodiversity), and various forms of human capital (e.g., financial, infrastructure, social benefits) interact to co-produce ES (Palomo et al., 2016). The benefits flowing from these services contribute to economic, health and social aspects of human well-being, although demand for certain services can vary among individuals and groups (Yahdjian et al., 2015). Changes in well-being influence system governance and management, which in turn affect the social and ecological structures and processes underpinning ES provision (Reyers et al., 2013) (Figure 2).

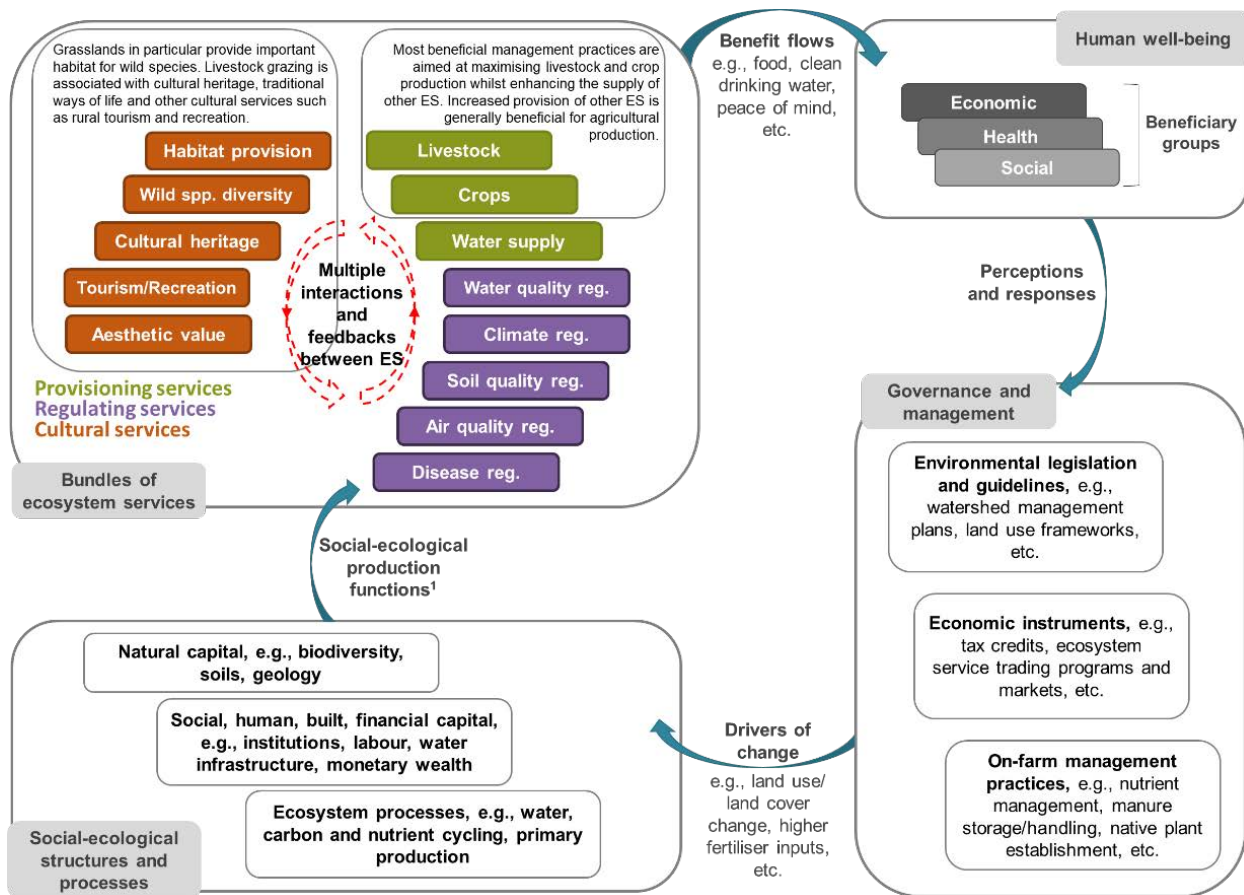


Figure 2. Conceptual ecosystem service framework for a livestock production system. Source: modified from Reyers et al. (2013). Social-ecological production functions link biodiversity, ecosystem processes and ecosystem services.

Agroecosystems not only produce food – a provisioning service – but also influence ES essential for food production including soil retention/erosion, pest control, and soil fertility and quality improvement, as well as other regulating (e.g., soil carbon storage in particular, refer to LEAP 2019b guidelines) and cultural services (e.g., aesthetic value such as the globally important agricultural heritage sites, FAO 2018b). In addition, agroecosystems interact with the surrounding landscape matrix, benefiting from services delivered by non-agricultural systems (e.g., pollination) or impacting these systems such as downstream water quality impacts due to nutrient runoff (Dale and Polasky, 2007; Zhang et al., 2007). There is often a trade-off between provisioning and other ES categories, in particular regulating and cultural services (Raudsepp-Hearne et al., 2010). Typically, intensive production systems make a high contribution to food production, but a low contribution to other ES categories, but impacts are usually even higher for services from non-agricultural systems.

As ES link to human well-being, they have great potential to influence decision-making (Villamagna et al., 2013). However, the integration of ES research into environmental decision-making has been limited by an incomplete understanding of how and for whom services are co-produced by social-ecological systems, and what the best management practices for ES governance are (Bennett et al., 2015). Central to understanding how services are produced is learning how biodiversity influences their provision (Bennett

et al., 2015), and how biodiversity and ES respond to management practices (Reyers et al., 2012; Rodriguez-Ortega et al., 2014). Integration of biodiversity and ES research is needed to address biodiversity conservation and ES management goals (Mace et al. 2012).

Biodiversity plays a key role in ES provision as a regulator of ecosystem processes, an ecosystem service, and a good or benefit (Mace et al. 2012). In grazed grasslands, many soil nutrient cycles are determined by soil biological community composition (Mace et al. 2012), including organisms that break down and integrate dung into the soil. Greater biodiversity is generally positively correlated with ecosystem functioning, as it is associated with a higher number of functional groups of species, and increases the efficiency by which ecological communities capture biologically essential resources, produce biomass, and decompose and recycle biologically essential nutrients (Cardinale et al. 2012). Biodiversity also increases ecosystem function resilience, essential for the maintenance of ES, especially under future predicted environmental change (Oliver et al. 2015). Resilience of ecosystem function to environmental perturbations will be higher when, for example, there is variation in response to change within and between species (i.e., species-level effects); there is greater functional redundancy (i.e., several species having the same function in an ecosystem); or there are mechanisms including landscape-level functional connectivity that facilitate the flow of biotic and abiotic components essential for ecosystem processes and services (i.e., landscape-level effects) (Oliver et al. 2015). However, the effect of landscape connectivity on service provision depends on the service and its relationships with biodiversity and ecosystem processes. For example, structurally diverse pastures sustained by livestock contribute to pollinator diversity, which in turn provides higher crop yields in adjacent fields (Hevia et al. 2016). Furthermore, increased pollinator movement can also increase disease vector movement, lowering disease regulation (Mitchell et al., 2013). As an ES, biodiversity at the species and gene level contributes directly to the generation of goods as, crop and livestock genetic diversity is important for the maintenance of crop and livestock populations (Rischkowsky and Pilling, 2007). As a good or benefit, biodiversity is valued by people and their well-being increases from simply knowing that certain species or habitats exist and are being conserved, with this existence value generating a cultural service (Reyers et al. 2012).

In livestock production systems, there can be synergies and trade-offs between biodiversity and ES, and between short-term performance and long-term resilience of the system. For instance, in South American Pampas and Campos grasslands, Modernel et al. (2016) reported that lower stocking rates were associated with higher plant, bird and mammal diversity and increased provision of services including soil organic carbon, soil erosion regulation, and meat production. Trade-offs can also occur, as the intensification of beef and dairy production can reduce greenhouse gas emissions per unit of product produced as grazing livestock are associated with higher enteric methane emissions (Beauchemin et al., 2010, 2011). However, sustainable grazing practices can promote carbon sequestration and contribute to the massive carbon stores within grassland soils (Chen et al. 2015; Wang et al. 2016; Hewins et al. 2018). Therefore, a holistic approach to biodiversity and ES assessment across the livestock sector is needed to understand the full range of impacts (Janzen 2011), and to identify management practices that maximise the social and ecological performance of the system.

To date, although considerable effort has been devoted to quantifying the life cycle impacts of products on ES (Appendix 3), key challenges remain unsolved (Bakshi and Small 2011; Othoniel et al. 2016; Maia de Souza et al. 2018). An additional challenge in this respect is to analyse the spatial and temporal configuration of ES supply and demand across scales. Moreover, the scale at which different groups of

people benefit from ES is directly linked to stakeholder interests, and mechanisms to address these relationships must be considered (Maia de Souza et al., 2018). At local scale, ecosystem services need to be qualified based on local population needs, considering rural land tenure and environmental legislation aspects, as well as on local natural conditions and disturbances. In peri-urban areas, the rural environment is the main source of ecosystem services that benefit cities (e.g., water provision) and is therefore a key partner in land-use planning. Fencing can be necessary to limit the access of livestock to ecosystems providing services such as water sources, floodplains or riparian forests. Larger scales also need to be considered to assess ecosystem services – landscapes, river basins, regions, biomes, countries etc.. Forests in particular contribute to important ecosystem services such as carbon storage and climate regulation, which can be quantified to inform forest protection policies. In relation to livestock production, plans for occupation organization such as ecological-economic zoning can be beneficial.

3.3 Livestock genetic diversity (breeds)

Key messages.

- The assessment of livestock genetic resources is beyond the scope of this document, but the following section provides background information and data sources on this aspect of biodiversity.
- More than 8800 livestock breeds have been recorded globally, representing a valuable resource with a high level of genetic biodiversity.
- Animal genetic resources remain at risk and many drivers of loss are shared with those of wild biodiversity, such as demand, intensification, degradation of natural resources and climate change.
- The Domestic Animal Diversity Information System (DAD-IS) collects information on animal genetic resources from 182 countries and provides a searchable database of information related to livestock breeds.

Around 15,000 national breed populations (representing more than 8,800 breeds) have been recorded globally (FAO, 2015). Diverse animal genetic resources underpin the capacity of livestock populations to provide a range of products and services across a diverse range of production environments. Livestock diversity makes a huge contribution to the adaptation of production systems and their resilience in the context of global environmental changes. Coping with climate change, diseases, changing markets and limited natural resources will require a diverse range of animal genetic resources. For instance, breeds that are disease tolerant or adapted to drought and other extreme climatic events will be of particular importance. Beyond their role in increasing resilience, diverse breeds are key for the livelihood of the poor. They not only produce food but also deliver a wider range of good and services such as draught power, manure for fertilization, environmental and socio-cultural services (i.e., employment, investment, insurance, social capital), often under conditions of limited feed and water resources.

Animal genetic resources remain at risk as the proportion of livestock breeds classified as being at risk of extinction increased from 15 percent to 17 percent between 2005 and 2014. A further 58 percent of breeds are classified as being of unknown risk status because no recent population data have been reported to FAO. The main drivers of loss of animal genetic resources include cross-breeding, changing market demands, weaknesses in animal genetic resources management programs, policies and institutions, climate controlled livestock production systems, degradation of natural resources, climate change and disease epidemics.

1 The Second state of the world animal genetic resources (FAO, 2015) provides a global overview of the
2 state and trend in livestock diversity, and identifies current capacities and strategies for conservation, as
3 well as needs and challenges.

4 The Domestic Animal Diversity Information System (DAD-IS¹) collects information on animal genetic
5 resources from 182 countries and provides a searchable database of information related to livestock
6 breeds, such as animal numbers, animal performance, management tools, references, links and contacts
7 of Regional and National Coordinators for the Management of Animal Genetic Resources.

8 DAD-IS was used in combination with climate models to develop a model predicting the potential impact
9 of climate change on breed distribution². Current breed distributions from DAD-IS were used to model
10 suitable areas for breeds under current and expected future conditions, taking several temperature and
11 humidity parameters into account.

12 The FAO's Commission on Genetic Resources for Food and Agriculture recognizes that biodiversity is
13 essential for food production and achieving nutritional diversity in the human diet. In a recent report,
14 biodiversity for food and agriculture has been shown to be declining and although biodiversity-friendly
15 practices exist, enabling frameworks for the sustainable use and conservation of biodiversity remain
16 insufficient (FAO, 2019a).

17

¹ <http://www.fao.org/dad-is/en/>

² <http://www.fao.org/breed-distribution-model/en/>

Part 2. Methodology

4 Definition of the assessment goal and method

Key messages

- Goal definition is the first step of the assessment and all further steps (i.e., scope, data, methods, results and conclusions) should align with this goal
- This document frames potential assessments within three major scales – global, regional and local using two main methods – Life Cycle Assessment (LCA) and Pressure, State, Response (PSR) indicators. Selection of the method depends on the overarching goal of the assessment, its scale and its constraints.

4.1 Goal of the assessment

The first step of any biodiversity assessment is to set the goal of the assessment and the intended use of the final results. The selected assessment method and all steps of the assessment should reflect the defined goal, so that the goal, scope, data, methods, selected indicators, results and conclusions are aligned. Engagement with multiple stakeholders can be extremely useful in defining goals that are relevant to the specific system under study (e.g., livestock system, geographical area). This is particularly relevant for livestock areas or systems where little specialized technical information is available. Here, stakeholder involvement can greatly contribute to the correct implementation of a successful biodiversity assessment.

During this phase, several aspects should be addressed and documented (European Commission, 2010), such as

- the subject of the analysis and key properties of the assessed system, such as name of the organization, its sector and location of production systems, dimension of the facilities, end products and by-products, and the position of the organization in the value chain;
- the reason for which the study is being performed and the decision-making context, if any, into which it is inserted;
- the intended use of the results, i.e., will they be used internally for decision-making or shared externally with third parties;
- limitations due to the method, assumptions, and choice of impact categories: in particular limitations to broad study conclusions associated with the exclusion of impact categories;
- the target audience of the results obtained;
- comparative studies to be disclosed to the public and need for critical review;
- the commissioner of the study and other relevant stakeholders.

In addition to defining the above-mentioned aspects, it is important to clarify the stated biodiversity goals of the sustainability initiative in question (i.e., what are the desired levels of biodiversity) or of the livestock system under study (i.e., which improvements are to be achieved in the system). Moreover, the goals of the assessment should consider overarching priority issues such as: (i) the extent to which critically endangered species are affected; (ii) the extent to which key ecosystems and habitats are affected (e.g., global ecoregions, biodiversity hotspots and corridors, IUCN red list of ecosystems and habitats, habitats of high ecological sensitivity, such as riparian habitats or areas of high risk of erosion);

(iii) the extent to which ecosystem functioning and services are maintained in areas of high conservation value; and (iv) the extent to which other biodiversity conservation goals within the study boundary are affected.

4.2 Scale of the assessment and method selection

This document frames potential assessments within three major scales – global, regional and local – using two main methods – Life Cycle Assessment (LCA) and Pressure, State, Response (PSR) indicators (Table 1).

Biodiversity assessments in LCA rely on a few currently available methods that have a number of constraints. In particular, they often focus on impacts through land use (i.e., other types of impacts through pollution or climate change are not considered); they consider broad land use classes (e.g., biodiversity impact of grassland vs. cropland); they have an intermediate level of biogeographical differentiation (e.g., 1 ha of grassland having the same impact anywhere within a 150000 km² ecoregion) and; they focus on species richness as a biodiversity indicator. Given the current state of knowledge, LCA approaches are not well suited to answering some questions such as ‘is livestock production practice A better than practice B for biodiversity?’, when both A and B occur within one of the broad land use classes of the current LCA approaches. Approaches that are based on large geographical scales are much more suited to assessing land use changes impacts across bioregions, and not suited to assessing other more qualitative changes (such as the impacts of overgrazing or undergrazing) within a bioregion. However, LCA is a very useful for broad assessment of impacts on biodiversity at large spatial scales and for finding impact hotspots along the supply chain or among spatial entities. LCA can be used to reveal supply-chain or spatial hotspots for further investigation with more detailed assessment methods.

Therefore, global and regional scale assessments can be addressed through LCA, which provides a framework to analyse the impacts of decisions along livestock supply chains on biodiversity (Chaplin-Kramer et al. 2017). LCA was specifically designed as a decision-making tool and is intended as a holistic assessment identifying the transfer of environmental burden among stages of the supply chain or among types of environmental impact. LCA application is ruled by a set of international standards (ISO 2006a; b) and is used by a wide variety of stakeholders, such as governments (e.g., for regulations or eco-labelling), companies (e.g., to adopt environmentally sound practices and to assess/increase eco-efficiency of products), and NGOs (e.g., to promote transparency and inform consumers). LCA quantitatively models cumulative impacts along environmental cause-effect chains using characterisation models and factors. Impacts can be characterized anywhere along the environmental cause-effect chain, either at the midpoint or endpoint level. The midpoint impact categories can be defined as part of a problem-oriented approach, translating impacts into environmental themes such as global warming, land use, acidification or human toxicity. Endpoint impact categories, such as biodiversity provide a damage-oriented or damage-avoidance approach (ISO, 2006b).

The third (local) scale can be addressed through indicator frameworks, such as the Pressure, State, Response (PSR) framework, which utilizes local data (Section 6). The type, amount, spatial and temporal distribution of the data needed will be determined by the combination of the goals of the study, analytical methods proposed and the scale at which the study will be conducted. The PSR model has been widely used to develop and structure biodiversity indicators (OECD, 1993), and can be a useful tool to monitor either biodiversity impacts or improvement in biodiversity performance. The model is based on causality: indicators are used to evaluate the pressures of human activities that lead to changes in environmental

states, causing responses (i.e., decisions and actions) of the stakeholders (i.e, political, socio-economic) to be undertaken to improve the state of the environment.

Table 1. Overview of the methods, possible applications, users and limitations associated with assessment methods at both scales.

Assessment scale	Large scale assessments	Local assessments
Method	Life Cycle Assessment	Pressure, State, Response indicators
Applications	<ul style="list-style-type: none"> • Identify hotspots of impacts: spatially or along the supply chain • Compare systems or scenarios • Prevent burden shifting among life cycle stages or environmental impacts 	<ul style="list-style-type: none"> • Improve impact on biodiversity by replacing or mitigating negative and supporting positive practices • Monitor improvement over time • Implement or test a biodiversity action plan or other biodiversity actions (including in key biodiversity or protected areas)
Users	Sector and sub-sector sustainability managers, trade/processors/other companies, policy makers	Farmers, pastoralists, land managers, communities, local companies, NGOs, policy makers
Constraints faced by the user	<ul style="list-style-type: none"> • Lack of resources to collect field data • Need to consider complex and globalized supply chains 	<ul style="list-style-type: none"> • Lack of information on off-farm processes • Need to consider specific biodiversity issues (e.g., protected areas or species, practices)
Limitations of current methods	Low ability to consider positive impacts, detailed practices or local improvements	Low ability to consider multiple supply chain steps, impact locations and to aggregate biodiversity impacts

5 LCA regional and global assessments

5.1 Impact pathway (Cause-effect chain) and steps of the LCA

Key guidelines

- The impact pathway is the conceptual cause-effect chain that links inventory flows associated with livestock production (e.g., land use, nutrient inputs, water use) to resulting impacts on biodiversity.
- LCIA models translate inventory flows into specific biodiversity indicators; the impact pathway points to the main LCIA model recommended by these guidelines (Chaudhary and Brooks 2018) and to alternative models.
- Not all inventory flows detailed in the impact pathway are required by all LCIA models, but it is recommended that as much information as possible on inventory flows is collected and reported.

The impact pathway is the conceptual cause-effect chain that links inventory flows associated with livestock production (e.g., land use or land transformation for pasture and /or crop production) to resulting impacts on biodiversity (e.g., changes in functional diversity, abundance and species composition) and finally to effects on ecosystem structure and function. The impact pathway is the basis for proposing measurable and simple quantitative indicators to assess the potential effects of livestock production on biodiversity (Curran et al. 2016; FAO 2016).

Figure 3 shows the impact pathway identified in these guidelines, which depicts the main causes of human interventions associated with livestock and feed production systems and their consequent impacts on biodiversity. The inventory flows outlined in dark grey are those that are required by the existing model(s) recommended in this document. Inventory flows in white, are those that should ideally be added to current land use parameters as defined by Koellner et al. (2013). This section on LCA proposes three degrees of grazing intensity (light, moderate, heavy), which could be quantitatively linked with effects on biodiversity through estimates of the number of grazing animals that an area can support without adversely impacting biodiversity or the productivity of land (i.e., the carrying capacity). These three categories roughly correspond to the minimal, light and intensive pasture land use categories defined by Newbold et al. (2015) and implemented by Chaudhary and Brooks (2018). In general, light to moderate grazing either maintains or improves biodiversity in grazed areas (e.g., promoting plant species richness), by creating niches and increasing spatial heterogeneity (Steinfeld et al. 2010). In contrast, heavy grazing (i.e, over-grazing) reduces vegetation and soil coverage. Grazing pressure also depends on animal species, body size, breed, sex and age (Rook et al. 2004), but due to data scarcity these factors have not been included in the proposed LCI.

Land management practices, such as nutrient input, pesticide application, and water management (e.g., irrigation, drainage), are also key to describing how land use will impact biodiversity and ecosystem services. These practices may be related to either on or off-farm areas that are used to produce feed for livestock (i.e., grasslands or croplands). For instance, irrigation practices may also influence return-flows of nutrients and pesticides to waterways, which in turn, may impact downstream aquatic ecosystems and affect biodiversity and ecosystem services. Large scale disturbance, such as fire, when adequately managed, can contribute to more heterogeneous habitats, and therefore high levels of biodiversity (Nekola 2002), whereas burning of crop residues can reduce biological activity and biodiversity in soils (Wallis et al. 2010).

