Food and Agriculture Organization of the United Nations

VERSION 1

Biodiversity and the livestock sector

Guidelines for quantitative assessment

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Preparation of this document

These guidelines are a product of the Livestock Environmental Assessment and Performance (LEAP) Partnership. The following groups contributed to their development.

LEAP BIODIVERSITY TECHNICAL ADVISORY GROUP

The Technical Advisory Group (TAG) on biodiversity, hereafter called Biodiversity TAG, is composed of 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 Biodiversity TAG conducted the background research and developed the core technical content of the guidelines.

The Biodiversity TAG was led by Tim McAllister (Agriculture and Agri-Food Canada and the University of Alberta), assisted by the technical secretary, Félix Teillard. Members of the TAG were Abhishek Chaudhary (ETH Zurich, Switzerland), Alejandra Martínez-Salinas (Tropical Agricultural Research and Higher Education Center – CATIE, Costa Rica), Arno Krause (Centre for Grassland, Germany), Assumpció Anton (Institute of Agrifood Research and Technology – IRTA, Spain), Bai Yongfei (Chinese Academy of Sciences, China), Danielle Maia de Souza (Université du Québec à Montréal, Canada), David McCracken (Scotland's Rural College, United Kingdom), Eyob Tenkir (Ministry of Environment, Ethiopia), Félix Teillard (FAO, Italy), Fernando Aiello (Faculty of Agricultural Sciences, National University of the Littoral – UNL, Argentina), Greg Thoma (University of Arkansas, United States of America), Jason Sircely (International Livestock Research Institute – ILRI, Kenya), John Finn (Agriculture and Food Development Authority – Teagasc, Ireland), Mario Barroso (The Nature Conservancy, Brazil), Marta Alfaro (Agricultural Research Institute – INIA, Chile), Michael Scarsbrook (Fonterra Co-operative Group, New Zealand), Nico Polman (Wageningen University and Research – WUR, the Netherlands), Olga Barbosa (Austral University of Chile, Chile), Oscar Blumetto (INIA, Uruguay), Philippe Jeanneret (Agroscope, Switzerland), Suiá Kafure da Rocha (Ministry of Environment, Brazil), Vânia Proença (University of Lisbon – ULisboa, Portugal), Vincent Manneville (French Livestock Institute – Idele, France). In addition, Sarah Pogue, Mohammad Reza (Agriculture and Agri-Food Canada) and Majid Iravani (Alberta Biodiversity Monitoring Institute, Canada) provided inputs on specific aspects of the document.

The Biodiversity TAG met in two workshops. The first workshop was held on 18–20 September 2017 at the Food and Agriculture Organization of the United Nations (FAO), Rome, Italy and the second workshop was held on 22–26 January 2018 at the International Livestock Research Institute (ILRI), Nairobi, Kenya. Between and after the workshops, the TAG worked via online communications and teleconferences.

LEAP SECRETARIAT

The LEAP Secretariat coordinated and facilitated the work of the TAG, guided and contributed to the content development and ensured coherence between the various guidelines. The LEAP Secretariat, hosted at FAO, was composed of: Camillo De Camillis (Technical officer and LEAP manager), Carolyn Opio (Technical officer and Coordinator), Aimable Uwizeye (Technical officer), Félix Teillard (Technical officer) and Maria Soledad Fernandez Gonzalez (Communication specialist). Félix Teillard and Camillo De Camillis coordinated technical input to the LEAP TAG.

LEAP STEERING COMMITTEE

The LEAP Steering Committee provided overall guidance for the activities of the Partnership and facilitated review and clearance of the guidelines for public release.

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MULTI-STEP REVIEW PROCESS

The initial draft guidelines developed by the TAG over 2017 and 2018 underwent an internal review by the LEAP Secretariat and Steering Committee, and an external peer review before being revised and submitted for public review. Gordon Smith (Ecofor, United States of America) and Karen Castaño Quintana (Centre for Research on Sustainable Agriculture – CIPAV, Colombia) peer reviewed these guidelines.

Before being submitted for both external peer review and public review, the guidelines were reviewed by the LEAP Secretariat. The LEAP Steering Committee also reviewed them at various stages of their development and provided additional feedback before clearing and releasing them for public review.

The public review lasted from July to September 2019 and was advertised through the FAO website. Scholars in life cycle assessment (LCA) were informed through announcements circulated via the mailing list on LCA held by PRé Consultants, and through the Life Cycle Initiative. Experts in ecology were reached through the networks of the TAG members and through various channels by LEAP partners such as IUCN and WWF. Livestock production system experts were reached through the Livestock Technical Network Newsletter and the Pastoralist Knowledge Hub. The following bodies were also asked for their input: FAO Collaborative Partnership on Sustainable Wildlife Management; Livestock Research Group of the Global Research Alliance on Agricultural Greenhouse Gases (GRA); Global Agenda for Sustainable Livestock (GASL); Global Alliance for Climate-smart Agriculture (GACSA); Mitigation of Climate Change in Agriculture (MICCA) Project; Standing Committee on Agricultural Research (SCAR); Joint Programming Initiative on Agriculture Food Security and Climate Change

(FACCE-JPI); European Commission's Environmental Footprint Technical Board members. Comments were also sought from relevant FAO technical units.

The following participated in the public review and hence contributed to improving the quality of this technical document: Paul Welcher (USDA, United States of America), Brad Fraleigh (Agriculture and Agri-Food Canada), John Erik Hermansen (Aarhus University, Denmark), Ashley McDonald (National Cattlemen's Beef Association, United States of America), Alexandra Marques (European Commission Joint Research Centre).

PERIOD OF VALIDITY

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

Contents

Foreword

These guidelines are a product of the Livestock Environmental Assessment and Performance (LEAP) Partnership, a multi-stakeholder initiative whose goal is to improve the environmental sustainability of livestock supply chains through better methods, metrics and data.

The aim of the methodology developed in these guidelines is to introduce a harmonized international approach for assessing the impacts of livestock on biodiversity. The livestock sector is a major user of natural resources (land in particular) and an important contributor to pollution (e.g. causing nutrient losses, increasing greenhouse gas emissions), which makes it one of the sectors with the highest impact on biodiversity. At the same time, livestock production is one of the few sectors with not only negative but also positive impacts on biodiversity; therefore, the sector can pull two levers to improve its biodiversity performance – mitigate harm and maximize benefits.

Many environmental assessments of the livestock sector have not addressed biodiversity because of its intrinsic complexity. These guidelines strive to include biodiversity in environmental assessments, in order to increase the understanding of the impacts of livestock on biodiversity and to reveal possible synergies or trade-offs with other environmental criteria or Sustainable Development Goals (SDGs). Several indicators in these guidelines are also of relevance for the UN Decade on Ecosystem Restoration.

The specific objectives of these guidelines are:

- To develop a harmonized, science-based approach resting on a consensus among the sector's stakeholders;
- To recommend a scientific, but at the same time practical, approach that builds on existing or developing methodologies;
- To promote approaches for assessing the impact of livestock on biodiversity at local to global scale, by various users and relevant to diverse global livestock supply chains;
- To leave room for the adaptation of the methodology to specific assessment goals and conservation priorities, while providing a common framework to ensure a minimum level of harmonization as well as robustness and transparency.

The Covid-19 pandemic has flagged to the public the findings of scientific studies showing how the rise in emerging infectious disease outbreaks (Smith *et al.*, 2014) is linked to pressures on ecosystems and biodiversity. Biodiversity exploitation and land use change such as agricultural encroachment, deforestation, urbanization and infrastructure development are important drivers of infectious disease emergence. This is because they increase contact between wildlife, humans and livestock, and consequently the spillover risk of emerging zoonoses (Plowright *et al.*, 2015). More generally, land use change leads to a cascade of factors exacerbating infectious disease emergence, including forest fragmentation, pollution, poverty and migration (Patz *et al.*, 2004). Emerging infectious disease events are dominated by zoonoses caused by viruses (Jones *et al.*, 2008) such as Covid-19, but land use change also influences vector-borne diseases (e.g. malaria linked to deforestation – Singh *et al.*, 2004). Assessments investigating how and to what extent biodiversity is associated with a risk or an impact on infectious disease emergence transmission are not in the scope of these guidelines. Studying the dynamics of infectious diseases requires much interdisciplinary expertise ranging from ecology to epidemiology, evolutionary biology, immunology, sociology and public health. Similarly, environmental risk assessment studies such as those targeting antimicrobial resistance (AMR) and ecotoxicity are also outside the scope of these guidelines. On the other hand, the guidelines do cover the ecological assessment of a number of livestock pressures on biodiversity that are also drivers of infectious disease emergence and toxicology – for example, land management and land use transformation, biodiversity exploitation, and contact between feed, livestock and ecosystems.

During the development process, these guidelines were submitted for technical review and public review. The purpose was to strengthen the advice provided and ensure that the technical document meets the needs of those seeking to improve biodiversity and environmental performance through sound assessment practice. This document is not intended to remain static. It will be updated and improved as the sector evolves and more stakeholders become involved in LEAP, and as new methodological frameworks and data become available.

The guidelines developed by the LEAP Partnership gain strength because they represent a multi-actor-coordinated, cross-sectoral and international effort to harmonize assessment approaches. Ideally, the harmonization leads to greater understanding, transparent application and communication of metrics, and, not least, real and measurable improvement in environmental performance.

Pablo Frere, World Alliance of Mobile Indigenous Peoples (LEAP chair 2020)

Caroline Emond, International Dairy Federation (LEAP chair 2019)

Ruy Fernando Gil, Uruguay (LEAP chair 2018)

Pablo Manzano, International Union for Conservation of Nature (IUCN) (LEAP chair 2017)

Hsin Huang, International Meat Secretariat (IMS) (LEAP chair 2016)

Henning Steinfeld, Food and Agriculture Organization of the United Nations (FAO) (LEAP co-chair)

Acknowledgements

FAO is very grateful for all valuable contributions provided at various levels by LEAP partners. Particular gratitude goes to the following countries that have continually supported the Partnership through funding and often in-kind contributions: France, Ireland, the Netherlands, New Zealand, Canada, Switzerland and Uruguay. Appreciation also goes to the French National Research Institute for Agriculture, Food and Environment (INRAe) for in-cash and in-kind contributions to the LEAP Partnership. Particularly appreciated were the in-kind contributions from the following civil society organizations and non-governmental organizations represented in the Steering Committee: the International Planning Committee for Food Sovereignty, the International Union for Conservation of Nature (IUCN), the World Alliance of Mobile Indigenous Peoples (WAMIP), World Vision and the World Wild Fund for Nature (WWF). The following international organizations and companies belonging to the LEAP private sector cluster also played a major role by actively supporting the project via funding and/or in-kind contributions: the International Dairy Federation (IDF), the International Egg Commission (IEC), the International Feed Industry Federation (IFIF), the International Meat Secretariat (IMS), the International Poultry Council (IPC), the International Council of Tanners (ICT), the International Wool Textile Organisation (IWTO), the European Union vegetable oil and protein meal industry association (FEDIOL), Health for Animals, the Global Feed LCA Institute (GFLI), DSM Nutritional Products AG and Novus International. Last but not least, the LEAP Partnership is also grateful for the advice provided by the International Organization for Standardization (ISO), UN Environment and the European Commission, and is honoured to network with the Global Research Alliance, the Life Cycle Initiative, the Global Soil Partnership, the "4 per 1000" Initiative, the Global Alliance for Climate-smart Agriculture (GACSA), and to share achievements in the context of the Global Agenda for Sustainable Livestock.

ADDITIONAL CONTRIBUTIONS

Although not directly responsible for the preparation of these guidelines, the other TAGs of the LEAP Partnership indirectly contributed to the preparation of this technical document.

Professional editing and proofreading was done by Ruth Duffy. Enrico Masci and Claudia Ciarlantini (FAO) were responsible for figure design and for formatting and layout of this publication. Administrative support was provided by Christine Ellefson (FAO).

Abbreviations and acronyms

BSI	British Standards Institution
CBD	Convention on Biological Diversity
CF	Characterization factor
CGRFA	Commission on Genetic Resources for Food and Agriculture
DAD-IS	Domestic Animal Diversity Information System
EBV	Essential biodiversity variable
ES	Ecosystem services
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GHG	Greenhouse gas
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
IUCN	International Union for Conservation of Nature
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LEAP	Livestock Environmental Assessment and Performance (Partnership)
MEA	Millennium Ecosystem Assessment
NGO	Non-governmental organization
OECD	Organisation for Economic Co-operation and Development

PDF	Potentially disappeared fraction (of species)
PNV	Potential natural vegetation
PSL	Potential species loss
PSR	Pressure-state-response
SAR	Species–area relationship
SDG	Sustainable Development Goal
SETAC	Society of Environmental Toxicology and Chemistry
SOC	Soil organic carbon
TAG	Technical Advisory Group
UN	United Nations
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
VS	Vulnerable species
WWF	World Wide Fund for Nature

Background information on Livestock Environmental Assessment and Performance Partnership and Technical Advisory Group on Biodiversity

LIVESTOCK ENVIRONMENTAL ASSESSMENT AND PERFORMANCE (LEAP) PARTNERSHIP

LEAP 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 (FAO), 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 the Partnership (2012–15) focused mainly on the development of guidelines to quantify the greenhouse gas (GHG) emissions, energy use and land occupation from feed and animal supply chains as well as the principles for biodiversity assessment. The second phase (2016–18), known as LEAP+, broadened the scope and focused on water footprinting, nutrient flows and impact assessment, soil carbon stock changes, quantification of the impact of livestock on biodiversity and assessment of 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 the 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 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.[1]

TECHNICAL ADVISORY GROUP 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 the technical secretary, Félix Teillard. Members of the TAG included Abhishek Chaudhary (ETH Zurich, Switzerland), Alejandra Martínez-Salinas (CATIE, Costa Rica), Arno Krause (Centre for Grassland, Germany), Assumpció Anton (IRTA, Spain), Bai Yongfei (Chinese Academy of Sciences, China), Danielle Maia de Souza (Université du Québec à Montréal, Canada), David McCracken (Scotland's Rural College, United Kingdom), Eyob Tenkir (Ministry of Environment, Ethiopia), Félix Teillard (FAO, Italy), Fernando Aiello (Faculty of Agricultural Sciences, UNL, Argentina), Greg Thoma (University of Arkansas, United States of America), Jason Sircely (ILRI, Kenya), John Finn (Teagasc, Ireland), Mario Barroso (The Nature Conservancy, Brazil), Marta Alfaro (INIA, Chile), Michael Scarsbrook (Fonterra Co-operative Group, New Zealand), Nico Polman (WUR, Netherlands), Olga Barbosa (Austral University of Chile, Chile), Oscar Blumetto (INIA, Uruguay), Philippe Jeanneret (Agroscope, Switzerland), Suiá Kafure da Rocha (Ministry of Environment, Brazil), Vânia Proença (ULisboa, Portugal), Vincent Manneville (Idele, France). In addition, Sarah Pogue, Mohammad Reza (Agriculture and Agri-Food Canada) and Majid Iravani (Alberta Biodiversity Monitoring Institute, Canada) provided inputs on specific aspects of the document.

The TAG met in two workshops. The first was held on 18–20 September 2017 at the Food and Agriculture Organization (FAO), Rome, Italy and the second was held on 22–26 January 2018 at the International Livestock Research Institute (ILRI), Nairobi, Kenya. Between and after the workshops, the TAG worked via online communications and teleconferences.

[1] More background information on the Partnership can be found at www.fao.org/partnerships/leap/en/.

Summary of key messages and guidelines

1. 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, together with other environmental criteria, to meet international commitments on biodiversity and avoid the risk of burden shifting among environmental criteria.

2. OBJECTIVE AND SCOPE

- The objective is to develop guidelines for quantitative assessment of the effects of livestock production on wild biodiversity, based on existing indicators and methods.
- The 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 discusses links with the ecosystem level. Livestock genetic diversity is outside the scope; therefore, "biodiversity" in this document refers to wild biodiversity unless specified otherwise.

3. GENERAL INFORMATION ON THE RELATIONSHIP BETWEEN LIVESTOCK, BIODIVERSITY AND ECOSYSTEM SERVICES

3.1 General ecological principles

- Biodiversity describes the variability of life on earth and is often positively affected by intermediate levels of disturbance opening 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 density)

systems and will determine impacts on biodiversity per unit of livestock product. In addition, several indicators can reflect negative impacts of extensive systems on biodiversity (e.g. soil erosion, degraded soil, livestock density).

3.2 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 and water quality regulation, climate regulation via soil carbon storage) and beneficiaries (e.g. forage production, pollination, pest control, water supply) of ES.
- Biodiversity plays a key role in ES provision as a regulator of ecosystem processes, an ecosystem service and the benefits that flow to people from ES.
- 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 ES, potentially damaging the natural processes that are essential for food production, leading to collapse of the system in 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.

3.3 Genetic diversity of livestock

- Livestock genetic diversity is beyond the scope of this document, but can be assessed using the work of the FAO Commission on Genetic Resources for Food and Agriculture.
- More than 8 800 livestock breeds have been recorded globally by the FAO Commission on Genetic Resources for Food and Agriculture, representing a valuable resource and a high biodiversity at the genetic level.
- Animal genetic resources remain at risk and share many drivers of loss with wild biodiversity (e.g. increased demand for animal products, intensification, degradation of natural resources, 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.

