Food and Agriculture Organization of the United Nations

A review of indicators and methods to assess biodiversity

Application to livestock production at global scale

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Recommended Citation
Teillard, F., Anton, A., Dumont, B., Finn, J.A., Henry, B., Souza, D.M., Manzano P., Milà i Canals, L., Phelps, C., Said, M., Vijn, S., White, S. 2016. *A review of indicators and methods to assess biodiversity – Application to livestock production at global scale.* Livestock Environmental Assessment and Performance (LEAP) Partnership. FAO, Rome, Italy.

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Acknowledgements

AUTHORSHIP AND DEVELOPMENT PROCESS

This review is a product of the Livestock Environmental Assessment and Performance (LEAP) Partnership. The initial objective of this review was to provide the members of the LEAP Technical Advisory Group (TAG) on biodiversity with a common ground of knowledge on the main biodiversity indicators and assessment methods applicable to livestock production. The objective of this group was to develop *Principles for the assessment of livestock impacts on biodiversity* (LEAP biodiversity principles, FAO, 2016). This document went indeed beyond its initial objective, providing an important body of information on biodiversity assessment which could be useful to stakeholders in the livestock and related sectors.

This technical review was initially drafted by Félix Teillard (FAO, Italy; INRA, France) as a preparatory document for the LEAP Technical Advisory Group (TAG) on biodiversity. Once the initial draft was shared with TAG members, the following task force led by Félix Teillard was formed for the finalization of this report: Assumpció Anton (IRTA, Spain); Bertrand Dumont (INRA, France); John Finn (Teagasc, Ireland); Beverley Henry (Queensland University of Technology, Australia); Danielle Maia De Souza (Swedish University of Agricultural Sciences, Sweden); Pablo Manzano (IUCN CEM, Kenya; UAM, Spain); Llorenç Milà i Canals (UNEP, France); Catherine Phelps (Dairy Australia, Australia); Mohammed Said (ILRI, Kenya); Sandra Vijn (WWF, USA); and Shannon White (Government of Alberta, Canada).

The LEAP Secretariat coordinated and facilitated the work of this task force within the biodiversity TAG, and guided and contributed to the development of its content. The LEAP secretariat, hosted at FAO, was composed of: Pierre Gerber (Coordinator); Camillo De Camillis (Manager); Carolyn Opio (Technical officer); Félix Teillard (Technical officer); and Aimable Uwizeye (Technical officer).

The LEAP Steering Committee provided overall guidance for the activities of the Partnership and helped review and clear the principles for public release. During development of the principles the LEAP Steering Committee was composed of:

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SPONSORS, ADVISORY AND NETWORKING

FAO is most grateful for all the valuable contributions provided at various levels by LEAP partners.

Special thanks go to France, Ireland, the Netherlands, New Zealand, and Switzerland who have continually supported the Partnership through funding and often through in-kind contributions.

Particularly appreciated were in-kind contributions from the following CSOs and NGOs 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 People (WAMIP); World Vision; and the World Wide 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 and Textile Organization (IWTO); FEDIOL, the federation representing the European Vegetable Oil and Proteinmeal Industry; the International Federation for Animal Health (IFAH); the Royal DSM Company and NOVUS International.

Substantial in-kind contribution also came from FAO staff and the Mitigation of Climate Change in Agriculture (MICCA) Programme.

Last but not least, the LEAP Partnership is grateful for the advice provided by ISO, UNEP, and the European Commission, and is glad to network with OIE and the Global Agenda for Sustainable Livestock.

Abbreviations

AES	Agri-Environmental Schemes
AFI	Agri-environmental Footprint Index
BDP	Biodiversity Damage Potential
CAP	Common Agriculture Policy (of the European Union)
CBD	Convention on Biological Diversity
CF	Characterization Factor
EDP	Ecological Damage Potential
EEA	European Environment Agency
EFBI	European Farmland Bird Index
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GAP	Good Agricultural Practices
GHG	Greenhouse Gas
HANPP	Human appropriation of Net Primary Production
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
NDVI	Normalized Difference Vegetation Index
NGO	Non-Governmental Organization
NPP	Net Primary Productivity
OECD	Organization for Economic Cooperation and Development
PAF	Potentially Affected Fraction (of species)
PDF	Potentially Disappeared Fraction (of species)
PNV	Potential Natural Vegetation
PSR	Pressure-State-Response
SAFA	Sustainable Assessment of Agriculture and Food systems
SETAC	Society for Environmental Toxicology and Chemistry
TAG	Technical Advisory Group
UN	United Nations
UNEP	United Nations Environment Programme
WWF	World Wide Fund for Nature

Glossary

Terms relating to biodiversity

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]

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]

Ecosystem A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit. [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]

Endemism Association of a biological taxon with a unique and well-defined geographic area. [The Encyclopedia of Earth, http://www.eoearth.org]

Endemic species See **Endemism**

Habitat The place or type of site where an organism or population naturally occurs. [Article 2 of the CBD]

Rangeland Land on which the indigenous vegetation (climax or natural potential) is predominantly grasses, grass-like plants, forbs, or shrubs and is managed as a natural ecosystem. If plants are introduced, they are managed similarly. Rangelands include natural grasslands, savannas, shrublands, many deserts, tundras, alpine communities, marshes and meadows. [International Society for Range Management]

Terms relating to life cycle assessment and environmental assessment

Acidification	Impact category that addresses impacts due to acidifying substances in the environment. Emissions of NOx, NH_3 and SOx lead to releases of hydrogen ions (H +) when the gases are mineralized. The protons contribute to the acidification of soils and water when they are released in areas where the buffering capacity is low. Acidification may result in forest decline and lake acidification. [Adapted from Product Environmental Footprint Guide, European Commission, 2013]
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 characterisation factors for each substance and impact category of concern. For example, with respect to the impact category "climate change", CO2 is chosen as the reference substance and kg CO2-equivalents as the reference unit. [Adapted from: Product Environmental Footprint Guide, European Commission, 2013]
Characterization factor	Factor derived from a characterization model which is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator. [ISO 14044:2006, 3.37]
Downstream	Occurring along a product supply chain after the point of referral. [Product Environmental Footprint Guide, European Commission, 2013]
Eco-toxicity	Environmental impact category that addresses the toxic impacts on an ecosystem, with damage to individual species and changes in the structure and function of the ecosystem. Ecotoxicity is a result of a variety of different toxicological mechanisms caused by the release of substances with a direct effect on the health of the ecosystem. [Adapted from: Product Environmental Footprint Guide, European Commission, 2013]
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 14044:2006, 3.12]
Emissions	Release of substance to air and discharges to water and land.

Endpoint impact category	Damage-oriented approach translating environmental impacts into issues of concerns such as biodiversity. [adapted from Guinée *et al.* (2002)]
Environmental impact	Any change to the environment, whether adverse or beneficial, wholly or partially resulting from an organization's activities, products or services. [ISO/TR 14062:2002, 3.6]
Eutrophication	Excess of nutrients (mainly nitrogen and phosphorus) in water or soil, from sewage outfalls and fertilized farmland. In water, eutrophication accelerates the growth of algae and other vegetation in water. The degradation of organic material consumes oxygen resulting in oxygen deficiency and, in some cases, fish death. Eutrophication translates the quantity of substances emitted into a common measure expressed as the oxygen required for the degradation of dead biomass. In soil, eutrophication favours nitrophilous plant species and modifies the composition of the plant communities. [Adapted from: Product Environmental Footprint Guide, European Commission, 2013]
Functional unit	Quantified performance of a product system for use as a reference unit. [ISO 14044:2006, 3.20]. It is essential that the functional unit allows comparisons that are valid where the compared objects (or time series data on the same object, for benchmarking) are comparable.
Greenhouse gases (GHGs)	Gaseous constituent of the atmosphere, both natural and anthropogenic, that absorbs and emits radiation at specific wavelengths within the spectrum of infrared radiation emitted by the Earth's surface, the atmosphere, and clouds. [ISO 14064-1:2006, 2.1]
Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned. [ISO 14044:2006, 3.39]
Land occupation	Impact category related to use (occupation) of land area by activities such as agriculture, roads, housing, mining, etc. [Adapted from: Product Environmental Footprint Guide, European Commission, 2013]
Land use change	Change in the purpose for which land is used by humans (e.g. between cropland, grassland, forestland, wetland, industrial land). [PAS 2050:2011, 3.27]

Life cycle	Consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal. [ISO 14044:2006, 3.1]
Life Cycle Assessment	Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle. [ISO 14044:2006, 3.2]
Life Cycle Impact Assessment (LCIA)	Phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential impacts for a product system throughout the life cycle of the product. [Adapted from: ISO 14044:2006, 3.4]
Life Cycle Inventory (LCI)	Phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle. [ISO 14046:2014, 3.3.6]
Midpoint impact category	Problem-oriented aproach translating impacts into environmental themes such as global warming, acidification, ecotoxicity. [Adapted from Guinée *et al*. (2002)]
Product(s)	Any goods or services. [ISO 14044:2006, 3.9]
Product system	Collection of unit processes with elementary and product flows, performing one or more defined functions, and which models the life cycle of a product. [ISO 14044:2006, 3.28]
Raw material	Primary or secondary material that is used to produce a product. [ISO 14044:2006, 3.1.5]
Soil organic matter	The measure of the content of organic material in soil. This derives from plants and animals and comprises all of the organic matter in the soil exclusive of the matter that has not decayed. [Product Environmental Footprint Guide, European Commission, 2013]
System boundary	Set of criteria specifying which unit processes are part of a product system. [ISO 14044:2006, 3.32]
Unit process	Smallest element considered in the life cycle inventory analysis for which input and output data are quantified. [ISO 14044:2006, 3.34]
Upstream	Occurring along the supply chain of purchased goods/services prior to entering the system boundary. [Product Environmental Footprint Guide, European Commission, 2013]

Water use Use of water by human activity.

Note 1 to entry: Use includes, but is not limited to, any water withdrawal, water release or other human activities within the drainage basin impacting water flows and/or quality, including in-stream uses such as fishing, recreation, transportation.

Note 2 to entry: The term "water consumption" is often used to describe water removed from, but not returned to, the same drainage basin. Water can be consumed because of evaporation, transpiration, integration into a product, or release into a different drainage basin or the sea. Change in evaporation caused by land-use change is considered water consumption (e.g. reservoir). The temporal and geographical coverage of the water footprint assessment should be defined in the goal and scope.