1 Burel et al. (1998) demonstrated that a simple linear relationship between land use intensification and
2 loss of species cannot be drawn, particularly if landscape patterns are not considered. This is mainly due
3 to the different temporal and spatial scales at which agricultural practices and biodiversity operate (i.e.,
4 context-dependency). Improved biodiversity levels are also usually associated with higher landscape
5 heterogeneity (Belfrage et al. 2015), and changes in species distribution may be determined by the nature
6 of the drivers (e.g., land management, stress on vegetation) and processes affecting landscape patterns
7 (e.g., intensity of process, history of land use). Understanding changes in landscape heterogeneity and
8 patterns help to better predict changes in biodiversity and ecosystem services (Lausch et al. 2015).

9

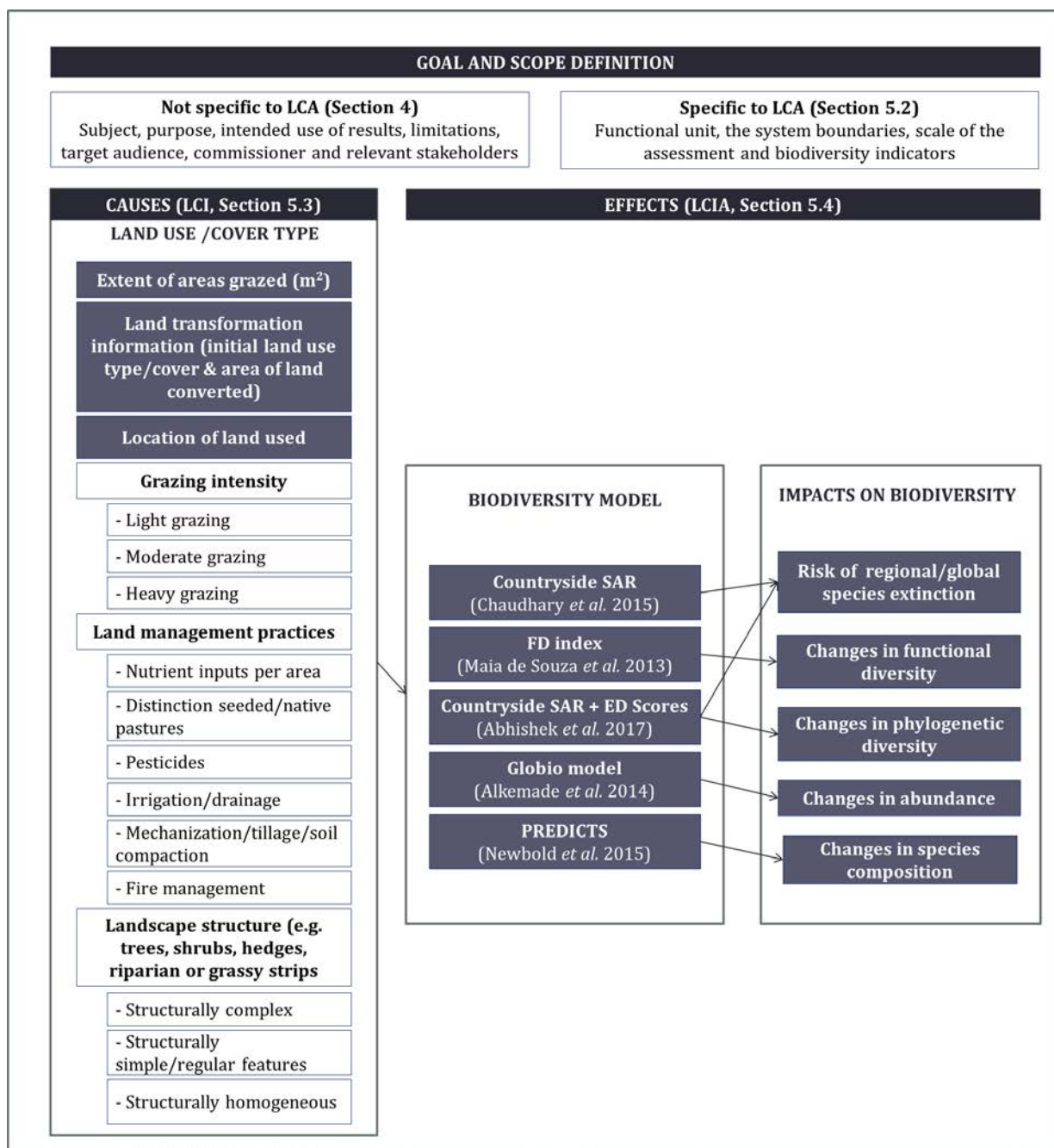


Figure 3. Schematic representation of the recommended cause-effect chain for the assessment of livestock impacts on biodiversity. The boxes in grey are those representing aspects already covered by life cycle inventory and LCIA models. Boxes in white are aspects that should ideally be added to current land use/cover flows, in order to provide a better representation of livestock systems. ED is ecological distinctiveness; FD is functional diversity and SAR is species-area relationship.

At the LCIA step, we have included the main indicators of impacts on biodiversity, which are currently addressed by existing LCIA models: (i) risk of regional/global species extinction (Chaudhary and Brooks 2018), (ii) changes in functional diversity (Souza *et al.* 2013), (iii) changes in phylogenetic diversity/evolutionary history (Chaudhary *et al.* 2017), (iv) changes in species abundance (Alkemade *et al.* 2013), and (v) changes in species composition/biodiversity intactness (Newbold *et al.* 2015).

Moreover, this document suggests that, to more comprehensively assess the impact of livestock-associated land use and land use change and feed production on biodiversity, LCIA should ultimately include impacts on ecological structure and function and consequent changes in ecosystem services. This is mainly because the diversity of species has a direct link to and importance for the generation of ecosystem services associated with livestock and feed production systems.

The complex LCIA pathway contains several interconnections and often needs simplification, as not all aspects can be represented by a single indicator. We therefore settled on representing the impact on biodiversity at local, regional and global levels for which global data is available and ecological models exist (Section 5.4).

5.2 Goal and Scope

Key guidelines

- In LCA as in other types of biodiversity assessment, the first step is to clearly set the goal for the study, the intended use of the results and to describe the scope in terms of depth and breadth of the study, the nature of the livestock system to be studied and the function of the system (see to Section 4).
- Several elements of goal and scope definition, however, are specific to LCA and are described in this section. They concern the functional unit, the system boundaries, scale of the assessment and biodiversity indicators/impact categories.

5.2.1 Functional unit

Key guidelines

- In LCA, impacts will always be expressed in relation to a functional unit – e.g., biodiversity loss per litre of milk or kg of carcass or protein – to ensure system definition and comparability.
- For LCA biodiversity assessments, the recommended functional units for different livestock supply chains (i.e. species, commodities) are the ones indicated in the LEAP sectoral guidelines (FAO 2015, 2016).

The functional unit in LCA shall be applicable and ensure commonality among the systems under study. The functional unit is a quantified description of the service delivered by a product system, according to the properties of the product, such as durability and functionality, and serves as a reference to which all inputs and outputs to the product system are related. Detailed information regarding the selection of functional units, as well as the necessary requirements to define the functional unit, can be found in previous LEAP Guidance documents (FAO 2015, 2016).

Alternative functional units, such as those based on land area required for the production system have also been used (Haas et al. 2001, Basset-mens & van der Werf 2005, Bartl et al. 2011). Although the aggregation of different types of products and services in a single functional unit is challenging, ecosystem services may be included in the functional unit as part of the services delivered by livestock production systems (Ripoll-Bosch et al., 2013).

The effect of intensification on biodiversity can also be captured through the complementarity of the LCA and PSR approaches which identify the appropriate characterization factors (CFs) which consider the impact of intensification from the perspective of the functional unit.

5.2.2 System boundaries

Key guidelines

- While carrying out an LCA, a flow diagram of all assessed processes in the livestock production system should be drawn, indicating system boundaries.
- The recommended system boundaries are the those indicated in the LEAP sectoral guidelines (FAO 2015, 2016) and typically encompass all stages of production, from raw material extraction to the primary processor gate.

Previous LEAP guidelines have most frequently defined two different downstream system boundaries for livestock production systems, i.e., the ‘farm gate’ and the ‘primary processor gate’. The upstream boundary shall extend to the ‘cradle’, or the point of initial extraction of the raw materials that serve as inputs to the supply chain. The inclusion of biodiversity metrics in LCA shall use the same system boundaries as for other ES, even though it is known that impacts on biodiversity are mainly captured up to the farm gate.

While carrying out an LCA, a flow diagram of all assessed processes in the livestock production system shall be drawn, indicating the system boundaries, the main life cycle stages, and all material flows. It shall be noted that material flows that are relevant to the production of the functional unit, as well as related activities that may affect biodiversity, may occur in off-farm locations. For example, the impact on biodiversity of exported soy or maize from Latin America to a dairy farm in Europe shall be attributed to the European dairy (Teillard et al. 2016).

The cradle-to-farm-gate stage includes feed and animal components. The LCA of feed is covered in detail in the LEAP Animal Feed Guidelines (FAO 2016b), which accounts for the cradle-to-animal-mouth stage for all feed sources including raw materials, inputs, production, harvesting, storage, loss and feeding. Feed may be grown on-farm, or animals may browse across a range of feed sources on land with multiple ownerships, and/or a proportion of the feed may be produced off-farm and transported to the farm for feeding animals.

5.2.3 Scale of assessment: global/regional/local

Key guidelines

- LCA is well suited to consider complex regional to global supply chains that are common to most livestock sectors.
- Currently, LCA is not well suited for assessing local biodiversity effects as global data lacks site-specific resolution.
- This document recommends several life cycle impact assessment models for regional to global assessment but the pressure-state-response approach is more adapted to local assessments – complementarities between the two approaches are also discussed.

In the context of a biodiversity assessment using LCA, the scale of the assessment depends on its goal and has several implications. At local scale, biodiversity loss is mainly considered at farm or field level. The region is a variable area which should be clearly defined (e.g., ecoregion, Olson et al. 2001) and the results of the impact assessment should be mainly relevant at this scale. Global assessment should consider the global vulnerability of species (e.g., IUCN red list) and impact should show damage to world scale.

LCA mainly aims to provide regional to global assessment, and CFs have therefore been developed mainly for these scales. On the contrary, LCA is not well suited for assessing local biodiversity effects due to the lack of sufficiently detailed data at the global scale to support the development of local or site specific CFs. Examples of models and CFs include de Baan et al. (2013) and de Souza et al. (2013) for the local to regional scale, and Chaudhary et al. (2015) and Chaudhary and Brooks (2018) for the regional to global scale.

LCIA based approaches should strive to link land use and land use change with effects on biodiversity through metrics such as species-area relationships, functional diversity, and extinction risk indicators. A relevant application of this approach to livestock systems still faces many challenges, such as the development of more scientifically robust models that describe local and regional changes in habitat structure, species function and composition, in relation to different livestock production systems (FAO-LEAP, 2015; Food SCP RT, 2013). Livestock and land management affect biodiversity primarily at regional and local scales, requiring spatial information capable of assessing species and ecosystem sensitivity at these levels (Potting and Hauschild, 2006).

There is an opportunity to take advantages of complementarities between the LCA and pressure-state-response approaches. For large scale, regional to global assessments of supply chains, the recommended approach Chaudhary and Brooks 2018) is based on countryside species area relationships linked to vulnerability scores based on IUCN data that is geographically explicit to ecoregions. This approach is well suited for identification of hotspots within supply chains, and therefore useful for global companies or national assessments. The identified hotspots should be further evaluated with PSR local/landscape approaches to fully assess management practices that are beneficial for biodiversity (Teixeira et al. 2016). Ideally, pressure or state indicators could be developed to be used as CFs in life cycle impact assessment.

5.2.4 Description of biodiversity indicators in LCIA

Key guidelines

- Different LCIA models can address different biodiversity indicators such as species richness, abundance or functional diversity.
- The main LCIA model recommended in this document (Chaudhary and Brooks 2018) uses the potentially disappeared fraction (i.e., impact on species richness) as an indicator.
- Other models can be used to address different biodiversity components, if justification and discussion is provided.

Previous LEAP documents reviewed potential biodiversity indicators (FAO 2015, 2016) that could be used to assess biodiversity impacts within the context of an LCA.

Most of the current models are based on compositional aspects of biodiversity (i.e., species richness and abundance) and only a few on functional diversity. While richness only considers the number of species that disappear locally, abundance-based models take into account population changes and have been shown to be more sensitive to land use change. Indicators of extinction risk explicitly translate changes in land cover and quality at the local scale into predicted regional or global losses of species.

Species extinction risk can be expressed at various scales, from national changes in the number of threatened species (Matsuda et al. 2003) to global species loss (de Baan et al. 2013, Lenzen et al. 2013). Regionally, abundance-based indicators simply entail summing up local abundance values across land use types in a region, with the assumption that losses in habitat are directly related to losses in species abundance (Alkemade et al. 2013, Alkemade et al. 2009, Scholes and Biggs, 2005).

Subjective measures of habitat quality include the “naturalness” of land cover and land use classes (Brentrup et al. 2002) or other scores based on a number of factors, such as proximity to habitat edge and neighbouring habitats (Leh et al. 2013). Functional indicators include Human Appropriation of Net Primary Productivity (HANPP) (Haberl et al. 2005, Haberl et al. 2004), functional trait diversity (Souza et al. 2013), and a range of ecosystem structural indicators summarized through meta-analysis (Gibson et al. 2011). Michelsen (2008) proposed to assess biodiversity indirectly by means of three factors: (i) Ecosystem Scarcity (ES) as a measure of the intrinsic rareness of an ecosystem; (ii) Ecosystem Vulnerability (EV) as a measure of the present condition of the structure; and (iii) Conditions for Maintained Biodiversity (CMB) as constructed from a suite of indicators. Coelho and Michelsen (2014) proposed the use of hemeroby values (levels of naturalness) as suggested by Brentrup et al. (2002) as a universally accepted indicator for the impact of human activities on a natural state.

Regarding habitat change, Larrey-Lassalle (2017) provided CFs to integrate fragmentation impact indexes into LCA. Taking forest fragmentation potential indicators and combining them with the species fragmented area relationship, new midpoint and endpoint indicators that consider the effects of fragmentation on biodiversity were generated. Unfortunately, these indicators are yet to be developed for grassland ecosystems.

From an endpoint perspective, biodiversity loss has usually been expressed as the potentially disappeared fraction (PDF) of species. This metric accounts for the fraction of species richness that may be potentially lost as a result of human activity (EU-JRC-IES 2011) and may be more suitable for impacts linked to midpoint indicators. The UNEP-SETAC Life Cycle Initiative (UNEP, 2017) has previously recommended that PDF be applied to regional and global levels based on Chaudhary et al. (2015) (see Section 5.4).

5.3 Life cycle inventory

Key guidelines

- Land use (in $\text{m}^2 \times \text{years}$) and land use change (in m^2 transformed from one land use class to another) are the main inventory flows that should be collected in the context of an LCA assessment using these guidelines.
- Inventory flows should be spatially differentiated – i.e., the location where they occur should be known – with the highest precision possible.
- These guidelines detail several levels of differentiation between land use categories; the highest level of differentiation possible should be used.

- For regional/global assessments, the main LCIA model recommended by these guidelines differentiates between 15 land use classes and spatially between more than 800 ecoregions.

The life cycle inventory (LCI) describes the first step of the environmental cause-effect chain (Section 5.1) and should include an analysis of land use and land use change registering the size and type of each land use. Relevant information for impact assessment regarding the quality and quantity of land use (i.e., type, intensity, management practices, location) shall be collected.

Ideally these data shall be collected based on GIS methodologies, but data on standardized classification and regionalization of land use may also be utilized (Koellner et al. 2013). Spatial differentiation (i.e., taking into account the accurate location of land use activity and using site-specific information on livestock and feed production) is important because land use impacts on biodiversity are spatially specific. For example, the occupation of pasture in biodiversity hotspots (e.g., Amazon, South East Asia, Congo Basin) is likely to have a higher impact on global biodiversity as compared to the same occupation in regions with lower levels of species richness and endemism.

Other important aspects include management practice and the intensity of land use. This is because the impact of land use on species can differ widely depending upon the management practices adopted on the farm (e.g., tillage or no tillage) and their intensity (e.g., low/high stocking density). Four levels of land use and management categorization have been suggested by Koellner et al. (2013):

- Level 1 uses very general land use and land cover classes (e.g., agriculture or grasslands),
- Level 2 builds on level 1 by describing annual crops or pasture/meadow,
- Level 3 provides information on land management (e.g., irrigated versus non-irrigated annual crops or extensive/intensive pastures), and
- Level 4 specifies the intensity of all land use (e.g., extensive/intensive irrigated annual crops).

Practitioners shall strive to develop the most detailed inventory of land use possible. Ideally this will achieve level 3 or level 4 categorisation and will involve obtaining data for the appropriate PSR indicators listed in Section 6. Practitioners should also be aware of the specific naming conventions used in assigning inventory flows, to ensure proper mapping of intended land use. This approach will ensure that the most appropriate CFs are applied. The methodologies used to develop CFs are continuously evolving and users shall seek out and use the most recent, precise and locally adapted CFs, or even develop specific CFs for the assessment under study.

In LCA, land occupation and land transformation can be distinguished as basic types of land use elementary flows (Koellner et al. 2013). In order to assess the impact of such land uses, it is necessary to at least register in the LCI the type of land use, its spatial and temporal extent, and its geographical location (Koellner et al. 2013). In LCIs, the elementary flows of land use are therefore specified as follows:

For land occupation: $m^2 \times \text{years}$, land use type i , and region k

For land transformation: m^2 , initial land use type i → final land use type j , and region k

5.4 Life cycle impact assessment models: impacts of land use on biodiversity

5.4.1 Global/Regional impact assessment

Key guidelines

- Chaudhary and Brooks (2018) model provides CFs reflecting potential regional or global species extinctions for different taxa; these guideline recommend using the global taxa-aggregated CFs.
- Recommendations in these guidelines are consistent with those of the UNEP-SETAC.
- One important limitation of the recommended LCIA model is its limited ability to reflect beneficial impacts on biodiversity – this shall be discussed as part of results interpretation and the use of complementary PSR indicators is recommended in an attempt to overcome this limitation.

For biodiversity assessments using LCA, this document recommends applying the method developed Chaudhary and Brooks (2018). These CFs were derived using the countryside SAR model, weighted with the vulnerability of the species in the region to assess the impacts on biodiversity due to land use and land use change associated with livestock production systems.

Two sets of CFs are available: regional and global. The former represent the biodiversity damage in terms of Potential Species Loss (PSL) from the ecoregion where the land use or land use change takes place (unit: PSL_{reg}/m^2). Note that regional species loss, also called species extirpation, is often reversible as the species might be present in other ecoregions. In contrast, the global CFs provide an estimate of irreversible global extinctions resulting from the land use/land use change activity (unit: PSL_{glo}/m^2). Both sets of CFs denote damage to different aspects of biodiversity. While preventing global extinctions is necessary to preserve the genetic diversity and ‘tree of life’ (Mace et al. 2003), avoiding a high number of regional species losses is necessary to ensure regional ecosystem function (Cardinale et al. 2012).

The global CFs for a particular taxon are derived by weighting the regional CFs with the vulnerability score ($0 < VS < 1$) of the taxon in a particular ecoregion. The VS is based on the proportion of endemic species in an ecoregion and the threat status of species hosted by the region. In detail, VS is calculated as the summed proportion of the range size for each species occurring in an ecoregion and weighted by its category of extinction risk (threat level) from the IUCN Red List (IUCN, 2017). The proportion of endemic species in an ecoregion is expressed as the ratio of area (km^2) for each species inside the ecoregion and the total (global) geographic area (km^2) coverage of this species and then aggregated for the total number of species or taxa found within the ecoregion. The endemic richness of a region can be interpreted as the specific contribution of the region to global biodiversity. The threat level is obtained by a linear rescaling of the IUCN red list to 0.2 representing the least concern, 0.4 near threatened, 0.6 vulnerable, 0.8 endangered, and 1 representing critically endangered (IUCN, 2017).

The CFs are available for five taxonomic groups: birds, mammals, reptiles, amphibians, and vascular plants as median and lower/upper 95% percentile. In addition, for ease of application, a taxa-aggregated set of CFs are also available representing the potentially disappeared fraction (PDF) of species per m^2 of a particular land use type. Taxa aggregation was performed using the following equation:

$$PDF_{global} = 0.5 \cdot \left(\sum_{t=1}^4 CF_t \cdot W_t \right) + 0.5 \cdot (CF_{plants} \cdot W_{plants})$$

where the CF of each animal taxa t and plants is multiplied by their respective weighting factor W . Equal weighting is given to plants and animals. Weighting factors are calculated as:

$$W = 1 / (S_{world} + VS_{median})$$

where S_{world} is the total species richness of the taxa and is equal to 5490 for mammals, 10104 for birds, 9084 for reptiles, 6433 for amphibians and 321212 for plants while VS_{median} is the median of all ecoregions VS and equal to 0.0158 for mammals, 0.0061 for birds, 0.0413 for amphibians, 0.0140 for reptiles and 0.012 for plants.

In terms of spatial coverage, the CFs are available for 804 terrestrial ecoregions (Olson et al. 2001) as well as aggregated to the country level and global average. We strongly recommend using ecoregions for processes in the foreground system rather than country averages.

Once the functional unit, location of production and the land use inventory associated with the livestock product in question are derived, the biodiversity impacts can be calculated by simply multiplying the inventory (e.g., 50 m² of light use pasture land in Canada per kg beef; Section 0) with the newly available CFs (e.g., 1.31×10^{-14} PDF/m² of light use pasture land in Canada).

Regional CFs do not include the species vulnerability considerations, so the by default the practitioner shall use global taxa-aggregated CFs in LCA studies. The complementary use of regional CFs is also encouraged especially if the supply chain activities are concentrated in certain regions. The global CFs include additional aspects such as endemism and species threat levels and therefore better reflect the potential damage caused to different aspects of biodiversity. The use of taxa aggregated CFs results in a single estimate of a product's impact on biodiversity rather than five different taxa-specific CFs which can complicate comparisons. However, this recommendation shall depend on the goal and scope of the LCA study and disaggregated CFs per taxon could aid the practitioner in results interpretation. It may also help identify practices that are a benefit rather than a detriment to biodiversity.