4. 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.

5. LIFE CYCLE ASSESSMENT REGIONAL AND GLOBAL ASSESSMENTS

5.1 Impact pathway (cause–effect chain)

- This section describes the impact pathway used in the document, that 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.
- 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.

5.2 Goal and scope

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 those indicated in the LEAP sectoral guidelines (FAO, 2016a, 2016b, 2016c, 2016d, 2018a), such as 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, 2016a, 2016b, 2016c, 2016d, 2018a) and typically encompass all stages of production, from raw material extraction to the primary processor gate.

Scale of assessment: global/regional/local

- LCA is well suited to consider complex supply chains encompassing geographically distributed locations that are common to most livestock sectors at both regional and global scale.
- Currently, LCA is not well suited for assessing local biodiversity effects as global data lack 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 life cycle impact assessment

- Different LCIA models can address different biodiversity indicators such as species richness, abundance or functional diversity.
- The main LCIA model recommended in this document for global/regional assessments (Chaudhary and Brooks, 2018) uses the potentially disappeared fraction of species (i.e. impact on species richness) as an indicator.

- Other models can be used to address different biodiversity components, if justification and discussion are provided.

5.3 Life cycle inventory

- Land use (in $m^2 \times$ 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 according to the impact characterization. When available, the location where they occur should be known and reported with as high a resolution 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.

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

- The Chaudhary and Brooks (2018) model provides characterization factors (CFs) reflecting regional or global species extinctions for different taxa; these guidelines recommend using the global taxa-aggregated CFs.
- Recommendations in these guidelines are consistent with those of the United Nations Environment Programme and the Society of Environmental Toxicology and Chemistry (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 the 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.
- 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 main LCIA model recommended in this document for global/regional assessments (Chaudhary and Brooks, 2018) uses 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.

6. LOCAL ASSESSMENTS USING PRESSURE-STATE-RESPONSE INDICATORS

6.1 The framework for local assessment

- When conducting a local biodiversity assessment using the PSR framework, five 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, together with stakeholder engagement, which should occur iteratively along the five steps.

Within the five major steps, it is important to consider the following key principles:

- 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 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 production'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 describe, respectively, the pressures from human activities on the environment, the resulting changes in environmental conditions and the societal response to environmental concerns, 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.

6.2 Recommended list of biodiversity indicators for local assessments

- This section (and Table 2) 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 potential interlinkages 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.

7. INTERPRETATION AND COMMUNICATION

7.1 Interpretation of results

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

7.2 Developing effective 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.

7.3 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 (e.g. 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. It is important to bear in mind that a specific indicator for one biodiversity level or dimension, such as species composition, may not be fully adequate to depict linkages between ecosystem function, biodiversity and ES.

8. DATA AND DATA QUALITY

8.1 Introduction

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

8.2 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.

8.3 Data quality assessment

- Data quality should be assessed by authoritative organizations (e.g. government, local agencies, research organizations, specialized NGOs), 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.

8.4 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 data sets are their spatial/temporal extent and resolution; there are frequent trade-offs among these dimensions and they shall be considered and justified when selecting data to match 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.

PART 1

Overview and general principles

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 together 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; Lenzen *et al.*, 2012; Secretariat of the CBD, 2014; Gibbs *et al.*, 2015). A first step in this direction is environmental footprinting, that is 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 percent of ice-free land on Earth is used for pastures (18%) and feed crops (4%) (Mottet *et al.*, 2017), resulting in habitat modification and biodiversity change. Livestock also contributes to climate change – the second most important driver of global biodiversity loss (MEA, 2005; Secretariat of the CBD, 2014) – by releasing about 14.5 percent of global anthropogenic greenhouse gases (GHG) (Gerber *et al.*, 2013, using 100-year Intergovernmental Panel on Climate Change [IPCC] global warming potential to convert CH_4 and N_2O into CO_2 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 (ES) in semi-arid (Milchunas *et al.*, 1989), tropical (Overbeck *et al.*, 2007) and temperate (Pogue *et al.*, 2018) grasslands. However, it does not apply to areas where the forest has been recently replaced with pasture (Nepstad *et al.*, 2008).

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

The inclusion of biodiversity in environmental assessment is an emerging, but increasingly important area of work. Several recent initiatives have attempted to address the relationship between biodiversity and livestock production. In the United Nations Environment Programme and the Society of Environmental Toxicology and Chemistry (UNEP–SETAC) life cycle initiative (Teixeira *et al.*, 2016), a specific task force worked on the inclusion of land use impacts on biodiversity in life cycle assessment (LCA), capitalizing on important past efforts and identifying future challenges (Curran *et al.*, 2016; Winter *et al.*, 2017). Other initiatives on biodiversity assessment exist at various levels:

- **Global** – Convention on Biological Diversity (CBD) biodiversity indicators for commodity production; the core initiative on biodiversity of the FAO-UNEP (United Nations Environment Programme) 10-Year Framework of Programmes.
- **Regional** – Product Environmental Footprint of the European Union.
- **Sectoral** – Cool Farm Tool; Sustainable Agriculture Initiative Platform; International Dairy Federation.

There is a need to ensure that the livestock sector – with its specific relationship with biodiversity (e.g. positive impacts) – is not left behind in recent developments on biodiversity assessment.

During LEAP 1 (2012–14), a first step was taken to tackle the challenge of biodiversity assessment in the livestock sector, with the formation of a dedicated TAG on biodiversity and the development of *Principles for the assessment of livestock impacts on biodiversity* (FAO, 2016e). The present document builds on this previous work and furthers it by moving from qualitative principles to guidelines for the quantitative assessment of livestock impacts on biodiversity.

1.2 THE NEED FOR QUANTITATIVE INDICATORS

LEAP has enabled a high level of methodological consensus on how to quantify GHG emissions and other environmental impacts, including nutrient cycles and water from livestock supply chains. It has allowed for a number of quantitative assessments and for technical and policy options to be proposed in order to mitigate the livestock contribution to climate change. In particular, increasing the efficiency and intensity of livestock production has been suggested as a mitigation option because more intensive mixed production systems where livestock are partially fed using crop by-products have lower GHG or nutrient emissions per unit of product compared to grassland-based systems (Gerber *et al.*, 2010, 2014). However, changing to high-input and intensively managed systems could result in higher impacts on biodiversity because of the associated habitat changes (e.g. natural to improved pastureland, grassland to feed crops) and negative effects through water withdrawal and the use of pesticides or inorganic fertilizers. On the contrary, extensively managed grassland-based systems can provide crucial biodiversity habitats and store vast quantities of carbon, but with higher GHG emissions per unit of product compared to intensively managed systems. “Units of product” usually focus on food or proteins and do not take into account other social benefits and ecosystem services.

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

Quantitative biodiversity assessments are needed to support international agreements that recognize the importance of biodiversity conservation, such as the 2020 Aichi Biodiversity Targets set by the Convention on Biological Diversity (CBD) and Sustainable Development Goals (SDGs) 14 and 15 on protecting, restoring and promoting sustainable use of terrestrial ecosystems. Furthermore, after the decision of the United Nations Framework Convention on Climate Change (UNFCCC) Conference of the Parties (COP23) 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, CBD and Global Plan of Action for Animal Genetic Resources; SDGs 13, 14 and 15) in the transition towards sustainable livestock production (FAO, 2018b).

A series of 13 case studies are cited throughout the document as examples to users of how the described methodologies were employed. The case studies are available in the forthcoming companion LEAP publication on quantitative case studies of livestock impacts on biodiversity.

2. Objective and scope

Key messages

- *The objective of this document is to develop guidelines for quantitative assessment of the impacts of livestock production on wild 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 discusses links with the ecosystem level. Livestock genetic diversity is outside the scope; therefore, "biodiversity" in this document refers to wild biodiversity unless specified otherwise.*

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 pressure-state-response (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.

2.2 OBJECTIVE AND INTENDED USERS

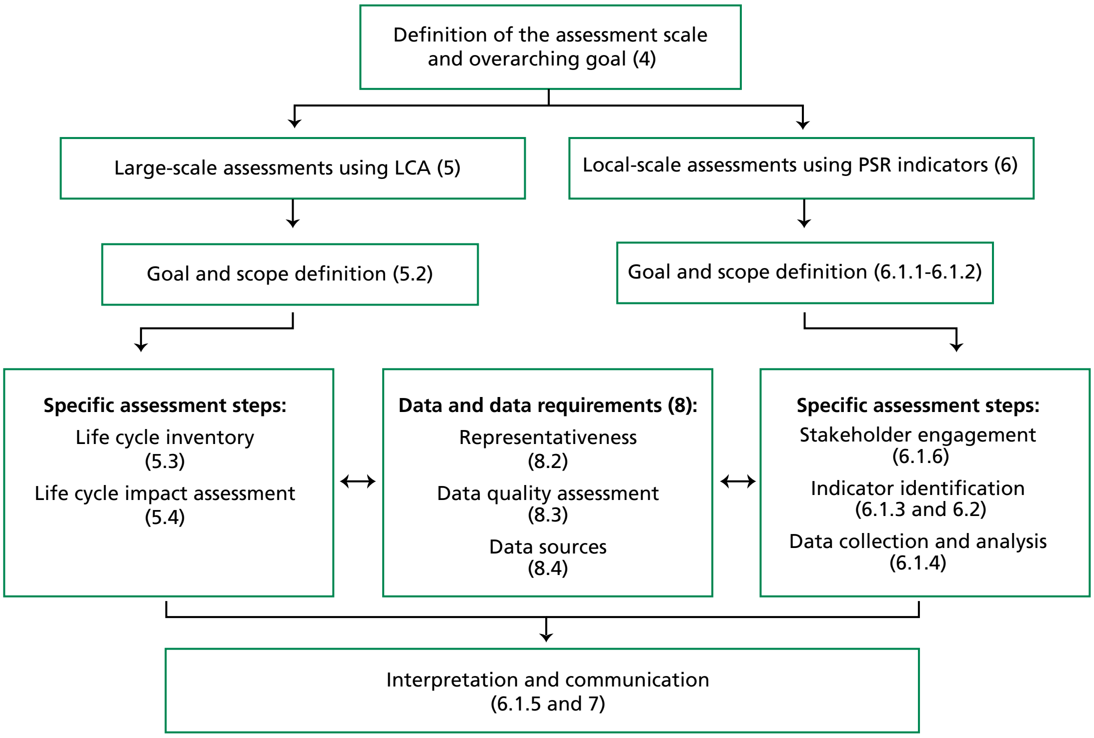
The objective of this document is to develop guidelines for quantitative assessment of the impacts of livestock production on wild biodiversity, based on existing indicators and methods.

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 the following:

- 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.
- Policymakers interested in developing biodiversity assessment and reporting specifications for livestock supply chains.
- Environmental organizations or land managers conducting biodiversity assessments for conservation objectives.

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



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. Section 3.2 describes the linkages between livestock, biodiversity and ES and highlights overlaps between assessment frameworks and methods. The assessment of livestock genetic resources is beyond the scope of this document, but section 3.3 provides background information and data sources on this aspect of biodiversity. In the absence of additional specifications, 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, chickens and buffaloes); however, 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 opening 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 density) systems and will determine impacts on biodiversity per unit of livestock product. In addition, several indicators can reflect negative impacts on biodiversity from extensive systems (e.g. soil erosion, degraded soil, livestock density).*

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 be linked to human-influenced management factors (e.g. deforestation, cropping, grazing by livestock, ploughing, breeding management, nutrient and pesticide input).

Disturbances linked to human influence often occur in more serious ways or at much higher levels, resulting in biodiversity losses.

Taking plant species richness as an example, situations subject to high disturbance, in terms of either 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 homogeneous vegetation structures, which limit biodiversity by restricting the growth and colonization 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 impact biodiversity are substantially more influenced by management and policy decisions. Hence, in livestock production systems, biodiversity is primarily driven by the nature and intensity of the management to which those habitats are subjected.

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

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

Adverse impacts on biodiversity can be associated with both extensive and intensive systems (FAO, 2016e). In extensive systems, unsuitable grazing management can lead to overgrazing which causes soil degradation and reduces plant diversity as well as productivity. Abandonment (i.e. inadequate or no grazing) can also lead to land degradation and biodiversity loss through high dominance of few species or shrub encroachment if wild herbivores are scarce or absent (Laiolo *et al.*, 2004; Koch *et al.*, 2016). In intensive or confined livestock production systems, a large proportion of the feed used is usually produced off-farm, in intensive farmland with simplified landscapes. Intensive livestock systems can also concentrate manure at the site of production, which if improperly managed, can adversely impact soil and water quality (Case studies 1 and 4).

In ruminant systems and beef production in particular, intensive and extensive systems are very often interlinked within the same supply chain. Grassland-based (extensive) cow-calf operations produce calves that are then sold to be fattened in confined (intensive) systems to produce final meat products and other by-products. Increasing the biodiversity performance will thus require an integrated supply chain perspective, as well as specific solutions targeting the extensive and intensive phases of the production cycle.

Reconciling food production and biodiversity conservation objectives could be achieved in contrasting ways in extensive versus intensive livestock production systems (Green *et al*., 2005). Intensive systems typically require a lower area of land for feed production per unit of livestock product produced and could therefore theoretically spare land for natural and semi-natural areas. By increasing the efficiency and integration of such systems in the circular bioeconomy, they maximize productivity while minimizing resource use and externalities (Godfray and Garnett, 2014). In some areas, intensification could also be achieved through an increase in biomass production by introducing shrubs and trees (silvopastoral systems) that broaden biodiversity habitats, sequester carbon and enhance the provision of ES (Chará *et al*., 2019; Case study 11).

Conversely, extensive systems use a higher area of land and actually exert less pressure on habitats. They can therefore be considered of high biodiversity value, even after some amount of transformation of natural habitats at a landscape level through the disruption of natural regimes occupied by wild herbivores which can be displaced by domestic herbivores (Appendix 2). Grazing influences and promotes biodiversity in grassland ecosystems (Watkinson and Ormerod, 2001) and several models have been proposed to describe the effect of grazing on vegetation (Cingolani, Noy-Meir and Díaz, 2005; Case study 8). Under adequate management and after a long history of livestock grazing, domestic animals can perform the ecological role of wild herbivores in maintaining this unique biodiversity in production systems (Eriksson, Cousins and Bruun, 2002; Bond and Parr, 2010). This is particularly the case where large herbivore guilds have undergone Pleistocene extinctions (Corlett, 2016). The biodiversity value of extensive livestock systems relates both to their spatial and temporal heterogeneity and to their 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 and 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, 2016f).