[ISO 14046:2014, 3.2.1]

PART 1

INTRODUCTION

1. Livestock & biodiversity

1.1 THE INFLUENCES OF LIVESTOCK ON BIODIVERSITY

Around 30 percent of the Earth's land surface is currently dedicated to livestock production through pastures (≈ 25%) and feed crops (≈ 5%) (Monfreda *et al.*, 2008; Ramankutty *et al.*, 2008). Some 30 percent of terrestrial habitats are therefore directly modified by livestock in various ways. At one extreme, undisturbed habitats can be destroyed, such as in conversion of primary forest to pastures or feed crops in the Brazilian Amazon (Lambin *et al.*, 2003; Wassenaar *et al.*, 2007; Nepstad *et al.*, 2009). It should be noted, however, that livestock are not the only driver and overall deforestation has been significantly reduced since 2004 (Bastos, 2013). At the other extreme, in some places with a long history of livestock grazing, a unique biodiversity has specifically adapted to habitats associated with the presence of domestic herbivores. Indeed, their grazing shapes biodiversity in many ecosystems (Frank, 2005) where livestock have, when under adequate management, taken over the role of wild herbivores (Eriksson *et al*., 2002; Bond & Parr, 2010). In Europe, extensive livestock grazing is key to maintaining permanent grassland habitats with high biodiversity levels (Bignal & McCracken, 1996; Atkinson *et al.*, 2002; Laiolo & Dondero, 2004; Rook *et al*., 2004). Similarly, in North American rangelands, cattle can play a similar ecological role to that of bison historically, and grazing has been shown to increase biodiversity in certain situations (Collins *et al.*, 1996). In African savannahs, pastoralism is often compatible with wildlife and can enrich landscapes and their biodiversity (Reid, 2012). Livestock producers can also help preserve biodiversity through control of feral animals and weeds and managing the risk of damaging wildfires. Other types of habitat modification by livestock lie between these two extremes. For instance, grazing can be a source of erosion and land degradation in areas where grazing history is recent and the indigenous vegetation is ill-adapted to grazing (e.g. in Iceland, Thórhallsdóttir *et al*., 2013). Overgrazing can also lead to rangeland degradation and biodiversity loss in humid and arid regions (Asner *et al.,* 2004). In temperate regions such as Europe, grassland management intensification has had very adverse effects on biodiversity during the past decades (Vickery *et al.*, 2001).

Livestock production influences biodiversity beyond these habitat changes. Fertilization and nutrient excretion significantly alters global nutrient cycles (Erisman *et al.*, 2007; Bouwman *et al.*, 2009) and causes important diffuse pollution from nitrogen and phosphorus (Jongbloed & Lenis, 1998). Diffuse nutrient pollution has a great impact on aquatic ecosystems, causing eutrophication and acidification (Carpenter *et al.*, 1998. Vörösmarty *et al.*, 2010). In soils, higher nutrient concentration and acidification modify species composition and the structure of terrestrial ecosystems in fertilized cropland and grassland, and also in forests (Clark *et al.,* 2007; Belsky & Blumenthal, 1997). GHG emissions related to livestock represent a significant share of human-induced emissions (14.5% according to Gerber *et al.,* 2013). These emissions contribute to climate change, an important driver of biodiversity loss at global scale (Millennium Ecosystem Assessment, 2005). However, it is complicated to isolate and to quantify the impact of livestock-related GHG emissions on biodiversity. Livestock can also have positive effect on biodiversity in the face of climate change. Klein *et al.* (2004) have shown on the Tibetan Plateau that grazing can mitigate the negative effects of global warming on rangeland species richness and that flexible and opportunistic grazing management may be required in a warmer future.

In the next decades, projected population growth and rising per capita incomes is predicted to lead to higher demand for livestock products, putting more pressure on land and resources (McMichael *et al.*, 2007; Wirsenius *et al.*, 2010). For instance, meat consumption in China and Indonesia has already increased significantly, and dairy consumption is increasing in India (FAOSTAT, 2014). With its burgeoning middle class, China, whose meat consumption in 1978 was one third that of the United States, now consumes twice as much as the US (Earth Policy Institute, 2012). With dietary shifts in emerging countries significantly increasing global demand for animal products, the livestock sector faces the challenge of increasing production while limiting its negative impacts on biodiversity.

1.2 THE IMPORTANCE OF BIODIVERSITY

The recognition of biodiversity as an important environmental issue emerged at the 1992 Rio de Janeiro Conference on Environment and Development. The conference opened the way toward the ratification of the Convention on Biological Diversity (CBD) in 2002, when 190 countries agreed to significantly reduce the rate of biodiversity loss. Biodiversity's essential role is now accepted internationally, not only because of its intrinsic value, but also because of the key role it plays in supporting ecosystem services that benefit human societies and economies. Biodiversity is essential to human wellbeing through the various ecosystem services it supports (MEA, 2005). These ecosystem services belong to four main categories: provisioning services (e.g., food, water, wood, fuel, fibre, medicines and genetic resources); supporting services (e.g., water cycling and soil formation); regulating services (e.g., climate and erosion); and cultural services (e.g., aesthetic, educational). In addition, biodiversity underpins the resilience and function of ecosystems, i.e. their capacity to sustain such services over time and in the face of various disturbances (Loreau *et al.*, 2001; Hooper *et al.*, 2005; Classen *et al.*, 2014). Regarding the contribution of biodiversity to the economy, Costanza *et al.* (1997) estimated that the value of 17 selected ecosystem services was higher than the entire global gross national product.

In many ecosystems or biomes, biodiversity and livestock play a role in shaping the landscape. The livestock sector is both a provider and a user of biodiversity and ecosystem services (Zhang *et al.*, 2007, Huntsinger & Oviedo, 2014). Key ecosystem services supporting livestock production include, among others, biomass production (provisioning service); micro-organism cycling of nutrients, soil formation, nitrogen fixation (supporting services); and pollination, pest control, climate regulation and water purification (regulating services). Other ecosystem services supporting livestock include climate change adaptation through greater heterogeneity (multi-species swards, agroforestry and habitat restoration) and protection from extreme weather (Oliver & Morecroft, 2014; UNEP 2010; Haines-Young, 2009). Several studies in grassland have shown how high plant species richness, niche complementarity and diversity of functional types result in significantly greater biomass production, carbon storage, and resistance to weed invasion than monoculture (Tilman *et al*., 2001; Finn *et al*., 2013).

Inter-specific differences in maturity and nutritive value also lead to a more stable digestibility of forage along the grazing season (Michaud *et al*., 2012). Legume and forb species that are rich in condensed tannins are known for their anthelmintic properties against parasitic nematodes (Hoste *et al*., 2006). In addition, there is a large body of evidence that plant diversity in semi-natural grasslands affects the nutritional and sensory quality of dairy products (Bosset *et al*., 1999; Chilliard *et al*., 2007; Sickel *et al*., 2012).

In rangelands, biodiversity is key to the resilience of pastoralist systems as heterogeneous landscapes are able to provide resources in a wider range of climatic situations (Krätli & Schareika, 2010). Changes in community composition are also used as ecosystem state indicators and, therefore, trigger critical management decisions (Oba, 2012). In croplands (including feed crops), biodiversity also supports ecosystem services that are crucial for agricultural production, such as soil fertility, pollination and pest control (Altieri, 1999; Klein *et al*., 2007; Classen *et al*., 2014).

As an intrinsic component of agro-ecosytems, livestock are not only a user but a provider of ecosystem services. The provisioning of food is the most obvious of these services. Livestock contribute to a wider range of ecosystem services such as encroachment control, maintenance of habitat for pollinators that benefit adjacent crops, soil fertility transfer and carbon sequestration in grasslands (Morandin *et al*., 2007; Soussana *et al.*, 2010; Janzen, 2011; Scohier *et al*., 2013).

1.3 THE NEED FOR QUANTITATIVE INDICATORS

There is a need for widely recognized frameworks for the assessment of the biodiversity performances of livestock in order to mitigate its negative impacts. Such frameworks could also foster synergies between the positive effects of livestock and the value of biodiversity for the sector. Assessment frameworks identify the most efficient systems and those requiring improvements. They also help develop evidence-based environmental policies targeting the livestock sector (Gill *et al.*, 2010).

Most existing assessments of livestock environmental performance have focused on GHG emissions. They mostly used a wide range of Life Cycle Assessment (LCA) approaches so as to provide a comprehensive assessment of the GHG emissions associated with several types of livestock products (de Vries & de Boer 2010, Roma *et al.,* 2014), by accounting for all stages of production (including feed production, livestock production, waste management and distribution). These quantitative assessments have made it possible to propose both technical (Smith *et al.*, 2008; Garnett, 2009) and policy (Gerber *et al.*, 2010; Steinfeld & Gerber, 2010) options to mitigate the livestock contribution to climate change. The influence of livestock production on the environment is not restricted to GHG emissions. Biodiversity is also strongly influenced both positively and negatively, but no consensus currently exists on the use of specific biodiversity assessment indicators or methods. Multi-criteria approaches in LCA would avoid shifting the environmental burden from one criterion to another. Expanding LCA approaches to include the interaction between livestock production, climate, habitat change, and biodiversity would also be an opportunity for these assessments to suggest more effective biodiversity management (Oliver & Morecroft, 2014).

Quantifying impacts of livestock systems on biodiversity (in addition to climate change) is crucial because GHG emissions mitigation options may have contrasting effects on biodiversity. For instance, intensifying livestock production in areas where it can be done most efficiently has been suggested as an option for mitigating global GHG emissions. Intensification reduces emissions per unit of product (Steinfeld & Gerber, 2010) and avoids the higher enteric emissions of CH_4 associated with per unit production in grassland-based systems (Eckard *et al.*, 2010).