The updated classes of the CFs can also be directly linked to the land use classes of existing inventory databases (e.g., Ecoinvent) to assess the impacts of background processes used in the LCA of the product (See Table S7 of Chaudhary and Brooks, 2018). The CFs are provided for assessing the impact of both land occupation (PSL/m²) and land transformation (PSL \times year/m²), or aggregated across taxa as global PDF/m² and land transformation in global PDF \times year/m². The model includes both average and marginal factors. The practitioner shall use the average factors for consistency with other indicator methods used in LCA.

Overall, the recommendations on the use of these CFs are consistent with that of UNEP-SETAC life cycle initiative. This initiative has provisionally recommended the use of Chaudhary et al (2015) CFs to assess land use driven biodiversity impacts within LCA. As part of recommendation of UNEP-SETAC it was advised: i) to expand land uses classes and intensities, ii) to include CFs for plants, iii) reduce uncertainty and iv) to conduct case studies to test feasibility, meanwhile the use of those CFs should be limited for hotspot analysis. Chaudhary et al (2018) method recommended by this document has addressed most of these suggestions for improvement. Further improvements such as better confidence intervals and finer intensity classes could allow to go beyond hotspot analyses and to partially differentiate the biodiversity impact of different production systems in the future.

The SAR approach is an evolving field and has its limitations (Halley et al. 2013) such as its assumption of contiguous habitat and the failure to account for the effects of habitat fragmentation that usually accompanies habitat loss (Hanski et al. 2013). However, the spatially explicit data needed to correct for the above factors are not available at the global scale for multiple taxa. Development of more sophisticated methods for use in LCA, including other species groups (e.g., arthropods) and additional indicators of biodiversity loss (e.g., functional or genetic) represent an important future research front.

The recommended CFs have a limited ability to reflect beneficial impacts of livestock production on biodiversity. This is mainly because of the small number of agricultural land use and intensity classes, and use of potential natural vegetation as a reference. A better distinction among land use intensity and management practices – including those with a positive impact on biodiversity (e.g., extensive grazing, Watkinson & Ormerod 2001) – is a priority for increasing the capability of LCA as an analytical and decision support tool for livestock products.

Finally, note that these CFs are only able to assess the impact on a single indicator of biodiversity (e.g., species richness) at the eco-regional (Olson et al. 2001), country or global level, but not at the local/landscape level. Because the CFs are only based on the area of a limited number of land use categories, they are not able to consider elements of landscape structure (e.g., heterogeneity of the landscape mosaic, micro habitats, biodiversity corridors) although those are crucial for biodiversity (refer to landscape scale conservation indicators in section 6.2). Local or landscape level biodiversity is also important for ecosystem service provision and therefore additional indicators shall be employed to understand the damage or benefits to local biodiversity whenever the required data or resources are available. To get a more comprehensive view of the potential impacts, we list several methods/tools that can be applied for local (farm) level biodiversity impact assessment (Section 5.4.2).

5.4.2 Regional/Local impact assessment

Key guidelines

- Regional/local impact assessment will often require the adaptation of existing CFs or the development of new CFs.
- Data availability for the development of local CFs is a challenge, which could be overcome in some contexts through stakeholder engagement.

Regional/local impact assessment often requires the use of a blend of CFs that are adopted from both the global and local scales. Ideally, specific biodiversity monitoring sites will be located over the region of interest in which detailed data has been collected over the time period of interest. Frequently, data of these quality are unavailable for impact assessment. In some instances regional biodiversity may be described in terms of potential loss of species (PSL) from the ecoregion as a result of land use change, but in this case it is more important to be aware that lost species may be replenished as a result of migration from adjacent ecoregions. Assessment of the degree of intactness of the ecosystem and the occurrence of corridor linkages should be considered as these connections may influence the likelihood of replenishment.

Often such detailed data will not be available at a regional scale. In this case the practitioner shall contact local stakeholders for regional information or the producer for the information needed to conduct local

or farm scale assessments of biodiversity. Compiling assessments at the farm or local scale could generate an overview of the impact of livestock on biodiversity at the regional scale. In this case the practitioner shall undertake the steps described in section 6.1 to identify appropriate PSR indicators for the study. If resources are available to collect additional data they shall be expended in a manner that compliments the local scale datasets that are already available within the region of interest.

5.4.3 Reference state

Key guidelines

- In LCIA, impacts on biodiversity are always expressed compared to a reference state.
- The potential natural vegetation (PNV, which describes the mature state of vegetation in the absence of human intervention) is often used as a reference state, but a historic reference or the current mix of land use can also be chosen.
- The CFs recommended by this document (Chaudhary and Brooks 2018) use the PNV as a reference state.
- The reference state decision has important implications for the results. Both the reference state and these implications should be reported and discussed, especially when using different methods.

In LCIA, the reference state shall be used as the basis to compare the environmental quality of the studied system (Milà i Canals et al. 2007). The choice of the reference strongly influences the final results and interpretation of the LCA. The reference state can be based on a variety of temporal points such as the Pleistocene, pre-industrial revolution or even the point prior to urbanization or the establishment of livestock production systems. Selection of the reference state shall also consider the richness of relevant data between comparative years.

Whether LCA methodologies are able to account for beneficial effects on biodiversity depends on the land use reference selected. Three main approaches to defining the reference state have been proposed: 1) the potential natural vegetation, 2) historic land use, and 3) current land quality states.

The concept of potential natural vegetation (PNV) has been used to describe the mature state of vegetation in the absence of human intervention (Chiarucci et al. 2010) and corresponds to the vegetation that would develop if all human activities were to cease, excluding changes in climatic conditions. Selecting PNV as the reference gives similar weight to land use impacts currently occurring (e.g., tropical deforestation) and land use impacts that occurred in the distant past (e.g., deforestation of European woodlands). With this methodology, species-rich, semi-natural grasslands that arise as a result of distant past deforestation are seen as having only a negative impact on biodiversity even though this ecosystem still contributes to biodiversity. An additional limitation of the PNV is that there may be multiple potential equilibria that constitute the mature state of vegetation (Souza et al. 2015). After recovery, the species richness and composition could differ from that of the original natural land cover. Factors such as the types of species present, intraspecific genetic variability and landscape structure in surrounding regions may influence the potential for biodiversity to return to the PNV state.

Alternative options for the reference state include a mix of current land uses, as proposed for Europe by Koellner and Scholz (2008a, b). It is important to recognize that the selection of recent land use states as a reference (e.g., land cover in the year 2000) shifts the emphasis of biodiversity impacts onto

1 contemporary as opposed to historical land use. Considering the state of the contemporary land use
2 reference is also important; for instance, although grassland generally host higher biodiversity levels than
3 cropland, certain crops such as soybean which can fix nitrogen could be used to recover degraded pasture
4 areas.

5 De Schryver et al. (2010) and De Baan et al. (2013) proposed a combination of both a current and a semi-
6 natural reference state. De Baan et al. (2013) defined a reference situation as the current natural mix of
7 natural land cover (i.e., forest, wetlands, shrubland, grassland, bare area, snow and ice, lakes and rivers).
8 Such an approach results in land use changes having a higher impact in areas of natural land cover.
9 Practically, the reference state should also consider short-term changes in land use and the need to
10 preserve traditional land uses that differ from the PNV, such as agricultural landscape, or extensively
11 managed forests. Land cover data do exist at the continental to global scale (Section 8.4.1) and can be
12 used to develop a proxy of the current vegetation state.

13 The selection of a suitable reference is therefore clearly a priority in discussions over how to make the
14 LCA methodology relevant to the livestock supply chain, particularly when making comparative
15 assessments of products or systems at the global to regional scale.

16

6 Local assessments using PSR indicators

6.1 The framework for local assessment

Key guidelines

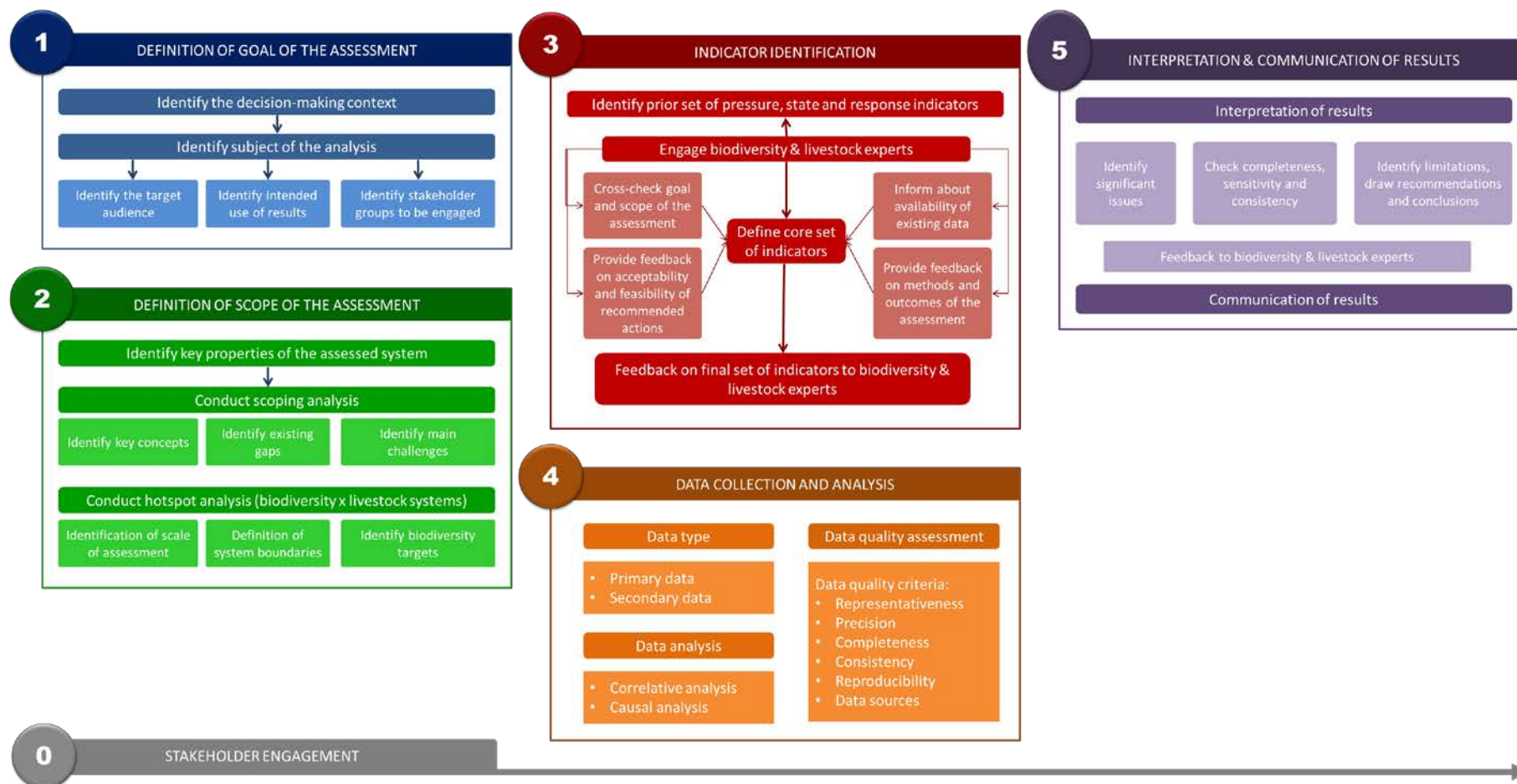
- When conducting a local biodiversity assessment using the PSR framework, 5 major steps can be considered – (1) definition of goal, (2) definition of scope, (3) indicator identification, (4) data collection and analysis and (5) interpretation and communication – along with stakeholder engagement that shall occur iteratively along those 5 steps.

Within the 5 major steps, key principles to consider include:

- The objectives of a biodiversity assessment and the objectives of any related initiatives shall be clearly stated, and appropriate indicators and methodologies chosen to reflect these objectives. The intended use of the results shall also be specified.
- A scoping and a hotspot analysis shall be conducted. The scoping analysis consists of a preliminary assessment of the scope and dimension of the study, in order to map key concepts and issues and to identify gaps and challenges related to biodiversity and livestock production. The hotspot analysis aims to provide a qualitative evaluation of the relative contribution of the livestock system to different biodiversity issues, and to identify the most prominent positive and negative impacts.
- The boundaries of the assessment shall be clearly defined. Processes such as feed production, in particular off-farm feed production, shall be included in the system boundaries of livestock systems. This is due to feed's substantial and increasing contribution to the overall impact on biodiversity.
- Under the PSR model, it will be necessary to select specific pressure, state and/or response indicators to describe, respectively, the pressures from human activities on the environment, the resulting changes in environmental conditions, and the societal response to environmental concerns, either to mitigate negative effects, to reverse damages, or to conserve habitats and biodiversity.
- Given the context-dependency of biodiversity conservation, engagement with multiple stakeholders (i.e., anyone who may be impacted by, or have an impact on, an issue) can improve several facets of the assessment, including goal definition, scoping/hotspot indicators, indicator selection, data collection/analysis and interpretation of results.
- Indicators are identified and prioritized for the biodiversity assessment, based on expert and stakeholder input and relevant resources.
- Relevant information shall be identified and a plan for data collection developed to be able to compute the selected indicators.
- The impacts on biodiversity can be identified through analysis of data collected for the chosen indicators and presentation of results. Data analysis and presentation of results shall be undertaken by personnel with appropriate expertise.
- Interpretation shall be aligned with the goal of the assessment, identify issues, guide decision making for improving biodiversity performance, and discuss limitations. Interpretation shall be undertaken by personnel with appropriate expertise and be dependent on data quality and results.
- Communication of the assessment results shall ensure transparency and be adapted to the target audience

Figure 4 outlines the major steps for a local biodiversity assessment. Details on each steps are provided in the following sections while a list of recommended indicators is provided in Section 6.2.

1 Figure 4. An operational framework for a local biodiversity assessment using pressure, state and response indicators.



6.1.1 Definition of the goal of the assessment

The first step in a PSR biodiversity assessment is to set the goal of the assessment. Guidelines on goal definition are provided in Section 4.1.

6.1.2 Definition of scope of the assessment

Scoping analysis – Several aspects of the scope of the assessment should be defined:

- What features of biodiversity are of concern such as protected habitat loss, habitat degradation and fragmentation, species' extinctions, declines in abundance of species, invasive species, aquatic biodiversity or landscape scale conservation (see Section 6.2).
- Scale – local scale assessments using a PSR approach would typically range from the farm to the landscape or other ecologically relevant unit (e.g., watershed, agricultural region, agroecosystem). Integration of scales is also recommended, (e.g., from soil to landscape).
- Whether the provision of ecosystem services is to be included.

Mapping or ecological zoning (e.g., biome or agroecological zones, ecological-economic context, pedoclimatic conditions, productive history, hydrological properties, pedoclimatic conditions, topography) are tools that can support the scoping analysis, and facilitate interpretation of the indicators later in the assessment. Consultation with stakeholders shall identify prominent biodiversity issues, suitable indicators and the spatial scales to be considered. This shall provide an adequate context for the study and the discussion and analysis of the results obtained by favouring the interest and participation of stakeholders, along with the information for the continuous improvement of biodiversity indicators in the system under study. Other information sources to be reviewed as part of the scoping analysis include the scientific literature, reports from environmental non-governmental organizations (NGOs) – local or international (e.g. WWF, IUCN) – laws and international frameworks. Some countries have agro-environmental programmes offering subsidies for the voluntary adoption of certain environmentally sound practices. The goals of these programmes may also indicate important effects of livestock systems on biodiversity issues and objectives. At the global scale, the Convention on Biological Diversity (CBD) is a multilateral treaty with the goal of the conservation of biodiversity and the sustainable use of its components. It includes the Aichi targets³, established to help reach this objective. These internationally agreed targets can be relevant to the user and may be included in the scoping analysis. When performing a scoping analysis, consideration shall be given to livestock impacts across multiple spatial scales from the local, regional, and national through to the global scale, where relevant to the user's activities. Not all countries have financial incentives to develop sound biodiversity practices. In these countries, the scoping analysis shall rely on emphasizing the value of conservation to local stakeholders and in promoting the functional values of biodiversity with an emphasis on its provision of ecosystem services.

Hotspot analysis - A hotspot analysis for biodiversity issues in livestock systems aims to provide a qualitative evaluation of the relative contribution of the livestock system to different biodiversity issues, and to identify the most prominent ones (e.g. habitat loss, invasive species or aquatic pollution). The spatial scale shall be clearly identified, and off-farm impacts shall be considered. Off-farm impacts occur

³ <https://www.cbd.int/sp/targets/>

when local pressures have an impact on biodiversity outside of the user's system, such as water pollution and GHG emissions, or when a local management action disrupts migratory routes. They also occur when the product's supply chain encompasses more than one geographical area (e.g. imported feed). The hotspot analysis should include this life-cycle perspective and qualitatively evaluate the relative contribution of the different stages of a production system.

System boundaries - The definition of system boundaries shall describe the scope of the assessment in terms of production processes and areas of impact on biodiversity. Regarding production processes, the minimum system boundaries shall include off-farm feed production (when relevant), on-farm feed production and animal husbandry, including grazing and land management. Additional processes may include both cow-calf and fattening operations, processing, transport etc. The geographical scope of all production processes shall be identified and areas overlapping with biodiversity hotspots or other high conservation value areas shall always be included. Even if a farm uses a small share of feed coming from a given high conservation value area, it could have a high impact on biodiversity. The system boundaries may be extended to areas of impact beyond production areas such as catchment and coastal areas impacted by pollution originating from livestock production.

The livestock and the commodity grain sectors should be encouraged to work together to measure and assess biodiversity throughout the supply chain. In this way, livestock farmers who buy (off-farm) feed from the market can be more informed, and better understand the off-farm (or landscape-scale) biodiversity impacts of the products that they buy.

6.1.3 Indicator identification

Scope of P, S and R indicators – The PSR indicator framework provides a way to structure indicators which facilitates interpretation and decision making. Pressure, State and Response indicators have complementary strengths and limitations and the user shall select the categories that best fit the goal and scope of the assessment.

Pressure indicators describe the link between human activities and biodiversity loss, such as habitat change, pollution or climate change. A list of pressure categories can be found in Appendix 4 and more details are provided in the LEAP biodiversity review (Teillard et al. 2016a). Pressure indicators shall be used when there is a significant contribution of the user to pressure categories and good scientific evidence of the link between these categories and biodiversity, as identified by the scoping and hotspot analyses (Section 6.1.2). They could also be used when the user does not have the capacity to collect data and calculate indicators that measure the state of biodiversity. The relative importance of the different pressure categories to the overall impact on biodiversity is difficult to quantify and this limitation shall be discussed when using pressure indicators.

State indicators provide a direct measure of the status of biodiversity and associated habitats/ecosystems, which is ultimately what the user shall act upon and improve. State indicators shall be used to assess change and provide evidence of improvement in the status of biodiversity. Determination of these indicators often requires a significant amount of time, financial resources and expertise. State indicators describing habitats rather than species may be measured more easily. The user shall also identify a specific target regarding the state of biodiversity such as reversing the decline of bird populations, or ensuring the

conservation of certain species or habitats. State indicators tend to be specific, to a given species or taxa, to a given level (e.g., species vs. ecosystems) or dimension (e.g., species composition vs. functional role) of biodiversity. Different state indicators can be used and their values will often be uncorrelated. In some cases, a specific state indicator can be a proxy for wider aspects of biodiversity, but it cannot be comprehensive and this limitation in scope shall be discussed. The choice of state indicators will have a substantial influence on the outcomes of the study; stakeholder engagement will therefore be very valuable in defining key biodiversity issues and selecting the corresponding state indicators (see Section 6.1.6). Essential Biodiversity Variables (EBV) are state variables derived measurements needed to study, report and manage biodiversity change. These variables are divided into 6 EBV classes that contain 21 EBV candidates which describe both the scale and dimensions of biodiversity from a biological perspective that is sensitive to change (Geo Bon 2019).

Response indicators are directly related to management decisions; therefore, the information required to estimate them are often already available. Response indicators shall be used as an indication of mitigation actions, strategies to reverse environmental damages, and/or conserve biodiversity and habitats. The link between the different response indicators and the positive influence on biodiversity outcomes shall be strongly supported by the scientific literature, legal frameworks or private audits or certification. There is no guarantee that responses will actually lead to biodiversity improvement as other factors may have a more important effect, responses may be taken at inadequate scale, or coordination could be lacking between the responses of different stakeholders. Under adaptive management regimes there is an expectation that assessment of response effectiveness leads into another cycle of pressure-state-response analysis and interpretation.

Combining several categories of indicators is strongly encouraged. Using response indicators in combination with pressure and state indicators allows a user to define response actions targeted towards environmental changes, such as biodiversity loss. Indeed, it allows a user to monitor whether societal responses actually result in lower environmental pressures, higher benefits, or improvements in the state of biodiversity. It is also useful to combine pressure and state indicators in order to demonstrate causal links, show the relative importance of the different pressures and to prioritize and catalyze action (Plantureux et al., 2014).

Identifying and prioritizing indicators – The identification of indicators will be strongly guided by the choice of assessment goals, and the scoping and hotspot analyses. There is typically a lot of experience available among key stakeholders to guide indicator selection, and this can help a user to choose indicators that are SMART (Specific, Measurable, Actionable, Relevant and Timely), economically feasible to measure and accepted by stakeholders. The engagement with stakeholders and experts (Section 6.1.6) makes a very important contribution to the process of indicator selection, and to ensuring that the indicators align with the goals and priority issues for livestock effects on biodiversity (both negative and positive). For example, if there is an internationally rare species or habitat(s) in a catchment or region, then the measurement of livestock effects on these should be a priority compared to measurement of parameters such as the length of riparian zones along drainage ditches.