3.2 ECOSYSTEM SERVICES

Key messages

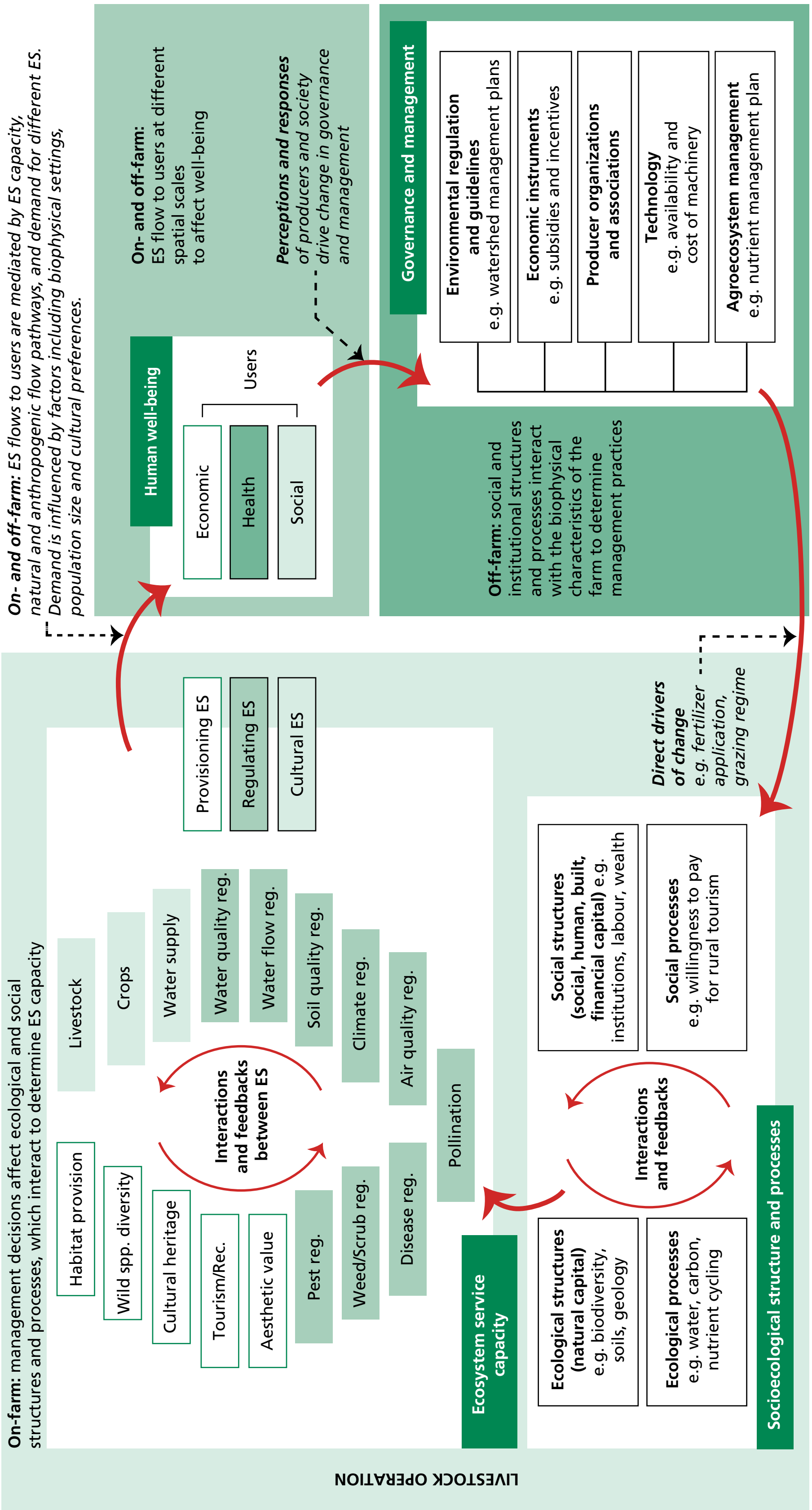
- *Ecosystem services (ES) are the outcomes from ecosystems that lead to benefits valued by people. Agroecosystems are both providers (e.g. food production, soil and water quality regulation, climate regulation via soil carbon storage) and beneficiaries (e.g. forage production, pollination, pest control, water supply) of ES.*
- *Biodiversity plays a key role in ES provision: as a regulator of ecosystem processes, as an ecosystem service and as the benefits flowing to people from ES.*
- *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 also with 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 ES, potentially damaging the natural processes that are essential for food production, leading to collapse of the system in 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.*

Ecosystem services (ES) are "nature's contribution to people" (Díaz *et al*., 2018), that is the outcomes from ecosystems that lead to benefits valued by people. In livestock production systems, ecological structures and processes (e.g. geographical and climatic conditions, wild and domestic biodiversity, water and nutrient cycling) and social structures and processes (e.g. financial and built capital) interact to co-produce ES (Palomo *et al*., 2016). The different categories of ES include provisioning services (e.g. water, wood, genetic resources, crop and livestock products), regulating services (e.g. soil, air and water quality, climate regulation) and cultural services (e.g. cultural identity, recreation and tourism). The flow of these services to people at different scales contributes to their economic, health and social well-being, although demand for certain services can vary among individuals and groups (Yahdjian, Sala and Havstad, 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).

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 – refer to the LEAP guidelines [FAO, 2019a]) and cultural services (e.g. aesthetic value such as the globally important agricultural heritage sites – FAO, 2018c). In addition, agroecosystems interact with the surrounding landscape matrix, benefiting from services delivered by non-agricultural systems (e.g. pollination) or impacting these systems as occurs when nutrient run-off impacts downstream water quality (Dale and

Figure 2

Conceptual ecosystem service framework for a livestock production system



Source: Reyers *et al.* (2013) (adapted).

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, Peterson and Bennett, 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, Angermeier and Bennett, 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 socioecological 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, Norris and Fitter, 2012).

Biodiversity plays a key role in ES provision as a regulator of ecosystem processes, as an ecosystem service and as a good or benefit (Mace, Norris and Fitter, 2012). In grazed grasslands, many soil nutrient cycles are determined by soil biological community composition (Mace, Norris and Fitter, 2012), including organisms that break down and integrate dung into 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 with 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 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, Bennett and Gonzalez, 2013). As an ES, biodiversity at the species and gene levels contributes directly to the generation of goods, as crop and livestock genetic diversity is important for the maintenance of crop and livestock populations (FAO, 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 GHG 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 maximize 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, ES 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 ES 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 ES (e.g. landscapes, river basins, regions, biomes, countries). Forests in particular contribute to important ES 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 (e.g. ecological-economic zoning) can be beneficial.

3.3 GENETIC DIVERSITY OF LIVESTOCK

Key messages

- *Livestock genetic diversity is beyond the scope of this document, but can be assessed using the work of the FAO Commission on Genetic Resources for Food and Agriculture (CGRFA).*
- *More than 8 800 livestock breeds have been recorded globally by the CGRFA, representing a valuable resource and high biodiversity at the genetic level.*
- *Animal genetic resources remain at risk and share many drivers of loss with wild biodiversity (e.g. increased demand for animal products, intensification, degradation of natural resources, 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.*

Livestock resulted from the domestication of wild ancestor species, few of which exist today. Differentiation of livestock breeds resulted from selective breeding for human needs. 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 tolerant to disease 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 goods and services such as draught power, manure for fertilization, and environmental and sociocultural 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 resource management programmes, unaligned policies and institutions, climate-controlled livestock production systems, degradation of natural resources, climate change and disease epidemics.

The Second Report on the State of the World's Animal Genetic Resources for Food and Agriculture (FAO, 2015) provides a global overview of the state and trend in livestock diversity and identifies current capacities and strategies for conservation, as well as needs and challenges.

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

DAD-IS was used in combination with climate models to develop a model predicting the potential impact of climate change on breed distribution (FAO, 2020a). Current breed distributions from DAD-IS were used to model suitable areas for breeds under current and expected future conditions, taking several temperature and humidity parameters into account.

The FAO Commission on Genetic Resources for Food and Agriculture (CGRFA) recognizes that biodiversity is essential for food production and achieving nutritional diversity in the human diet. In a recent report, biodiversity for food and agriculture was shown to be declining and although biodiversity-friendly practices exist, enabling frameworks for the sustainable use and conservation of biodiversity remain insufficient (FAO, 2019b). The role of the CGRFA and the work of its Intergovernmental Technical Working Group on Animal Genetic Resources should be recognized in describing and assessing the genetic diversity of livestock.

[5] More information at: http://www.fao.org/dad-is/en/

During its 17th Session in February 2019, the FAO Commission on Genetic Resources for Food and Agriculture took note of a review of methods for identification and valuation of the ES provided by livestock breeds (FAO, 2019b). This review defined the role of livestock production systems, and livestock breeds in particular, in the delivery of ES. It outlined the main steps involved in valuing these services and identified potential ES provided by livestock breeds and associated agro-ecosystems. The review identified the main methodologies for valuing ES in specific socio-economic and biophysical contexts.

PART 2

Methodology

4. Definition of the assessment goal and method

4.1 GOAL OF THE ASSESSMENT

Key guidelines
• *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.*

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 (EC, JRC and IES, 2010), including:

- the subject of the analysis and key properties of the assessed system (e.g. name of the organization, its sector and position in the value chain, location of production systems, dimension of facilities, end products, by-products);
- 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. 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 the need for critical review; and
- 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. the desired levels of biodiversity) or of the livestock system under study (i.e. the improvements to be achieved in the system). Moreover, the goals of the assessment should consider overarching priority issues such as the effect on:

- critically endangered species;
- key ecosystems and habitats, including global ecoregions, biodiversity hotspots and corridors, International Union for the Conservation of Nature (IUCN) Red List of ecosystems and habitats, and habitats of high ecological sensitivity (e.g. riparian habitats, areas at high risk of erosion);
- maintenance of ecosystem functioning and services in areas of high conservation value; and
- other biodiversity conservation goals within the study boundary.

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). The local scale typically corresponds to a farm, landscape or territory. It involves local biodiversity issues and conservation priorities – such as protected areas or locally endangered/patrimonial species. Even though maintaining a given rare species can be a conservation priority for local stakeholders, its local extinction will not necessarily translate into a regional extinction if it is not locally endemic or if it has a wider distribution area. Regional to global assessments need to take this into account. When considering globalized supply chains, it is impossible to consider multiple local conservation priorities, and a common unit of impact (i.e. regional or global rather than local species extinction) needs to be adopted.

Existing methods for including biodiversity assessments in LCA currently rely on a few available models – for example, Huijbregts *et al*. (2017) – that estimate the impact of different emissions on species richness. Together with these models – but often not integrated in the same assessments – several methods focus on impacts on biodiversity through land use. Available LCA models to compute land use impacts on biodiversity have limitations as they:

- consider broad land use classes (e.g. biodiversity impact of grassland vs cropland);
- have an intermediate level of biogeographical differentiation (e.g. 1 ha of grassland having the same impact anywhere within a 150 000 km^2 ecoregion); and
- focus on species richness as a biodiversity indicator.

Currently no LCA approach available on a global scale is well suited to answering 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. Currently, global-scale LCA approaches are not available to assess the impacts on biodiversity of a gradient of extensive to intensive production practices. Approaches that are based on large geographical scales are much more suited to assessing land use change impacts across bioregions and not suited to assessing other more qualitative changes such as the impacts of over- or undergrazing within a bioregion. LCA is very useful both for broadly assessing impacts on biodiversity on large spatial scales and for finding impact hotspots among life cycle steps and geographical locations along the supply chain. 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 on biodiversity of decisions

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

Assessment scale	Large-scale assessments	Local assessments
Method	Life cycle assessment	Pressure-state-response indicators
Applications	• Identify hotspots of impacts: spatially or along the supply chain • Compare systems or scenarios • Prevent burden shifting among life cycle stages or environmental impacts	• 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 subsector sustainability managers, trade/processors/other companies, NGOs, policymakers	Farmers, pastoralists, land managers, communities, local companies, NGOs, policymakers
Constraints faced by the user	• Lack of resources to collect field data • Need to consider complex and globalized supply chains	• 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 and impact locations, and to aggregate biodiversity impacts

along livestock supply chains (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, 2006b) 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 assess the 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 characterization models and factors. Impacts can be characterized anywhere along the environmental cause–effect chain, at either the midpoint or end-point 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. End-point 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 PSR framework, which utilizes local data (Chapter 6). The type, amount, spatial and temporal distribution of the data needed is determined by the combination of the goals of the study, analytical methods proposed and the scale on which the study is 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) from the stakeholders – both political and socio-economic – necessary to improve the state of the environment. Finally, data collected in the context of PSR assessments can also be used for the development of local characterization factors and specific LCA models that would be able to differentiate between management intensities and practices (Knudsen *et al.*, 2017; subsection 5.4.2).

5. Life cycle assessment regional and global assessments

5.1 IMPACT PATHWAY (CAUSE–EFFECT CHAIN)

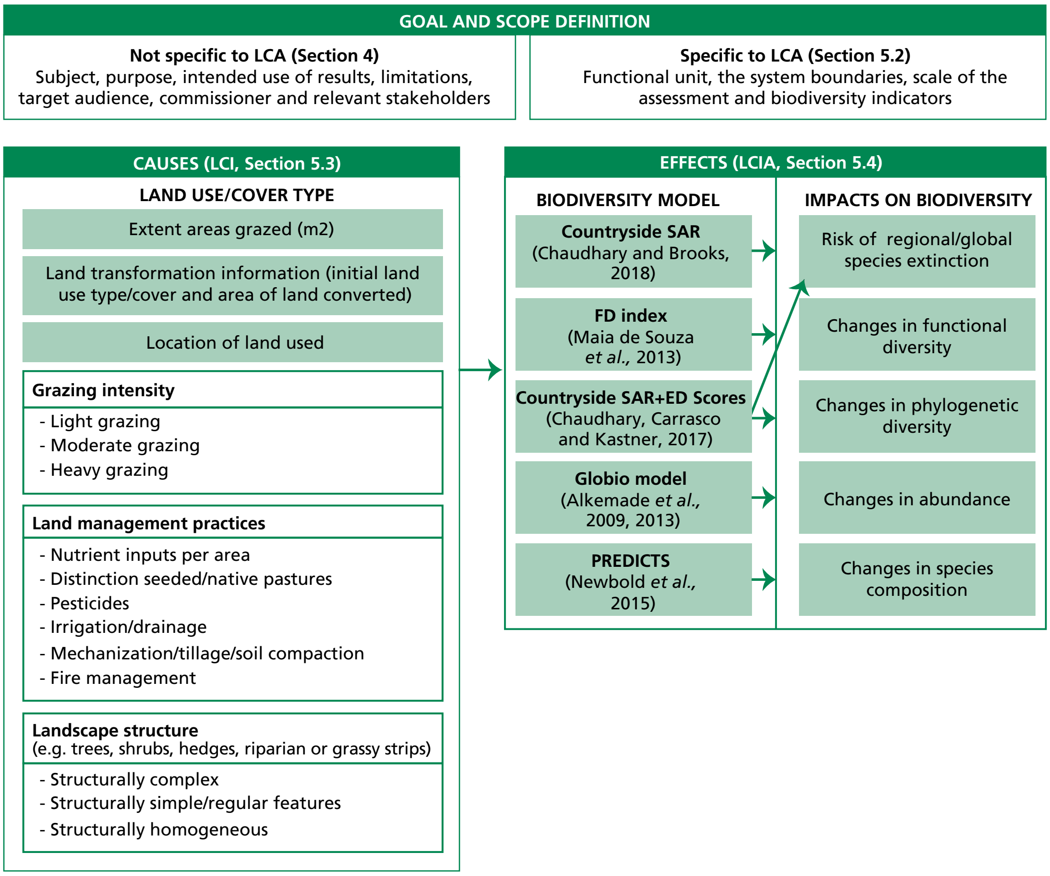
Key guidelines

- *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.*
- *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 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 (Case study 6). 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, 2016f).

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). The LCIA method recommended in this document (section 5.4) proposes three degrees of grazing intensity (light, moderate, heavy), which could be quantitatively linked with effects on biodiversity through estimates of the carrying capacity, that is the number of grazing animals that an area can support without adversely impacting biodiversity or the productivity of land. These three categories roughly correspond to the minimal, light and intensive pastureland 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*., eds, 2010). In contrast, heavy grazing (i.e. overgrazing) reduces vegetation and soil coverage. Grazing pressure also depends on animal

Figure 3
Schematic representation of the recommended cause–effect chain for the assessment of livestock impacts on biodiversity



Notes: ED – ecological distinctiveness; FD – functional diversity; SAR – species–area relationship.
The boxes in green represent aspects already covered by life cycle inventory (LCI) and life cycle impact assessment (LCIA) models. Boxes in white represent aspects that should ideally be added to current land use/cover flows in order to provide a better representation of livestock systems.

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 life cycle inventory (LCI). Such classification into discrete classes of grazing intensity is a simplification and will have important limitations especially in rangeland and semi-natural systems where factors that are not linked to livestock production practices (aridity/precipitation, time lags in natural forage dynamics) have an important influence on plant community composition (Case studies 3 and 4). PSR indicators and multivariate models disentangling the effect of those livestock and non-livestock factors will be more suited to local assessments, especially in semi-natural systems.

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 ES. These practices may be related to either on- or off-farm areas that are used to produce feed for livestock (i.e. grassland or cropland). 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 ES. Large-scale disturbance such as fire, when adequately managed can contribute to more heterogeneous habitats and therefore high levels of biodiversity (Nekola, 2002). In contrast, burning of crop residues can reduce biological activity and biodiversity in soils (Wallis *et al*., 2010).

Burel *et al*. (1998) demonstrated that a simple linear relationship between land use intensification and loss of species cannot be drawn, particularly if landscape patterns are not considered. This is mainly due to the different temporal and spatial scales at which agricultural practices and biodiversity operate (i.e. context dependency). Improved biodiversity levels are also usually associated with higher landscape heterogeneity (Belfrage, Björklund and Salomonsson, 2015) and changes in species distribution may be determined by the nature of the drivers (e.g. land management, stress on vegetation) and processes affecting landscape patterns (e.g. intensity of process, history of land use). Understanding changes in landscape heterogeneity and patterns helps to predict changes in biodiversity and ES (Lausch *et al*., 2014).

At the LCIA step, the main indicators of impacts on biodiversity are included, currently addressed by existing LCIA models:

- risk of regional/global species extinction (Chaudhary and Brooks, 2018);
- changes in functional diversity (Maia de Souza *et al*., 2013);
- changes in phylogenetic diversity/evolutionary history (Chaudhary, Pourfaraj and Mooers, 2017);
- changes in species abundance (Alkemade *et al*., 2013); and
- changes in species composition/biodiversity intactness (Newbold *et al*., 2015; Case study 13).

Moreover, this document suggests that, to 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 ES. This is mainly because the diversity of species has a direct link to and importance for the generation of ES 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. The guidelines therefore aim to represent the impact on biodiversity at the local, regional and global levels for which global data are available and ecological models exist (section 5.4).