But intensification often results in lower biodiversity levels because of the associated habitat changes and negative effects of nutrient pollution and chemical inputs. Intensifying production in already cultivated areas may spare land surface for biodiversity conservation, although this is debated (Borlaug, 2007; Ewers *et al.*, 2009). However, extensive livestock production systems are crucial biodiversity habitat and ecosystem services providers (e.g. carbon sequestration, Soussana *et al.*, 2010). Grass-fed livestock and pastoral systems not only are the primary methods for converting plant biomass into food edible by humans in many marginal lands (Rodriguez, 2008), they also play an important role in socio-economic and sustainable development terms (Dedieu *et al.*, 2008; Ickowicz, 2010). Extensive livestock production systems usually show higher direct GHG emissions per unit of protein produced. However, focusing on protein products alone does not do justice to extensive systems' contribution to ecosystem services and the maintenance of biodiversity. If biodiversity and ecosystem services are considered as a product, emission intensity can be similar or even lower in extensive systems than in intensive ones (Ripoll-Bosch *et al.*, 2013). The example of intensification shows that trade-offs exists between the performances on GHG emissions vs. biodiversity; therefore, assessing both criteria is needed to understand what mitigation options will best serve to improve the overall sustainability of livestock production.

2. Objective

The objective of this report is to review biodiversity indicators and assessment methods applicable to the livestock sector at global scale.

We conducted a systematic review of scientific articles (details in Appendix A1) describing biodiversity indicators, assessment, and footprinting methods in the context of livestock production or agriculture. Specific searches of publications were used to complete this systematic review.

We detail biodiversity indicators and assessment methods within two main frameworks: Pressure-State-Response (PSR), and Life Cycle Assessment (LCA) These were selected because they are widely recognized and used to assess environmental impacts, and because they allowed the development of many indicators and methods. In addition, these frameworks are well adapted to the context of livestock production. In Part 2, we describe biodiversity indicators and structure them using the PSR framework (OECD, 1993). This framework has been broadly used to structure biodiversity indicators and facilitate their interpretation. Part 2 also provides an overview of the different categories of pressures (and benefits) that livestock place on biodiversity (Section 4) and of the different levels and dimensions of biodiversity that can be described (Section 5). In Part 3, we describe several methods for including biodiversity impacts in the LCA framework. LCAs are a key tool for conducting environmental impact assessments and an increasing number of methods are being developed to address biodiversity loss.

The Conclusion (Part 4) highlights complementarities between the PSR and LCA approaches. Indicators and methods have been developed separately within each of the two frameworks. Although they formalize it differently and use a different terminology, the PSR and LCA frameworks describe the same environmental cause-effect chain, from livestock production to drivers of biodiversity changes, and biodiversity changes themselves.

While the review focuses on the PSR and LCA frameworks, Appendix A3 mentions other environmental assessment and management tools that are available from academia, NGOs and intergovernmental organizations, and the private sector.

Throughout the review, we discuss whether the various indicators and methods presented are well-suited to the context of livestock production (where both negative and positive impacts have to be considered) and to the global scale, in order to further develop the LEAP Principles for the assessment of livestock impacts on biodiversity (LEAP, 2016)

This review focuses on wild biodiversity: the genetic, domestic diversity of livestock breeds and crop varieties are not within its scope.

PART 2

THE PRESSURE-STATE-RESPONSE INDICATOR FRAMEWORK

3. The Pressure-State-Response framework

Indicators are a crucial tool for monitoring biodiversity impacts or the improvement in biodiversity performance. Indicators share certain properties: they must be rigorous, repeatable, widely accepted and easily understood (Balmford *et al.*, 2005). Making a selection among the many existing biodiversity indicators (more than 600 of them were identified at the European scale, EEA, 2003) should be based on logical frameworks (EEA, 2007). The Pressure-State-Response framework (OECD, 1993) has been widely used to develop and structure biodiversity indicators. Several frameworks have been derived from the original PSR, including Driver-Pressure-State-Impact-Response (Smeets *et al.* 1999) and Use-Pressure-State-Response-Capacity (CBD, 2003).

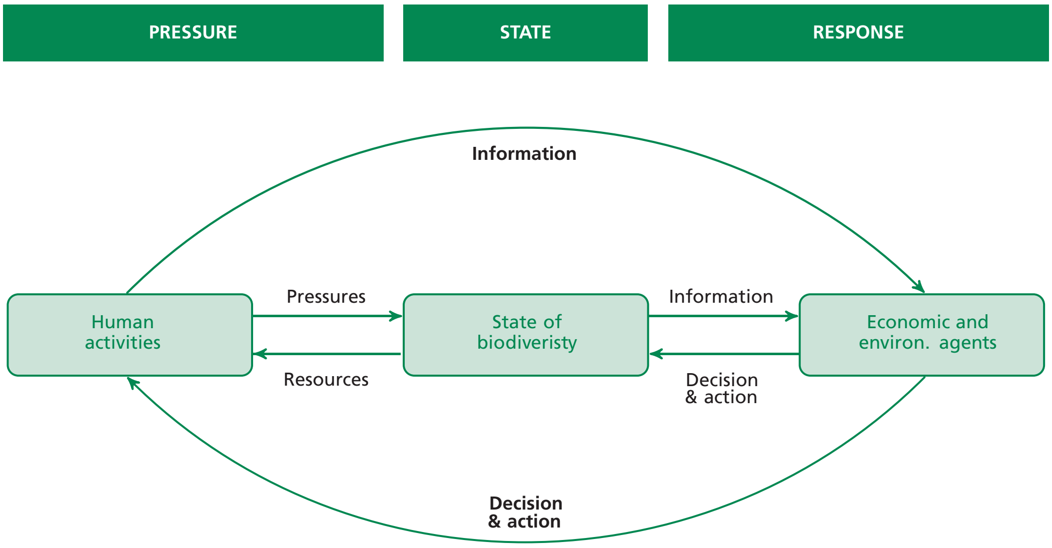
The PSR framework is based on causality. Indicators evaluate the *pressures* of human activities that lead to changes in environmental *states*, causing *responses* (decision and actions) from the stakeholders (political, socio-economic), aimed at reaching a more sustainable state. Focusing on livestock production among other human activities and on biodiversity among other environmental components is a straightforward application of the PSR framework to this specific context (Figure 1).

The PSR framework helps inform policy-makers by providing indicators that are structured and easy to interpret (Smeets *et al.*, 1999). The linear cause-effect relationships it posits may be too simplified to describe complex socio-ecological interactions, especially at local scale; however, the intuitive design of the PSR indicator system makes it a useful tool at larger scales (Levrel *et al.*, 2009). Using the PSR framework, Butchart *et al.* (2010) analyzed the performance of global biodiversity indicators and indicated an overall increase in pressure indicators and decline in state indicators, despite increasing political responses. The CBD has encouraged the development of biodiversity indicators to monitor progress in reducing the rate of biodiversity loss.

At the global and European levels, the Convention on Biological Diversity (CBD, 2006) and the European Environmental Agency (EEA, 2007) proposed headline biodiversity indicators covering the pressure, state and response components. Although these indicators do not focus on agricultural pressures, several of them could be relevant to the context of livestock production. Other initiatives have developed indicators for agriculture with a wider environmental scope than biodiversity (i.e. OECD 2001). They provide indicators of the biodiversity state; moreover, state indicators for certain environmental components (e.g. pesticides) can correspond to pressure indicators for biodiversity.

In the following sections, we use the PSR framework and identify crucial indicator themes in the context of livestock production and biodiversity. We also review existing indicators and identify gaps in indicator and data availability.

Figure 1
The Pressure-State-Response framework applied to livestock production and biodiversity.



Source: adapted from OECD (1993)

4. Pressures, benefits and their indicators

The Millennium Ecosystem Assessment (2005) recognizes five main direct drivers of biodiversity loss: habitat change, climate change, pollution, overexploitation and invasive species. Steinfeld *et al*., (2006) showed how livestock contributed directly or indirectly to each of these drivers. No comprehensive indicator framework exists to measure the pressure of livestock on biodiversity within each of these drivers. For the five drivers, we identified key categories of biodiversity pressures that are more specific to the context of livestock production (Figure 2).

The influence of livestock on biodiversity is not restricted to pressures as several types of benefits also exist. Indeed, pressure and benefits are often two sides of the same coin. For almost every category of biodiversity pressure identified in Figure 2, environmentally sound livestock production practices can lead to a corresponding benefit.

In this section, we detail the specific categories of pressures and benefits that link livestock to biodiversity. For each category, the following structure is used:

Context – provides key elements on the environmental mechanisms linking (i) livestock production and the pressure/benefit category and (ii) the pressure/benefit category and biodiversity. These relationships have already been described; for more detail, see cited references or Steinfeld *et al.* (2006).

Scope – discusses the relative importance of the pressure/benefit category among the different global regions and livestock production systems.

Indicators – gives the main examples of indicators existing to describe the pressure/benefit category.

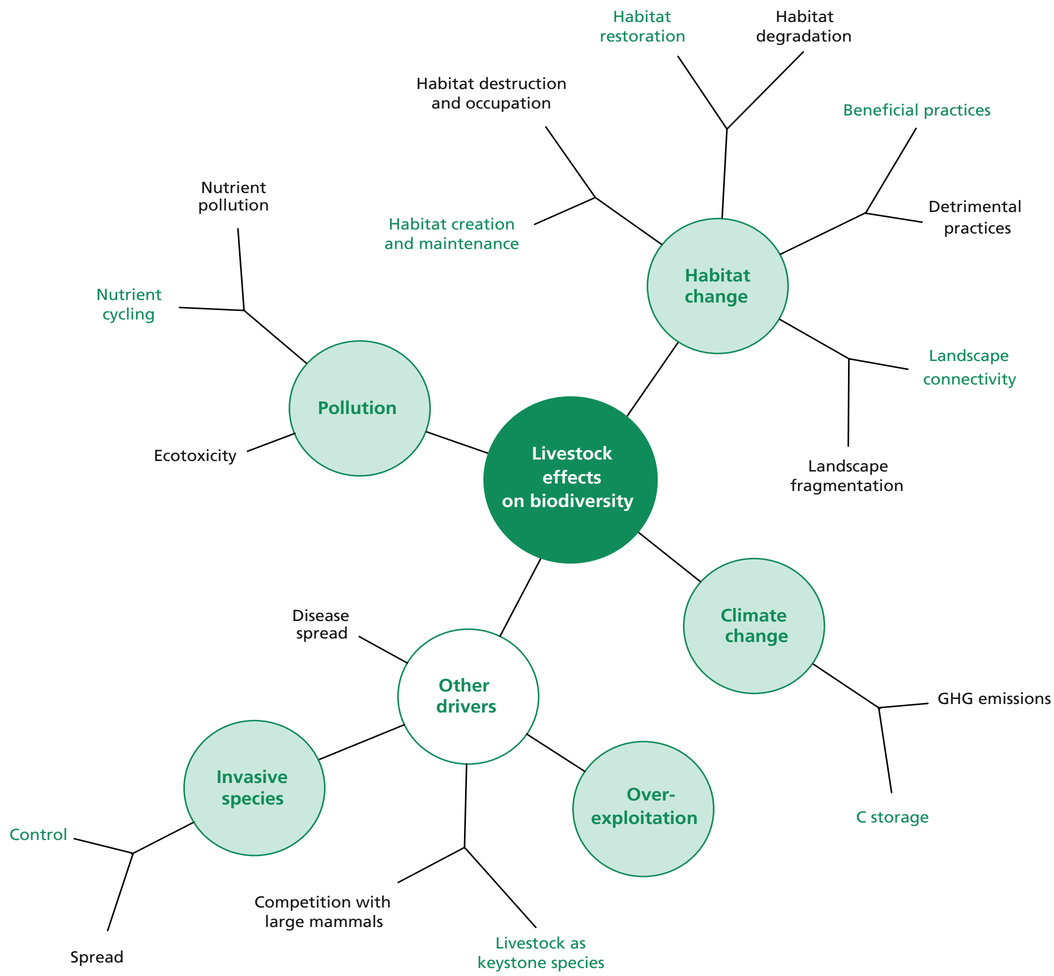
Data availability – reviews the data potentially available to compute indicators of the pressure/benefit category at large (global) scale.