A list of recommended indicators is provided in Section 6.2. It addresses pressure, state and response indicators and a range of categories including habitat protection, habitat degradation, wildlife conservation, invasive species, aquatic biodiversity, off-farm impacts and landscape scale conservation. The list outlines recommendations and not a requirements, users should consider each of the indicators

in turn, and provide a short justification of why an indicator is selected or not, or why an alternative indicator is used. An assessment is not expected to include all indicators. However, the credibility and transparency of the assessment is enhanced when there is a clear justification for decision-making about indicators.

The assessment goals will determine a group of relevant indicators needed to conduct the assessment. An example of such a group is the assessment of species with high conservation values (a response indicator) and their status in a particular livestock farming context. Such an assessment will require picking multiple indicators from the list so that relevant information will be available at the end of the assessment for outcomes such as management decision-making. In this particular case, the multiple indicator list would include: 1) as state indicator, the abundance of species with high conservation value, the % of semi-natural habitats; 2) as pressure indicator, the rate of conversion of semi-natural habitats, livestock density on grazed semi-natural habitats, and action-driven indicators; 3) as response indicators to be monitored, natural and semi-natural habitats are maintained and correspond to species of high conservation value.

The list of recommended indicators is not intended to be exhaustive, and the user, stakeholders or experts can modify these indicators, or identify more appropriate indicators that they consider to better address the goals and impacts as determined by the contribution of stakeholders. When a user declines to use these proposed indicators, they need to provide a written justification (in the viewpoint analysis, see above) of why these were not implemented, and whether this was agreed to by the stakeholders and experts involved.

When indicators are relevant to the livestock system, but there is no information available to quantify the indicator, then a reason should be provided for omission of information in its communication, and possible ways to collect the relevant information shall be identified. Note however, that while indicators should be economically feasible to measure, this does not mean that users cannot expect to devote some budget to data collection and analysis, especially for high priority effects on biodiversity. Indeed, the willingness of users to allocate funds for this purpose is a key test of their commitment to the biodiversity dimension of sustainability.

In addition to the selection of indicators, it is usually also appropriate to define quantitative targets for the indicators. For example, it is one thing to have an indicator 'proportion of area of wildlife habitat on a farm', and it is another to have an associated target of 'not less than 7.5% of farm area occupied by wildlife habitat'. The selection of quantitative targets may be prescribed in the case of some species and habitats such as those that have defined targets associated with their legal protection status. They may be also be defined by the user to reflect their degree of commitment, or may be more qualitative in nature (e.g., increasing trend in population abundance within five years). Undoubtedly, the setting of targets can be difficult and contentious, but it is an important process for the involvement of experts and stakeholders. Indeed, target setting may be a valid Response to a Pressure-State analysis.

6.1.4 Data collection and analysis

Data collection – Because of the inherent complexity of biodiversity and due to the need of simplification in order to provide clear and feasible indicators for the livestock sector, a concerted effort is needed to identify relevant information. Once the goal of the assessment has been established and indicators

selected, it will be necessary to focus on relevant information to provide quantification of the recommended indicators. Depending on the specificity of the indicator, default global values could be provided (e.g., Soil Organic Carbon content, HWSD - Harmonized Soil Database viewer⁴; land use impact on biodiversity at ecoregion level, Chaudhury et al 2015), or site-specific values (e.g., Soil Organic Carbon Content derived from site-specific analysis) could be used for the assessment. The identification of specific data for specific farms or small areas of assessment reduce the uncertainty of the assessment. This specific information could be acquired from previous studies of the area. However, if data are not available, it may be advisable to collect them through a new monitoring study.

Limited data availability should not be used as a reason for excluding important pressure/benefit categories if users have the capacity and financial resources to collect additional data. In some cases, there may be options for structured and organized self-reporting by farmers, although more specialized biodiversity monitoring will essentially require the use of specialist expertise. The willingness of an organisation to commit resources to an effective monitoring programme that collects quantitative information is viewed by many stakeholders as a strong test of commitment to a sustainability programme. In any event, it is imperative that the data are collected in a way that is fit for the purpose and scope of the assessment.

The design of a monitoring programme and data collection (and quality control) protocols is a key activity that shall be undertaken by personnel with the appropriate specialist expertise in this area (e.g., NGOs, researchers, local conservation groups). Thus, for example, there should be a stratification of the sample of farms and randomised selection of farms from the relevant suite of farms. Stratification based on habitat extent, quality, sensitivity, connectivity (at landscape scale), and capacity to monitor or implement practice change and/or location relevant to off-site impacts may provide more information and greater improvements. Important questions will need to be answered along with identification of the relevant data and information such as the temporal and spatial scales at which the indicators have been or will be assessed and the precision level of the assessment required to answer the questions posed. Many universities, NGOs and other local conservation groups concerned with biodiversity have relevant expertise that can contribute to the valid design of a monitoring programme.

Several data sources are indicated in section 8.4.1 and additional guidance on data collection is provided in section 8.4.2.

Data analysis – Users should ensure that several aspects of data collection and analysis have been taken into consideration when carrying out an assessment. These aspects are detailed in Section 8.2 (representativeness) and 8.3. (precision, error, completeness, consistency, reproducibility and uncertainty).

Two types of data can be collected to compute pressure, state or response indicators:

- primary data: defined as directly measured or collected data representative of the livestock operation at a specific facility (for pressure and response indicators) or local biodiversity of a specific area (for state indicators)
- secondary data: defined as information obtained from sources other than direct measurement. Secondary data are used when primary data are not available or it is impractical to obtain them.

⁴ <http://www.fao.org/soils-portal/soil-survey/soil-maps-and-databases/harmonized-world-soil-database-v12/en/>

For example, some data might be calculated from a model, and are therefore considered secondary data.

Primary data should preferably be used to describe foreground processes, that is those that are under the direct control of the user. Secondary data can be used for background processes. In that case, they shall be as specific as possible, that is specific for the supplier of a given input and communicated by that supplier, as well as product-specific or country-specific.

Biodiversity data collection can be very demanding in terms of time, cost and expertise so that users are more likely to use secondary data. Such data are often collected for other purposes and can vary greatly in quality. However, even with secondary data, quality should be assessed and reported according to the recommendations provided in Section 7.

Data analysis will strongly depend on the goal of the biodiversity assessment, the indicators selected, the source of data (primary and/or secondary), and the design of the data collection (including scale). Two main approaches may be highlighted:

- Correlative analysis: pressure and state indicators are or have been recorded in a biodiversity monitoring program. The indicators selected are monitored with the aim of showing a trend over time in a time series analysis. For example, the European Grassland Butterfly Indicator used in 19 European countries (van Swaay et al. 2015). Pressure indicators such as the decline of semi-natural grasslands during the same period of time may be used to assume possible cause-and-effect relationships, but the analysis will be purely correlative. Secondary data will be analysed in this way when pressure and state indicators have been assessed independently. For instance, the change in vegetation (plant diversity, community turnover, etc.) might be measured over a period of 20 years in a particular region, with the stated purpose of assessing the relationship to the change of livestock density in the same region. To this purpose, secondary data on livestock density may be used. The analysis will be correlative and show co-variation;
- Causal analysis: pressure and state indicators are or have been recorded with the aim of explaining causal relationships. This type of data analysis requires specific data recording design within a controlled study. For instance, one may want to know what the contribution of livestock density was to vegetation change over a 20 year period, besides the effects of climate change and atmospheric nitrogen deposition. The causal analysis requires that vegetation is recorded in a set of replicated sites before any grazing occurs, and then along a gradient of livestock densities over time.

The methods and approach to data analysis should be defined early in the design of the assessment. It is extremely important to establish whether a correlative or causal relationship analysis is required and the selected practice should be applied throughout the assessment. This goes together with accurate protocols of methods and techniques for data collection and quality control. Seeking the advice of a biostatistician from the beginning of a biodiversity monitoring program is highly recommended as it helps achieve the appropriate design.

6.1.5 Interpretation and communication

Guidelines on results interpretation and communication are provided in Section 7.

6.1.6 Stakeholder engagement

The role of stakeholders may include, but is not limited to (i) contributing to more effective goal definition (Section 6.1.1); (ii) improving awareness of traditional knowledge and practices about biodiversity; (iii) contributing to the selection of indicators; (iv) informing about the availability of existing knowledge and data; (v) providing feedback on the goal, methods and outcomes of an assessment; and (vi) providing feedback on the acceptability and feasibility of recommended actions.

Depending on the assessment, a formal process of stakeholder analysis may be required to ‘systematically gather and analyse qualitative information to determine whose interests should be taken into account when developing and/or implementing a policy or program’ (Schmeer 2000). It may also be appropriate to conduct a viewpoint analysis⁵ that recognises the existence of multiple perspectives and provides a structured framework for capturing the various requirements of different stakeholders. Stakeholder and viewpoint analyses should result in a written report that documents the agreements and disagreements, and justifies the final decisions on the different steps of the assessment.

It is important to engage stakeholders, consult experts and access relevant information from other resources to identify the current or past biodiversity state within the system boundaries. Stakeholders can help to verify whether any plans or projects might be in place or in development to improve the state of biodiversity, and to advise on the mitigation of biodiversity impacts. Stakeholders can also support the selection of assessment methods and tools, as well as the identification of solutions for the mitigation of impacts. Experts can also provide such information, and have a more important role in providing specialized skills that can assist the validity, efficiency and effectiveness of an assessment. Depending on the goal of an assessment, there may be a need to include stakeholders with specialised expertise (‘experts’) to conduct part of the initiative (e.g., measuring population trends in a threatened species, conducting habitat surveys, analysing ecological data). If it is to be effective and credible, engagement with stakeholders and experts should be continuous, with regular interaction at key points in the planning, implementation and interpretation of a biodiversity assessment.

Where an assessment results in recommended actions, stakeholder engagement is necessary to achieve acceptance, especially if there is need for a coordinated response involving the user and multiple stakeholders, which is often required to improve the state of biodiversity. For instance, coordination of several farmers or groups of farmers can provide a response at the landscape level, and coordination along the supply chain can ensure that both on-farm and off-farm feed cultivation lead to biodiversity improvements. Stakeholders are also able to provide a good indication of the wider response to an assessment, and whether it has sufficient content and clarity of communication to be trustworthy and is likely to be accepted.

⁵ <http://ldc.usb.vt/~abianc/materias/ci4712/Principles-ProblemsRequirementsEngineering-Sommerville.pdf>
Other reference?

6.2 Recommended list of biodiversity indicators for local assessments

Key guidelines

- This section provides a list of recommended pressure, state and response indicators addressing key thematic issues: habitat protection, habitat degradation, wildlife conservation, invasive species, aquatic biodiversity, off-farm impacts and landscape scale conservation.
- The indicators in the list are recommendations and not requirements, users shall consider each of the indicators in turn, and provide a short justification of why an indicator is selected or not, or why an alternative indicator is used.

As a good practice, the selected indicators shall include:

- All indicators related to 'procedural checks';
- At least one indicator from each category (pressure, state and response) to show if actions do have an effect on decreasing pressure and improving the state of biodiversity;
- At least one indicator for each of the thematic issues identified as relevant during the scoping and hotspot analyses;
- Indicators reflecting potential
- inter-linkages and trade-offs identified during the scoping and hotspot analyses;
- Indicators reflecting both positive and negative impacts on biodiversity;
- Indicators covering off-farm impacts, when relevant.

Table 2 provides an overview of the recommended indicators. More details about the indicators (along with formulas where relevant) are provided in Appendix 5 and an extended indicator list is provided in Appendix 6. Indicators are mainly structured by key thematic issues:

- **Habitat protection** – When livestock affect terrestrial habitats, impacts are not restricted to biodiversity losses; the modifications can also be beneficial to biodiversity. Grazing shapes grassland ecosystems and can increase plant species richness under adequate management (Section 3.1). Farmlands can also provide a variety of habitats (e.g., soil, grass, fallow, shrubs, trees, wetlands) and resources (e.g., seeds, flowers) for a variety of species. Supporting such habitat variety generates high nature farmlands (Baldock et al. 1993) hosting a high biodiversity of farmland species.
- **Habitat change** – Habitat change is the most important global driver of biodiversity loss (MEA 2005). Livestock production has an important contribution to this driver as it is estimated that 30% of the Earth's land surface is dedicated to livestock production through pastures ($\approx 25\%$) and feed crops ($\approx 5\%$) (Ramankutty et al. 2008; Monfreda et al. 2008). Livestock therefore affect some 30% of terrestrial habitats but their intervention can occur in very different ways, from protection to degradation or destruction. The most drastic pressure leading to negative habitat modification (e.g. habitat destruction) is the transition from one land cover (such as forest or grassland) to another (such as grassland or cropland). Degradation refers to soil and vegetation degradation and to slow transitions between land cover classes (e.g. encroachment from rangeland to shrubland, desertification from rangeland to bare soil).
- **Wildlife conservation** – Many wildlife species are under the direct influence of land managers as wildlife habitat is often intertwined with farmlands. These farmlands may serve as habitat or food resource for wildlife, or act as linkages between natural habitats enabling population movements and genetic variation. Farmers can thus play a direct role in protecting those species and their

1 habitats. Information is a key factor to achieve this protection. The species and habitats under the
2 direct influence of the land managers need to be identified, mapped and monitored. This
3 information can be used to establish a biodiversity action plan where detrimental practices are
4 avoided and practices protecting or promoting wildlife species are adopted.

- 5 • **Invasive exotic species** – Invasive exotic species are defined by the CBD as species whose
6 introduction and/or spread threaten biological diversity. They are a major threat to biodiversity
7 at global scale. Along with other vertebrates, livestock contribute to the seed dispersal of invasive
8 and native plant species (Rejmanek et al., 2005). Among other human activities, livestock
9 production contributed to the dispersal of invasive species around the world. Species invasion is
10 a complex phenomenon influenced by a wide range of factors. The introduction of an alien species
11 is a common starting point, but whether invasive species are a cause or a consequence of
12 ecosystem degradation is often unclear (MacDougall & Turkington, 2005; White et al., 2013).
13 While native species can also show excessive and harmful population increase (e.g. bush
14 encroachment in rangelands), this is a different process treated in the ‘habitat change’
15 (degradation) category.
- 16 • **Pollution & aquatic biodiversity** – Livestock production is responsible for two main types of
17 pollution having, in turn, negative impacts on biodiversity: nutrient pollution and ecotoxic
18 pollution. Nutrient pollution can be caused by fertilization at the feed production stage; however,
19 it is often most important at the farm stage. As nutrient capture by animals is quite inefficient, a
20 large amount of nutrients is concentrated in urine and manure. With improper management
21 practices, excess nutrients can enter soils and surface water where it causes eutrophication (i.e.,
22 the growth of nuisance species of algae and aquatic weeds harmful to other native freshwater
23 species). The LEAP nutrient guidelines describe how to account for the nutrient flows and
24 associated environmental impacts in livestock supply chains (FAO 2018c). At the feed production
25 stage, ecotoxic pollution is caused by pesticides. Hormonally active pesticides have adverse
26 effects on a wide range of organisms (Colborn, Saal & Soto 1993). Ecotoxic substances can also be
27 used at the animal production stage in the form of veterinary products, antibiotics, anthelmintics
28 and hormones that can contaminate water and impact aquatic biodiversity.
- 29 • **Off-farm feed** – As stressed in the LEAP biodiversity principles, the impacts of a livestock farm on
30 biodiversity do not only concern on-farm wildlife species under the direct influence of the farmer.
31 Agricultural supply chains are increasingly globalized, with production sites connected by complex
32 international trade routes. For instance, in 2011, 58 million tonnes of soybean meal were
33 exported by 86 countries and imported by 114 countries to feed livestock (FAOSTAT 2013). An
34 important share of this meal is produced in areas that were previously Amazonian savannahs or
35 rainforests, i.e. biodiversity hotspots. Natural ecosystems are usually converted first to cattle
36 pastures, and this land can then be sold for soybean production, with a risk of profits being
37 reinvested in buying and clearing forests in other areas (Gollnow & Takes 2014). In some cases,
38 the off-farm impacts on biodiversity associated with imported feed can be more important than
39 on-farm impacts (Teillard et al. 2016). Such off-farm impacts should always be included in
40 biodiversity assessments of livestock production, except when use of off-farm feeds is negligible.
- 41 • **Landscape scale conservation** – When livestock cause habitat destruction, negative effects on
42 biodiversity are often worsened by fragmentation because a given area of original habitat
43 fragmented into small and distant patches will sustain fewer species than a single, continuous
44 patch of the same area. Conversely, if patches of original habitat are large and in proximity to one

1 another, connecting them with wildlife corridors provides a conservation opportunity. By
2 reducing the possibilities for organism mobility, fragmentation impacts gene exchanges as well as
3 biodiversity at species and ecosystem levels.

4 Table 2 also indicates the P, S or R category of each indicator, whether their measures are quantitative or
5 qualitative, if they are relevant for feed production or animal husbandry stages and if they are directly link
6 to ecosystem services.

7

- 1 Table 2. Overview of possible indicators and their characteristics that can be used in biodiversity assessments. In the first column, bold text indicate the themes used to group
2 indicators (rows) while italic text describe actual metrics that can be used for each indicator. The other columns indicate the indicator category (Pressure, State or Response);
3 whether the measures used for the indicator are usually qualitative or quantitative; the stage of livestock production (animal husbandry, feed production or both) that can be
4 targeted by the indicator; and if the indicator can describe the ecosystem level in addition to the species level.

Thematic issues Indicators <i>Examples of measures (when several are possible)</i>	P, S, R category	Qualitative (QI) or quantitative (Qt)	Relevance to feed (F) production and animal husbandry (A)	Strong link to ecosystem services
Procedural checks				
A scoping analysis was conducted	R	QI		
Regulatory constraints and extrinsic value are considered. <i>Regulatory constraints include:</i> <ul style="list-style-type: none"> • <i>International frameworks such as international biodiversity hotspot, WWF ecoregions with outstanding biodiversity features, IUCN red listed species.</i> • <i>National regulations such as protected areas and species. If national regulations are not met, their perverse or ill-informed nature should be justified.</i> <i>The extrinsic use value defined by local stakeholder should also be considered.</i>	R	QI		
Progress is monitored	R	QI		
Stakeholder engagement <i>Perspective/stakeholder analysis</i> <i>Iterative stakeholder engagement</i>	R	QI		
Data quality <i>Refer to Section 8.3</i>				
Habitat protection				
Wildlife habitats under the farm influence are inventoried (mapped) and protected	R	QI/Qt	A/F	Yes
Semi-natural habitats in the landscape <i>Area or proportion (relative to the area controlled by the user)</i>	P	Qt	A/F	Yes
Grassland restoration	R	Qt	A/F	Yes

Area of degraded grassland restored through improved grazing management				
Habitat change				
Soil erosion and soil erosion risk are mapped and a management plan is implemented <i>Information related to soil erosion risk: soil type, slope, burning, precipitation, wind, bare soil cover, vegetation type. Key information will depend on the ecosystem (e.g. slope not relevant everywhere, bare soil particularly relevant in dryland rangelands).</i>	R	Ql/Qt	A/F	Yes
Degraded soil <i>Area or proportion (relative to the area controlled by the user) of degraded soil, including bare soil or areas with bush encroachment</i> <i>Soil organic matter content</i> <i>Modelling of carbon, nitrogen and phosphorus cycles</i>	P	Qt	A/F	Yes
Livestock density <i>Livestock density in number of animals or other livestock units (e.g. tropical livestock units) per ha</i> <i>Where relevant (for instance in more humid grazing lands where rainfall > 800 mm / year) livestock density can be compared to carrying capacity, i.e. the maximum livestock density for which livestock requirements (based on their live weight) can be fulfilled by grassland biomass productivity (in kg of dry matter).</i>	P	Qt	A	Yes
Habitat conversion <i>Area or rate of conversion of natural and semi-natural habitats</i>	P	Qt	A/F	Yes
Wildlife conservation				
Priority actions promoting species with high conservation value are listed and implemented <i>High conservation value includes national and international designations, but also the functional role of the species, and the perspective of local stakeholders.</i>	R	Ql	A/F	Yes (depending on the species)
Particular species (with high conservation value) <i>Presence/absence, abundance and/or distribution</i>	S	Qt	A/F	Yes (depending

				on the species)
Species richness or diversity <i>Number of species</i> <i>Shannon or Simpson index of diversity</i> <i>Functional diversity, Mean Trophic Index</i>	S	Qt		
Invasive exotic species				
A management plan is in place for the control of invasive species	R	Ql	A/F	Yes
Invasive exotic species <i>Presence/absence, abundance and/or distribution</i>	P	Ql/Qt	A/F	Yes
Pollution & aquatic biodiversity				
A management plan is in place for the application of ecotoxic agrochemicals <i>Pesticides, veterinary products</i>	R	Ql	A/F	Yes
A nutrient management plan is in place to rationalize fertilizer application	R	Ql	A/F	Yes
Protected waterways <i>Length or proportion (relative to length controlled by the user, or to the length in need of protection)</i>	R	Qt	A/F	Yes
Biological indicators of water quality	S	Qt	A/F	Yes
Off-farm feed				
An inventory of the off-farm feed being used is established	R	Ql/Qt	F	
Traceability systems for feedstuff is implemented	R	Ql	F	
Share of imported feed <i>Share of imported feed from areas that are certified/deforested/of high conservation value</i>	P	Qt	F	Yes
Landscape scale conservation				
Measures to promote connectivity between habitat patches and between water bodies identified and implemented	R	Ql	A/F	Yes
Landscape heterogeneity	R	Qt	A/F	Yes

<i>Spatial Shannon index of diversity</i> <i>Landscape diversity conservation index, Brillouin Diversity index</i> <i>Connectivity measures</i> <i>Average area and distance between patches of habitats</i>				
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7 Interpretation and communication

7.1 Interpretation of results

Key guidelines

- The interpretation stage makes use of available evidence to evaluate, draw conclusions, and inform specific decision- and policy-making contexts.
- Interpretation should be aligned with the goal and scope of the assessment.
- The limitations to robustness, uncertainty and applicability of the assessment results also need to be explicitly discussed.