5.2 GOAL AND SCOPE

Key guidelines

- *In LCA, as in other types of biodiversity assessment, it is first necessary to clearly set the goal for the study, specify the intended use of the results, and 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 (Chapter 4).*
- *Several elements of goal and scope definition are specific to LCA and are described in this section. They concern the functional unit, system boundaries, scale of the assessment and biodiversity indicator/impact categories.*

5.2.1 Functional unit

Key guidelines

- *In LCA, impacts are always expressed in relation to a functional unit (e.g. biodiversity loss per litre of milk or kg of carcass/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 those indicated in the LEAP sectoral guidelines (FAO, 2016a, 2016b, 2016c, 2016d, 2018a).*

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 the LEAP sectoral guidelines (FAO, 2016a, 2016b, 2016c, 2016d, 2018a).

Alternative functional units such as those based on land area required for the production system have also been used (Haas, Wetterich and Köpke, 2001; Basset-Mens and van der Werf, 2005; Bartl, Gómez and Nemecek, 2011; Case study 13). Although the aggregation of different types of products and services in a single functional unit is challenging, ES may be included in the functional unit as part of the services delivered by livestock production systems (e.g. through monetization, Ripoll-Bosch *et al*., 2013).

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 those indicated in the LEAP sectoral guidelines (FAO, 2016a, 2016b, 2016c, 2016d, 2018a) 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: the “farm gate” and the “primary processor gate”. The upstream boundary shall extend to the “cradle” – 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*., 2016a; Case study 11).

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, 2016d), which account 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 suited to assess local biodiversity effects, as global data lack 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. On a local scale, biodiversity loss is mainly considered at the farm or field level (Case study 6). The region is a variable area and should be clearly defined (e.g. ecoregion – Olson *et al*., 2001); the results of the impact assessment should be mainly relevant on this scale. Global assessment should consider the global vulnerability of species (e.g. IUCN Red List) and impact should show damage on a world scale.

LCA tends to focus on regional to global assessment and CFs have therefore been developed mainly for these scales (Case study 5). On the contrary, LCA is not well suited for assessing local biodiversity effects due to the lack of sufficiently detailed data on a global scale to support the development of local or site-specific CFs (Case study 13). Examples of models and CFs include De Baan, Alkemade and Koellner (2013) and Maia 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 (SAR), 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 (Food SCP RT, 2013; FAO, 2016e). Livestock and land management affect biodiversity primarily on 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 PSR approaches. For large-scale, regional to global assessments of supply chains, the recommended approach of Chaudhary and Brooks (2018) is explicit to ecoregions. It is based on vulnerability scores derived from IUCN status, and on countryside SAR (Pereira and Daily, 2006) that quantify how species richness responds to changes in native habitat area taking into account the ability of species to live in both altered and natural habitats. 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 should be developed to be used as CFs in life cycle impact assessment. For instance, Nemecek *et al.* (2011) used farm-scale experiments collecting several pressure indicators (e.g. yield, fertilizer and pesticide use, soil quality) to calculate CFs for intensive and extensive production in crops and grasslands.

5.2.4 Description of biodiversity indicators in life cycle impact assessment

Key guidelines

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

Previous LEAP documents reviewed potential biodiversity indicators (FAO, 2016e, 2016f) 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 on a 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, Alkemade and Koellner, 2013; Lenzen *et al.*, 2012). 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 (Scholes and Biggs, 2005; Alkemade *et al.*, 2009, 2013).

Subjective measures of habitat quality include the "naturalness" of land cover and land use classes (e.g. Brentrup *et al.*, 2002; Geyer *et al.*, 2010) 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 production (HANPP) (Haberl *et al.*, 2004, 2005), functional trait diversity (Maia de 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:

- ecosystem scarcity – as a measure of the intrinsic rareness of an ecosystem;
- ecosystem vulnerability – as a measure of the present condition of the structure; and
- conditions for maintained biodiversity – 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 and IES, 2011) and may be more suitable for impacts linked to midpoint indicators.

5.3 LIFE CYCLE INVENTORY

Key guidelines

- *Land use (in m^2 × 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 (Case study 2).

Ideally, these data shall be collected based on Geographic Information System (GIS) methodologies, but data on standardized classification and regionalization of land use may also be utilized (Koellner *et al*., 2013). Spatial differentiation 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, Southeast Asia, Congo Basin) is likely to have a higher impact on global biodiversity compared with 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 on 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 are suggested by Koellner *et al.* (2013):

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

Practitioners shall strive to develop the most detailed possible inventory of land use. Ideally, this means level 3 or level 4 categorization and will involve obtaining data for the appropriate PSR indicators listed in Chapter 6. Practitioners should also be aware of the specific naming conventions used in assigning inventory flows (Koellner *et al.*, 2013), 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 continually 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 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 specified as follows:

- For land occupation: $m^2 \times$ years, land use type i and region k
- For land transformation: m^2, initial land use type i $\rightarrow$ 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

- *The model provided by Chaudhary and Brooks (2018) provides CFs reflecting potential regional or global species extinctions for different taxa; these guidelines recommend using the global taxa-aggregated CFs.*
- *Recommendations in these guidelines are consistent with the UNEP-SETAC recommendations.*
- *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 the interpretation of results, 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 by Chaudhary and Brooks (2018). These CFs were derived using the countryside species–area (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 (Case studies 2 and 5).

Two sets of CFs are available: regional and global. The former represent 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 genetic diversity and "tree of life" (Mace, Gittleman and Purvis, 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 vulnerable species (VS) is based on the proportion of endemic species in an ecoregion and the threat status of species hosted by the region (Case study 2). 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, 2020). 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) geographical area (km^2) coverage of this species, 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 (least concern), 0.4 (near threatened), 0.6 (vulnerable), 0.8 (endangered) and 1 (critically endangered) (IUCN, 2020).

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:

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

where S_{world} is the total specie richness of the taxa and is equal to 5 490 for mammals, 10 104 for birds, 9 084 for reptiles, 6 433 for amphibians and 321 212 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. It is strongly recommended to use ecoregions for processes in the foreground system rather than country averages.

Once the functional unit, location of production and 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 m^2 of light use pastureland in Canada per kg beef; subsection 5.2.1) by the newly available CFs (e.g. 1.31×10^{-14} PDF/m^2 for light use pastureland in Canada), resulting in a final impact of 65.5×10^{-14} PDF per kg beef in the same example. More detailed examples are described in Case studies 2 and 5.

Regional CFs do not include the species vulnerability considerations, so 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 interpretation of results. 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 (Chaudhary and Brooks, 2018, Table S7). The CFs are provided for assessing the impact of both land occupation (PSL/m^2) and land transformation (PSL × year/m^2), or aggregated across taxa as global PDF/m^2 and land transformation in global PDF × year/m^2.

Overall, the recommendations regarding the use of these CFs are consistent with that of the UNEP-SETAC life cycle initiative (Teixeira *et al.*, 2016). This initiative provisionally recommended the use of Chaudhary *et al.* (2015) CFs to assess land use-driven biodiversity impacts within LCA. The UNEP-SETAC recommendations include (UNEP, 2017):

- expanding land use classes and intensities;
- including CFs for plants;
- reducing uncertainty; and
- conducting case studies to test feasibility.

Meanwhile, the use of those CFs should be limited to hotspot analysis. Chaudhary and Brooks (2018) address most of these suggestions for improvement. Further improvements such as better confidence intervals and finer intensity classes could allow to go beyond hotspot analysis 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, Sgardeli and Monokrousos, 2013). For instance, countryside SARs recognize the ability of species to use human-modified habitats but still focus on contiguous habitat and fail to account for the effects of habitat fragmentation and metapopulation dynamics that usually accompany 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, genetic), represents 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 (subsection 5.4.3). Improving the distinction among land use intensity and management practices – including those with a positive impact on biodiversity (e.g. extensive grazing – Watkinson and Ormerod, 2001) – is a priority in order to increase 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 ecoregional (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, microhabitats, biodiversity corridors), despite the fact that they 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, several methods/tools that can be applied for local-level (farm-level) biodiversity impact assessment are listed in subsection 5.4.2.

5.4.2 Regional/local impact assessment

Key guidelines

- *Regional/local impact assessment often requires the adaptation of existing CFs or the development of new CFs.*
- *Data availability for the development of local CFs is a challenge, which can 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 have been collected over the period of interest. Frequently, data of this quality are unavailable for impact assessment (Case study 13). 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 (Case study 5). 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 (Case study 13).

Often such detailed data are not available on 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 (Case studies 1 and 6). Compiling assessments at the farm or local scale could generate an overview of the impact of livestock on biodiversity at a 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 complements the local-scale data sets that are already available within the region of interest.

5.4.3 Reference state

Key guidelines

- *In LCIA, impacts on biodiversity are always expressed in comparison to a reference state.*
- *Potential natural vegetation (PNV), which describes the mature state of vegetation in the absence of human intervention, is often used as a reference state; a historic reference or the current mix of land use can also be chosen.*
- *The main LCIA model recommended herein for global/regional assessments (Chaudhary and Brooks, 2018) uses 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, the 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:

- potential natural vegetation;
- historic land use; and
- current land quality states (Case study 7).

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). PNV 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 can be seen as having only a negative impact on biodiversity, despite the fact that this ecosystem contributes to biodiversity. The use of PNV as a reference state does not prevent it from identifying positive impacts of livestock on biodiversity. However, high resolution data (both spatially and in terms of production practices) would be needed to detect systems where livestock systems achieve biodiversity levels similar to or higher than those in PNV; currently, most LCIA models do not have access to such data. An additional limitation of PNV is that there may be multiple potential equilibria that constitute the mature state of vegetation (Maia de Souza, Teixeira and Ostermann, 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 historical state or a mix of current land uses, as proposed for Europe by Koellner and Scholz (2007, 2008). The use of old historical land cover types as reference states gives similar weights to land use impacts that are currently occurring (e.g. contemporary tropical deforestation) and land use impacts that occurred a long time ago (e.g. deforestation of European woodlands). For instance, such reference sees species-rich grasslands in Europe as deforested areas, and the impact on biodiversity appears to be mostly negative. Alternatively, the selection of recent land use states as the reference state (e.g. land cover in 2000) shifts the emphasis of biodiversity impacts onto contemporary as opposed to historical land use.

De Schryver *et al.* (2010) and De Baan, Alkemade and Koellner (2013) proposed a combination of a current and a semi-natural reference state. De Baan, Alkemade and Koellner (2013) defined a reference situation as the current natural mix of natural land cover (i.e. forestland, wetland, shrubland, grassland, bare area, snow and ice, lakes and rivers). Such an approach results in land use changes having a higher impact in areas of natural land cover. In practice, the reference state should also consider short-term changes in land use and the need to preserve traditional land uses that differ from the PNV, such as agricultural landscape or extensively managed forests. Land cover data exist at the continental to global scale (subsection 8.4.1) and can be used to develop a proxy of the current vegetation state.

The selection of a suitable reference is therefore clearly a priority in discussions over how to make the LCA methodology relevant to the livestock supply chain, particularly when making comparative assessments of products or systems at the global to regional scale. In most cases, using PNV, semi-natural or historic vegetation states will result in livestock having a negative biodiversity impact compared to the reference. However, it can be interesting to compare the impact of different scenarios (using the same reference), showing the relative impact of livestock compared to alternative ways to produce proteins, calories or other micronutrients (White and Hall, 2017).

6. Local assessments using pressure-state-response indicators

6.1 THE FRAMEWORK FOR LOCAL ASSESSMENTS

Key guidelines

When conducting a local biodiversity assessment using the PSR framework, five major steps can be considered: 1) definition of goal; 2) definition of scope; 3) identification of indicators; 4) data collection and analysis; and 5) interpretation and communication – in addition to stakeholder engagement occurring iteratively along the five steps.

Within the five steps, key principles to consider include 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 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, are 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 is 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 and/or 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.*

(cont.)

- *Indicators shall be 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 enable computation of 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, in order to guide decision-making for improving biodiversity performance, discuss limitations and identify issues. Interpretation shall be undertaken by personnel with appropriate expertise and shall 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 for each step are provided in the subsections 6.1.1–6.1.5, while a list of recommended indicators is provided in section 6.2.

6.1.1 Definition of 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

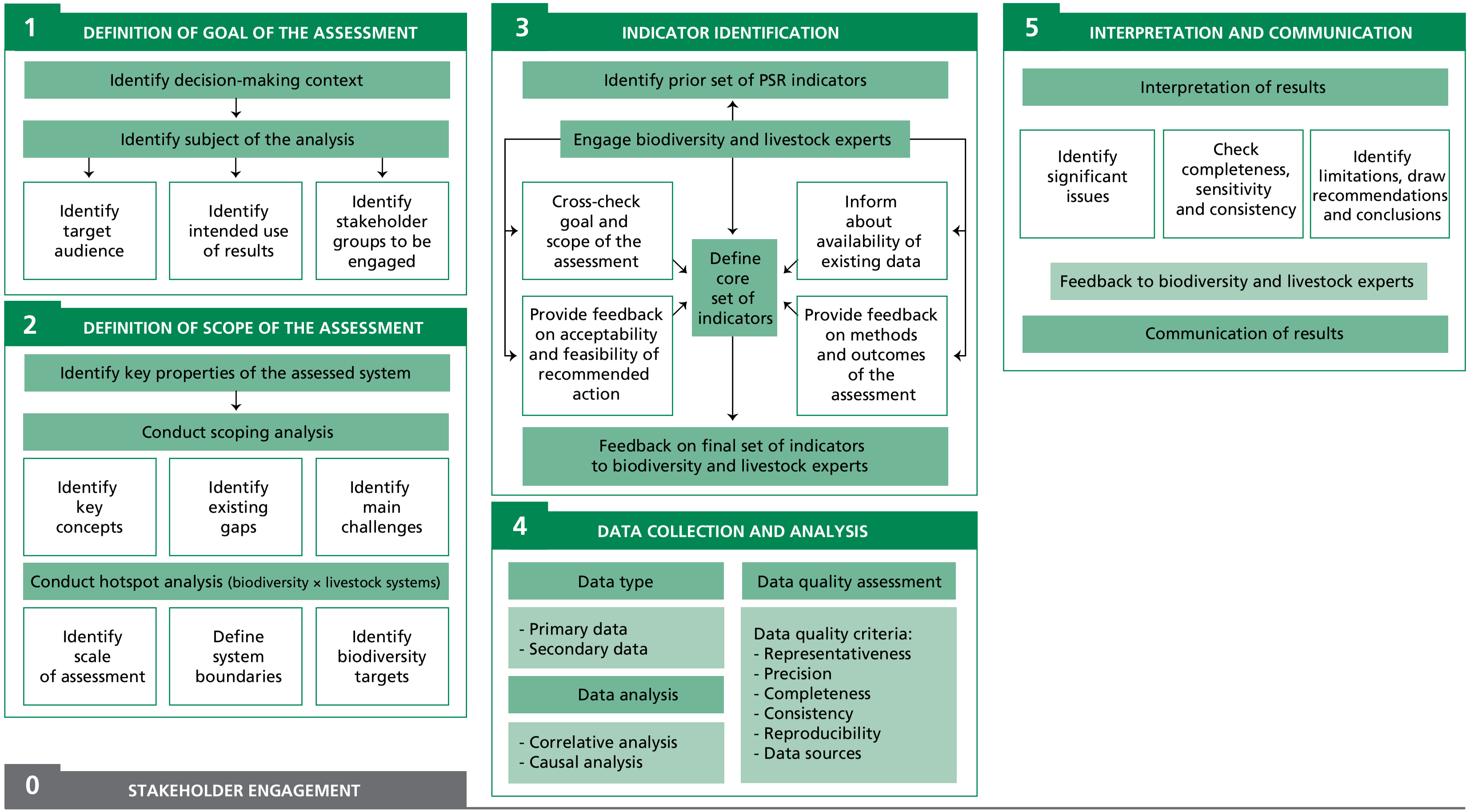
Scoping analysis – It is important to define several aspects of the scope of the assessment:

- Features of biodiversity of concern (e.g. protected habitat loss, habitat degradation and fragmentation, extinction of species, decline in abundance of species, invasive species, aquatic biodiversity, landscape scale conservation – section 6.2).
- Scale – local-scale assessments using a PSR approach 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).
- Inclusion (or not) of the provision of ES.

Mapping or ecological zoning (e.g. biome or agroecological zones, ecological-economic context, pedoclimatic conditions, productive history, hydrological properties, 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. The resulting information 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, and allow for the continual improvement of biodiversity indicators in the system under study. Other information sources to be reviewed as part of the scoping analysis include the scientific

Figure 4

An operational framework for a local biodiversity assessment using PSR indicators



literature, reports from environmental non-governmental organizations (NGOs) – local or international (e.g. WWF, IUCN) – laws and international frameworks. Some countries have agri-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 Biodiversity Targets, (CBD Secretariat, 2018) 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 this case, the scoping analysis shall rely on emphasizing the value of conservation to local stakeholders and on promoting the functional values of biodiversity with an emphasis on its provision of ES.

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, aquatic pollution). The spatial scale shall be clearly identified and off-farm impacts shall be considered. Off-farm impacts occur when local pressures have an impact on biodiversity outside of the user's system (e.g. water pollution, 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 – System boundaries need to be defined to 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, for example, both cow–calf and beef fattening operations, processing and transport. 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 PSR 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 (e.g. habitat change, pollution, climate change) (Case study 7). An overview of the categories of pressures and benefits on biodiversity derived from livestock can be found in Appendix 4 and more details are provided in the LEAP biodiversity review (Teillard *et al*., 2016b). 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 (subsection 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 (Case study 9).