4.1 HABITAT CHANGE

4.1.1 Pressure: habitat destruction and fragmentation

Context – Livestock production has a strong impact on land cover and land use change, driven by both global factors (e.g. demand, market opportunities, policy interventions, climate) and local ones (e.g. resource scarcity, social organization) (Lambin *et al.*, 2003; Wassenaar *et al.*, 2007). Changes in land use can lead to the destruction or modification of biodiversity. One striking example is the conversion of the Amazonian rainforest to pastures and arable crops for animal feed. In the Amazon, pasture is the predominant new land use in the deforested regions, representing 85 percent of all agricultural lands (Steinfeld *et al.*, 2006). Deforested land area in the Brazilian Amazon totalled 58.7 million ha in 2000 (Kaimowitz *et al.*, 2004). Soybean has also been a driver of deforestation, mainly due to increased global demand. Land for soybean production more than doubled between 1990 and 2010 in Brazil (Boucher *et al.*, 2011). Although livestock production is the main driver of deforestation in the Brazilian Amazon, biofuel crop production and the illegal timber industry also play their part.

Figure 2
Overview of the categories of influences that livestock have on biodiversity. The five main drivers of biodiversity loss recognized by the Millennium Ecosystem Assessment (2005) appear in green circles. However, for most of these drivers, livestock can either put pressure (brown) or provide benefits (green) to biodiversity.

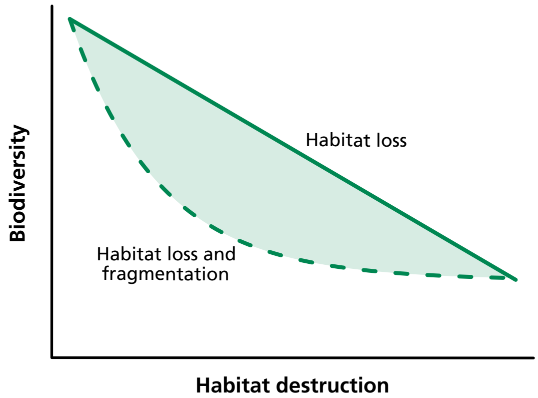


There is often a strong, linear correlation between habitat destruction and biodiversity loss (Figure 3). Amazonian rainforests are biodiversity hotspots, and they may host up to a quarter of the world's terrestrial species (Dirzo & Raven, 2003). A mass extinction of species is projected if deforestation rates are not contained (Soares-Filho *et al.*, 2006; Wright & Muller-Landau, 2006). In terms of ecosystem services, Amazonian rainforests account for about 15 percent of global terrestrial photosynthesis, they represent a considerable carbon sink, and the evaporation and condensation over Amazonia influences global atmospheric circulation (Grace *et al.*, 1995; Field, 1998; Werth, 2002).

Most of the pasture expansion into tropical forest occurs in a diffuse manner, causing fragmentation of the original forest habitat (Wassenaar *et al.*, 2007), which exacerbates the negative impact on biodiversity (Figure 3). Fragmentation leads to

Figure 3

Theoretical effect of habitat loss and fragmentation on biodiversity. The negative effects of habitat loss on biodiversity are exacerbated by fragmentation, i.e. for the same area of habitat loss, negative effects on biodiversity are more important if the remaining habitat is fragmented.



Source: adapted from Andren & Andren (1996)

smaller and more isolated patches of the original habitat, for which the island biogeography theory predicts reduced number of species (MacArthur, 1967; Levins, 1969). While limiting the number of species (as also shown by the species-area relationship, Connor & McCoy, 1979), small habitat size also increases local species extinctions, while isolation is acting as an obstacle to colonization. In practice, evidence of reduced biomass (Laurance, 1997) and species loss in fragmented habitats has been widely described (review in Turner 1996).

Since 2004 in Brazil, enforcement of laws, interventions in soy and beef supply chains, expansion of indigenous reserves, and the creation of protected and sustainable use areas have contributed to a 70 percent decline in the annual deforestation rate (Nepstad *et al.*, 2014). More than 85 000 km^2 of Amazon rainforest were saved, with a consequent strong reduction in GHG emissions (Boucher *et al.*, 2014) while at the same time Brazil was able to increase its beef and soybean production. Particular practices such as sylvo-pastoralism can also contribute to preventing further deforestation and provide additional benefits for biodiversity conservation and economic profitability (McDermott & Rodewald, 2014; Paciullo *et al.*, 2011; 2014). Mechanisms to fight deforestation in Latin America also exist in soybean-importing countries. For instance, several European dairy companies have committed to the Roundtable on Responsible Soy certification (2014), which encourages responsible soybean production so as to reduce its social and environmental impacts while maintaining or improving the economic returns for producers.

Scope – Today, deforestation driven by livestock production mainly occurs in humid regions, with some other regions such as Australia (Bradshaw, 2012) and sub-Saharan Africa (Davidson *et al.*, 2003) also involved. The main drivers differ from one region to another (Boucher *et al.* 2011). Deforestation for pastures and soybean cultivation is predominant in Latin America, which is why the Brazilian

Forest Code now includes interventions that specifically target soy and beef supply chains. Logging is the main driver in Africa and Southeast Asia, and palm plantations in Indonesia and Malaysia, where palm kernel cakes and other palm oil by-products are made into feed for livestock. The palm oil industry is also targeted by sustainability initiatives such as the Roundtable on Sustainable Palm Oil (2014) which has developed a Certified Sustainable Palm Oil standard.

Pastures cover 70 percent of global agricultural land (Steinfeld *et al.*, 2006) and most of the deforested areas. Among livestock production systems, grass-fed ruminant systems are globally major users of land and potentially make a significant contribution to deforestation. Ruminants are able to digest cellulose in grass, which is inedible to most other species, and convert it into meat and milk that humans can eat. This makes it possible to make use of large areas of rangeland that would be too dry or infertile for growing crops. The conversion of grass to beef is an inefficient process, however, meaning that grazed beef production requires a greater land area than food produced from monogastric species such as poultry or pork (Wirsenius *et al.*, 2011). However, in some cases, grass-fed ruminants are the only way to maintain semi-natural habitats in the face of two opposing pressures – conversion to arable land or abandonment leading to conversion to forests – which can both result in biodiversity loss (details in Section 4.1.2). Because soybeans are also grown on deforested areas, intensive systems and species other than cattle can also play a part in deforestation when they use this type of feed crop.

Deforestation is not the only type of habitat destruction caused by livestock production. In certain regions such as North America or Europe, grassland is being turned into arable land to produce feed crops, among other crops. (Gibson, 2009). Urban sprawl encroaching on agricultural land is another type of habitat destruction. Although not directly driven by livestock production, it can displace agricultural production (onto new lands in the same country or even in other regions of the world) and lead to further habitat loss (Paül & Tonts, 2005; Chen, 2007, Wang *et al*. 2014).

Indicators

- Habitat destruction can be calculated as a rate of conversion over time. The rate of deforestation in the Amazon has been estimated by several studies (Skole & Tucker, 1993; Achard *et al.*, 2002; Kaimowitz *et al.*, 2004). A similar metric – the trend in habitat/ecosystem extent – is part of the headline biodiversity indicators of the CBD (2006).
- Alternatively, the extent of original habitat remaining at time *t* can be used to describe habitat destruction in a static framework. Such metrics have been applied to tropical rainforests but also to European forest/agriculture mosaics (Heikkinen *et al.*, 2004; Radford *et al.*, 2005). Habitat/ecosystem extent is a core indicator of the EEA (2007).
- A wide range of metrics exist to describe habitat fragmentation (Turner, 2001). The patch size is a simple metric but it has been shown to be important for species diversity. Because edge effects can be important for certain organisms, the patch shape can also be calculated (e.g., ratio between perimeter and area). Isolation and connectivity also relate to fragmentation although they are more complex to describe. They can be computed from the distance between patches and take into account the "friction" (i.e. the resistance to the movement of a given organism) of the matrix composition (e.g. Sutcliffe *et al.*, 2003).

Table 1: Examples of global and continental land cover datasets.

Data	Based on	Year	Resolution	Nr. of classes
GLC	SPOT VEGETA-TION	2000	0.05^{O}	23
Global Map	MODIS	2003	1km	20
IGBP Land Cover	NOAA-AVHRR	1992-1993	1km	17
Corine Land Cover (only Europe)	LANDSAT	2000, 2006	25ha	44
Global Forest Cover Change 2000 – 2012	Landsat, ETM+	2000, 2012	30m	4

Data availability – Several free datasets of land cover are available at the global or continental level (Table 1). These datasets can be used to compute habitat extent. However, they are not directly comparable because they use different classifications and none of them provide the long time-series needed to compute rates of habitat destruction. Their resolution – although quite fine considering that the global scale is covered – is also too coarse to describe the scale at which fragmentation mechanisms take place. They often include a fewer number of land cover classes, which makes it impossible to distinguish between fine land use categories (e.g. different grassland types or cropland intensities).