In LCA, life cycle interpretation is the phase in which the outcomes and various steps of the life cycle study are evaluated, quantitatively and qualitatively, in order to provide robust recommendations to inform policy- and decision-makers, and stakeholders (ISO, 2006). Within the LCA framework, there are clear guidelines for the interpretation of results, which are (Section 7):

- identification of significant issues based on the results of the LCI and LCIA steps;
- completeness, sensitivity and consistency checks; and
- conclusions, limitations and recommendations.

In PSR, the interpretation stage will similarly make use of available evidence to evaluate, draw conclusions, and inform specific decision- and policy-making contexts. In particular, qualitative information is used to provide supplementary information to explain the linkages between drivers, changes in state and potential response actions. The interpretation of the results will be closely linked with data quality issues, including measurability issues, over different time and spatial scales.

For both LCA- and PSR-based approaches, the interpretation phase should be aligned with the goal and scope of the assessment. This means it should deliver answers to the question(s) raised and the assumptions made during the goal and scope definition, and provide knowledge to the intended audience, in order for them to develop appropriate decision-support strategies and conservation actions. The limitations to robustness, uncertainty and applicability of the assessment results also need to be explicitly discussed. Stakeholders can provide important inputs and feedback on the interpretation of the evidence.

The desired outcomes of an assessment may not be apparent because of long delays (and sometimes distances) between practice change and measurable change in state indicators. Therefore, a lack of apparent response in state indicators cannot always determine whether the response practices have been successful or not. An understanding of the underlying cause-and-effect relationships can help guide expectations on the temporal scale over which responses should be evident (this is where experts can make a valuable contribution).

7.2 Developing effective communication

Key guidelines

- A major success factor in maintaining and improving sustainability (including biodiversity) is the successful transfer of information, and the achievement of cultural awareness and appreciation of biodiversity.

- Information provided should be transparent about the aims and methods of an assessment.
- For transparent communication, the limitations of an assessment should be clearly described and discussed.

A major success factor in maintaining and improving sustainability (including biodiversity) is the successful transfer of information, and the achievement of cultural awareness and appreciation of biodiversity. As part of a wider set of activities to foster such awareness and appreciation, the results of monitoring programmes should also be communicated externally. This can help to illustrate successes where they occur, and help motivate farmers, consumers and other stakeholders. Where appropriate, the wider public should be kept informed of progress in biodiversity initiatives. Where monitoring indicates a lack of success, such quantitative information should also be useful in guiding and justifying the introduction of management actions that are more likely to be successful.

Information provided should be transparent about the aims and methods of an assessment. This should include: the methods chosen, the outcomes, the action plans following the assessment and any limitations related to the assessment or information. In particular, information should be communicated in a clear and understandable form, be complete, reliable, comparable (over time) and accurate. Communication should include information about boundaries, timelines, assumptions, resources consulted and stakeholders engaged. Tools may include guidance about communication of biodiversity assessment outcomes.

For transparent communication, the limitations of an assessment should be clearly described and discussed. First, a completeness check should ensure consistency between the goals of the assessment, its scope, its system boundaries and the assessment methods selected. Secondly, sensitivity checks should assess the extent to which the study outcomes are affected by methodological choices such as system boundaries, data sources, and the choice of indicators. If relevant, a quantitative sensitivity analysis can be performed. Biodiversity is a complex issue and its assessment will always involve simplifications and assumptions; the consequences of these should be discussed.

7.3 Policy implications

Key guidelines

- LCA has arisen as a structured, comprehensive, internationally standardized tool that is capable of offering objective data for use as an environmental decision support tool; however, there is a risk for policy-makers to assume that LCA generates simple answers to complex environmental questions, especially with non-climatic impacts like biodiversity for which describing the complexity with models remains a challenge.
- It is critically important to model impacts at adequate spatial and temporal scales, particularly by using more accurate local and regional data, and to use appropriate indicators to address policy- and decision-making processes, bearing in mind that specific indicators for one biodiversity level or dimension such as species composition are not fully adequate to depict linkages between ecosystem function, biodiversity and ecosystem services.

1
2 With continued global biodiversity loss, there is a strong societal demand to measure the environmental
3 impacts of livestock production at the global scale and devise strategies to address these effects. The
4 ecological footprint is an easy-to-grasp concept, rooted deeply in popular culture and is gaining increasing
5 prominence in the scientific literature (Wiedmann & Barrett 2010; Hoekstra & Wiedmann 2014).
6 However, given that it is limited to land use analyses and is difficult to extrapolate to implications for other
7 ecosystem services, its usefulness for setting policy can be limited (Kovacic & Giampietro 2015).

8 LCA has arisen as a structured, comprehensive, internationally standardized tool that is capable of offering
9 objective data for use as an environmental decision support tool (Čuček et al 2014). Consequently, LCA
10 has emerged in the regional and global scenarios as a key element in assessing potential environmental
11 impacts of products and services to support decision making at industry and government levels (Hellweg
12 and Milà i Canals 2014). LCA provides an opportunity to provide a transparent comparative analysis of the
13 effects of livestock production across a range of production systems and environmental conditions. This
14 exercise can point out key hotspots in the supply chain and identify strategies for improvement such as
15 the preservation and proper management of grasslands. Biodiversity assessments can be coupled with
16 the analysis of other social and economic attributes that consider animal health and welfare, and other
17 economic performance indicators of the production system (Maia de Souza et al. 2017). These
18 assessments are well aligned with the increasing need for the livestock industry to provide transparent
19 information to the consumer regarding its efforts to continuously improve production standards and meet
20 sustainability goals.

21 However, the LCA scientific community has been raising the alarm for the last two decades on the danger
22 of not grasping the analytical complexity in attempting to use LCA to generate policy (Bras-Klapwijk 1998;
23 De Benedetto & Klemeš 2009; Wardenaar et al 2012). Often policy-makers make the mistake of assuming
24 that LCA generates simple answers to complex environmental questions. There is a clear danger of
25 oversimplification of messages and derived decisions. This is particularly true for non-climatic impacts,
26 such as biodiversity and ecosystem services where model complexity is increased across temporal and
27 spatial scales and the development of robust models remains a challenge (Chaplin-Kramer et al. 2017).

28 Most sustainable development initiatives have fallen short at appropriately identifying and selecting
29 indicators that adequately describe biodiversity and ecosystem service losses, often as a result of a lack
30 of proper scale factors (Bunnell and Huggard 1999). It remains critically important to address this issue by
31 modeling impacts at adequate spatial and temporal scales, particularly by using more accurate local and
32 regional data. Increased accuracy at the local and regional scales will help to further recognize the impact
33 of hot spots on biodiversity and to allocate these impacts to the correct components within the production
34 chain. In addition, appropriate indicators are required to address policy- and decision-making processes
35 and, while species-based approaches have been the norm, they are not fully adequate to depict linkages
36 between ecosystem function, biodiversity and ecosystem services (Flynn et al. 2009). This is particularly
37 true for livestock production systems where local and regional LCI data may be available, but must be
38 coupled with ecological data generated on different temporal and spatial scales.

39 Decision-making in relation to food systems is greatly affected by social factors, which are central in the
40 wider biodiversity-related land-sharing vs. land sparing debate (Fischer et al. 2013) – freeing land for
41 conservation uses through agricultural intensification can potentially have a direct negative outcome on
42 food sovereignty because of increased capital needs. Interactions are complex, with family farming

1 systems apparently yielding more food than industrial ones (Samberg et al 2016), possibly because of their
2 multifunctional design (Altieri et al 2012). In the concrete case of livestock, the picture is further
3 complicated because of the proven beneficial effects that some low-yielding production types can have
4 by providing key ecosystem functions (Teillard et al 2016, Manzano-Baena & Salguero-Herrera 2018), but
5 it may depend on the ecosystem and the type of grazing provided (e.g. functional similarity to wild
6 herbivores, Bond et al 2004, Bond & Silander 2007).

7 There is considerable debate with regard to promoting intensified livestock systems that exhibit high
8 levels of production with more focused impacts on biodiversity over extensive livestock production
9 systems that usually have less serious environmental impacts, but produce less livestock meat and milk.
10 These systems generally need to occupy more land area to produce the same amount of livestock product.
11 The optimal system likely involves a trade-off between the two extremes that is tailored to local
12 environmental conditions and ensures the prudent use of available natural resources (Garnett et al. 2013),
13 including integrated (e.g. crop-livestock, sylvopastoral) systems.

14 Assessing these impacts and promoting the livestock sector's environmental improvements is important
15 to reach the Sustainable Development Goals (SDGs) in local and regional economies, in particular those
16 of developing countries, where livestock contributes to ca. 40% of the agricultural GDP (Veerasamy et al.
17 2016). At a global scale, compliance with the UN SDGs, in particular SDG 12 (responsible consumption and
18 production), 13 (climate action) and 15 (life on land), has required information on the environmental
19 performance of economic activities, including livestock production worldwide. Furthermore, the
20 application of LCA to identify scenarios of further development and/or intensification, provides additional
21 information for policy decision-making at different scales, ranging from local (e.g. regional, basin) to global
22 (e.g. national) levels.

23 A policy scenario approach can highlight the importance that livestock systems have, both in terms of
24 negative and positive impacts. This is something that the current state of opinion doesn't seem to be
25 grasping, with widespread attacks to extensive livestock systems that have been flagged as emission-
26 intensive, even if other environmental benefits including biodiversity conservation and the provision of
27 certain ecosystem services in part compensate for higher emissions (Ripoll-Bosch et al. 2013).

28 The results of biodiversity assessments can bring scientific evidence to support policies for sustainable
29 rural development at landscape to regional scale, promoting the conservation of natural areas and their
30 connectivity within the landscape, in support of ecological processes. The cost of protecting, restoring and
31 maintaining protected areas established within rural property boundaries is most often bared by rural
32 producers, especially in developing countries. Policies emphasizing biodiversity conservation at landscape
33 scale can provide a tax/incentive framework for natural resource preservation, stimulate and enable the
34 protection of endangered species, raise the interest of companies' investments in the preservation of
35 areas, and may even generate income from tourism.

8 Data and data quality

8.1 Introduction

Key guidelines

- Biodiversity data should either be aligned with the scale at which the analysis will be conducted, when relevant, and be scalable to enable cross-scale analyses.
- When using data at a large geographic scale the risk of simplification, lack of specificity and not considering all aspects and interactions should be minimized.
- When using data at small geographic scale, the risk of lacking representativeness and over generalization potential should be minimized.

The study of biodiversity is complex in nature and requires a deep understanding of different factors interacting and shaping wild species communities (e.g. animal, plant, fungi, soil microbial organisms, etc.). Measuring the impacts of livestock production decisions on biodiversity is challenging, particularly when moving across scales (Levin 1992, Poiani et al. 2000) (Figure 5). Biodiversity data should be aligned with the scale at which the analysis will be conducted, when relevant, and/or be scalable to enable cross-scale analyses. Data on drivers and pressures should consider both the scale of potential impacts and the scale of the underlying mechanism of impact. Finding enough data with the necessary quality and geographical extent that can fit the needs of each scale and analytical method is a major task and should be addressed accordingly. Data needed when focusing on larger geographical scales will suffer from simplification of nature's complexity, lack of specificity and from not considering important ecological factors, ecosystem processes and functions and species interactions shaping wild species communities (Bunnell and Haggard 1999). For example, most data available for large-scale analyses are usually based on simplistic information such as number of species in a given region, when it is widely accepted that biodiversity is much more than just species richness (Marchese 2014). On the other hand, data needed when focusing on smaller scales will suffer from the amount of effort necessary to generate enough information to be able to extrapolate and have a more comprehensive view of the issues at larger geographical scales. It is important to understand beforehand that each approach will have limitations and thus the relevant scale should be carefully selected depending on the scope of the study and data availability.

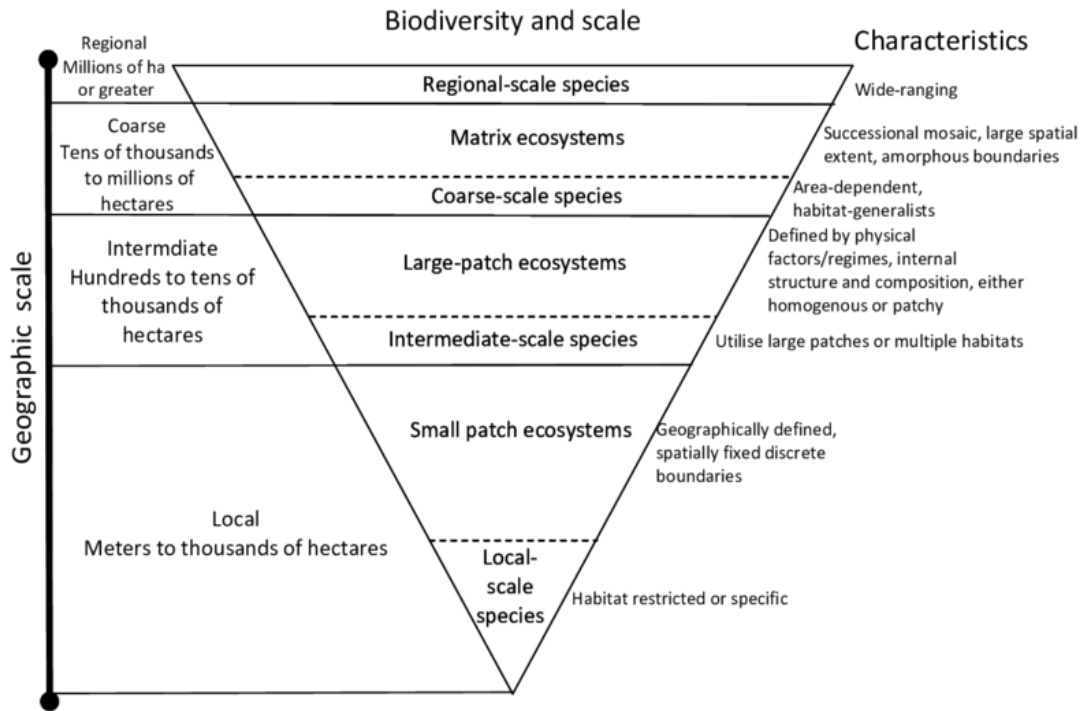


Figure 5. Biodiversity at various spatial scales (from Poiani et al. 2000).

8.2 Representativeness

Key guidelines

- Data used in biodiversity assessment should be representative regarding three main aspects: time, space and taxa.
- Representativeness should be considered when designing the sampling procedure for data collection.

This section covers three aspects of the representativeness of data for biodiversity assessments:

- Temporal representativeness:** The age of the data and the length of time over which it was collected, and data ability to describe the system or to capture a specific event or change of interest;
- Spatial representativeness:** The geographical area from which data for unit processes was collected to satisfy study goals; and
- Taxonomic representativeness:** The taxonomic breadth that is deemed appropriate to the scope of the assessment.

Representativeness should be considered in the study design when collecting primary data, where the data is collected directly by the investigator using collection methods suitable for the objectives of the study, as well as parameters of data quality and fitness for secondary data that has been collected by others.

1 In determining the fitness for purpose of available data, representativeness in both spatial and temporal
2 scales needs to be considered. For instance, sampling of a natural population seeks to provide a
3 representative assessment of that population. If the data fails to represent the population then the
4 analysis may generate a biased outcome that will impact the level of confidence in any interpretation,
5 directly affecting the relevance of the results for decision making purposes. Hence, confidence in the
6 interpretation of data requires knowledge of the underlying data quality, including how well the data
7 sample represents the underlying natural population. Furthermore, depending on the objective of the
8 study, data age and temporal spread could be equally as important as spatial coverage. In general, data
9 sources for temporal aspects of biodiversity (e.g. trends in state over time) are more scarce than spatial
10 data for assessments of state, which reflects the general lack of long-term monitoring programs. However,
11 long-term data, with regular monitoring (daily, monthly, annual, bi-annual, decadal, etc., depending on
12 the variable and expected rate of change), are often essential to assess biodiversity responses to
13 environmental change, be it as a result of land use or climate change, and to define reference states that
14 are more meaningful for management and representative of the desired state. “Surrogate” options for
15 the reference state such as pre-human state often provide little direction, especially if a change in current
16 human activities to the benefit of biodiversity is the desired outcome.

17 In relation to taxonomic representativeness, if the defined scope of an investigation encompasses a broad
18 taxonomic inventory, then data representativeness will be weakened if data sources only cover certain
19 taxonomic groups. For example, an assessment of Coleoptera may provide a relatively poor sample if the
20 project scope requires an inventory of all Insecta. Within the scientific literature there is a marked bias in
21 taxonomic representativeness, with vertebrates being over-represented and invertebrates under-
22 represented in many global data sources and in global assessments of extinction risk (Newbold et al. 2015;
23 Proença et al. 2017).

24 Some description of data representativeness is recommended for any biodiversity assessment, regardless
25 of scale. Using ‘best available’ data for an assessment, or undertaking a new survey for a specific objective,
26 almost inevitably requires some compromises on data quality and hence the representativeness of the
27 sample taken from a natural population. Representativeness is often described qualitatively, but
28 quantitative assessment is preferred.

29 A number of studies have identified spatial and temporal biases arising from non-representative
30 biodiversity data sources. For example, biodiversity data richness and time series availability is frequently
31 strongly skewed at global (Collen et al. 2008; Boakes et al. 2010; Proença et al. 2017) and regional (biome)
32 scales (Martin et al. 2012). In general, assessments at global scales show a strong ‘temperate’ bias,
33 particularly in the northern hemisphere. This affects the representativeness of both species richness and
34 time series datasets for global assessments.

35 At a regional level (biome/anthrome) there are also clear biases in the availability of published data, with
36 under-representation of modified habitats in biodiversity assessments. For example, ecologists tend to
37 work in relatively unmodified habitats, even though these make up a relatively small percentage of the
38 landscape (Martin et al. 2012). The latitudinal biases seen at the global scale also have significant
39 consequences for data representativeness at the regional level. In mid-latitudes, available monitoring
40 data to inform regional assessments are scarce and greater investment is required to fill gaps in data
41 coverage.

Livestock production systems are globally distributed across the latitudinal range and by their nature are associated with anthropogenically-modified landscapes. Hence, the spatial and taxonomic biases identified in data sources globally create issues for the assessment of biodiversity impacts, particularly with regard to livestock production. Representativeness of reference conditions is of specific interest to the assessment of impacts from livestock production systems on biodiversity. Measurement of impact requires a point of reference, and the choice of reference point is critical to robust assessment of impacts. Spatial and temporal bias in datasets can impact the ability to use consistent reference points. For example, in some livestock production landscapes, near-natural habitat reference points may be under-represented in datasets, or absent altogether. Assessments using near-natural or “pre-human” reference points may force non-representative or ‘surrogate’ reference points to be used, introducing a potentially significant source of sampling error.

Understanding the spatial, temporal, taxonomic or thematic gaps in available data can allow researchers to develop appropriate strategies to compensate for these gaps (Proença et al. 2017).

There are a range of strategies to account for biases in data representativeness:

- If the level of bias is quantifiable then the known bias can be controlled for in modelling (Newbold et al. 2015);
- A greater effort can be made to obtain data (or digitize existing records) that are more spatially representative (Feeley and Silman 2011);
- Available datasets can be sub-sampled to increase representativeness (e.g. rarefaction methods)
- Indicators that are informed by more coarse data can be used, to reduce sensitivity to bias in sampling (e.g. changes in occupancy versus changes in abundance);
- Data can be captured at coarser scales using methodologies that are less sensitive to spatial bias (e.g. remote sensing versus field surveys).

Appropriate techniques for reducing spatial, temporal and taxonomic bias in datasets should be considered as an integral part of data quality assessment.

8.3 Data quality assessment

Key guidelines

- Data quality should be assessed, reported and discussed.
- Data quality assessment should include several key criteria – precision, error, completeness, consistency, reproducibility and uncertainty – which are described in the following sections.
- Databases supporting biodiversity assessment in livestock should ideally be made open-access.

As part of any biodiversity assessment, data quality should also be assessed, reported, and its potential impact on results should be discussed. Data quality assessment should consider the six criteria described in the following sections. In addition, it is strongly recommended that databases supporting biodiversity assessments in livestock are made publically available. This raises the credibility of the results, strengthens the data quality assessment through an open discussion, and improves the primary data use, which is often generated with public (national or international) funds. Open-access databases have benefits for

transparency and data collection continuity, and they can support evidence base public policies for sustainable rural development.

8.3.1 Precision

Precision is a measure of the data's variability for each expressible data point (e.g. standard deviation). According to Graham et al. (2004), there are three major issues surrounding the utility of biodiversity databases for spatial modelling; (i) error, including error in taxonomic identification and spatial error; (ii) bias, primarily the geographical and environmental biases associated with *ad hoc* data collection; and (iii) presence only versus presence-absence data, which influences the type of modeling algorithm that can be used.

In biodiversity collections, presence data indicate that researchers observed a species in a given location at the time of sampling, but it does not provide information on the abundance of the species, only that it was present in that place at that moment. Limitations associated with presence data include species that might no longer be present at a historic local collection, or sampling locations might represent a demographic sink for the species. Contrary to presence data, absence records do not necessarily inform on species absence at a certain location and time. Absence might indicate that a particular species was truly absent at a site or there was a failure to detect the species. If the latter, occupancy models (MacKenzie et al. 2006) can be used to correct for false zeros (false absences). These types of statistical models are extremely useful when working in regions where cryptic species are present but are usually recorded as absent.

For modeling techniques requiring real absence data, surrogate 'pseudo-absence' points can be created using several approaches:

- Sampling of locations from which collections have been made, but the species is not recorded (with reference to field notes);
- Sampling of habitat types or regions judged not to include the species in question;
- Sampling across the region, but excluding sites with presence records.