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 (subsection 6.1.6). Essential biodiversity variables (EVB) are state indicators needed to study, report and manage biodiversity change. These variables are divided into six EBV classes containing 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 is often already available. Response indicators shall be used as an indication of mitigation actions, that is 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 the user to define response actions targeted towards environmental changes such as biodiversity loss. Indeed, it allows the 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 prioritize and catalyse action (Plantureux *et al*., 2014).

Identifying and prioritizing indicators – The identification of indicators will be strongly guided by the choice of assessment goals and by 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, achievable, relevant and time-bound), economically feasible to measure and accepted by stakeholders (Case study 10). The engagement with stakeholders and experts (subsection 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 both positive and negative effects of livestock on biodiversity. For example, if there is an internationally rare species or habitat(s) in a catchment or region, the measurement of livestock effects on these should be a priority compared with the 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 PSR 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 requirements; users should consider each of the indicators in turn and provide a short justification for 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 to support management decision-making. In this case, the multiple indicator list would include:

- state indicators: the abundance of species with high conservation value, the percentage of semi-natural habitats;
- pressure indicators: the rate of conversion of semi-natural habitats, livestock density on grazed semi-natural habitats and action-driven indicators;
- response indicators: 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, written justification must be provided (e.g. in a stakeholder analysis – subsection 6.1.6) explaining 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, 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 measurement of indicators should be economically feasible, 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 percent 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 (Case study 12). They may 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 – Given the inherent complexity of biodiversity and due to the need for 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 [SOC] content [FAO, 2020b], land use impact on biodiversity at the ecoregion level [Chaudhary *et al.*, 2015]) or site-specific values (e.g. SOC 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 reduces 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 organization 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 randomized 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 and the relevant data and information will have to be identified, 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 subsection 8.4.1 and additional guidance on data collection is provided in subsection 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 section 8.3 (precision, error, completeness, consistency, reproducibility and uncertainty).

Two types of data can be collected to compute PSR indicators:

- **Primary data** – defined as directly measured or collected data representative of the livestock operation at a specific facility (pressure and response indicators) or of local biodiversity in a specific area (state indicators).
- **Secondary data** – defined as information obtained from sources other than direct measurement. Note that secondary data are used when primary data are not available or it is impractical to obtain them. 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; 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; for this reason, 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 Chapter 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 programme. The indicators selected are monitored with the aim of showing a trend over time in a time series analysis (e.g. 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 may be used to assume possible cause–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 aim of assessing the relationship to the change of livestock density in the same region. For this purpose, secondary data on livestock density may be used. The analysis will be correlative and show covariation.

- **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 the contribution of livestock density 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; the selected practice should then 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 bio-statistician from the beginning of a biodiversity monitoring programme 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 Chapter 7.

6.1.6 Stakeholder engagement

The role of stakeholders may include, but is not limited to:

- contributing to more effective goal definition (subsection 6.1.1);
- improving awareness of traditional knowledge and practices about biodiversity;
- contributing to the selection of indicators;
- informing about the availability of existing knowledge and data;
- providing feedback on the goal, methods and outcomes of an assessment; and
- 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 stakeholder analysis (Brugha and Varvasovszky, 2000) that recognizes the existence of multiple perspectives and provides a structured framework for capturing the various requirements of different stakeholders. Stakeholder analysis 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 specialized 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 continual, 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 and adopted.

6.2 RECOMMENDED LIST OF BIODIVERSITY INDICATORS FOR LOCAL ASSESSMENTS

Key guidelines

- *This section provides a list of recommended pressure-state-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 for why an indicator is selected or not, or why an alternative indicator is used.*

As 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 interlinkages 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 (together 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). Farmland 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; Case study 7). Livestock production has an important contribution to this driver as it is estimated that 30 percent of the Earth's land surface is dedicated to livestock production through pastures (≈ 25%) and feed crops (≈ 5%) (Ramankutty *et al*., 2008; Monfreda, Ramankutty and Foley, 2008). Livestock therefore affect some 30 percent 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 (e.g. forest, grassland) to another (e.g. grassland, cropland – Case studies 8 and 11). 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 habitats. Information is a key factor to achieve this protection. The species and habitats under the direct influence of the land managers need to be identified, mapped and monitored (Case studies 1 and 12). This information can be used to establish a biodiversity action plan where detrimental practices are avoided and practices protecting or promoting wildlife species are adopted.
- **Invasive alien species.** Invasive alien species are defined by the CBD as species whose introduction and/or spread threaten biological diversity. They are a major threat to biodiversity on a global scale. As well as other vertebrates, livestock contribute to the seed dispersal of invasive and native plant species (Rejmanek *et al*., 2005). Species invasion is a complex phenomenon influenced by a wide range of factors. The introduction of an alien species is a common starting point, but whether invasive species are a cause or a consequence of ecosystem degradation is often unclear (MacDougall and Turkington, 2005; White *et al*., 2013). Excessive and harmful population increase in native species (resulting in, for example, bush encroachment in rangelands) is a different process and comes under the "habitat change" (degradation) category.
- **Pollution and aquatic biodiversity.** Livestock production is responsible for two main types of pollution having, in turn, negative impacts on biodiversity:

nutrient pollution and ecotoxic pollution. Nutrient pollution can be caused by fertilization at the feed production stage; however, it is often most important at the farm stage. As nutrient capture by animals is quite inefficient, a large amount of nutrients are concentrated in urine and manure. With improper management practices, excess nutrients can enter soils and surface water where they cause eutrophication (i.e. the growth of nuisance species of algae and aquatic weeds harmful to other native freshwater species). The LEAP nutrient guidelines describe how to account for the nutrient flows and associated environmental impacts in livestock supply chains (FAO, 2018d). At the feed production stage, ecotoxic pollution is caused by pesticides. Hormonally active pesticides have adverse effects on a wide range of organisms (Colborn, vom Saal and Soto, 1993). Ecotoxic substances may also be used at the animal production stage in the form of veterinary products, antibiotics, anthelmintics and hormones; these can contaminate water and impact aquatic biodiversity.

- **Off-farm feed.** As stressed in the LEAP biodiversity principles, the impacts of a livestock farm on biodiversity do not only concern on-farm wildlife species under the direct influence of the farmer. Agricultural supply chains are increasingly globalized, with production sites connected by complex international trade routes. For instance, in 2011, 58 million tonnes of soybean meal were exported by 86 countries and imported by 114 countries to feed livestock (FAO, 2013). An important share of this meal is produced in areas that were previously Amazonian savannahs or rainforests, that is biodiversity hotspots. Natural ecosystems are usually converted first to cattle pastures; this land can then be sold for soybean production, with a risk of profits being reinvested in buying and clearing forests in other areas (Gollnow and Lakes, 2014). In some cases, the off-farm impacts on biodiversity associated with imported feed can be more important than on-farm impacts (Teillard *et al*., 2016a). Such off-farm impacts should always be included in biodiversity assessments of livestock production, except when use of off-farm feeds is negligible.
- **Landscape-scale conservation.** When livestock cause habitat destruction, negative effects on biodiversity are often worsened by fragmentation, because a given area of original habitat fragmented into small and distant patches will sustain fewer species than a single, continuous patch of the same area. Conversely, if patches of original habitat are large and in proximity to one another, connecting them with wildlife corridors provides a conservation opportunity. By reducing the possibilities for organism mobility, fragmentation impacts gene exchanges as well as biodiversity at the species and ecosystem levels.

Table 2 indicates: the category of each indicator – pressure (P), state (S) or response (R); whether measures are quantitative or qualitative; the stage of livestock production (animal husbandry, feed production or both) that can be targeted by the indicator; and whether or not there is a direct link to ecosystem services.

Table 2: Overview of possible indicators and their characteristics for use in biodiversity assessments

Thematic issues Indicators *Examples of measures (when several are possible)*[1]	Category[2]	Qualitative (Ql) or quantitative (Qt)	Relevance to feed production (F) and animal husbandry (A)	Strong link to ecosystem services
Procedural checks				
Scoping analysis conducted	R	Ql		
Regulatory constraints considered *Regulatory constraints include:* • *International frameworks (e.g. international biodiversity hotspot, WWF ecoregions with outstanding biodiversity features, IUCN Red List species).* • *National regulations (e.g. protected areas and species). If national regulations are not met, their perverse or ill-informed nature should be justified.*	R	Ql		
Extrinsic use value of biodiversity considered *The extrinsic use value should be defined through the involvement of local stakeholders.*				
Progress monitored	R	Ql		
Stakeholder engagement *Stakeholder analysis* *Iterative stakeholder engagement*	R	Ql		
Data quality *Refer to section 8.3*				
Habitat protection				
Wildlife habitats under farm influence inventoried (mapped) and protected (Case study 1)	R	Ql/Qt	A/F	Yes
Semi-natural habitats in the landscape *Area or proportion (relative to the area controlled by the user)*	P	Qt	A/F	Yes
Grassland restoration *Area of degraded grassland restored through improved grazing management*	R	Qt	A/F	Yes
Habitat change				
Soil erosion and soil erosion risk mapped and management plan implemented *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).*	R	Ql/Qt	A/F	Yes
Degraded soil *Area or proportion (relative to the area controlled by the user) of degraded soil, including bare soil or areas with bush encroachment* *Soil organic matter content* *Modelling of carbon, nitrogen and phosphorus cycles*	P	Qt	A/F	Yes
Livestock density *Livestock density in number of animals or other livestock units (e.g. tropical livestock units) per ha* *Where relevant (e.g. in more humid grazing lands with 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).*	P	Qt	A	Yes
Habitat conversion *Area or rate of conversion of natural and semi-natural habitats*	P	Qt	A/F	Yes

(cont.)

Thematic issues **Indicators** *Examples of measures (when several are possible)*[1]	Category[2]	Qualitative (Ql) or quantitative (Qt)	Relevance to feed production (F) and animal husbandry (A)	Strong link to ecosystem services
Wildlife conservation				
Priority actions promoting species with high conservation value listed and implemented *High conservation value includes national and international designations, but also the functional role of the species and the perspective of local stakeholders.*	R	Ql	A/F	Yes (depending on the species)
Particular species (with high conservation value) *Presence/absence, abundance and/or distribution*	S	Qt	A/F	Yes (depending on the species)
Species richness or diversity *Number of species* *Shannon or Simpson diversity index* *Functional diversity, Trophic index*	S	Qt		Yes
Invasive alien species				
Management plan in place for the control of invasive species	R	Ql	A/F	Yes
Invasive alien species *Presence/absence, abundance and/or distribution*	P	Ql/Qt	A/F	Yes
Pollution and aquatic biodiversity				
Management plan in place for the application of ecotoxic agrochemicals *Pesticides, veterinary products*	R	Ql	A/F	Yes
Nutrient management plan in place to rationalize fertilizer application	R	Ql	A/F	Yes
Protected waterways *Length or proportion (relative to length controlled by the user, or to the length in need of protection)*	R	Qt	A/F	Yes
Biological indicators of water quality	S	Qt	A/F	Yes
Off-farm feed				
Inventory of the off-farm feed being used established	R	Ql/Qt	F	
Traceability systems for feedstuff implemented	R	Ql	F	
Share of imported feed *Share of imported feed from areas that are certified/deforested/ of high conservation value*	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 *Spatial Shannon diversity index* *Landscape diversity conservation index, Brillouin index* *Connectivity measures* *Average area and distance between patches of habitats*	R	Qt	A/F	Yes

[1] Themes used to group indicators are in bold; indicators are in normal type; metrics that can be used for each indicator are in italics.

[2] Category of each indicator – pressure (P), state (S) or response (R).

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 policymaking contexts.* • *Interpretation should be aligned with the goal and scope of the assessment.* • *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, 2006b). Within the LCA framework, there are clear guidelines for the interpretation of results:

- 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 policymaking 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. High uncertainty levels could also lead to the revision of the goal of the assessment, moving, for example, from a quantitative assessment to a qualitative evaluation of issues to be considered or overcome.

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–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 with regard to 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 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 with regard to the aims and methods of an assessment and should include: methods chosen, outcomes, and action plans following the assessment, as well as any limitations related to the assessment or information. In particular, information should be communicated in a clear and understandable form, and 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. Second, 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 of policymakers assuming 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 imperative to model impacts on 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. It is important to appreciate that specific indicators for one biodiversity level or dimension (e.g. species composition) are not fully adequate to depict linkages between ecosystem function, biodiversity and ecosystem services.*

With continued global biodiversity loss, there is a strong societal demand to measure the environmental impacts of livestock production on the global, regional and local scales and devise strategies to address these effects. The ecological footprint is an easy-to-grasp concept, rooted deeply in popular culture, and it is gaining increasing prominence in the scientific literature (Wiedmann and Barrett, 2010; Hoekstra and Wiedmann, 2014). However, given that it is limited to land use analysis and is difficult to extrapolate to implications for other ES, its usefulness in terms of setting policy may be limited (Kovacic and Giampietro, 2015).

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 (Čuček, Klemeš and Kravanja, 2012). Consequently, LCA has emerged in the regional and global scenarios as a key element in assessing potential environmental impacts of products and services to support decision-making at the industry and government levels (Hellweg and Milà i Canals, 2014). LCA represents an opportunity to provide a transparent comparative analysis of the effects of livestock production across a range of production systems and environmental conditions. This exercise can point out key hotspots in the supply chain and identify strategies for improvement (e.g. preservation and proper management of grassland). Biodiversity assessments can be coupled with the analysis of other social and economic attributes that consider animal health and welfare and other economic performance indicators of the production system (Maia de Souza *et al*., 2013). These assessments are well aligned with the increasing need for the livestock industry to provide transparent information to the consumer regarding its efforts to continually improve production standards and meet sustainability goals.

However, for the last two decades, the LCA scientific community has been raising the alarm about the danger of failing to grasp the analytical complexity when attempting to use LCA to generate policy (Bras-Klapwijk, 1998; De Benedetto and Klemeš, 2009; Wardenaar *et al*., 2012). Often policymakers make the mistake of assuming that LCA generates simple answers to complex environmental questions.

There is a clear danger of oversimplification of messages and derived decisions. This is particularly true for non-climatic impacts such as biodiversity and ES where model complexity is increased across temporal and spatial scales and the development of robust models remains a challenge (Chaplin-Kramer *et al*., 2017).

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

Decision-making in relation to food systems is greatly affected by social factors, which are central in the wider biodiversity-related land sharing versus land sparing debate (Fischer *et al*., 2014) – freeing land for conservation uses through agricultural intensification can potentially have a direct negative outcome on food sovereignty because of increased capital needs. Interactions are complex, with family farming systems apparently yielding more food than industrial ones (Story *et al*., 2016), possibly because of their multifunctional design (Altieri, Funes-Monzote and Petersen, 2012). In the concrete case of livestock, the picture is further complicated because of the proven beneficial effects that some low-yielding production types can have by providing key ecosystem functions (Teillard *et al*., 2016a; Manzano-Baena and Salguero-Herrera, 2018), but it may depend on the ecosystem and the type of grazing provided (e.g. functional similarity to wild herbivores) (Bond, Lee and Craine, 2004; Bond and Silander, 2007).

There is considerable debate about the promotion of intensified livestock systems (exhibiting high levels of production, but greater impacts on biodiversity) over extensive livestock production systems (producing less livestock meat and milk, but with less serious environmental impacts). Extensive systems generally need to occupy more land area to produce the same amount of livestock product. The optimal system is likely to involve a trade-off between the two extremes, tailored to local environmental conditions and ensuring the prudent use of available natural resources (Garnett *et al*., 2013), including integrated (e.g. crop–livestock, silvopastoral) systems.

Assessing these impacts and promoting the livestock sector's environmental improvements is important in order to reach the Sustainable Development Goals (SDGs) in local and regional economies, in particular in developing countries, where livestock contributes to approximately 40 percent of the agricultural gross domestic product (GDP) (FAO, 2018b). At a global scale, compliance with the SDGs, in particular SDG 12 (responsible consumption and production), SDG 13 (climate action) and SDG 15 (life on land), has required information on the environmental performance of economic activities, including livestock production worldwide.

Furthermore, the application of LCA to identify scenarios of further development and/or intensification provides additional information for policy decision-making on different scales, ranging from the local (e.g. regional, watershed) to the global (e.g. national) levels.

A policy scenario approach can highlight the importance of livestock systems, in terms of both negative and positive impacts. However, this tends to be refuted by the current state of opinion, with widespread attacks directed at extensive livestock systems – flagged as emission-intensive, despite the fact that other environmental benefits, including biodiversity conservation and the provision of certain ES, in part compensate for higher emissions (Ripoll-Bosch *et al*., 2013).