4.1.2. Benefit: extensive use, habitat creation and maintenance

Context – Europe has a long history of farming, which has enabled a large pool of species to become specialized and to adapt to agricultural land uses (Bignal & McCracken, 2000, Benton *et al.*, 2002). Today, extensively managed, permanent grasslands are among the habitats with the highest biodiversity. Baldock *et al.*, (1993), and Bignal & McCracken, (1996), estimated that more than 50 percent of Europe's most highly valued biotopes for biodiversity occur in low-intensity farmland. Without livestock, semi-natural grasslands would be lost to ecological succession habitats of lower conservation value; these open habitats would gradually "close" into shrubland and ultimately forest – habitats with lower conservation value. Such habitat loss involves the disappearance of many specialized species. In certain areas such as Eastern Europe, abandonment of agricultural activities can be as threatening to biodiversity as agricultural intensification (Verhulst *et al.*, 2004). Steppe-like grassland in Eastern Europe is a regional biodiversity hotspot that hosts an extremely high diversity of endemic plant and arthropod species and is considered a refuge for many threatened open-habitat species (Cremene *et al.*, 2005). In both Eastern and Western Europe, calcareous grasslands are extraordinarily species-rich but are particularly threatened by the abandonment of grazing activities (Poschlod & WallisDeVries, 2002). In the Mediterranean region, grassland-shrubland mosaics also suffer from the abandonment of traditional grazing activities, resulting in a loss of species diversity and endemicity (share of species native to the area) (Verdu *et al.*, 2000; Plieninger *et al.*, 2014). Abandonment in Mediterranean grassland-shrubland mosaics also threatens species with patrimonial value such as the Iberian lynx (Palomares, 2001). The ecological optimization of the grazing process is tightly related to following the growing times of the vegetation, which is best done by mobile livestock (Manzano Baena & Casas, 2010, McAllister 2010).

The maintenance of European grassland habitat and its rich biodiversity depends on its wide use for extensive livestock production (moderate grazing, mowing)

(Watkinson & Ormerod, 2001). Moreover, moderate grazing can have a direct positive impact on a variety of taxa (plants, arthropods, birds, (Verdu *et al.,* 2000; Watkinson & Ormerod, 2001; Donald *et al.,* 2002), and can increase species richness in vegetation (WallisDeVries *et al.*, 2002). Livestock can eat competitively dominant grassland species which make it possible for rarer species to persist (Olff & Ritchie, 1998). Livestock creates small disturbances across the landscape, facilitated by trampling effects and deposition of dung and urine. It favours intermediate disturbance and heterogeneity, which enhance species diversity (Benton *et al.*, 2003; Bakker *et al*., 2006; Olofsson *et al*., 2008; Dumont *et al*., 2012). Certain traditional cultural practices, such as customary water infrastructure, promote the habitat of organisms such as amphibians (Canals *et al.*, 2011) that are of high conservation value.

The European Union (EU) recognizes that certain farming systems – extensive livestock production in particular – have a high biodiversity value. The European Environment Agency has put time and resources into characterizing and mapping High Nature Value (HNV) farmland (Baldock *et al.*, 1993; Beaufoy *et al.*, 1994; Pointereau *et al.*, 2010). The presence of permanent grassland maintained by extensive livestock production plays a key role in defining such HNV areas. At the policy level, the EU has developed Agri-Environment Schemes (AESs): these offer subsidies to farmers, based on voluntary compliance, for adoption of management practices that reduce environmental pollution and preserve biodiversity and landscapes. Many AESs promote biodiversity in livestock farms, e.g. maintenance of permanent grasslands, reduced fertilization or stocking rates (see Table 5). More generally, AESs are part of the rural development goal of the EU Common Agricultural Policy. Other measures include subsidies to agricultural activities in "less favoured areas" where abandonment of farming would pose a threat to both rural development and biodiversity.

Research into livestock's beneficial effects on biodiversity has also been conducted outside Europe. In China, Akiyama & Kawamura, (2007), report situations where moderately grazed sites have higher species diversity than either heavily grazed and non-grazed sites. Work on the grazed steppes of Inner Mongolia also demonstrates the beneficial effects of moderate grazing, with severe degradation following inappropriate grazing levels (Ren *et al*., 2008, Renzhong & Ripley, 1997). In tundra ecosystems, more productive and resilient grassland is created and maintained by large herbivores (Van der Wal, 2006). In Bolivia, biodiversity loss has been observed following supression of traditional grazing by domestic camelids in high-conservation-value, high-altitude meadows, which has led to the restoration of grazing practices (WISP 2008). In Australia, ecosystems that have evolved with fire can benefit from livestock grazing. When used strategically, livestock maintain biodiversity values by influencing vegetation structure and composition to create habitats for particular plants or animals, and by maintaining fire regions (Adler *et al.,* 2001, Lunt & Territory, 2005). In the United States, the effects of grazing on biodiversity in rangelands have been extensively studied. Grazing impacts on the biomass, composition and structure of vegetation, reducing thatch and vegetation height to support native annual forbs (Hayes & Holl, 2003). This in turn modifies the habitat and affects other taxa such as grassland birds (Rao *et al.*, 2008). At ecosystem level, Chaplin-Kramer *et al.* (2011) recently highlighted how crucial rangeland beef systems are for supplying pollination ecosystem services to adjacent agricultural fields. Rangelands provide foraging and nesting habitats to several species of wild bees, which contribute to 15.3% of total pollination services in the

US, valued at $3.07 billion in 2003. Beside pollination, rangelands in the US and elsewhere provide many types of ecosystem services such as safe food, higher water yields and clean water supplies (Pyke & Marty 2005), carbon sequestration, fire control (Nader *et al.*, 2007), cultural services (Havstad *et al.*, 2007, see also Section 1.2). Livestock production managed sustainably is crucial in order to prevent rangelands encroachment, erosion and degradation.The presence of livestock maintains high levels of biodiversity, ensures the healthy functioning of rangeland ecosystems.

Scope – Positive effects of livestock on biodiversity mainly concern extensive grazing systems (see details in the Context paragraph above). Semi-natural grasslands have replaced previously forested areas after deforestation in many areas of the world (Watkinson & Ormerod, 2001). When semi-natural grasslands are old, it means there has been sufficient time for species to adapt, and that grazing by livestock is the only way to maintain them in the unique biodiversity of their habitat.

Natural grasslands also occur extensively around the world, as shown by the World Wide Fund for Nature (WWF) ecoregion map, classified by biome type. Natural grasslands typically occur in areas with long dry seasons such as African savannahs (Watkinson & Ormerod, 2001) inhabited by wild species of grazing herbivores. In such areas, grazing livestock herds are not required to maintain natural grassland and indeed sometimes compete with wild herbivores (Homewood *et al.,* 2001, Madhusudan, 2004 – see also Section 4.4.4). However, careful livestock management can also lead to neutral or positive effects on biodiversity (DuToit & Cumming, 1999). Depending on the type of management, livestock may determine oak savannah degeneration, as in the case of sedentary herds (López-Sánchez *et al* 2014), or oak savannah sustainability, as in the case of seasonally mobile herds (Carmona *et al.,* 2013).

Indicators

- Area of semi-natural grassland or rangeland can be used as an indicator of the positive effects of livestock, when it is a key biodiversity habitat.
- Similarly, the proportion of farm area with field margins (or other agro-ecological infrastructures), or the relative area on which biodiversity-friendly practices such as late grazing or exclusion of animals at flowering peak are applied, can be used as an indicator of the compliance of farm management with biodiversity conservation.
- Within semi-natural grassland, those practices that promote biodiversity, such as moderate livestock density and absence of fertilization can also be measured. Based on the literature and expert interviews, Plantureux *et al.* (2014) proposed a set of practice-based response indicators for evaluating the impact of grassland management on butterflies, moths, bumblebees, domestic bees, grasshoppers, spiders and earthworms at the plot level. The methodology combines multi-criteria decision trees with fuzzy partitioning, and makes it possible to deal with different types of information (qualitative or quantitative, moderately accurate knowledge). Biodiversity indicators were calculated from simple and easily accessible input variables: management practices, sward botanical composition, and to some extent plant functional traits and Ellenberg indices. The e-Flora-Sys website (e-Flora-Sys, 2014) was used for mean functional traits and other plant species characteristics, such as whether they were food resources for the different insect taxa.
- Management type, where mobility incorporates an important sustainability factor (Carmona *et al.*, 2013; Manzano Baena & Casas, 2010).

- In order to measure the positive effects of livestock on biodiversity, state indicators (e.g. grassland species richness, water provision in a rangeland) can be very useful. They are described in Section 5.

Data availability – Differentiating between semi-natural grassland (benefiting biodiversity) and heavily grazed, artificial grasslands (detrimental to biodiversity) is not possible through global land cover datasets such as those cited in Table 1. However, comparing these sources with data on livestock density (Section 4.1.5), fertilization (Section 4.2.1) or potential vegetation maps could make it possible to identify the areas of semi-natural grassland and rangelands benefiting biodiversity.

4.1.3 Pressure: habitat degradation

Context – Habitat destruction driven by livestock mainly concerns the conversion of forest to agricultural land uses. In contrast, inappropriate grazing management in existing pastures can be responsible for slower processes that result in the degradation (as opposed to destruction) of habitats. The main processes of habitat degradation are desertification and woody encroachment (Asner *et al.*, 2004). They both result from a combination of factors, including overgrazing (i.e. when livestock density exceeds the carrying capacity of the rangeland), climatic factors, and changes in fire regime (Table 2). Desertification concerns arid and semi-arid rangelands where excessive grazing combined with climatic factors (e.g. drought, large fluctuations of temperature, strong wind) decrease the herbaceous cover and increase bare soil. Woody encroachment takes place in semi-arid rangelands: excessive grazing and fire suppression shift the equilibrium in favour of woody species and ultimately turn grasslands into woodlands. It also occurs in semi-arid grazed woodland, woody proliferation being particularly important in the grazed savannahs (Burrows *et al*., 2002). In Australia, ecosystems evolved with fire, and the change in fire regimes associated with the introduction of livestock grazing has significant impacts on biodiversity (Perrings & Walker, 1997; Bowman & Murphy 2010). Conversely, in temperate eco-regions where forest is the potential vegetation, livestock grazing is important to prevent reversion to forest (details in Section 4.1.2). It maintains open grassland habitats and has an important role in fire suppression.