Although there is a possibility of including some false absence (i.e., presence undetected), pseudo-absence points can serve to increase the range and statistical power of applicable methods (Cerasoli et al. 2017).

8.3.2 Error

The identification of species can be: correct (no error), incorrect (misidentification), correct but based on incomplete knowledge (cryptic species), or correct but based on outdated knowledge (synonyms). Identification errors can be detected based on conflicting name usage across collections, or distribution records that are suspect because they exist in a different geographical or environmental space than the rest of the records of a given species (Graham et al. 2004).

To avoid such errors, data from biodiversity collections should be used in the context of a thorough knowledge of the study group's taxonomic history, in many cases requiring physical examination of the

specimens themselves. Spatial error includes georeferencing error, inaccuracy of location of a record, and error in the original location of a record.

Records with these types of error can often be detected because they represent outliers in geographical or environmental space or because discrepancies exist between the georeferenced location and the collector field notes.

Spatial errors can usually be corrected by checking specimens and archived notes, eliminating or down weighting suspect records, and including precision estimates in georeferencing.

8.3.3 Completeness

Completeness is a dimension of the data that indicates sufficiency for a given task. Completeness can be defined intuitively (i.e., data perspective), theoretically (i.e., real-world perspective) or empirical (i.e., user perspective). An example of lack of completeness is when essential information for an analysis is missing from the data. For example, in a potential distribution analysis, incompleteness would be associated with a lack of geographic coordinates or a misidentification of a species. Completeness problems are generally associated with missing values, incorrect data values and non-atomic data values (i.e., occurrence of multiple values when there should be a single value).

8.3.4 Consistency

Consistency: qualitative assessment of whether the study methodology is applied uniformly to the various components of the analysis.

The scale (spatial and temporal) in which some data are collected can strongly affect the results. Ecological data can deal with organisms, and how they are affected by their environment, but when scaled up to populations (group of individuals of the same species), data will reflect the presence or absence of a particular species, along with their abundance and trends in population numbers. When dealing with communities (several populations that coexist in space and time), common measures that describe their composition (e.g. species identity, relative abundance or cover of these species, similarities and dissimilarities between communities) and structure (species richness, species diversity and its indexes) will be assessed in response to abiotic factors, interactions among species, and the level of disturbance as a result of random environmental effects (Townsend et al. 2010). Regardless of the metric selected, data needs to be representative of the spatial scale the population or community of interest inhabits.

The temporal scale of data collection also needs to be considered. For instance, species that are seasonally absent (e.g. migratory species), or not easily detectable due to their cryptic nature during specific stages of their life cycle could skew the data if the temporal scale for data collection is not carefully considered. These same factors can also lead to an underestimation of species abundance (see section 8.3.1). In contrast, sampling in periods where species are at their peak activity and detectability (e.g. mating season), may overestimate populations and produce misleading results.

Because data vary in terms of spatial and temporal scales comparisons between ecosystems need to be approached with caution or in some instances avoided. There are some general patterns in biodiversity

that showcase the importance of dealing correctly with scales and data sources. Biological diversity, for instance, increases with the area sampled, decreases from the equator towards the poles, and is generally higher in hot and humid regions. Indexes that describe communities frequently increase with estimated total abundance of individuals as a result of greater turnover of compositional species of local communities that contribute to habitat heterogeneity and species aggregation (Storch et al. 2007).

Data sources are important considerations. Data may come from different sources as they have usually been collected to answer different questions. The main data types are a) observations, b) field experiments, c) laboratory experiments, and d) (mathematical and statistical) models. The optimal source depends on the nature of the questions that are being addressed. For example, a temporal census of a specific population of *Puma concolor* may be able identify fluctuations in the population of this cat (observational evidence). However, these data alone are insufficient to infer what causes the fluctuation in the population and other data must be collected to address this question. Alternatively, controlled laboratory experiments are often more adept at addressing mechanisms that influence biodiversity, but extrapolation of these observations to field conditions needs to be undertaken with caution. Laboratory base models are incapable of replicating the complexity of natural ecosystems. Finally, mathematical models can be used to inform influences on the biodiversity of a simulated species (e.g. automats, etc.) as when combined with time series modelling and other techniques can prove useful for predicting species distributions and population dynamics.

Use of specific terms in different contexts can also dramatically alter the interpretation of biodiversity assessments. For example, the term “forest” can refer to a natural forest community of species that interact in a specific space and time, but this same term has also been used to describe forest plantations that are used to generate products such as palm oil. This ambiguity can have a huge impact on inventory data and can result in overestimation of biodiversity given that monoculture commercial forest plantations host lower species richness and less complex communities (Hanzelka et al. 2016; Peralta et al. 2018).

8.3.5 Reproducibility

Reproducibility: qualitative assessment of the extent to which information about the methodology and data values would allow an independent practitioner to reproduce the results reported in the study.

Having said the above, reproducibility is an essential goal to generate adequate comparisons and the use of data. Scientific rigor is based to a large part on the detailed description of methods that lead to specific results. Reference to scientific sources of data where methods are detailed enough to be reproduced with sources that are compatible and comparable is a key part of this process. Metadata should always be assessed for completeness and accuracy. At the very least, metadata should outline when (seasonality, periodicity), where (spatial scale and representativeness of the study object), and how the data was collected (source type and quality), with consideration of the relevance of the data to the scope, breadth and depth of the inventory that is being populated.

8.3.6 Uncertainty

All science has uncertainty.

Data uncertainty assessment methods: Because it is virtually impossible to define all species within a given spatial area, data are generated from a sample that is hopefully representative of the ecosystem of interest. Ecological data relies on estimates (indexes, mean, median, standard errors, etc.) and these estimates cannot be derived if criteria for data quality as mentioned above are not met.

Collected data needs to meet at least three criteria so as to increase accuracy and precision and reduce uncertainty (Townsend et al. 2010): (a) the estimate should be accurate and unbiased meaning that it is neither systematically too high nor too low as a result of some flaws in the previous steps that lead to the estimate; (b) the estimate should have narrow confidence limits being as precise as possible (significant differences are usually achieved when data is more consistent and where the number of samples is high); and (c) the time, money and human effort invested in the program (e.g. inventory) should be used as effectively and efficiently as possible (make choices based on limitations and the specific questions). To reduce uncertainty in field studies, representativeness of the samples needs to be maximized and biases avoided. One way to do that is by using stratified random sampling where sampling sites are divided into equal parcels (remember species richness increases with size), and then a number of random samples are taken from each parcel, in this way the coverage of the field is greater (includes more variability) and biases are minimized.

Although uncertainty needs to be minimized, its existence also needs to be acknowledged and discussed, especially in decision making processes that involve multiple stakeholders.

8.4 Existing data sources

Key guidelines

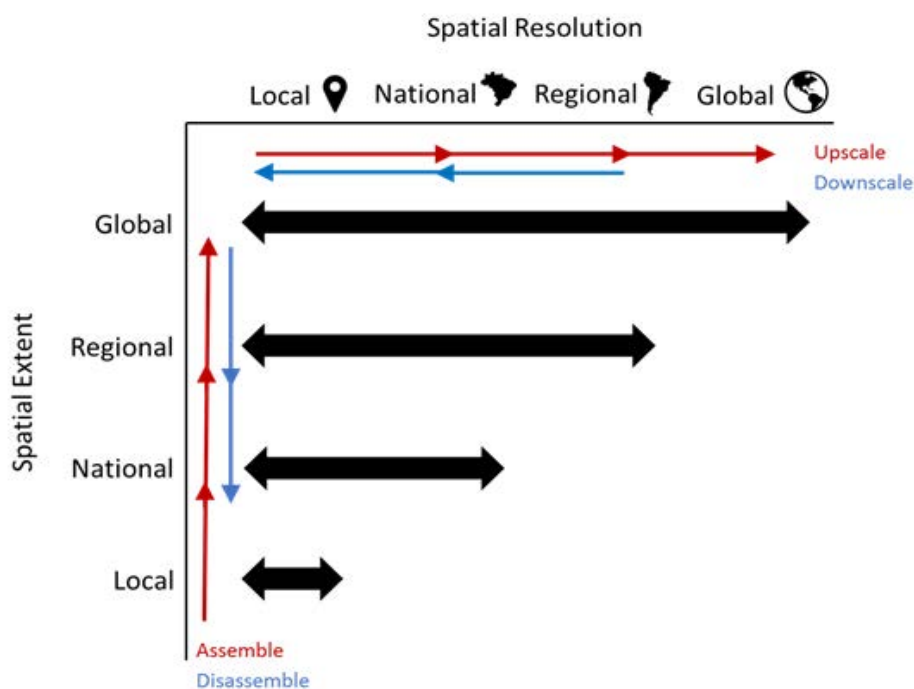
- This section provides a number of sources of global and regional data; other sources can also be used if sufficient information is provided to assess their representativeness and quality (refer to previous sections).
- Key aspects of global and regional datasets are their spatial/temporal extent and resolution; there are frequently trade-offs among these dimensions which should be considered and justified when selecting data matching the assessment goals.
- With local data, accessibility is an important issue and engagement of data owners as stakeholders in study design, including data handling provisions, is likely to aid data access.

Overall, biodiversity information in the world is fragmented, scattered and often difficult to access, especially when only available in non-indexed literature or in non-digital formats, such as dissertations, monographs and reports. Even when such data are published, there are limited opportunities to use it to improve public policies as it is often temporally dependent. The relevance of datasets to assess biodiversity depends on the scale of assessment used to measure the impacts of livestock production (Table 3).

1 Table 3. Different sources of data depending on scale of assessment.

Spatial scale	Impacts	Sources of data / methods
Global Level	Impacts of global issues such as climate change and issues related to aggregate impacts of human resource use on the planet.	<ul style="list-style-type: none"> - Global and regional databases and models - Peer-Reviewed Articles and Technical Reports - Global and regional maps - Remote sensing-derived information
Regional level (agro climatic microregions)	Impacts of regional issues such as deforestation and desertification.	<ul style="list-style-type: none"> - Databases / specific capture models - Peer-Reviewed Articles and Technical Reports - Intermediate supply chain - Remote sensing-derived information
Landscape, farm and field level	Impacts such as habitat fragmentation and loss of locally endemic species.	<ul style="list-style-type: none"> - Direct data (primary) - Use of detailed calibrated and validated model (if direct measurements are not possible) - Interviews - Remote sensing-derived information

2



3

4 Figure 6. The spatial scale of data sources is defined by their extent and resolution. A change in the scale of analysis requires a
5 change in either the resolution or the extent, or both (Farina 2006). Changes in resolution require upscaling or downscaling

approaches, while changes in extent may be achieved by assembling new data to increase the extent of the survey area, or disassembling data to focus on a specific area.

Data management plans that consider data publication, long-term curation and the generation of meta-data are seen as increasingly important steps in research projects, with the generation of such datasets being an important output of research projects. The internet has become an essential platform for data publication and sharing, requiring the development of computational analysis tools and big data approaches that make use of the increasing amount of available data.

Limitations in access to primary data and data-holder's reluctance to share information remain a critical barrier to global and cross-scale biodiversity monitoring (Han et al. 2014; Geijzendorffer et al. 2016). Publication of biodiversity data is critical for a timely assessment of biodiversity state and change and should be encouraged (Costello et al. 2013). Required actions include the implementation of publishing mechanisms that reward data providers, ensure data quality standards and the sustainability of public databases (Costello et al. 2013; Costello & Wieczorek 2014).

8.4.1 Global and regional sources

Global and regional datasets are produced and made available by international and regional organizations, national agencies with an international scope, research institutes, and academic research groups and networks (IPBES 2015). Global and regional sources of data on wild species and biodiversity conservation are often informed by field observations, models and national reports. Remote sensing sources are particularly suited to deliver input data to indicators and models of habitat condition and of some ecosystem services, especially some regulating services (e.g. carbon storage and sequestration). However, while there are several major land cover-land use mapping initiatives (Table 4), their mapping products are better able to inform aspects of vegetation and habitat structure and phenology metrics that are essential for assessing livestock impacts (Appendix 7 describes global and regional data sources focusing on a wider diversity of taxa). Thus, the use of these products for impact assessments on a global scale requires the support of information on environmental pressures (e.g., stocking rates), data which are often only available at the regional or local scale. National statistics may constitute a useful source of data for pressure indicators, but also for provisioning ecosystem services (Balvanera et al. 2016). Data can be available in their primary form (i.e., raw measures or observations), as in the case of species occurrence points or boundaries of protected areas, or as secondary data, after data processing and transformation, such as averaging, interpolation, or modelling to cover data gaps, as in the case of remotely sensed vegetation indexes.

Moreover, global and regional datasets can assemble data from a single source (e.g., land cover from Landsat imagery) or from different sources (e.g., species occurrences assembled from atlases, scientific papers, museum collections, or national statistics provided by different countries). The use of multiple sources is often needed to build regional and global databases and to increase the spatial, temporal and thematic coverage of data. However, the quality of data assembled from multiple sources may be affected by differences in monitoring methods (e.g., effort and data collection design), affecting precision, and reporting accuracy. For instance, national statistics on forest cover change can be affected by countries' monitoring capacity and even by their concept of forest (Rudel et al. 2005; Chazdon et al. 2016), which

ultimately affects data comparability. Similarly, the accuracy of secondary data will depend on the estimation methods and models used.

Because global and regional data sources tend to be affected by some level of uncertainty (due to loss of accuracy or precision), the selection and use of data sources should be preceded by an assessment of data quality based on the existing information on the underlying sources and methodologies used in data production, and on the existence of metadata that follows accepted standards (IPBES 2015). The lack of sufficient information that enables the assessment of data quality parameters, such as representativeness, accuracy, error, and comparability, affects the robustness of findings and should be acknowledged when presenting results and using that information to support policy.

Key aspects of global and regional datasets are their spatial and temporal extent (the size of the area or the time range in which the data is distributed), spatial and temporal resolution (the minimum spatial or temporal unit used to measure the variable of interest), and spatial and temporal coverage (the proportion of the spatial extent or time range for which information - primary, estimated or modelled - exists (i.e. data completeness) (Table 4, Figure 6). There are frequently trade-offs among these three dimensions. For instance, datasets may have full spatial coverage but coarse resolution (e.g. national statistics at FAOSTAT database; Table 4) or incomplete spatial coverage but deliver data collected at the local level (e.g. PREDICTS database; Table 4). The exceptions are remotely sensed data that can have a global extent, high spatial and temporal resolution and, virtually, full spatial completeness. Data with high spatial resolution (i.e. small spatial unit, such as local data) are better suited for cross-scale or cross-regional assessments; as these data can be used in smaller or larger spatial extents by directly assembling or disassembling data (e.g., Potter et al. 2010; Brooks et al. 2016). Nevertheless, data quality may be affected by this process, and should always be reassessed, as spatial coverage and other data attributes may not be homogenous among regions or countries. In other cases, data produced at lower resolutions may need to be downscaled to be used across smaller spatial extents or in cross-regional or cross-national comparisons (e.g., Araujo et al. 2005; Sánchez-Ruiz et al. 2014; Hoskins et al. 2016). Likewise, data may need to be up scaled to a lower resolution to reduce data complexity or to be combined with data at coarser scales (e.g. Dalgaard et al. 2011, Marcer et al. 2012). In the case of categorical data, such as land cover, changes in spatial resolution may be associated with changes in thematic resolution (e.g. resolution of land cover classes).

More generally, the use of different data products in assessments and modelling approaches requires data to share the same spatial extent and resolution. This may be achieved by assembling or disassembling data to adjust to the required spatial extent, and through upscaling or downscaling methods to adjust to the desired spatial resolution (Figure 6).

Table 4. Examples of global and regional data sources.

Data source / product ¹	Type of data	Spatial extent	Spatial resolution	Spatial coverage	Time range	Temporal resolution
Global Biodiversity	Species occurrences	Global	Local	High to low (depending on	Historical data to present	High to low (daily to

Information Facility (GBIF)				region and taxa)		scattered records)
PREDICTS	Species abundance (in relation to land uses)	Global	Local	> 26 000 sites (>75 countries)	1997-present	Depends on studies
Global Invasive Species Database	Invasive species occurrences	Global	National	Depends on data available (from different sources)	Started in 2000	Data continuously added
European Alien Species Information Network	Invasive species occurrences	Europe	10 km grid-cell	Depends on data available (from different sources)	Started in 2012	Data continuously added
IUCN Red List of Species	Species conservation status	Global	Summaries by country; RL status assessed globally	Depends on data available per species	(1964)-1986-present	At least twice a year
IUCN Red List of Ecosystems	Ecosystem conservation status	Global	Summaries by country	Depends on countries having done the evaluation	2013-present	Depends on studies
Plant trait database (TRY)	Plant functional traits	Global	Local (field studies)	Depends on data available per species	NA	NA
Global Forest Change	Forest loss and change	Global	30 m pixel	Full extent	2000-2014	Annual
Landsat	NDVI	Global	30 m pixel	Full extent	1982-present	16 days resolution
GlobCover	Land cover	Global	300 m pixel	Full extent	NA	12/2004 – 06/2006; 01 – 12/2009
Corine Land Cover	Land cover	Europe	25 ha (MMU)	Full extent	1990 - 2012	1990, 2000, 2006, 2012
Prototype LC map Africa	Land cover	Africa	20 m pixel	Full extent	NA	12/2015 – 12/2016

World Database Protected Areas	Protected Areas	Global	Local (Protected Area)	Full extent	NA	Monthly
FAOSTAT	Grassland emissions	Global	National	Depends on reporting country	1990 – 2014 (present)	Annual
FAOSTAT	Pesticides use	Global	National	Depends on reporting country	1990 – 2014 (present)	Annual
EUROSTAT	Irrigated area	Europe	NUTS 2 (sub-national)	98 % of the UAA and the livestock of each country	2003-2013 (EU 27)	Census every 10 years

1 GBIF: <https://www.gbif.org/>; PREDICTS: <http://www.predicts.org.uk/>; GISD: <http://www.iucngisd.org/gisd/>; EASIN: <https://easin.jrc.ec.europa.eu/>; IUCN RL: <http://www.iucnredlist.org/>; TRY: <https://www.try-db.org/>; GFC: <https://earthenginepartners.appspot.com/science-2013-global-forest>; Landsat: <http://landsat.usgs.gov/>; GlobCover: http://due.esrin.esa.int/page_globcover.php; CLC: <https://land.copernicus.eu/pan-european/corine-land-cover>; Prototype LC map Africa: <http://2016africallandcover20m.esrin.esa.int/>; WDPA: <https://www.protectedplanet.net/>; FAOSTAT: <http://www.fao.org/faostat/>; EUROSTAT: <http://ec.europa.eu/eurostat>

8.4.2 Local sources

Biodiversity can be measured at different geographical scales, which is important in conservation planning. On a local scale, it may correspond to the number of species found in a relatively small area of homogeneous ecosystem, which can be in the form of either a farm or landscape. This kind of diversity is very sensitive to how habitats are delimited (e.g. nearest neighbor) and how intensely a community is sampled.

Local data sources include farm-scale data generated by, or for, individual producers for a range of purposes, including producer benchmarking, landscape assessments, processor reporting requirements and regulatory compliance (e.g. local government). As the scale of geographic interest increases from field or farm-scale to landscape-scale, additional data will be required beyond aggregation of farm-scale data. For example, within a defined area (e.g. a freshwater catchment), local data covering other land uses (e.g. urban, horticulture, production forestry, conservation estate) will be required to provide a representative assessment at a landscape scale.

Livestock production systems utilize natural resources (common pool resources), so there is a reasonable expectation that the impacts of production systems are monitored and practices continually improved to minimize impacts. However, until recently producer-level assessments have tended to focus on issues such as water (ref LEAP guidelines) and nutrient use (ref LEAP guidelines). Farm-level data on biodiversity have received far less attention (hence the need for these guidelines). With an increasing focus on biodiversity and ecosystem services as ‘integrators’ of a diverse array of human pressures, the availability of local, producer-level data is likely to increase significantly.

1 One of the fundamental issues with producer-level data is accessibility. Private individuals or businesses
2 can be very reluctant to share their data, particularly if the end use of the data is poorly-defined or it
3 results in other groups gaining access. For many producers and processors, farm-level data is commercially
4 sensitive. Engagement of data owners as stakeholders in study design, including data handling provisions,
5 is likely to aid data access. If data is available only for a small number of farms, then issues of bias,
6 especially relating to a positive skew of data (i.e. the best farmers are those most willing to share data)
7 need to be considered.

8 Where data from individual producers may not be available, then remote sensing data can be used to
9 generate local spatial and temporal datasets. However, it is recommended that assessments relying on
10 remote sensing of biodiversity data also incorporate on-site validation (ground-truthing) to ensure the
11 remotely-sensed information is providing appropriate data. For example, if an assessment requires the
12 mapping of habitats of high conservation value, then remote sensing techniques may be a reliable and
13 cost-effective method of data collection, but only if there is confidence that the technique can reliably
14 distinguish the high value species from other similar species.

15 Government agencies, including local government, conservation and resource management organizations
16 are often an important repository for primary and secondary biodiversity data. However the availability
17 of such data may vary extensively between countries and regions. Much of this data resides in the public
18 domain and has robust data handling, quality control and access systems. Monitoring for regulatory
19 compliance also provides access to farm-level data that might otherwise be considered commercially
20 sensitive and of limited accessibility. One disadvantage of regulatory compliance data is that it can be
21 skewed towards representing the poor performers.

22 Performance benchmarking of individual producers is becoming an important generator of local data. For
23 many pressure indicators (e.g. nutrient inputs, GHG emissions, water use) individual performance can be
24 readily compared against standards or peer group performance norms. In contrast, biodiversity
25 assessment is highly context-dependent and the point of reference against which to measure the
26 biodiversity state at local scales is more problematic. An 'undisturbed' reference condition may not be an
27 appropriate comparison for biodiversity condition, as livestock systems often involve fundamental shifts
28 in land cover (e.g. vegetation) and the opportunity to avoid or remedy biodiversity impacts associated
29 with this change in land cover are often limited. An alternative approach is to identify the 'best available'
30 exemplars of a livestock production system at a relevant local scale as a more realistic point of comparison.
31 Identifying 'best practice' associated with these exemplars then provides a more complete Pressure-State-
32 Response model, as producers gain clarity on where they sit relative to realistic expectations and they
33 have a greater understanding of what changes are required to achieve it.