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

8. Data and data quality

8.1 INTRODUCTION

Key guidelines
• *Biodiversity data should be aligned with the scale at which the analysis is to be conducted, when relevant, and/or be scalable to enable cross-scale analyses.* • *When using data on a large geographical scale, the risk of simplification, lack of specificity and failure to consider all aspects and interactions should be minimized.* • *When using data on a small geographical scale, the risk of lacking representativeness and of overgeneralizing 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). Measuring the impacts of livestock production decisions on biodiversity is challenging, particularly when moving across scales (Levin, 1992; Poiani *et al.*, 2000) (Figure 5, Case studies 10 and 12). 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 failure to consider important ecological factors, ecosystem processes and functions and species interactions shaping wild species communities (Bunnell and Huggard, 1999). For example, most data available for large-scale analyses are based on simplistic information (e.g. 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, more detailed data available for small-scale analyses would require an enormous effort to generate enough information to be able to extrapolate and have a more comprehensive view of the issues at larger geographical scales (Case study 13). 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.

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:

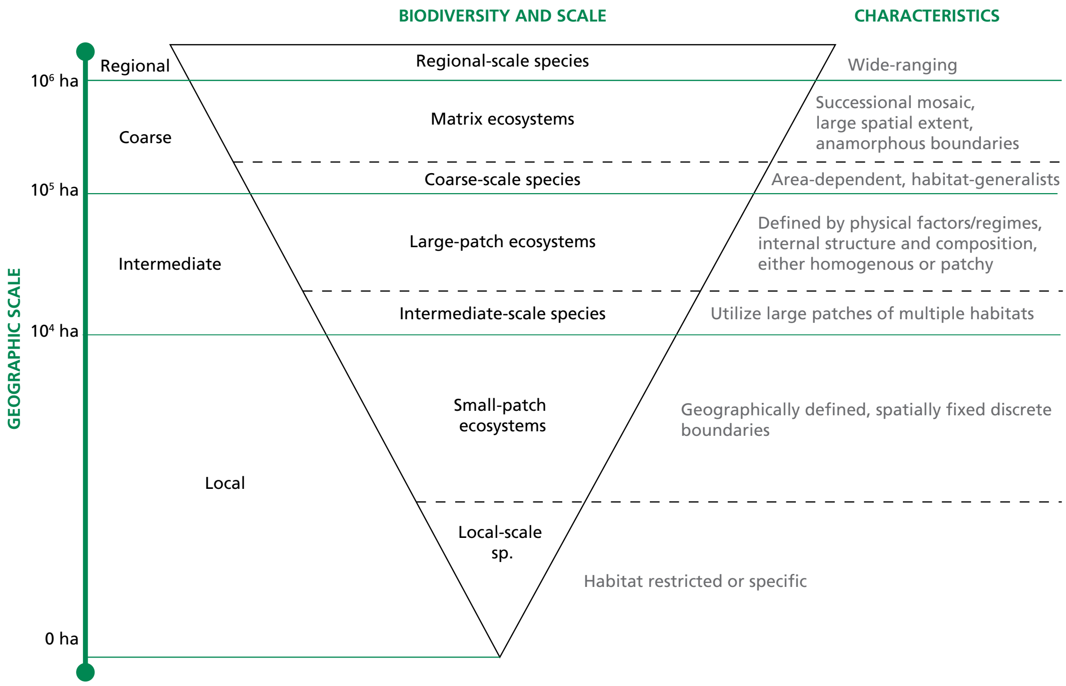
i) **Temporal** – age of the data and the length of time over which they were collected and the ability of the data to describe the system or to capture a specific event or change of interest.
ii) **Spatial** – geographical area from which data for unit processes were collected to satisfy study goals.
iii) **Taxonomic** – taxonomic breadth deemed appropriate to the scope of the assessment.

Representativeness should be considered in the study design when collecting primary data, where the data are 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 collected by others.

When determining the fitness for purpose of available data, both spatial and temporal representativeness need to be considered. For instance, sampling of a natural population seeks to provide a representative assessment of that population. If the data fail to represent the population, the analysis may generate a biased outcome that will impact the level of confidence in any interpretation, directly affecting the relevance of the results for decision-making purposes. Hence, confidence in the interpretation of data requires knowledge of the underlying data quality, including how well the data sample represents the underlying natural population. Furthermore, depending on the objective of the study, data age and temporal spread could be as important as spatial coverage. In general, data sources for temporal aspects of biodiversity (e.g. trends in state over time) are scarcer than spatial data for assessments of state; this is a reflection of the general lack of long-term monitoring programmes. However, long-term data with regular monitoring (daily, monthly, annual, biannual, decadal etc., depending on the variable and expected rate of change) are often essential to assess biodiversity responses to environmental change, whether as a result of land use or of climate change, and to define reference states that are more meaningful for management and representative of the desired state. "Surrogate" options for the reference state (e.g. pre-human state) often provide little direction, especially if a change in current human activities for the benefit of biodiversity is the desired outcome.

In relation to taxonomic representativeness, if the defined scope of an investigation encompasses a broad taxonomic inventory, data representativeness will be weakened if data sources only cover certain taxonomic groups. For example, an assessment of Coleoptera may provide a relatively poor sample if the project scope requires an inventory of all Insecta. Within the scientific literature, there is a marked

Figure 5
Biodiversity at various spatial scales



Source: Poiani *et al.* (2000).

bias in taxonomic representativeness, with vertebrates being over-represented and invertebrates under-represented in many global data sources and in global assessments of extinction risk (Newbold *et al.*, 2015; Proença *et al.*, 2017).

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

A number of studies have identified spatial and temporal biases arising from non-representative biodiversity data sources. For example, biodiversity data richness and time series availability are frequently strongly skewed at global (Collen *et al.*, 2008; Boakes *et al.*, 2010; Proença *et al.*, 2017) and regional (biome) scales (Martin, Blossey and Ellis, 2012). In general, assessments at global scale show a strong "temperate" bias, particularly in the Northern hemisphere. This affects the representativeness of both species richness and time series data sets for global assessments.

At the regional level (biome/anthrome), there are also clear biases in the availability of published data, with under-representation of modified habitats in biodiversity assessments. For example, ecologists tend to work in relatively unmodified habitats, even though these comprise a relatively small percentage of the landscape (Martin, Blossey and Ellis, 2012). The latitudinal biases seen at the global scale also have significant consequences for data representativeness at the regional scale. In mid-latitudes, available monitoring data to inform regional assessments are scarce and greater investment is required to fill gaps in data 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 data sets 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 data sets 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, the known bias can be controlled for in modelling (Newbold *et al.*, 2015).
- More effort can be made to obtain data (or digitize existing records) that are more spatially representative (Feeley and Silman, 2011).
- Available data sets can be subsampled 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 vs changes in abundance).
- Data can be captured on coarser scales using methodologies that are less sensitive to spatial bias (e.g. remote sensing vs field surveys).

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

8.3 DATA QUALITY ASSESSMENT

Key guidelines

- *Data quality should be assessed by authoritative organizations (e.g. government, local agencies, research organizations, specialized NGOs), reported and discussed.*
- *Data quality assessment should include several key criteria – precision, error, completeness, consistency, reproducibility and uncertainty.*
- *Databases supporting biodiversity assessment in livestock should ideally be on open access.*

As part of any biodiversity assessment, data quality should also be assessed, reported, and its potential impact on results discussed. Data collection on biodiversity requires specific expertise and should be conducted by competent individuals (taxonomic identification in particular). Therefore, authoritative validation of data quality is of particular importance to ensure reliability. Data quality assessment should consider the six criteria described in the subsections 8.3.1 to 8.3.6. In addition, it is strongly recommended that databases supporting biodiversity assessments in livestock are made publically available. This enhances the credibility of the results, strengthens 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-based public policies for sustainable rural development.

8.3.1 Precision

Precision: a measure of the data's variability for each 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:

- Error – including error in taxonomic identification and spatial error.
- Bias – primarily the geographical and environmental biases associated with ad hoc data collection.
- Presence only versus presence–absence data – influencing the type of modelling 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 do 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 in a historic local collection, or sampling locations that might represent a demographic sink for the species. Contrary to presence data, absence records do not necessarily inform species absence at a certain location and time. Absence might indicate that a particular species was truly absent at a site or could signify a failure to detect the species. In the latter case, occupancy models (MacKenzie *et al.*, 2002) can be used to account for imperfect detection of organisms and to correct for false zeros (false absences). They determine the probability of the true presence of a species at a site based on detection probability, which is modelled through generalized linear mixed effects models where characteristics of the sites are the fixed effects while random effects reflect the true state of occurrence. These types of statistical models are extremely useful when working in regions where cryptic species are present, but usually recorded as absent.

For modelling 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 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

Error: a measure of the estimated difference between the observed or calculated value of a quantity and its true value.

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 on distribution records that are suspect because they exist in a different geographical or environmental space from 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 a record location 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: 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 empirically (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 geographical coordinates or with 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) at which certain 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 (i.e. groups of individuals of the same species), data will reflect the presence or absence of a particular species, as well as their abundance and trends in population numbers. When dealing with communities (i.e. 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 (e.g. 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, Scarsbrook and Dolédec, 1997). Regardless of the metric selected, data need to be representative of the spatial scale the population or community of interest inhabits (Case study 13).

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 (subsection 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, Marquet and Brown, 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:

- observations;
- field experiments;
- laboratory experiments; and
- (mathematical and statistical) models.

The optimal source depends on the nature of the questions being addressed. For example, a temporal census of a specific population of *Puma concolor* may be able to identify fluctuations in the population of this species (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-based models are incapable of replicating the complexity of natural ecosystems. Finally, mathematical models can be used to simulate population or ecosystem dynamics and to predict influences on biodiversity. When combined with time series, modelling and other techniques, ecological modelling can prove useful for predicting species distributions and population dynamics (Case study 13).

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 and Reif, 2016; Peralta, Frost and Didham, 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.

Reproducibility is an essential goal in order to generate adequate comparisons and make use of data. Scientific rigour is largely based 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 (i.e. seasonality and periodicity), where (i.e. spatial scale and representativeness of the study object) and how (i.e. source type and quality) the data were collected, taking into consideration the relevance of the data to the scope, breadth and depth of the inventory that is being populated.

8.3.6 Uncertainty

Uncertainty: the degree to which data are inaccurate, imprecise, untrusted and unknown.

Data uncertainty arises 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 rely 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 need to meet at least three criteria in order to increase accuracy and precision and reduce uncertainty (Townsend, Scarsbrook and Dolédec, 1997):

i) The estimate should be accurate and unbiased, meaning that it is neither systematically too high nor too low as a result of flaws in the previous steps that led to the estimate.
ii) The estimate should have narrow confidence limits being as precise as possible (significant differences are usually achieved when data are more consistent and where the number of samples is high).
iii) The time, money and human effort invested in the programme (e.g. inventory) should be used as effectively and efficiently as possible.

To reduce uncertainty in field studies, representativeness of the samples needs to be maximized and biases avoided. One way to do this is by using stratified random sampling where sampling sites are divided into equal parcels and then a number of random samples are taken from each parcel; in this way, the coverage of the field is greater, accounting for more variability, and biases are minimized.

Although uncertainty needs to be minimized, its existence also needs to be acknowledged and discussed (Case study 13), especially in decision-making processes that involve multiple stakeholders.

Table 3: Different sources of data depending on scale of assessment

Spatial scale	Impacts	Sources of data / methods
Global level	Impacts of global issues (e.g. climate change) and issues related to aggregate impacts of human resource use on the planet	• Global and regional databases and models • Peer-reviewed articles and technical reports • Global and regional maps • Remote sensing-derived information
Regional level (agroecological zones)	Impacts of regional issues (e.g. deforestation, desertification)	• Databases / specific capture models • Peer-reviewed articles and technical reports • Local value chains (e.g. cooperatives, collectors, primary processors) • Remote sensing-derived information
Landscape, farm and field level (Case study 6)	Impacts such as habitat fragmentation and loss of locally endemic species.	• Direct data (primary) • Use of detailed calibrated and validated model (if direct measurements are not possible) • Interviews • Remote sensing-derived information (Case study 1).

8.4 EXISTING DATA SOURCES

Key guidelines

- *This section provides 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 data sets 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 (e.g. dissertations, monographs, reports). Even when such data are published, there are limited opportunities for use to improve public policies, as they are often temporally dependent. The relevance of data sets to assess biodiversity depends on the scale of assessment used to measure the impacts of livestock production (Table 3).

Data management plans that consider data publication, long-term curation and the generation of metadata are seen as increasingly important steps in research projects, with the generation of such data sets being an important research output. 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 holders' 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 and ensure data quality standards and the sustainability of public databases (Costello *et al.*, 2013; Costello and Wieczorek, 2014).

8.4.1 Global and regional sources

Global and regional data sets 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 certain ES, especially regulating services (e.g. carbon storage and sequestration). However, while there are several major land cover–land use mapping initiatives (Table 4), the respective mapping products are better able to inform aspects of vegetation and habitat structure and phenology metrics that are essential for assessing livestock impacts (see Appendix 7 for 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 a regional or local scale. National statistics may constitute a useful source of data for pressure indicators, but also for provisioning ES (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 (e.g. averaging, interpolation or modelling to cover data gaps), as in the case of remotely sensed vegetation indexes.

Moreover, global and regional data sets 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). Multiple sources are 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, data collection design), affecting precision and reporting accuracy. For instance, national statistics on forest cover change can be affected by a country's monitoring capacity and even by the definition of what constitutes a 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 following 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 data sets are their:

- spatial and temporal extent (i.e. the size of the area or the time range in which the data are distributed);
- spatial and temporal resolution (i.e. the minimum spatial or temporal unit used to measure the variable of interest); and
- spatial and temporal coverage including 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, data sets may have full spatial coverage, but coarse resolution (e.g. national statistics in the FAOSTAT database – Table 4), or incomplete spatial coverage, but deliver data collected at the local level (e.g. PREDICTS database – Table 4). Remotely sensed data are an exception: they 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 they can be used in smaller or larger spatial extents by directly assembling or disassembling data sets (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 homogeneous 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 (Araújo *et al*., 2005; Sánchez-Ruiz *et al*., 2014; Hoskins *et al*., 2016). Likewise, data may need to be upscaled to a lower resolution to reduce data complexity or to be combined with data on coarser scales (Dalgaard *et al*., 2011; Marcer *et al*., 2012). In the case of categorical data (e.g. land cover), changes in spatial resolution may be associated with changes in thematic resolution (e.g. resolution of land cover classes).

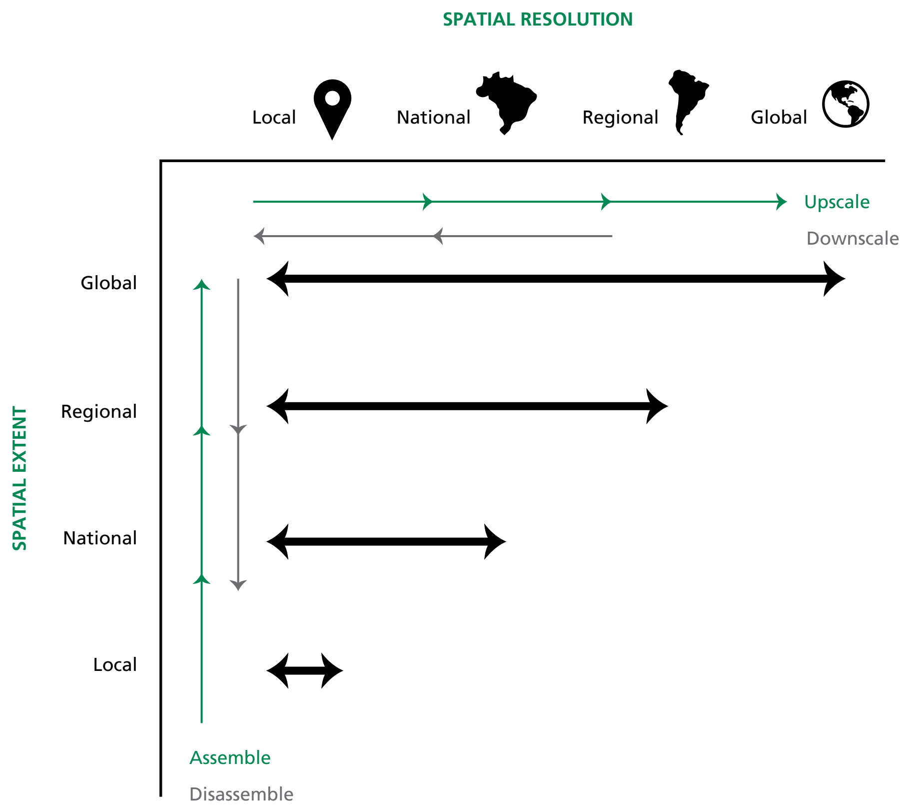
More generally, the use of different data products in assessments and modelling approaches requires data to share the same spatial extent and resolution. This may be achieved by assembling or disassembling data to adjust to the required spatial extent and through upscaling or downscaling methods to adjust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

8.4.2 Local sources

Biodiversity can be measured on 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 a landscape. This kind of diversity is very sensitive to how habitats are delimited (e.g. nearest neighbour) 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 geographical 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 on a landscape scale (Case study 6).

Figure 6
Spatial scale of data sources defined by their extent and resolution



Note: A change in the scale of analysis requires a change in the resolution or the extent, or both (Farina, 2006). Changes in resolution require upscaling or downscaling approaches, because small-scale heterogeneity may not be detected with a coarse resolution while general patterns at larger scale may not be detected using fine resolution. 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.