In arid regions, desertification is associated with three main processes: increase in bare soil, decrease in herbaceous cover and increase in woody shrub clusters (Asner *et al.*, 2004). It thus leads to a loss of biodiversity because a few dominant woody species replace a richer pool of herbaceous species (Milton & Dean, 1995). Desertification

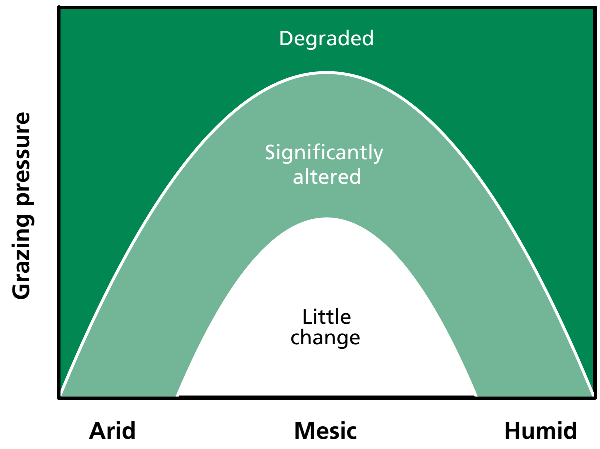
Table 2: Pathways of habitat destruction and degradation across global bioclimatic conditions.

Deforestation (humid climate)	Woody encroachment (semi-arid climate)	Desertification (arid climate)
Forest	Grassland	Grassland & Steppe
↓	↓	↓
Clear cutting, grazing, poor soils	Heavy grazing, fire suppression, drought	Heavy grazing, drought
↓	↓	↓
Grassland & pasture	Savannah & woodland	Desert shrub

Source: adapted from Asner *et al.* (2004)

Figure 4

Effect of grazing pressure on habitat degradation across global bioclimatic conditions.



Source: adapted from Asner *et al.* (2004)

and the associated biodiversity loss is a concernnot only in Africa but also in other arid regions of Australia and North America. About 2.1–2.6 million km^2 are also affected by desertification in northern China (Yang *et al.*, 2005). Increased grazing pressure has led to a substantial reduction in soil cover in the Inner Mongolian grassland (Renzhong & Ripley, 1997). Much of this steppe ecosystem now demonstrates severe degradation (Renzh *et al.*, 2008).

In semi-arid regions, overgrazing of herbaceous species favours woody species. Associated with fire suppression, it enhances the survival of woody plants and leads to woody encroachment (Asner *et al.*, 2004). It is clear that woody encroachment modifies the species composition, but a net negative effect on species richness is less clear than in the case of desertification. However, it does modify key ecosystem functions such as nutrient cycling, biomass production and soil and water conservation (Steinfeld *et al.*, 2006). Overall, it tends to reduce the quality of land for animal production (Schlesinger *et al.*, 1999).

Scope – Among systems, those relying on grazing make the most important contribution to habitat degradation, with overgrazing playing a leading role.

Among geographical areas, all ecosystems do not have the same sensitivity to grazing pressure and habitat degradation. The difference is made by bioclimatic and edaphic conditions (Figure 4). The grazing intensity at which the habitat becomes degraded is lower in arid and humid conditions than in temperate climates. Grazed areas are more vulnerable to degradation pressure at the extremes of the climatic gradient (Table 2). Milchunas & Lauenroth (1993) propose that the relationship between grazing and biodiversity is also a function of the evolutionary grazing history of an ecosystem. In a global analysis, Díaz *et al*., (2007) concluded that grazing history, along with climate, was essential for understanding the functional response of plant communities to grazing.

Indicators

- The Normalized Difference Vegetation Index (NDVI) is an indicator that can be remotely sensed by satellites measuring wavelengths of the light absorbed and reflected by vegetation. It gives an indication of the vegetation state of an ecosystem (Alcaraz *et al.*, 2006) and can thus be used to characterize habitat degradation (e.g. in Jepson 2005, Thompson *et al.,* 2009). As a measure of the state of vegetation, its computation, use and limitations are detailed in Section 5.4 on state indicators.
- Recent studies have used the ratio of net primary production to rainfall, or rain-use efficiency (RUE), to map the occurrence and severity of land degradation. RUE is used for this purpose because it relates plant productivity to rainfall, which is a primary factor controlling plant growth. Plant productivity may be assessed by mapping vegetation cover using satellite images (e.g. with the NDVI) (Bai *et al*., 2008, Prince *et al.,* 1998).
- Overgrazing can also be used as an indicator of habitat degradation pressure. MacLeod (2011) computed a measure of grazing pressure corresponding to the ratio between forage yield and livestock feed demand. Livestock demand was based on the live weight of the grazing animals.
- The OECD (2001) proposed several indicators of soil degradation. Soil degradation is part of the habitat degradation process. It is accompanied by biological (decrease in organic matter content and in soil biodiversity), chemical (salinization, acidification) and physical (erosion, compaction) degradations. The OECD indicators measure erosion risk (by water and by wind), acidification, salinization, compaction, fertility and chemical pollution.
- The UNEP Global Assessment of Human-induced Soil Degradation (GLASOD, 2014)[1] project has produced a world map of the status of soil degradation, based on expert judgment. Four major degradation types are considered: water and wind erosion, and physical and chemical deterioration. Limitations include the low resolution and the data year (1990).

Data availability – Unlike habitat destruction, habitat degradation is not a process that immediately translates into land cover changes. Characterizing it with indicators is less straightforward and cannot be achieved with large-scale land cover data whose categories are too coarse to describe a gradient of degradation. Section 4.1.5 details data that could describe overgrazing and Section 5.4.1 lists data sources for computing NDVI. One limitation of remotely sensed NDVI data is that they can be too technically demanding to be used by relatively small institutions. In addition, there may be a mismatch between the spatial scale of data availability, which could be too large compared to the spatial scale required for an environmental assessment or for decision making.

4.1.4 Benefit: habitat restoration

Context – In addition to clearing shrubs and trees, the reintroduction of livestock grazing can be used to restore abandoned grasslands and the high biodiversity levels associated with these open habitats (see Section 4.1.2). Some instances of restoration through livestock grazing have resulted in increased species richness of vascular plants (Pykälä, 2003) or arthropods (Pöyry *et al.*, 2004). But restoration of abandoned

[1] http://www.isric.org/projects/global-assessment-human-induced-soil-degradation-glasod

grassland is not always successful in recovering high species richness so that preventing abandonment may be a better solution than restoration (Muller *et al.*, 1998).

Inadequate livestock management (e.g. overgrazing) in combination with ecological and pedo-climatic factors can lead to grassland/rangeland degradation (Section 4.1.3). Conversely, if managed well, livestock can prevent degradation or promote restoration. Rational livestock management or rotational grazing in the initial stages of grassland degradation can be a viable option (Yong-Zhong *et al.*, 2005; Zhou *et al.*, 2005). However, depending on the context, removal of livestock for several years can be necessary for restoration.

Scope – Livestock are a relevant tool for restoration in all grassland and rangeland areas. In areas affected more strongly by climate change, there is an emerging tension around promoting the restoration of the initial vegetation state which may not be adapted to the evolution of climate. In Australia, several years of extreme variability have highlighted the need to be flexible as local provenance species are dying. Recent studies detailed how land managers could select species for revegetation based on the expected climate in the future (Booth & Williams, 2012a; 2012b). It may be important to allow restoration of 'function' rather than particular species.

Indicators and data availability

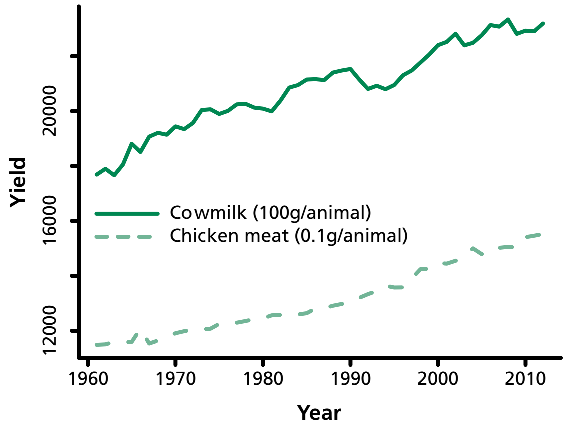
- For the restoration of abandoned grassland, state biodiversity indicators (Section 5) will be the most useful in order to ensure that the reopening of the habitat does result in biodiversity gains. Indicators of low management intensity (Sections 4.1.5, 4.1.2) can also be used.
- At local scale, restoration of degraded grasslands is often an active process and progression towards the target needs to be monitored. Akiyama & Kawamura (2007) proposed several indicators to monitor management and restoration, including measures based on spectral reflectance, soil respiration, indicator plant species and links between grazing pressure and species distribution.
- At large scale, see Section 4.1.3 for indicators and data reflecting the degradation-restoration gradient.

4.1.5 Pressure: detrimental practices and intensity

Context – Agricultural intensity can be defined as increased utilization or productivity of land (Netting, 1993); either output-oriented (production) or input-oriented (utilization) measures can therefore be used to describe it (Turner & Doolittle, 1978). In terms of output, livestock production yields have greatly increased during the past decades (e.g. cow milk and chicken meat, see Figure 5). Increased yields have led to higher per-capita production, reduced hunger, and improved nutrition (Tilman *et al.*, 2002; Bank, 2007). Yield increases have been made possible by the adoption of highly productive breeds and by mechanization, which boosted work efficiency. They have also been accompanied by more intensive use of inputs in feed crops (e.g., fertilizers, pesticides, irrigation water) and for animal production (e.g., energy, veterinary products, concentrate feeds) (Tilman *et al.*, 2002).

The changes associated with intensification have had major effects on biodiversity. The higher use of inputs (pesticides and fertilizers in particular) has had direct negative effects on plant and arthropod species, decreasing both their diversity and their abundance (McLaughlin & Mineau, 1995; Firbank *et al.*, 2008). However, species loss from anthropogenic eutrophication can be improved to some extent in

Figure 5
Evolution of the global cow milk and chicken meat yield.