34

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Appendix

1 Links between the different LEAP guidelines document

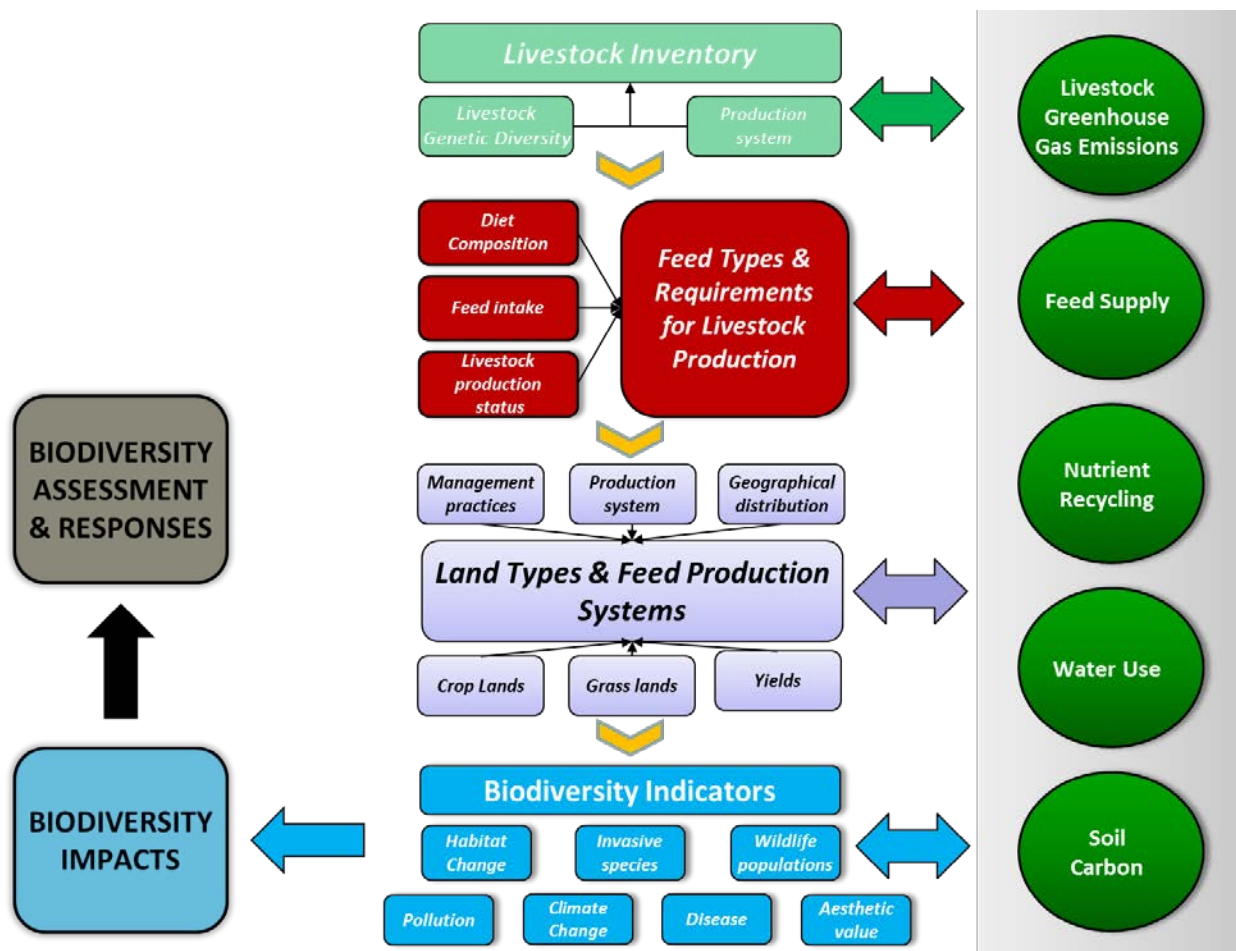


Figure A2. A conceptual model to assess biodiversity responses in livestock production supply chains. The model relies heavily on methodologies already used in existing LEAP guidance documents. Firstly, an inventory of the livestock chain of interest is undertaken. Corresponding feed requirements to maintain the livestock population are then estimated. Feed requirements are then extrapolated to define the land use needs that will satisfy feed demand for the livestock population. The impact of land use requirements for feed production on biodiversity are then defined through the selection of relevant biodiversity indicators. These indicators are used to assess potential impacts on biodiversity and to assess associated responses.

Livestock production systems are complex, with negative or positive impacts on biodiversity, with consequential influences on a wide range of ecosystem services. LEAP has developed a number of guidance documents that outline approaches to characterize the environmental performance and greenhouse gas emissions from pig, poultry, small and large ruminant and animal feed supply chains. Additional guidance documents on nutrient cycling, water use assessment and soil carbon stocks in livestock production chains are also being drafted and reviewed (<http://www.fao.org/partnerships/leap/overview/the-partnership/en/>). Combined, the information in these documents provides a valuable foundation to the assessment of the impacts of livestock production

on biodiversity (Figure 1). Several of these documents share a common methodological approach in the environmental assessment of livestock supply chains. Firstly, an inventory of livestock is undertaken to characterize the livestock (species, numbers) and feed (type, areas, quantities). The production system is further characterized in terms of its intensive or extensive nature and scale (i.e., local, regional, global) that it occupies. At local or regional scales, it may be possible to gather detailed information on the production status of livestock (i.e., growing, mature, gestating, lactating), factors that all influence the level of feed intake. Defining the types and amount of feed that satisfies the productivity of the livestock population provides the basis to estimate the amount of crop or pasture needed to produce the required feed. Estimation of the amount of land required to produce the feed requires estimates of crop and pasture yields within the region that the crops are produced. Some feeds may be produced regionally, while others may be imported from distant global locations. Once land use requirements have been defined, the impacts of this use on biodiversity can be assessed through the selection of multiple relevant biodiversity indicators. Selection of these indicators should be undertaken with an appreciation for the geographical location of where the feed was produced. Interpretation and integration of these indicators leads to assessment of the impact of the livestock production on biodiversity and associated responses. Data on livestock populations, feed types and crop / pasture land production systems also provides valuable information that is relevant to other ecosystem services including air, water and soil quality, nutrient cycling and climate.

2 The high nature value of extensive livestock grazing systems

Many extensive livestock grazing systems can still be considered of high biodiversity value because:

- They continue to utilise and maintain a high proportion of natural and/or semi-natural vegetation managed at relatively low levels of intensity. This may be largely by default in areas where climatic and topographic constraints limit the intensification of the vegetation management and grazing practices that can be applied. However, the outcome is a greater range of ecological niches over much of the area utilised by the livestock grazing system.
- The constraints imposed on the vegetation by climate and topography control not only the type but, just as importantly, the timing of the management that is applied to the vegetation. Hence, livestock grazing practices in particular are generally synchronised with the annual natural growth cycle of the vegetation and so are not imposed at a time when it would be detrimental to a wide range of the plant species involved.
- For most of the year, the nutritional value of much of the natural or semi-natural vegetation is generally low which places limits on the number of livestock or wild herbivores and hence the intensity and duration of grazing intervals in a given area. It also leads to a need for larger areas to be utilised by these animals. Hence, grazing pressure on any one area is generally either low or (in closely managed herd or flocks) only high for a very short period, which leads to a greater heterogeneity of vegetation structures.
- The habitats of many wildlife species are naturally unstable and it is common for populations to disappear from one area and for these or new ones to appear when a suitable niche becomes available elsewhere. Extensive livestock grazing systems, and associated practices and natural processes, are maintained at a scale and intensity which ensures a sufficient area of potentially suitable habitat is available within relatively close proximity to each other (i.e., in terms of the distance that the species can move) and thereby facilitates these cycles of colonisation and re-colonisation.
- By the same token, extensive livestock grazing systems are much more favourable than intensive systems, to a wider range of wildlife species because they are practised over a wider scale and therefore (a) the conditions required at any one time of year, particularly by more mobile species can be found at a wide variety of locations and (b) the different requirements by these species at different times of year are catered for, i.e., through changes in the mix of structures and habitats in any one area through the year.

Additionally, extensive livestock grazing systems contribute to maintain balanced water and carbon cycles, which in turn provide better conditions for the maintenance of a wide range of niches in the landscape, contributing also to aesthetic values and ecosystem services.

3 Methods to include impacts on ecosystem services in LCA

Table A3. Overview of selected studies that have included ecosystem service impacts in life cycle analysis

Impact indicators	Description	Methodology	Spatial scale of assessment/ Regionalisation	Indicator	Source
Biotic production potential	Represents soil fertility, i.e., the capacity of soils to produce biomass	LU inventory flows linked to biophysical indicators via midpoint CFs provided by LULCIA project for biotic production, groundwater recharge, erosion regulation, water purification and climate regulation soil potentials. Biophysical midpoint indicators converted to economic units based on economic valuation of ES reduction (product of economic conversion factor, exposure factor and adaptation capacity).	Global (between biomes or climatic regions)	Productivity loss (\$ ha ⁻¹ yr ⁻¹)	Cao et al. (2015)
		CFs describing expected SOC changes due to LU calculated as a function of SOM change, area and time. CFs based on IPCC SOC values per soil type, climatic condition and management option. Impact measured as C deficit or credit compared to reference system ((quasi-)natural land cover).	Global (between biomes or climate regions)	Soil carbon deficit (or credit) (kg SOC yr ⁻¹ m ⁻²)	Brandão and Milà i Canals (2013)
Water supply (consumption)	Irrigation (blue) water consumed by crop production	Spatially-explicit land change modelling based on logistic regression with climatic and soil suitability, followed by land use change translated to ES impacts using spatially-explicit InVEST models.	23,000, 86,000, 321,000 t HDPE production volumes*	Water consumption (m ³ water t HDPE ⁻¹)	Chaplin-Kramer et al. (2017)
	Quantity of water withdrawn for production processes	A herd-level, cradle-to-farm gate life cycle livestock feed requirements model, adapted and applied within ISO-compliant LCA to estimate the environmental burden of grass-fed beef vs. management-intensive grazing vs. confined dairy beef. LCIA conducted in openLCA software.	Northeast region of the US	Water depletion (m ³ water kg HCW ⁻¹)	Tichenor et al. (2017)
Water supply (freshwater recharge potential)	Capacity of soils to recharge groundwater			Urban water supply (\$ ha ⁻¹ yr ⁻¹)	Cao et al. (2015)
		Impacts on terrestrial green water flow and surface blue water production due to decreased runoff from LU = product of effective net green water flow and CF of each area under analysis. Net green water flow = difference between total green water flow (green ET) of actual crops and total green water flow of PNV (ET of PNV); CF is a function of actual and PNV ET.	Global (per climatic criteria) Case study: <i>Eucalyptus globulus</i> stands in Portugal	Terrestrial green water flow and surface blue water production (m ³ ha ⁻¹ yr ⁻¹)	Quintero et al. (2015)

Freshwater regulation potential		LANCA model used to compute soil ecological function impact indicators. The difference between the baseline reference state (PNV) and the outputs yielded a set of CFs for each biome and land use type.	Global, Holdridge life regions/zones, terrestrial biomes	Groundwater recharge (mm water yr ⁻¹)	Saad et al. (2013)
Water purification potential	Ecosystem's chemical, physical and mechanical capacity to filter water			Cation exchange capacity (cmol _c kg _{soil} ⁻¹)	Saad et al. (2013)
				Water purification process costs (\$ ha ⁻¹ yr ⁻¹)	Cao et al. (2015)
	Soil's capacity to mechanically filter water			Rate of water passing (cm water day ⁻¹)	Saad et al. (2013)
	Capacity of willow to purify water via nutrient buffering	Estimation of environmental loading changes using attributional LCA (ALCA) of heat system burdens, and consequential LCA (CLCA) of environmental loading changes using an adapted LCAD tool.	Landscape scale: Skåne (Sweden)	P export (g PO ₄ eq. MJ _{th} ⁻¹ ; kg PO ₄ eq. ha ⁻¹ yr ⁻¹)	Styles et al. (2016)
Erosion regulation potential	Capacity of terrestrial ecosystem to withstand soil loss through erosion			Tons of soil eroded (ton soil ha ⁻¹ yr ⁻¹)	Saad et al. (2013)
				Sediment export (m ³ sediment t HDPE ⁻¹)	Chaplin-Kramer et al. (2017)
				Erosion mitigation costs (\$ ha ⁻¹ yr ⁻¹)	Cao et al. (2015)
Climate regulation potential	Capacity of ecosystem (soils) to uptake carbon from the air			Social cost of C (\$ ha ⁻¹ yr ⁻¹)	Cao et al. (2015)
		Proxy-based approach that assigns terrestrial C stock and C stock change values due to land use change to different land use types, and compares these with the reference condition (PNV).	Global (between biomes)	Vegetation/soil to atmosphere C flows (t C m ⁻² yr ⁻¹)	Muller-Wenk and Brandão (2010)
				Soil C seq. (g CO ₂ eq. MJ _{th} ⁻¹)	Styles et al. (2016)

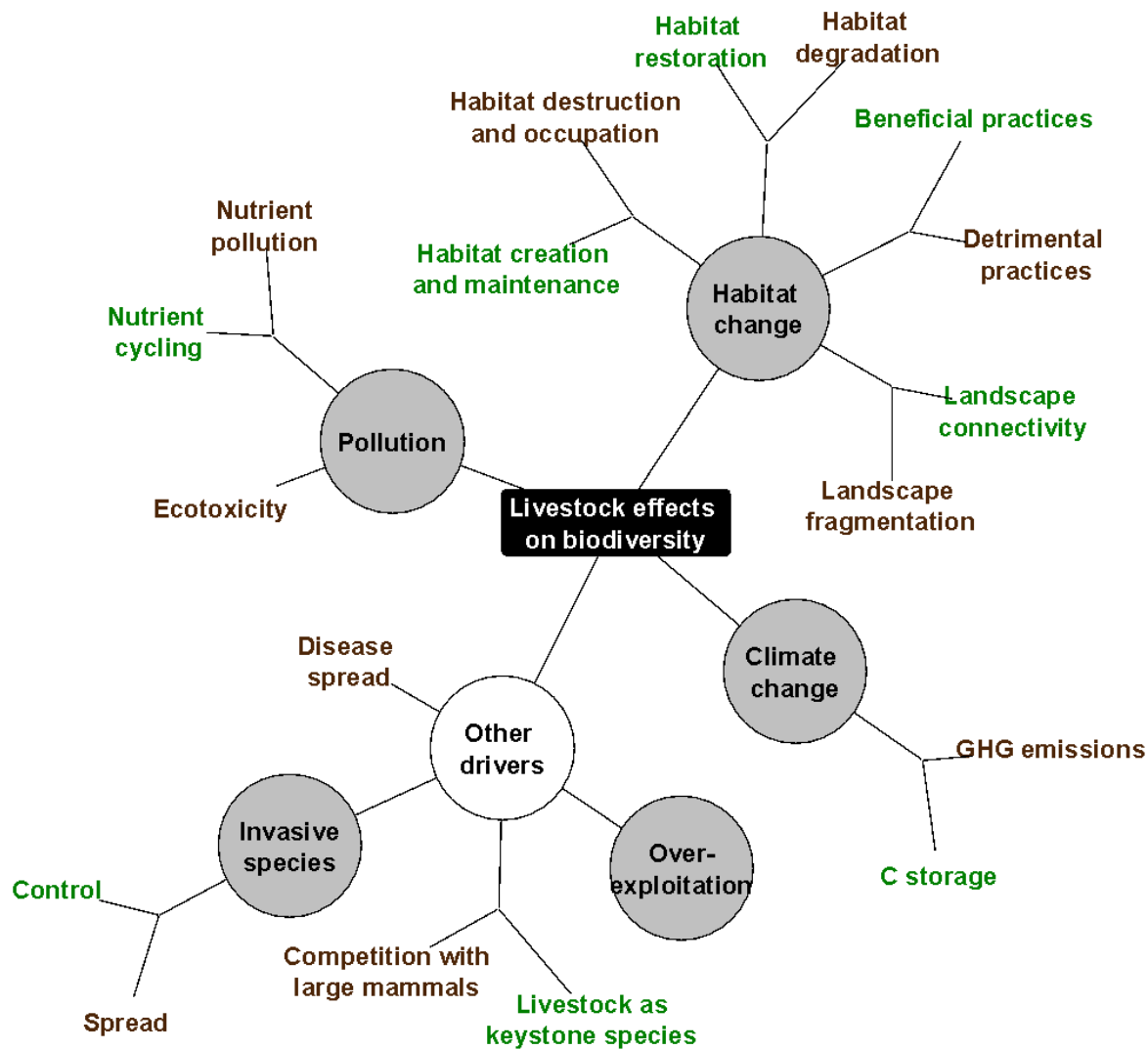
	Carbon losses due to land cover change			CO ₂ emissions (t CO ₂ eq. t HDPE ⁻¹)	Chaplin-Kramer et al. (2017)
	Ecosystem's capacity to limit or regulate emissions of GHGs to the atmosphere			Net GHG emissions (Mg CO ₂ eq. ha ⁻¹)	Styles et al. (2016)
				GHG emissions (kg CO ₂ eq. kg HCW ⁻¹)	Tichenor et al. (2017)
		LCA conducted using FAO LCA guidelines for small ruminants.	Ireland: cradle-to farm gate case study sheep farms	GHG emissions (kg CO ₂ eq. kg LW ⁻¹)	O'Brien et al. (2016)
Nutrient regulation potential	Ecosystem's chemical, physical and mechanical capacity to adsorb nutrients and prevent N and/or P loss to the environment			Nutrient export (t N t HDPE ⁻¹)	Chaplin-Kramer et al. (2017)
				Nutrient export (kg N kg HCW ⁻¹)	Tichenor et al. (2017)
				Potential P loss to waterways (kg PO ₄ eq. kg LW ⁻¹)	O'Brien et al. (2016)

CF: 1 characterization factor; GHG: greenhouse gas; HCW: hot carcass weight; HDPE: high-density polyethylene; LANCA: LAND use indicator value CA l culation; LCAD tool: LCA model for net environmental and
2 economic effects of farm scale anaerobic digestion and bioenergy scenarios; LCIA: life cycle impact assessment; LU: land use; LULCIA: Land Use Life Cycle Impact Assessment; LW: live weight; MJ_{th} =
3 megajoule of useful heat output; PNV: potential natural vegetation; SOM/SOC: soil organic matter/carbon; Cells in grey: information already provided.

* The LCA of bio-based HDPE production is provided as an example as it includes the agricultural stage of production, i.e., the production of maize and sugarcane feedstock.

4 Categories of pressures and benefits

The figure below provides an overview of the categories of influences that livestock have on biodiversity. The five main drivers of biodiversity loss recognized by the Millennium Ecosystem Assessment (2005) appear in grey circles. However, for most of these drivers, livestock can either put pressure (brown) or provide benefits (green) to biodiversity.



1 5 Detailed description of recommended indicators

2 Table A4. Recommended list of Pressure, State and Response indicators and detailed description.

Category	
<i>Indicator</i> (P: pressure or benefit, S: state, R: response; QI: qualitative, Qt: quantitative)	<i>Description</i>
Procedural checks Elements in this category aim to demonstrate that key requirements of the assessment process are met.	
A scoping analysis was conducted	<p>The scoping analysis should (i) set the context by identifying biodiversity features of concern, legal and designation frameworks, due diligence etc. and (ii) conduct a pre-assessment for the relevant scale/territory, identifying potential hotspots of impacts including downstream and off-farm.</p> <p>More details on the scoping analysis are provided in Section 6.1.2</p>
Regulatory constraints are met	<p>How regulatory constraints related to biodiversity are met should be discussed, including those on protected areas and species, banned or regulated biocides (fungicide, herbicide, pesticide) and other ecotoxic agrochemicals (hormones, antibiotics).</p> <p>The extrinsic use value defined by local stakeholder should also be considered.</p>
Progress is monitored	<p>Indicators should be monitored over time or against a clearly defined reference (more details on the reference are provided in Section 8.2).</p>
Stakeholders are engaged	<p>Iterative stakeholder engagement should be documented at all steps of the assessment (in design, scoping, hotspot, selection of goals and indicators, data assessment and communication). A stakeholder or perspective analysis could also be performed.</p>
Data quality	<p>Data quality should be ensured, refer to specific guidelines in Section 8.3.</p>
Habitat protection	
Wildlife habitats under the farm influence are inventoried (mapped) and protected (R, QI)	<p>Across the agri-food industry, many sustainability assessments for formal accreditation require that farmland wildlife habitats are inventoried and mapped. Because accreditations are usually for individual farms, this is often achieved by indicating the spatial location of farmland habitats on a map. The approach could, however, also be used for larger spatial scales to indicate the spatial location of wildlife habitats across larger spatial scales that encompass multiple farms (and involve remote sensing).</p> <p>The inventory should identify and include (semi-)natural (e.g. grassland, grassy strips, flowering plants, isolated trees, hedgerows, woodland patches, shrubs, wetlands and waterways – ideally differentiating between native and exotic woody/shrub/grass species), protected and priority habitats (both terrestrial and aquatic).</p> <p>The inventory should cover habitats within the farm but also in the surrounding area (e.g. landscape, watershed) when potentially impacted by</p>

	<p>farming practices (e.g. pesticides drift, nutrient runoff, farm in a corridor between natural habitats).The territorial scope may be different if it is justified for the existence of delimitation of protected areas or other administrative units of management or conservation.</p> <p>The inventory should also include habitats (potential distribution) of species of high conservation interest e.g. protected at local, national or higher level, endangered or threatened (CR, EN, VU and NR IUCN categories), migratory wildlife. The inventory includes the presence of priority species for conservation at the national or regional level if it exists. If not, it must include species considered by IUCN at the international level as critically endangered (CR), endangered (EN), vulnerable (VU) or near threatened(NT).</p> <p>An inventory will usually be the end-product of several other stages that may include the following:</p> <ul style="list-style-type: none"> - definition of a clear spatial boundary for the area of interest - an indication of the different areas that are occupied by wildlife habitats - an indication of the relative conservation priority of the habitats and species in the area of interest
Percentage (or area) of semi-natural habitats in the landscape (P, Qt)	Of particular importance in areas that have a mosaic of wildlife habitats and areas of intensively managed agricultural land use. Changes over time in this indicator provide important information on large-scale trends in habitat quantity in the landscape or boundary area.
Grassland restoration (R, Qt)	Degraded grasslands can be restored through improved management of grazing (e.g., adapting the timing and intensity of grazing to biomass availability, rotational grazing, temporary grazing exclusion) and grassland (e.g., light fertilization, liming).
Habitat change	
Soil erosion and soil erosion risk are mapped and a management plan is implemented (R, Ql/Qt)	<p>Factors influencing soil erosion risk include soil type, slope, burning, precipitation intensity, wind, bare soil cover, vegetation type.</p> <p>Local NGOs or other stakeholders should be involved in selecting indicators of soil erosion which can include compaction, low organic matter content, bare soil, encroachment or change in plant species composition.</p>
Area/proportion of degraded soil, including bare soil and areas with bush encroachment (P, Qt)	This indicator can be computed from the map of soil erosion and soil erosion risk. Soil degradation from grazing is a particularly strong threat in heavily stocked grazing systems. Very dry and very humid systems tend to be more sensitive to land degradation, i.e. it can occur at lower livestock densities. Processes and indicators of degradation will depend on the ecosystem. Bare soil can be a simple and effective indicator of degradation especially in dryland rangelands. Bush encroachment is also an indicator of land degradation.
Livestock density (P, Qt)	High livestock densities cause land degradation in terms of declining range productivity, soil degradation, and woody invasion of grasslands. The degrading effect of livestock density varies widely, depending on climate, soils, management practices including grazing regimes, and rangeland fragmentation.