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 (FAO, 2019c) and nutrient use (FAO, 2018d). Farm-level data on biodiversity have received far less attention, hence the need for these guidelines. With an increasing focus on biodiversity and ES as "integrators" of a diverse array of human pressures, the availability of local, producer-level data is likely to increase significantly.

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

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

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

Performance benchmarking of individual producers is becoming an important generator of local data. For many pressure indicators (e.g. nutrient inputs, GHG emissions, water use), individual performance can be readily compared against standards or peer group performance norms. In contrast, biodiversity assessment is highly context dependent and the point of reference against which to measure the biodiversity state on local scales is more problematic (Case study 9). An "undisturbed" reference condition may not be an appropriate comparison for biodiversity condition, as livestock systems often involve fundamental shifts in land cover (e.g. vegetation) and the opportunities to avoid or remedy biodiversity impacts associated with this change in land cover are often limited. An alternative approach is to identify the "best available" exemplars of a livestock production system on a relevant local scale as a more realistic point of comparison. Identifying the "best practice" associated with these exemplars then provides a more complete pressure-state-response model, as producers gain clarity on where they sit relative to realistic expectations and they have a greater understanding of what changes are required to achieve it.

Table 4: Examples of global and regional data sources

Data source/product	Type of data	Spatial extent	Spatial resolution	Spatial coverage	Time range	Temporal resolution
Global Biodiversity Information Facility (GBIF)[1]	Species occurrences	Global	Local	High to low (depending on region and taxa)	Historical data to present	High to low (daily to scattered records)
PREDICTS[2]	Species abundance (in relation to land use)	Global	Local	> 26 000 sites (> 75 countries)	1997–present	Depends on studies
Global Invasive Species Database (GISD)[3]	Invasive species occurrences	Global	National	Depends on data available (from different sources)	Started in 2000	Data continually added
European Alien Species Information Network (EASIN)[4]	Invasive species occurrences	Europe	10-km grid cell	Depends on data available (from different sources)	Started in 2012	Data continually added
IUCN Red List of Threatened Species[5]	Species conservation status	Global	Summaries by country; Red List status assessed globally	Depends on data available per species	(1964)–1986–present	At least twice a year
IUCN Red List of Ecosystems[5]	Ecosystem conservation status	Global	Summaries by country	Depends on countries having done the evaluation	2013–present	Depends on studies
Plant trait database (TRY)[6]	Plant functional traits	Global	Local (field studies)	Depends on data available per species	n.a.	n.a.
Global Forest Change (GFC)[7]	Forest loss and change	Global	30-m pixel	Full extent	2000–2014	Annual
Landsat[8]	NDVI[15]	Global	30-m pixel	Full extent	1982–present	16-day resolution
GlobCover[9]	Land cover	Global	300-m pixel	Full extent	n.a.	12/2004–06/2006; 01–12/2009
Corine Land Cover (CLC)[10]	Land cover	Europe	25 ha (MMU)[16]	Full extent	1990–2012	1990, 2000, 2006, 2012
Prototype LC map Africa[11]	Land cover	Africa	20 m pixel	Full extent	n.a.	12/2015–12/2016
World Database Protected Areas (WDPA)[12]	Protected areas	Global	Local (protected area)	Full extent	n.a.	Monthly
FAOSTAT[13]	Grassland emissions	Global	National	Depends on reporting country	1990–2014 (present)	Annual
FAOSTAT[13]	Pesticide use	Global	National	Depends on reporting country	1990–2014 (present)	Annual
EUROSTAT[14]	Irrigated area	Europe	NUTS 2 (subnational)	98% of the UAA[17] and the livestock of each country	2003–2013 (EU 27)	Census every 10 years

1 GBIF: https://www.gbif.org/
2 PREDICTS: http://www.predicts.org.uk/
3 GISD: http://www.iucngisd.org/gisd/
4 EASIN: https://easin.jrc.ec.europa.eu/
5 IUCN Red List: http://www.iucnredlist.org/
6 TRY: https://www.try-db.org/
7 GFC: https://earthenginepartners.appspot.com/science-2013-global-forest
8 Landsat: http://landsat.usgs.gov
9 GlobCover: http://due.esrin.esa.int/page_globcover.php
10 CLC: https://land.copernicus.eu/pan-european/corine-land-cover
11 Prototype LC map Africa: http://2016africalandcover20m.esrin.esa.int/
12 WDPA: https://www.protectedplanet.net/
13 FAOSTAT: http://www.fao.org/faostat/
14 EUROSTAT: http://ec.europa.eu/eurostat
15 NDVI – Normalized Difference Vegetation Index
16 MMU – Minimum mapping unit
17 UAA – Utilized agricultural area

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10. Glossary

Agro ecological zones (agroclimatic zones) Land resource mapping unit, defined in terms of climate, landform and soils, and/or land cover, and having a specific range of potentials and constraints for land use (FAO, 1996).

Background processes (background system) Processes on which no or, at best, indirect influence may be exercised by the decision maker for which an LCA is carried out (UNEP–SETAC life cycle initiative [Teixeira *et al.*, 2016]).

Biodiversity Variability among living organisms from all sources including, *inter alia*, 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).[2]

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 Union, 2013).

Characterization factor Factor derived from a characterization model that is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator (ISO, 2006a, 3.37). For instance, a characterization factor converting land use of 1 km^2 of grassland to an impact on potential species extinction on a global scale.

[2] For this and other definitions taken from Article 2 of the Convention on Biodiversity (CBD), refer to the full text at: https://www.cbd.int/convention/text/

Conservation value (high)	A concept used to prioritize conservation efforts. Several factors can determine if a species (or ecosystem) is of high conservation value, for example: 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).
Cultivated grassland	Forage established with domesticated introduced or indigenous species that may receive periodic cultural treatment such as renovation, fertilization or weed control.
Data quality	Characteristics of data that relate to their compliance with stated requirements (ISO, 2006a, 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, 2006, 3.12).
Endemism	Association of a biological taxon with a unique and well-defined geographical area (The Encyclopedia of Earth, 2019).

Endpoint impact category	Attribute or aspect of natural environment, human health or resources, identifying an environmental issue giving cause for concern (ISO, 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, 2002, 3.6).
Foreground processes (foreground system)	Processes that are under the control of the decision-maker for which an assessment is carried out (UNEP–SETAC life cycle initiative [Teixeira *et al.*, 2016]).
Functional unit	Quantified performance of a product system for use as a reference unit (ISO, 2006b, 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.
Grassland	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.
Grassland meadow	A natural or semi-natural grassland often associated with the conservation of hay or silage.
Habitat	The place or type of site where an organism or population naturally occurs (Article 2 of the CBD).

Hotspot	In ecology, a hotspot of biodiversity is a biogeographical region that is both a significant reservoir of biodiversity with a high density of endemic species and threatened with destruction. A more restrictive and quantitative definition is that a hotspot should have lost 70 percent or more of its primary vegetation and host at least 0.5 percent of the world's plant species as endemics (Myers *et al.*, 2000). In life cycle assessment, a hotspot analysis is an assessment of the relative contribution of different elements (locations, steps of supply chains, types of pressure), with the aim of identifying those that make the strongest contribution to biodiversity loss (LEAP Biodiversity TAG). The two concepts are not directly related; hotspots of impact revealed by life cycle assessment may or may not coincide with hotspots of biodiversity.
Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned (ISO, 2006a, 3.39).
Invasive alien species	An alien species whose introduction and/or spread threaten biological diversity (CBD).[3]
Land occupation	Life cycle inventory flow related to use of a land area by activities such as agriculture, roads, housing and mining (adapted from Product Environmental Footprint Guide, European Union, 2013).
Land use change	Change in the purpose for which land is used by humans (e.g. between cropland and grassland, forestland, wetland or industrial land) (BSI, 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, 2006a, 3.1).
Life cycle assessment (LCA)	Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO, 2006a, 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, 2006a, 3.4).

[3] Refer to the CBD Glossary of Terms at: https://www.cbd.int/invasive/terms.shtml

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, 2006a, 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 example, climate change or acidification.
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.
Pressure-state-response (PSR) framework	A means for structuring indicators which facilitates interpretation and decision-making and 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.
Primary data	Data directly measured or collected for specific activities within a particular product's life cycle (e.g. energy used for the production of 1 kg of a specific feed additive in a particular production plant) or for a specific area (for state indicators).
Rangeland	Land on which the indigenous vegetation (climax or subclimax)[4] 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.
Secondary data	Information obtained from sources other than direct measurements, or from activities other than those specifically assessed.

[4] Many rangelands are not in climax or subclimax state and indigenous vegetation exists together with non-native or even invasive species.

Semi-natural grassland	Managed ecosystem dominated by indigenous or naturally occurring grasses and other herbaceous species.
Semi-natural habitat	Permanent woody or herbaceous area in agricultural landscapes (e.g. permanent grassland, grassy field margins or ditch banks, tree or shrub hedgerows, woodland).

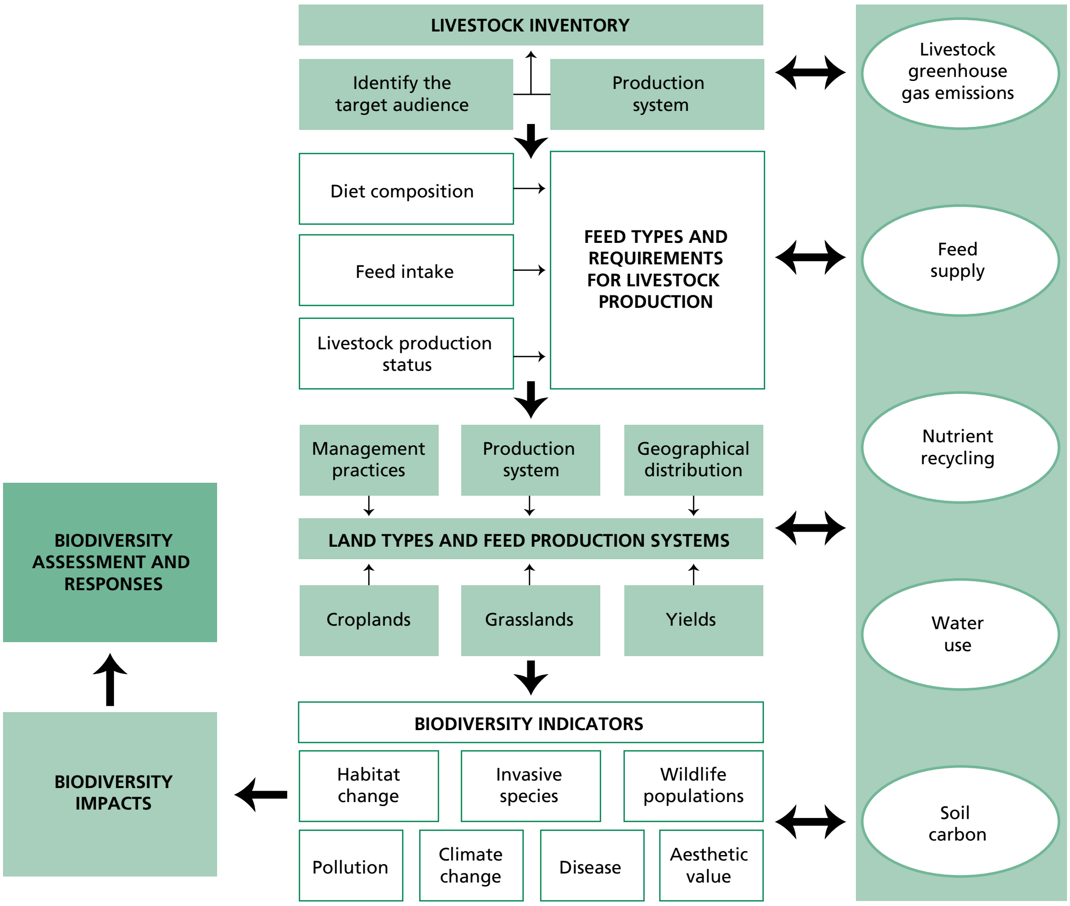
Appendices

Appendix 1

Links between the different LEAP guidelines documents

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 have been published or are under review (http://www.fao.org/partnerships/leap/overview/the-partnership/en/). Combined, the information in these documents provides a valuable foundation for the assessment of the impacts of livestock production on biodiversity (Figure A1.1). Several of these documents share a common methodological approach in the environmental assessment of livestock supply chains. First, 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 its scale (local, regional or global). 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) – all factors that influence the level of feed intake. Defining the types and amounts of feed that satisfy the productivity of the livestock population provides the basis for estimating the amount of crop or pasture needed to produce the required feed. Estimation of the amount of land needed to produce the feed requires estimates of crop and pasture yields in the region where 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/pastureland production systems also provide valuable information that is relevant to other ecosystem services, including air, water and soil quality, nutrient cycling and climate.

Figure A1.1
A conceptual model to assess biodiversity responses in livestock production supply chains[1]



[1] The model relies heavily on methodologies already used in existing LEAP guidance documents. First, 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 impacts 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 as well as associated responses.

Appendix 2

The high nature value of extensive livestock grazing systems

Many extensive sustainable livestock grazing systems can still be considered of high biodiversity value for the following reasons:

- They continue to utilize 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 utilized 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 synchronized 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 utilized by these animals. Hence, grazing pressure on any one area is generally either low or (in closely managed herds 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 that a sufficient area of potentially suitable habitat is available within relatively close proximity (i.e. in terms of the distance that the species can move) and thereby facilitates these cycles of colonization and recolonization.
- Extensive livestock grazing systems are more favourable than intensive systems to a wider range of wildlife species. They are practised over a wider scale and, therefore: i) the conditions required at any one time of year, particularly by more mobile species, can be found in a wider variety of locations; and ii) the different requirements of 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 during 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 ES.

Appendix 3

Methods to include impacts on ecosystem services in life cycle analysis

Table A3.1: Overview of selected studies that have included ecosystem service impacts in life cycle analysis

Impact indicators	Description	Methodology	Spatial scale of assessment/ Regionalization	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 (USD 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 tonnes HDPE production volumes[1]	Water consumption (m^3 water tonne $HDPE^{-1}$)	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 USA	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 (USD ha^{-1} yr^{-1})	Cao *et al.* (2015)
		Impacts on terrestrial green water flow and surface blue water production due to decreased run-off 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: *Eucalyptus globulus* 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, Koellner and Margni (2013)

(cont.)

Impact indicators	Description	Methodology	Spatial scale of assessment/ Regionalization	Indicator	Source
Water purification potential	Ecosystem's chemical, physical and mechanical capacity to filter water			Cation exchange capacity ($cmol_c\ kg_{soil}^{-1}$)	Saad, Koellner and Margni (2013)
				Water purification process costs (USD $ha^{-1}\ yr^{-1}$)	Cao *et al.* (2015)
	Soil's capacity to mechanically filter water			Rate of water passing (cm water day^{-1})	Saad, Koellner and Margni (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			Tonnes of soil eroded (tonne soil $ha^{-1}\ yr^{-1}$)	Saad, Koellner and Margni (2013)
				Sediment export (m^3 sediment tonne $HDPE^{-1}$)	Chaplin-Kramer *et al.* (2017)
				Erosion mitigation costs (USD $ha^{-1}\ yr^{-1}$)	Cao *et al.* (2015)
Climate regulation potential	Capacity of ecosystem (soils) to uptake carbon from the air			Social cost of C (USD $ha^{-1}\ yr^{-1}$)	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 (tonne C $m^{-2}\ yr^{-1}$)	Müller-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 (tonne CO_2 eq. tonne $HDPE^{-1}$)	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 (tonne N tonne $HDPE^{-1}$)	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)

Notes: GHG – greenhouse gas; LU – land use; CF – characterization factor; LULCIA – land use life cycle impact assessment; ES – ecosystem services; SOC – soil organic carbon; SOM – soil organic matter; IPCC – Intergovernmental Panel on Climate Change; ISO – International Organization for Standardization; LCIA: life cycle impact assessment; ET – evapotranspiration; PNV – potential natural vegetation; LANCA – land use indicator value calculation; ALCA – attributional life cycle assessment; CLCA – consequential life cycle assessment; LCAD tool – LCA model for net environmental and economic effects of farm scale anaerobic digestion and bioenergy scenarios; HDPE – high-density polyethylene; HCW – hot carcass weight; MJth – megajoule of useful heat output; LW – live weight.

Cells in grey: information already provided.

[1] 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 sugar cane feedstock).