Source: data from FAOSTAT (2013)

grasslands by using grazing animals to crop fast-growing grasses, increasing the availability of ground-level light for other plant species (Borer *et al.,* 2014). Decrease of plant and arthropod abundance causes, in turn, a decline of species at higher trophic levels (e.g. birds) which depend on arthropods and plants for food resources and habitat (Fuller & Gregory, 1995; Robinson & Sutherland, 2002). Large-scale monitoring of bird populations strongly evidences this decline, showing how specialists of the farmland habitat have been particularly affected compared to species living in other habitats (Gregory *et al.*, 2005, Jiguet *et al.*, 2011).

Livestock have an effect on grassland biodiversity (plants and fauna) as they shape sward structure and composition as a result of selective grazing. Grazing impact varies according to livestock species and interacts with sward productivity and stocking rates (Öckinger *et al.*, 2006; Sebastià *et al.*, 2008; Dumont *et al.*, 2011; 2012). In semi-natural pastures with moderate or low soil fertility, increasing stocking rates can also have detrimental effects on biodiversity (Pöyry *et al.*, 2006; WallisDeVries *et al.*, 2007; Dumont *et al.*, 2009), as does grazing abandonment at the other end of the disturbance gradient (Loiseau *et al.*, 1998; Sebastià *et al.*, 2008; Tocco *et al.*, 2013: details in Section 4.1.2). Increasing the grazing stocking rate in improved grassland also has direct adverse effects on birds, such as disturbance or nest destruction by trampling (Paine *et al.*, 1996; Sabatier *et al.*, 2014).

The process of agricultural intensification can be accompanied by habitat destruction (e.g. forests converted to grassland or grassland converted to cropland, (Wassenaar *et al.*, 2007; Ogutu *et al.*, 2011). Here, we differentiate between the intensity pressure and the habitat destruction pressure (Section 4.1.1). We define intensity as relating to practices and input use within the same land use, without involving land use transformation. Unlike habitat destruction, intensity pressure is reversible over a short timescale without long-term ecological restoration.

Scope – The adverse effects of production intensity on biodiversity concern all livestock species and a wide range of systems. While extensive systems can benefit biodiversity, limited intensity increases in grazing systems such as grassland fertilization can lead to biodiversity damage (Vickery *et al.*, 2001; Kleijn *et al.*, 2009). At the other extreme of the gradient, intensive landless production systems (e.g feedlots in the United States, intensive pig farming in Europe) also involve biodiversity changes through manure management and intensive cultivation of feed crops.

During the past decades, intensification has been greatest in developed countries. In these countries, very few or no pristine habitats remain and conversion to agricultural land use rarely occurs, meaning that intensity has been the predominant pressure category, rather than habitat destruction. In Europe, intensification directly threatens the pool of species specifically adapted to old, extensive agricultural habitats (Benton *et al.*, 2002; Kleikn *et al.*, 2009; Poschlod & WallisDeVries, 2002; Kleijn *et al.*, 2009). More recently, livestock production has also undergone important intensification in rapidly developing countries (e.g. China, Brazil). Intensity may thus place increasing pressure on biodiversity in those countries. However, policies that foster resource-efficient intensification with high levels of agricultural knowledge, science and technology could reduce the overall pressure on rangeland biodiversity, in particular in Africa (Alkemade *et al.*, 2012).

Unlike intensity, which represents a form of pressure on biodiversity, the concept of sustainable intensification could benefit biodiversity. Its objective is to increase production from the same area of land while reducing environmental impacts (The Royal Society 2009; Godfray *et al.*, 2010). It mainly relies on technological solutions to increase resource-use efficiency and mitigate negative externalities. By increasing yields in already cultivated areas, sustainable intensification could ease the pressure to convert new land to agriculture, and spare land for nature (Borlaug, 2007; Phalan *et al.*, 2011). Whether increased yield do spare land is subject to debate, however (Perfecto *et al.*, 2009; Ewers *et al.*, 2009; Rudel *et al.*, 2009). Moreover, it has been argued that the concept of sustainable intensification often focuses on yields and technical solutions, and that this narrow definition lacks engagement with the key principles of sustainability (Loos *et al.*, 2014).

Indicators

- As a direct output-oriented measure, yield has been used to describe farming intensity. Donald *et al.,* (2006) showed how farmland bird populations were negatively correlated to cereal yields across European countries. Yield can be computed with different functional units – e.g. output *per* unit area – although area seems particularly relevant to biodiversity, which needs land for habitat and resources.
- The Human Appropriation of Net Primary Production (HANPP) is also an output-oriented measure of agricultural intensity. It reflects the reduction of energy availability in natural ecosystems and could be an important pressure on biodiversity (Vitousek *et al.*, 1986; Wright, 1983). The species-energy hypothesis states that the energy available in ecosystems is a factor determining species diversity (Wright, 1983). Haberl *et al.,* (2005) showed a negative relationship between HANPP and the species diversity of birds. HANPP has been computed for terrestrial ecosystems at global scale (Haberl *et al.,* 2006).
- Livestock density (expressed as animals, live weight or livestock units per unit area) has been used as a proxy for agricultural intensity. It relates to output intensity because at large scale, higher livestock densities are found in systems with higher productivity. It also relates to input intensity because increased density of animals is associated with higher grassland fertilization and nutrient excretion (Herzog *et al.*, 2006). The adverse effect of high livestock densities

on biodiversity have been widely evidenced (Fleischner, 1994; Dorrough *et al.*, 2004). At smaller scale such as within natural, unfertilized grasslands or rangelands, small variations of livestock density may not correlate with agricultural intensity.

- Input-oriented intensity measures compute the ratio between various categories of inputs (e.g. pesticides, fertilizers, but also water or energy) and area. Some, such as the nitrogen input/ha, have been widely used to describe intensity (Atkinson *et al.*, 2005; Billeter *et al.*, 2008; Durant *et al.*, 2008; Kleijn *et al.*, 2009). Input-oriented intensity measures are useful because they reflect management practices that have a direct impact on biodiversity, while output-oriented measures like yield correlate with these management practices, but also depend on pedoclimatic conditions.
- More complex intensity indicators have been proposed; they aggregate different categories of inputs (Teillard *et al.*, 2012) or intensity components (Herzog *et al.*, 2006; Pointereau *et al.*, 2010).

Data availability – Data on inputs are quite rare, however: more datasets exist to describe output-oriented measures of intensity. FAO and other sources provide yield data at global level (FAOSTAT, 2013; LUGE, 2013). Models have also been developed to estimate HANPP at global scale (Haberl *et al.*, 2007). The Gridded Livestock of the World (GLW, 2007; also refer to http://livestock.geo-wiki.org/) models the global distribution of livestock at global scale at high resolution and can be used to compute livestock densities (Figure 6). As for habitat degradation, the land cover classes of global data do not permit differentiating between several classes of intensity. But although data on inputs are rare, certain countries conduct agricultural censuses which include variables that can be used to compute input-oriented intensity measures. In Europe, the Farm Accountancy Data Network (FADN), a network of farms, provides statistically representative data at infranational (NUTS 2) scale. FADN has already been used to compute input-oriented intensity indicators in various European countries (Reidsma *et al.*, 2006; Teillard *et al.*, 2012).

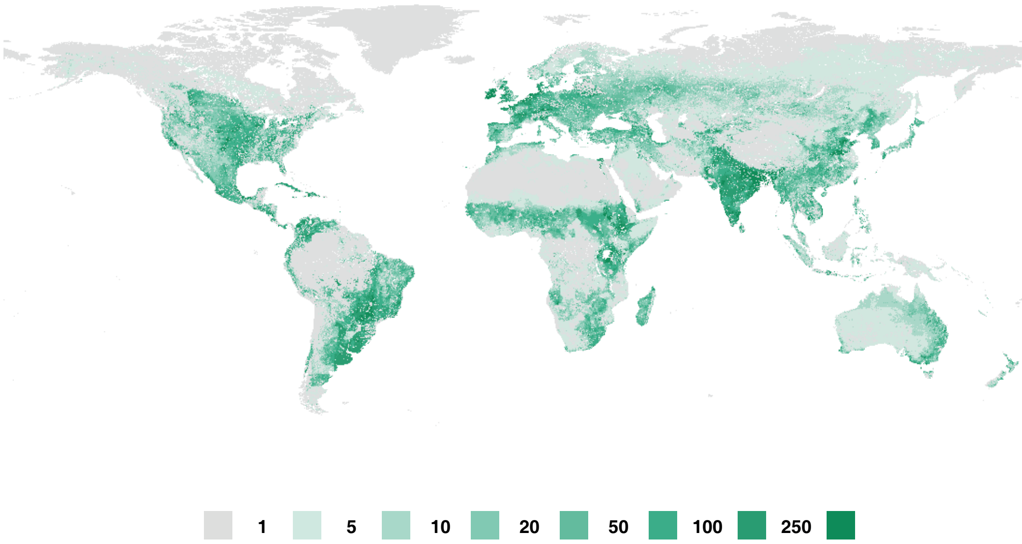
4.1.6 Pressure: Landscape homogenization

Context –Important landscape homogenization has been associated with intensification, increased efficiency of labour, and use of farming machinery (Björklund *et al.*, 1999; Robinson & Sutherland, 2002; Sutherland, 2004).

Two components of landscape heterogeneity are recognized: composition and configuration (Duelli, 1997; Fahrig *et al.*, 2011). A landscape has high *compositional heterogeneity* if it contains a large variety of land uses in even proportions. Spatial arrangement of land uses in a complex pattern leads to high *configurational heterogeneity*. Among all land uses, semi-natural habitats (e.g. grassland or grassy strips, tree or shrub edges, tree clumps) are of particular importance because they offer crucial resources to wild species. A high proportion of such semi-natural habitats relative to agricultural land uses, and their complex spatial arrangement in order to favour connectivity, are a key component of heterogeneity in agricultural landscapes. For many wild species, the quality and diversity of the matrix of agricultural land uses around these semi-natural habitat is also important (Donald & Evans, 2006).

A negative correlation between landscape homogenization and biodiversity has been widely evidenced (Benton *et al.*, 2003). Both the composition and the structure

Figure 6
Cattle density across the globe.



Source: data from GLW (2007), see also Robinson *et al.* (2014)

components of landscape heterogeneity are important to biodiversity. As a consequence, they also influence ecosystem services (Tscharntke *et al.*, 2005).