	<p>Dry rangelands (e.g., < 800 mm yr⁻¹) are less vulnerable to degradation from over-grazing, although all rangelands face a potential threat from over-grazing, or more precisely, “under-resting” of grazing lands.</p> <p>In regions with higher rainfall (e.g., > 800 mm yr⁻¹), this indicator should compare livestock density to the carrying capacity of rangeland to indicate if there is over- or under-stocking. It can be estimated with local stakeholders and experts, or quantified using measures of vegetation productivity (e.g. NDVI, DMP), energy content, compared with energy requirements from livestock.</p>
Area or rate of habitat conversion (P, Qt)	<p>Three main types of habitat conversion should be considered:</p> <ul style="list-style-type: none"> - Deforestation, i.e. conversion from forest to grassland or feed crops - Permanent grasslands that are tilled, i.e. converted to temporary grasslands or to cropland - Abandoned grasslands which slowly convert to shrublands and forest through ecological succession
Wildlife conservation	
Priority actions promoting species with high conservation and functional value are listed and implemented (R, QI)	<p>Priority actions to promote and sustain species with high conservation value should be identified, listed and implemented. A preliminary review of legal frameworks and other initiatives (from the private sector, NGOs) can be conducted to identify good practices. Local NGOs or other stakeholders should then be involved in the identification and selection of both high conservation values and priority actions.</p> <p>High conservation value can derive from threat status, patrimonial aspects, national and international designations, but also the functional role of the species, and their extrinsic value from the perspective of local stakeholders.</p> <p>Priority actions may include the protection of habitats and other key features (e.g. breeding sites, food resources) for species with high conservation value, or the adoption of offset areas. Priority actions may be of the normative or incentive type. The efficiency of priority actions should be assessed.</p> <p>The area (or proportion) of protected habitat or feature can be calculated as a quantitative indicator.</p>
Abundance, presence/absence or distribution of species (with high conservation value) (S, Qt)	<p>The abundance and distribution of species (with high conservation value, including from the perspective of local stakeholders, and because of their functional role) should be monitored over time through ecological surveys, whether by simple checks or sophisticated protocols. Species with high conservation value may be defined locally or internationally, and should be addressed as a priority using this indicator. In some cases, local NGOs or other stakeholders may be able to provide information for the indicator calculation from existing monitoring programs. Species with a key ecological role or with added value as an indicator (e.g., keystone species, umbrella species, ecosystem engineers, species with trophic level) may also be considered. For species with high conservation values, this indicator may need to be combined with several associated pressure and response indicators to be developed with stakeholders.</p>

Species richness, diversity (of species or functional) (S, Qt)	Species richness corresponds to the number of species and is a relatively simple and widely used indicator. Species diversity is maximal when there is a high number of species and when the number of individuals is even across species. These indicators do not reflect possible differences of conservation value across the species (including negative value, e.g. invasive species) and the abundance of species with high conservation value should ideally be reported separately. Diversity indices can be calculated at the species level but also for functional groups (e.g., functional diversity, mean trophic index). Higher functional diversity at the species level is often linked to the provision of ecosystem services.
Invasive exotic species	
A management plan is in place for the prevention and control of exotic species (R, QI/Qt)	<p>The development of management plans to prevent or control invasive species (at the local level, property or establishment), allows the execution of strategic actions with a systemic view, and multi-year planning. The support of specialists in different technical areas can improve the qualitative performance of the plan. A first step is the mapping of invasive species in the area under management and the measurement of the area that they occupy, followed by an analysis of the factor causing invasion and the persistence of invasive species.</p> <p>Qualitative component: Existence or creation of an Invasion Management Plan (at the local level, property or establishment).</p> <p>Quantitative component: area under management plans for invasion control, baseline and historical progression</p>
Presence of exotic invasive species (P, QI/Qt)	A regional list of exotic invasive species should be established, with an assessment according to degrees of threat. The list may include naturalized exotic species for which negative impacts on native communities have not been document, but could happen in the future and under climate change scenarios (Koch et al. 2016). The aim is to identify the species that pose the greatest risk to local communities and populations, with an evaluation of the impact on the area they occupy, the potential risk and the degree of threat to objects of high conservation value. Foreign species can exert displacement and threaten native biodiversity. For any region of the world there are lists of the most dangerous invasive species, and the degree of threat to native species, ecosystems, and production systems is generally known. It is also important to appreciate that a species does not need to be exotic to impact biodiversity as is the case when native bush encroachment can contribute to a reduction in biodiversity as well as livestock productivity (addressed under the habitat change category and degradation indicator).
Distribution (abundance) of exotic invasive species (P, Qt)	The spatial distribution (abundance) of invasive species should be mapped (as a result of surveys, inventories, census) and include a description of the type of environments, or native communities within which the invasive species occur, and the level of disturbance they exert. The percentage of the reference areas where invasive species are present may be calculated.

	This indicator should be measured over time (e.g. progression of the area occupied by invasive species) to evaluate the actions of control and/or eradication of invasive species
Pollution & aquatic biodiversity	
A management plan is in place for the application of ecotoxic agrochemicals (R, QI)	Livestock production systems utilize a range of biocides (e.g. fungicides, herbicides and pesticides) and other potentially ecotoxic agrichemicals (e.g. animal health remedies and fertilisers) that can have direct and indirect effects on aquatic ecosystems. For example, herbicide use to maintain drainage performance in pastoral landscapes can reduce fish and macroinvertebrate diversity. Management of these chemicals, including application following manufacturer's guidelines, safe storage and measures to avoid application in sensitive habitats should be described in a farm management plan. Integrated pest management can be included in the management plan as a useful way to reduce utilization of ecotoxic biocides.
A nutrient management plan is in place to rationalize fertilizer application (R, QI)	Loss of nitrogen and phosphorus sourced from livestock production systems to freshwater ecosystems is inevitable. However, there is a wealth of robust science that has identified mitigation measures that can significantly reduce the risk of nutrient loss through leaching and runoff. A farm nutrient management plan should identify areas of risk for nutrient loss to waterways and identify and track implementation of actions to minimize these risks. This plan may also include adequate animal nutrition strategies to adjust nutrient intake to requirements and reduce losses.
Length/proportion of protected waterways (R, Qt)	<p>Waterways – as indicated in the inventory of wildlife habitats – can be protected through livestock exclusion (e.g. fencing), edges or buffer strips.</p> <p>Direct access of livestock to waterways has a significant and usually deleterious effect on aquatic biodiversity, particularly for larger animals such as cattle, deer and pigs. In addition to direct habitat damage, stock access can increase bank erosion and the deposition of fine sediment downstream. Generation of direct faecal contamination can also be an issue relating to disease spread and organic loading. Assessment of the length/proportion of protected waterway needs to explicitly define 'waterway' (e.g. minimum size; hydrologic permanency). If there are specific exclusions (e.g. ephemeral waterways that flow after heavy rain) these should be spelt out. The width of the riparian area should also be considered and sufficient to ensure waterway protection. Specify that this action is targeted at reduction of phosphorus and sediment in waterways. The extent of protection should also be described. For example, a temporary fence might be erected close to a stream during periodic grazing, but it will provide less protection than a permanent fence, and much less than a fenced waterway that has a well-managed riparian zone.</p>
Ecological indicators of eutrophication or water quality (S, Qt)	Eutrophication of freshwater and marine systems is one of the most serious and far-reaching environmental impacts that livestock production systems can have. Excess nitrogen and phosphorus leads to nuisance growths of aquatic plants and algae, causing fundamental shifts in aquatic ecosystems, loss of aquatic biodiversity, die-offs, and poor water quality. The LEAP

	<p>Nutrient guidelines (FAO 2018c) recommend a method to account for eutrophication.</p> <p>Aquatic communities integrate a range of anthropogenic stressors and shifts in community composition are largely predictable and repeatable. This has led to a wide range of aquatic taxa being used as biological indicators (e.g. fish, macroinvertebrate or autotroph assemblage indicators, functional indicators, habitat quality metrics). These indicators can include simple diversity-based indices through to predictive modelling linking stressor levels to expected communities. A number of simple indices lend themselves to producer or citizen-science monitoring. When biological indicators are being monitored it is important that these should be capable of revealing whether aquatic habitat quality is increasing, decreasing or meeting target levels. As a target, livestock systems should not reduce the health of aquatic ecosystems.</p> <p>This indicator is of particular importance in sensitive catchments, where there may be additional management and monitoring requirements.</p>
Off-farm feed	
An inventory of the off-farm feed being used is established (R, QI/Qt)	The inventory should include the composition and volume (weight) of the off-farm feed, and whether it is internationally traded or locally/nationally produced. When the information can be found, the inventory should also include the production origin of the imported feed.
Traceability systems for feedstuff is implemented (R, QI)	Off-farm feed production can be related to deforestation of tropical rainforest and other forests and woodlands of high biodiversity value, which represents a major impact on biodiversity. This indicator aims to track such impacts. In practice, it can be difficult to trace purchased feed to specific areas of origin, but good practice in livestock systems would generally include such traceability.
Share of imported feed – total, from areas that are certified/deforested/of high conservation value	<p>The share of imported feed related to the total amount of feed used should be computed.</p> <p>When used, the share of accredited feedstuffs produced in ways that mitigate or avoid land use and associated biodiversity impacts should be computed. The greater the reliance on such feed, the lower the expected impact on biodiversity. Accredited feedstuffs represent improved knowledge of the origin of imported feed and avoidance of recently deforested areas or removal of wildlife habitats.</p> <p>In addition, when the production origin of the imported feed is known, the share of imported feed coming from specific areas should also be computed, including:</p> <ul style="list-style-type: none"> - Recently deforested areas - Areas with high conservation value (e.g. Conservation International biodiversity hotspot, WWF ecoregions with outstanding biodiversity features, IUCN red list of ecosystems) <p>Deforestation of tropical rainforest and other forests and woodlands of high biodiversity value represents a major impact on biodiversity. This indicator aims to track such impacts. In practice, it can be very difficult to trace purchased feed to specific areas of origin.</p>

Landscape scale conservation	
Measure to promote connectivity identified and implemented (R, QI)	<p>Local NGOs or other stakeholders should be involved in the identification of measures to promote connectivity, which may include sufficient size and close distance between patches of (semi-)natural vegetation, or creation of corridors between natural areas. Implementing such measures most often requires coordination between different farms and stakeholders at the landscape scale.</p> <p>There are physical elements that prevent the mobility of organisms along natural corridors or habitats, and measures should be implemented to overcome them. This is especially relevant for freshwater organisms (fish, amphibians, etc.) that need the continuity of the watercourse to carry out their migrations upstream or downstream according to their life cycle. Therefore dams or deviations can be insurmountable barriers.</p>
Landscape heterogeneity (P, Qt)	<p>Farmlands are often mosaic landscapes with (semi-)natural habitats, various agricultural land uses and other activities. Such landscape heterogeneity tends to increase opportunities for diverse species to find resources and occupy different niches. Even small portions of natural habitats such as hedgerows can provide significant ecosystem services such as pollination and control of pests and erosion.</p> <p>Indicators reflecting the heterogeneity and structural complexity of the landscape include the number and relative areas of land uses, the (spatial) Shannon diversity index, edge length or perimeter/area ratio of natural patches.</p> <p>Certain production systems can achieve high levels of heterogeneity and improve biodiversity while being consistent with livestock production objectives. They include agroforestry/sylvopastoralisms (e.g., with hedgerows, shelterbelts, windbreaks, live hedges) and integrated crop-livestock production systems.</p>

- 1 **6 Extended list of indicators**
- 2 **6.1 Habitat protection**
- 3 **6.2 Habitat change**
- 4 - Permanent area of bare soil or under desertification process
- 5 - Area of bare soil between cropping seasons
- 6 - Livestock density
- 7 - Area of irrigated feed crops
- 8 - Feed crop yield
- 9 - Seeding of grasslands
- 10 - Shallow or no-tillage is used
- 11 **6.3 Wildlife conservation**
- 12 - Key functional or engineer species
- 13 - Use of fishmeal in feed rations
- 14 - Number of conflict with wildlife including number of large predator kills
- 15 - Overexploitation of wildlife species (e.g. hunting, fishing, collecting) on farm is prohibited
- 16 - Mowing (and grazing) is delayed until after the nesting season of ground-nesting birds in part of the
- 17 grassland area, particularly around wetlands
- 18 - Stubbles are left over winter
- 19 **6.4 Invasive species**
- 20 **6.5 Pollution & aquatic biodiversity**
- 21 **6.5.1 Pollution by nutrients**
- 22 - Quantity (kg) of N and/or P applied in grassland or feed crops
- 23 - Inland or coastal water in a state of eutrophication
- 24 - Presence of plant species characteristic of nutrient-rich (eutrophic) conditions
- 25 - Critical load exceedance for nitrogen (from nitrogen deposition) in the soil
- 26 - Emissions of gasses leading to nitrogen deposition and acidification
- 27 - Nitrogen balance or nutrient use efficiency (accounting for inputs and outputs) (see LEAP guidelines
- 28 on nutrients)
- 29 - Nutrients in transitional, coastal and marine water
- 30 - Animal diet is balanced to meet requirements and reduce nutrient excretion
- 31 - Manure management is optimized to minimize nutrient leaching and optimize nutrient recycling
- 32 - Crop and livestock productions are integrated to optimize nutrient recycling
- 33 - Manure is used to produce biogas (with leakage issues being controlled)
- 34 **6.5.2 Pollution by ecotoxic substances**
- 35 - Application of specific pesticides molecules with high ecotoxicity
- 36 - Amount of toxic substance used, weighted by factors reflecting their toxicity (including half-life,
- 37 mobility in the environment)
- 38 - Pesticide application (number of application or quantity of active ingredient) in feed crops

- 1 - Presence of faecal anthelmintic residues
- 2 - Use (and quantity) of toxic veterinary products: antibiotics, anthelmintics, hormones
- 3 - Water contamination by hormones
- 4 - Biological control is used
- 5 - Crop rotation is used to break weed and pest life cycles and avoid disease buildup
- 6 - Crop rotation is used to break weed and pest life cycles and avoid disease buildup
- 7 - Mechanical control is used when relevant
- 8 - No preventive spraying is used and only affected areas are sprayed
- 9 - Precision spraying is used and drift is minimized
- 10 - Products targeting specific species are used rather than generalist products
- 11 - Semi-natural habitats are created and maintained for natural pest predators
- 12 - Spatial intercropping is used to limit pest propagation
- 13 **6.6 Off-farm feed**
- 14 **6.7 Landscape connectivity**
- 15 - Area of patches of semi natural habitats
- 16 - Distance between patches of semi natural habitats
- 17 - A diversity of crops and crop varieties is grown
- 18 - Crop rotations are long
- 19 **6.8 Additional categories**
- 20 **6.8.1 Large scale indicators**
- 21 - Farmland Bird Index
- 22 - IUCN red list indices
- 23 - Living Planet Index (LPI)
- 24 - Mean Species Abundance (MSA)
- 25 **6.8.2 Ecosystem services**
- 26 - Production/yield of animal food products
- 27 - Other livestock products: hides, skins, fiber, manure and urine for fertilizer, manure and methane for energy
- 28 - Vegetation indices (remotely sensed): Dry Matter Productivity (DMP), Net Primary Productivity (NPP), Normalized Difference Vegetation Index
- 29 - Above-ground biomass
- 30 - Human Appropriation of Net Primary Production (HANPP)
- 31 - Crop production/yield
- 32 - Ground water, stream flow, water abstracted
- 33 - Forest area or biomass
- 34 - Soil erosion risk/erodibility or erosion protection
- 35 - Pollination potential
- 36 - Vegetation type
- 37 - Flood events

- 1 - Fire events
- 2 - Soil organic carbon depletion
- 3 - Nutrient flux
- 4 - Nutrient cycling (soil fertility, nutrients and organic matter distribution)
- 5 - Soil organic carbon storage
- 6 - Weed control
- 7 - Shrub control and fire regulation
- 8 - Presence of species/landscapes with aesthetic, cultural or religious importance
- 9 - Aesthetic value of livestock-maintained landscapes
- 10 - Livestock contribution of cultural heritage and identity
- 11 - Contribution of cultural heritage and identity
- 12 - Role in social events, relations, status
- 13 - Extent of protected areas or high nature value farmlands
- 14 - Recreation and tourism
- 15 - Loss of biodiversity habitats
- 16 - Erosion of livestock genetic resources (loss of breeds)
- 17 - Creation and maintenance of biodiversity habitats
- 18 - Maintenance of livestock genetic resources (breeds)
- 19

7 Regional and global data sources for specific taxonomic groups

Table A7. Examples of relevant data sources on biodiversity available on the web, with global and regional coverage of a wide range of taxonomic groups and other specific taxonomic groups.

Data source	Geographic coverage	Network	Groups of species	Website
Biodiversity Heritage Library (BHL)	United States, South Africa, Australia, China, Egypt, Europe, Brazil	More than 45 million pages	Broad	http://biodivlib.wikispaces.com/About
BIOTA-FAPESP Program (BIOTA)	Brazil	10 databases	Broad	http://www.biota.org.br/
Global Biodiversity Information Facility (GBIF)	Global	96 participant countries, economies and international organizations	Broad	https://www.gbif.org/
The Brazilian Biodiversity Information Facility – SiBBr	Brazil	93 institutions	Broad	http://www.sibbr.gov.br/
The World Information Network on Biodiversity (REMIB)	146 Countries	25 databases from various countries	Broad	http://www.conabio.gob.mx/remib_ingles/doctos/remib_ing.html
European Natural History Specimen Information Network Facility (ABIF)	Europe	Seven institutions from Europe, three data providers representing various natural history collections	Broad	http://www.nhm.ac.uk/science/research/european_natural_history_specimen_information_network_facility/
European Bird Census Council	Europe	~100 research institutions and NGOs from European countries	Birds	https://www.ebcc.info/
Integrated Biodiversity Assessment Tool (IBAT) for Research & Conservation Planning	Global	Tool aggregating biodiversity data from BirdLife International, Conservation International, IUCN, UN Environment	Broad	https://www.ibat-alliance.org/ibat-conservation/
The Biota of Canada Information Network (BCIF)	Canada	40 Canadian databases	Broad	http://www.durable.gc.ca/group/biota/index_e.phtml
Distributed Information Network for Biological Collections (SpeciesLink)	Brazil	12 databases in São Paulo state, Brazil	Broad	http://splink.cria.org.br/index?&setlangZen
SiB Colombia	Colombia	Network with more than 90 institutions	Broad	https://www.sibcolombia.net/el-sib-colombia/
Instituto Nacional de Biodiversidad (INBio/Atta)	Costa Rica	One institute	Broad	http://atta.inbio.ac.cr/attaing/atta03.html
Mammal Networked Information System (MANIS)	Global	32 institutions	Mammals	http://dip.cs.berkeley.edu/manis/

Fishnet	North America	24 North American fish databases	Fish	http://habanero.nhm.ku.edu/fishnet/
HerpNET (HerpNet)	Global	37 databases	Broad	http://herpnet.org/
Missouri Botanical Garden (Tropicos)	Global	One institution	Plants	http://mobot.mobot.org/W3T/Search/vast.html
Living Planet Index (LPI)	Global	19 institutions	Broad	http://www.livingplanetindex.org/
Red Interamericana de Información sobre Biodiversidad (IABIN)	American continent	4 thematic networks	Broad	http://www.oas.org/es/sedi/dsd/iabin/default.asp
ASEAN Regional Centre for Biodiversity Conservation (ARCBC)	South east Asia	31 databases	Broad	https://www.arcbc.org.ph/default.html
Conservation Evidence	Global	Review of conservation actions and their effects	Broad	https://www.conservationevidence.com/

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