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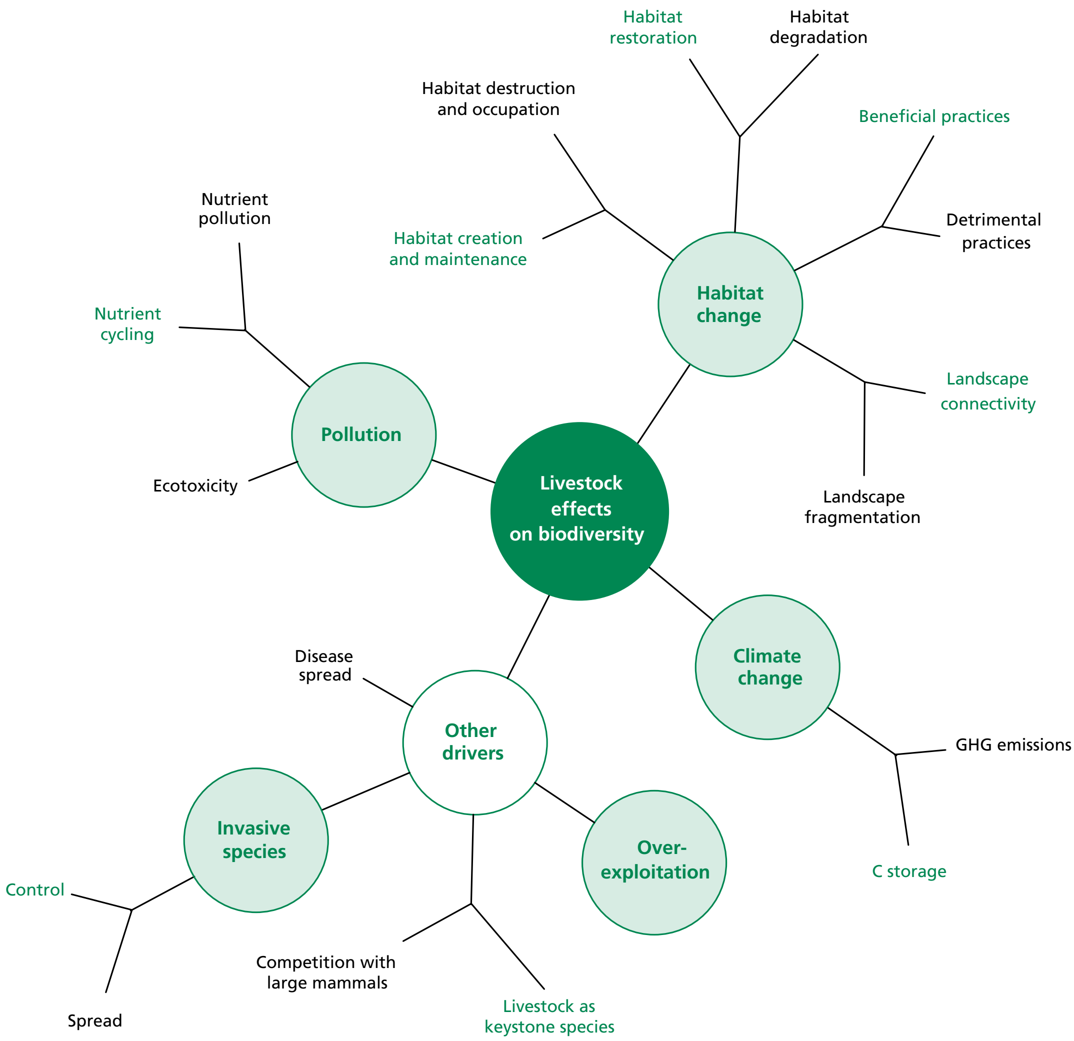
Appendix 4

Categories of pressures and benefits

Figure A4.1 provides an overview of the categories of influences that livestock have on biodiversity.

Figure A4.1

Categories of influences that livestock have on biodiversity



Notes: The five main drivers of biodiversity loss recognized by the Millennium Ecosystem Assessment (MEA) appear in light green circles. However, for most of these drivers, livestock can either exert negative pressure (black) on, or provide benefits (green) to, biodiversity. See MEA (2005) for a detailed description of all categories.

Source: LEAP Biodiversity Review (Teillard *et al.*, 2016).

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Appendix 5

Detailed description of recommended indicators

Table A5.1: Recommended list of pressure-state-response indicators and detailed description

Indicator[1] (subdivided by category)	Description
Procedural checks Elements aim to demonstrate that key requirements of the assessment process are met	
Scoping analysis is conducted	The scoping analysis should: • set the context by identifying biodiversity features of concern, legal and designation frameworks, due diligence etc.; and • conduct a pre-assessment for the relevant scale/territory, identifying potential hotspots of impacts including downstream and off-farm. (see subsection 6.1.2)
Regulatory constraints are met	How regulatory constraints related to biodiversity are met should be discussed, including those on: • protected areas and species; • banned or regulated biocides (fungicides, herbicides, pesticides); and • other ecotoxic agrochemicals (hormones, antibiotics).
Extrinsic use value of biodiversity is considered	The extrinsic use value should be defined through the involvement of local stakeholders.
Progress is monitored	Indicators should be monitored over time or against a clearly defined reference.
Stakeholders are engaged	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 analysis could also be performed.
Data quality	Data quality should be ensured. (see section 8.3)
Habitat protection	
Wildlife habitats under the farm influence are inventoried (mapped) and protected (R, Ql)	Across the agri-food industry, many sustainability assessments for formal accreditation require that farmland wildlife habitats are inventoried and mapped. Since 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). The inventory should: • identify and include (semi-)natural habitats (e.g. grasslands, grassy strips, flowering plants, isolated trees, hedgerows, woodland patches, shrubs, wetlands and waterways – ideally differentiating between native and exotic woody/shrub/grass species), and protected and priority habitats (both terrestrial and aquatic). • cover habitats within the farm, but also in the surrounding area (e.g. landscape, watershed) when potentially impacted by farming practices (e.g. pesticide drift, nutrient run-off, 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. • comprise habitats (potential distribution) of species of high conservation interest (e.g. protected at the local, national or higher level, endangered or threatened – CR, EN, VU and NT IUCN categories[2] – migratory wildlife). • include 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 CR, EN, VU or NT. An inventory will usually be the end product of several other stages that may include the following: • definition of a clear spatial boundary for the area of interest; • indication of the different areas that are occupied by wildlife habitats; • 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 grassland 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).

(cont.)

Indicator[1] (subdivided by category)	Description
Habitat change	
Soil erosion and soil erosion risk are mapped and management plan is implemented (R, Ql/Qt)	Factors influencing soil erosion risk include soil type, slope, burning, precipitation intensity, wind, bare soil cover, vegetation type. 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.
Area/proportion of degraded soil, including bare soil and areas with bush encraochement (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 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 and management practices (e.g. grazing regimes, rangeland fragmentation). Dry rangelands (e.g. < 800 mm yr^{-1}) are less vulnerable to degradation from overgrazing, although all rangelands face a potential threat from overgrazing, or more precisely, "under-resting" of grazing lands. 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.
Area or rate of habitat conversion (P, Qt)	Three main types of habitat conversion should be considered: • Deforestation (i.e. conversion from forest to grassland or feed crops) • Permanent grassland that is tilled (i.e. converted to temporary grassland or to cropland) • Abandoned grassland slowly converting to shrubland and forest through ecological succession
Wildlife conservation	
Priority actions promoting species with high conservation and functional value are listed and implemented (R, Ql)	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. 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. 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. The area (or proportion) of protected habitat or feature can be calculated as a quantitative indicator.
Abundance, presence/absence or distribution of species (with high conservation value) (S, Qt)	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 programmes. Species with a key ecological role or with added value as an indicator (e.g. keystone species, umbrella species, ecosystem engineers, species with defined 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.
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); 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.

(cont.)

Indicator[1] (subdivided by category)	Description
Invasive alien species	
Management plan is in place for the prevention and control of exotic species (R, Ql/Qt)	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. Qualitative component: Existence or creation of an invasion management plan (at the local level, property or establishment). Quantitative component: area under management plans for invasion control, baseline and historical progression
Presence of exotic invasive species (P, Ql/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 documented, 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 and aquatic biodiversity	
Management plan is in place for the application of ecotoxic agrochemicals (R, Ql)	Livestock production systems utilize a range of biocides (e.g. fungicides, herbicides, pesticides) and other potentially ecotoxic agrichemicals (e.g. animal health remedies, fertilizers) 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 macro-invertebrate 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.
Nutrient management plan is in place to rationalize fertilizer application (R, Ql)	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 run-off. 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)	Waterways – as indicated in the inventory of wildlife habitats – can be protected through livestock exclusion (e.g. fencing), edges or buffer strips. Direct access of livestock to waterways has a significant and usually deleterious effect on aquatic biodiversity, particularly for larger animals (e.g. cattle, deer, 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 spelled out. The width of the riparian area should be sufficient to ensure waterway protection and this action should be specified as 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 with a well-managed riparian zone.
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 nutrient guidelines (FAO, 2018) recommend a method to account for eutrophication. 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, macro-invertebrate 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. This indicator is of particular importance in sensitive catchments, where there may be additional management and monitoring requirements.

(cont.)

Indicator[1] (subdivided by category)	Description
Off-farm feed	
Inventory of off-farm feed used is established (R, Ql/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 is available, the inventory should also include the production origin of the imported feed.
Traceability systems for feedstuff are implemented (R, Ql)	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 generally includes such traceability.
Share of imported feed – total, from areas that are certified/deforested/of high conservation value (P, Qt)	The share of imported feed related to the total amount of feed used should be computed. 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. 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: • recently deforested areas; and • areas with high conservation value (e.g. CI biodiversity hotspot, WWF ecoregions with outstanding biodiversity features, IUCN Red List of Ecosystems). 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.
Landscape scale conservation	
Measure to promote connectivity identified and implemented (R, Ql)	Local NGOs or other stakeholders (e.g. scientists, geographers, local/regional land planners) should be involved in the identification of measures to promote connectivity, which may include maintenance of 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. 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 (e.g. fish, amphibians) 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.
Landscape heterogeneity (P, Qt)	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 (e.g. pollination, control of pests and erosion). Indicators reflecting the heterogeneity and structural complexity of the landscape include: • number and relative areas of land uses; • (spatial) Shannon diversity index; and • edge length or perimeter/area ratio of natural patches. Certain production systems can achieve high levels of heterogeneity and improve biodiversity while being consistent with livestock production objectives. They include agroforestry/silvopastoralism (e.g. with hedgerows, shelterbelts, windbreaks, live hedges) and integrated crop–livestock production systems.

[1] P – pressure or benefit; S – state; R – response; Ql – qualitative; Qt – quantitative.

[2] IUCN – International Union for Conservation of Nature. IUCN categories: CR – critically endangered; EN – endangered; VU – vulnerable; NT – near threatened.

Note: NGO – non-governmental organization. NDVI – Normalized Difference Vegetation Index; DMP – dry matter productivity; CI – Conservation International; WWF – World Wide Fund for Nature.

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Appendix 6

Extended list of indicators

1.1 HABITAT CHANGE

- Permanent area of bare soil or under desertification process
- Area of bare soil between cropping seasons
- Livestock density
- Area of irrigated feed crops
- Feed crop yield
- Seeding of grassland
- Shallow or no-tillage is used

1.2 WILDLIFE CONSERVATION

- Key functional or engineer species (e.g. earthworms, dung beetles/amount of soil removed by dung beetles – Giraldo *et al.*, 2011)
- Use of fishmeal in feed rations
- Number of conflicts with wildlife including number of large predator kills
- Over-exploitation of wildlife species (e.g. hunting, fishing, collecting) on farm prohibited
- Mowing (and grazing) delayed until after the nesting season of ground-nesting birds in part of the grassland area, particularly around wetlands
- Stubbles left over winter

1.3 INVASIVE SPECIES

1.4 POLLUTION AND AQUATIC BIODIVERSITY

1.4.1 Pollution by nutrients

- Quantity (kg) of N and/or P applied in grassland or feed crops
- Inland or coastal water in a state of eutrophication
- Presence of plant species characteristic of nutrient-rich (eutrophic) conditions
- Critical load exceedance for nitrogen (from nitrogen deposition) in the soil
- Emissions of gases leading to nitrogen deposition and acidification
- Nitrogen balance or nutrient-use efficiency (accounting for inputs and outputs) (see LEAP guidelines on nutrients)
- Nutrients in transitional, coastal and marine water
- Animal diet balanced to meet requirements and reduce nutrient excretion
- Manure management optimized to minimize nutrient leaching and optimize nutrient recycling
- Crop and livestock productions integrated to optimize nutrient recycling
- Manure used to produce biogas (with leakage issues being controlled)

1.4.2 Pollution by ecotoxic substances

- Application of specific pesticide molecules with high ecotoxicity
- Amount of toxic substance used, weighted by factors reflecting their toxicity (including half-life, mobility in the environment)
- Pesticide application (number of applications or quantity of active ingredient) in feed crops
- Presence of faecal anthelmintic residues
- Use (and quantity) of toxic veterinary products: antibiotics, anthelmintics, hormones
- Water contamination by hormones
- Biological control used
- Crop rotation used to break weed and pest life cycles and avoid disease build-up
- Mechanical control used when relevant
- No preventive spraying used and only affected areas sprayed
- Precision spraying used and drift minimized
- Products targeting specific species used rather than generalist products
- Semi-natural habitats created and maintained for natural pest predators
- Spatial intercropping used to limit pest propagation

1.5 OFF-FARM FEED

1.6 LANDSCAPE HETEROGENEITY

- Area of patches of semi-natural habitats
- Distance between patches of semi-natural habitats
- Diversity of crops and crop varieties grown
- Crop rotations long
- Structural complexity of the vegetation (e.g. trees – McElhinny *et al.*, 2005 – or grass)

1.7 ADDITIONAL CATEGORIES

1.7.1 Large-scale indicators

- Farmland Bird Index
- IUCN Red List indices
- Living Planet Index (LPI)
- Mean Species Abundance (MSA)

1.7.2 Ecosystem services

- Production/yield of animal food products
- Other livestock products: hides, skins, fibre, manure and urine for fertilizer, manure and methane for energy
- Vegetation indices (remotely sensed): dry matter productivity (DMP), Net Primary Productivity (NPP), normalized difference vegetation index
- Above-ground biomass
- Human appropriation of net primary production (HANPP)
- Crop production/yield
- Groundwater, streamflow, water abstracted

- Forest area or biomass
- Soil erosion risk or erosion protection
- Pollination potential
- Vegetation type
- Flood events
- Fire events
- Soil organic carbon depletion
- Nutrient flux
- Nutrient cycling (soil fertility, nutrients and organic matter distribution)
- Soil organic carbon storage
- Weed control
- Shrub control and fire regulation
- Presence of species/landscapes with aesthetic, cultural or religious importance
- Aesthetic value of livestock-maintained landscapes
- Livestock contribution of cultural heritage and identity
- Contribution of cultural heritage and identity
- Role in social events, relations, status
- Extent of protected areas or high nature value farmlands
- Recreation and tourism
- Loss of biodiversity habitats
- Erosion of livestock genetic resources (loss of breeds)
- Creation and maintenance of biodiversity habitats
- Maintenance of livestock genetic resources (breeds)

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Appendix 7

Regional and global data sources for specific taxonomic groups

Table A7.1: Examples of relevant data sources on biodiversity available on the internet, with global and regional coverage of a wide range of taxonomic groups and other specific taxonomic groups

Data source	Geographical coverage	Network	Groups of species	Website
Biodiversity Heritage Library (BHL)	USA, South Africa, Australia, China, Egypt, Europe, Brazil	> 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/
Brazilian Biodiversity Information System – SiBBr	Brazil	93 institutions	Broad	http://www.sibbr.gov.br/
World Biodiversity Information Network (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 (ENHSIN)	Europe	7 institutions from Europe, 3 data providers representing various natural history collections	Broad	http://www.nhm.ac.uk/science/rco/enhsin/
European Bird Census Council (EBCC)	Europe	~100 research institutions and NGOs from European countries	Birds	https://www.ebcc.info/
Integrated Biodiversity Assessment Tool (IBAT) for Research and Conservation Planning	Global	Tool aggregating biodiversity data from BirdLife International, Conservation International, IUCN, UN Environment	Broad	https://www.ibat-alliance.org/ibat-conservation/
Distributed Information System 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 > 90 institutions	Broad	https://www.sibcolombia.net/el-sib-colombia/
National Institute of Biodiversity (INBio/Atta)	Costa Rica	1 institute	Broad	http://atta.inbio.ac.cr/attaing/atta03.html
Mammal Networked Information System (MANIS)	Global	32 institutions	Mammals	http://dlp.cs.berkeley.edu/manis/
Fishnet	North America	24 North American fish databases	Fish	http://habanero.nhm.ku.edu/fishnet/
HerpNET	Global	37 databases	Broad	http://herpnet.org/
Missouri Botanical Garden (Tropicos)	Global	1 institution	Plants	http://mobot.mobot.org/W3T/Search/vast.html

(cont.)

Data source	Geographical coverage	Network	Groups of species	Website
Living Planet Index (LPI)	Global	19 institutions	Broad	http://www.livingplanetindex.org/
Inter-American Biodiversity Information Network (IABIN)	American continent	4 thematic networks	Broad	http://www.oas.org/es/sedi/dsd/iabin/default.asp
ASEAN Regional Centre for Biodiversity Conservation (ARCBC)	Southeast 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/
Domestic Animal Diversity Information System (DAD-IS)	Global	199 countries and territories	Livestock genetic diversity	http://www.fao.org/dad-is/en/)

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