SCOPE – The relationships between landscape heterogeneity and biodiversity have been extensively studied in Europe but they are of global importance (Fischer & Lindenmayer, 2007). In Africa, studies have concentrated on East African savannahs and on southern Africa (McNaughton, 1976; Sinclair & Norton-Griffths, 1982; Obadi, *et al.*, 2013; Ogutu *et al.*, 2014). In tropical regions, studies show that maintaining heterogeneity in agricultural landscapes could be a way of preserving biodiversity, including forest species (Hughes *et al.*, 2002; Chazdon *et al.*, 2009; Gardner *et al.*, 2009). In Europe, the importance of heterogeneity is recognized at the policy level, with specific Agri-Environment Schemes (AESs) aiming to favour biodiversity in farmlands through enhancement of landscape heterogeneity. In arid and tropical countries, the loss and fragmentation of previously natural landscape may have a stronger effect on biodiversity, compared to the homogenization of recent agricultural landscapes. But striving for high landscape heterogeneity is not relevant to every context. Certain grassland or rangeland environments are historically homogeneous and heterogeneity is associated with disturbance. Such disturbance may be associated with elevated biodiversity coming in from new, and not necessarily desirable, species (e.g. invasive species, non-native species replacing endemic species, generalists replacing specialists). Conservation measures promoting heterogeneity in these historically homogeneous landscapes may even be associated with biodiversity losses (Batary *et al.*, 2007; Batáry *et al.*, 2011).

INDICATORS

- The percentage of semi-natural habitats has been the most widely used indicator to describe composition heterogeneity (Billeter *et al.*, 2008; Batáry *et al.*, 2010), as has been the proportion of arable land (Ekroos *et al.*, 2010; Filippi-Codaccioni *et al.*, 2010). In a few studies, multiple land uses (agricultural and semi-natural) have been considered in a Shannon index or in other diversity indices (Chiron *et al.*, 2010; Pointereau *et al.*, 2010).

- Several indicators exist to describe the configuration component of heterogeneity, such as the probability of adjacency, the spatial Shannon index or the length of edges, the perimeter or perimeter/area ratio, the patch isolation (Turner, 2001; Butet *et al.*, 2010).

Data availability – Experimental studies addressing the effect of landscape heterogeneity on biodiversity usually considered landscape at the scale of a few hundreds to a few thousand meters. Within these landscapes, they described the composition of several land uses or landscape elements and their spatial arrangement. Such description requires data at a spatial resolution that is a lot higher than what is available through global and indeed most global statistics. At a lower resolution of 1×1*km* but on the global scale, Kadoya *et al.* (2011) calculated an index of habitat-type diversity using the open-access *Global* Map data on land cover (International Steering Committee for Global Mapping, 2009). The authors showed that their index correlated well with the occurrence of a birds of prey and with amphibian species richness in Japan. It also correlated with the pattern of traditional and high-conservation-value agricultural landscapes in other countries (e.g. the Iberian Peninsula and Central America).

4.1.7 Benefit: Landscape connectivity

Context – Landscape heterogeneity is a key factor for maintaining biodiversity in agricultural landscapes (Benton *et al*., 2003) and can partly compensate for the negative effects of management intensity (Tscharntke *et al*., 2005). Heterogeneous agricultural landscapes host more semi-natural habitats (e.g. semi-natural grasslands, fragments of forests or tree clumps/isolated trees, tree or shrub edges, grassy strips, ponds). Some species can only persist in patches of semi-natural habitats: their metapopulation dynamics consist of local population dynamics within these patches and spatial dynamics of migrations and colonization among them. For such species, landscape heterogeneity through a higher abundance and connectivity of these semi-natural habitats ensures a viable metapopulation dynamics (Verboom *et al*., 1991; Andren, 1994). Even for these species, the quality of the surrounding agricultural habitat matrix is important for dispersal or because it can be used as a lower quality habitat in some cases (Baillie *et al*., 2000; Donald & Evans, 2006). For other species, the presence of various land uses is key because they play different roles throughout the life cycle (e.g. nesting and foraging habitats [Blomqvist & Johansson 1995]). Heterogeneous agricultural landscapes exhibit higher diversity of land uses, which favour such species.

Landscape connectivity not only depends upon a favourable landscape structure, but also on the conservation of certain ecosystem functions associated with livestock mobility, essentially dispersal and landscape structures like drove roads. Dispersal is necessary for sessile organisms (plants) to respond to environmental changes which are becoming more critical with climate change (Travis *et al*., 2013). Additionally, fragmentation and resulting inbreeding can lead to reproductive depression, further deterioration in dispersal capacity, and local extinction (Mix *et al*., 2006). Long-distance dispersal may be particularly affected and is a key component of the issue (Ouborg *et al*., 2006). Mobile livestock have been identified as a key vector in long-distance dispersal, both for endozoochory (Milton & Dean, 2001; Manzano *et al*., 2005) and for epizoochory (Manzano & Malo, 2006). Drove roads are a general feature of landscapes where mobile livestock are present, their conservation value has been shown for plants both in Australia (Williams *et al*., 2006; Lentini *et al*., 2011) and for Spain (Azcárate *et al.*, 2013a), serving both as corridors through, and as refuges from, the surrounding matrix of intensive land use. They are also valuable for the conservation of birds (Lentini, 2011) and arthropods (Azcárate *et al*., 2013b). Their structure is fractal, network-like (Manzano Baena & Casas, 2010; Lentini *et al.*, 2013), thus fulfilling an important connectivity role that is linked to sustained use, however.

European Agri-Environmental Schemes promote extensification and habitat maintenance (Section 6.1), and also contain measures specifically targeting the enhancement of landscape heterogeneity (Table 5). Although the current effectiveness of AESs is still debated (Kleijn, 2001; 2006; Princé *et al*., 2012), they show that the EU recognizes that farmers can act to promote landscape heterogeneity, with potential key benefits for biodiversity. Studies in Australia have found that remnant vegetation and/or revegetated patches on private land are as effective, and in some case more effective in preserving bird species biodiversity, than the best managed national parks (Rayner *et al*., 2014). Wildlife corridors have been established in some countries with the aim of providing habitat connectivity at large scales (Worboys *et al*., 2010). Through retention, restoration and management of lands that are naturally interconnected, they provide opportunities for conservation and movement of species. Such ecological connectivity and management may involve activities such as: retention of isolated trees on grazing lands, which can act as stepping stones; retaining or restoring natural riparian vegetation and remnant patches through active management; and controlling fire, pest animals and weeds through livestock managers, community groups, governments and others (Fischer *et al*., 2006). Australia initiated a National Wildlife Corridor Plan as a strategy to harness voluntary networks of landowners, communities and organizations to enhance landscape conservation. Besides biodiversity conservation, the objectives included improvements in water quality and greater resilience to climate variability and change, (Worboys & Mackey, 2013).

Scope – See Section 4.1.6. In addition, Couvreur *et al*., (2005) show that livestock have the capacity to disperse most of the plant community in a grassland, confirming how important mobile livestock can be in ensuring dispersal mechanisms that guarantee connectivity. Mobile livestock also play the additional important role of maintaining drove roads. The degree of ecosystem functionality maintained by livestock is given by the scale of movements undertaken by the herd.

Indicators

- Indicators describing the composition and configuration components of landscape heterogeneity are shown in Section 4.1.6.
- At the landscape scale, the existence/enhancement of wildlife/biodiversity corridors linking fragments in the landscape can be used as an indicator. Livestock farmers in Australia are participating in Biodiversity Corridor schemes (2014). In eastern Africa, there are concerted efforts to set up conservancies, or wildlife management areas, connected parks and dry-season ranges for wild herbivores (Galvin and Reid 2014).

- In Europe, Pointereau *et al.* (2010) developed a score reflecting High Nature Value and taking into account three components: land use diversity, landscape elements and low-intensity land uses.
- The development of indicators measuring the scale of herd movement is desirable.

Data availability – See Section 4.1.6

4.2 POLLUTION

4.2.1 Pressure: Soil and water pollution

Context – Nutrient (mainly nitrogen and phosphorus) pollution occurs at several stages of livestock production. Upstream, it is related to the fertilization of feed crops. A striking example of nutrient pollution at this stage is the nitrogen loading in the Mississippi River due to broad fertilizer use in the central US croplands, which are mainly used for animal feed (Donner, 2007). Nutrient pollution can also be important at the farm stages. Nutrient conversion from plant proteins being quite inefficient in the animal rumen, very large amounts of nutrients are concentrated in urine and manure. When urine and manure are not directly excreted on pasture soil, the management of manure and waste waters is a complex issue. Nutrient pollution at both stages (feed cultivation and animal excretion) can lead to nutrient leaching and run-off, mainly in the form of nitrate (NO_3^-) (Rischkowsky & Pilling, 2007).

Fertilization and nutrient loading have a direct effect on terrestrial communities (Billeter *et al.*, 2008; Hellawell 1986; Schofield *et al.*, 1990; Haslam *et al.*, 1990; Withrington *et al.*, 1991). The modification of the community structure, and the decrease in species richness and abundance of these taxa, influence other species groups such as birds which need them as habitat and food resources (Vickery *et al.*, 2001). The negative effects of nutrient pollution are critical for aquatic biodiversity. As nutrients flow from land-based livestock production to lakes, rivers and coastal waters, excess of nutrient loads in water results in increased growth of nuisance species of algae and aquatic weeds. Those species compete with, and are harmful to, other native species of algae. Their senescence and decomposition cause hypoxia (oxygen shortage) and they can release toxins detrimental to various aquatic organisms (Carpenter *et al.*, 1998). Eutrophication has been shown to be a factor in the loss of aquatic biodiversity, including in coral reefs (Seehausen *et al.*, 1997).

Scope – Within extensive systems, manure and urine excretion by grazing animals goes directly to the soil and can lead to nutrient loss and water pollution depending on grazing management, slope, geology and soil cover (O'reagain *et al.*, 2005). In intensive systems, animals and nutrients are concentrated, which leads to important sources of N and P pollution. At country scale, Peyraud *et al.*, (2012) showed the strong correlation between the concentration of animals and nutrient loading in France. At farm scale, however, manure management practices have a strong influence on the extent of nutrient pollution in soil or water.

In terms of habitat change drivers, the relative importance of the different pressure categories varies strongly among regions because original habitats are different. Although the initial concentration of nutrients in the soil also varies among global regions, the effects of nutrient pollution on biodiversity are likely to be more homogeneous.

Indicators

- A simple indicator that is used to describe nutrient pollution is the amount of N or P used for fertilization (Billeter *et al.*, 2008; Kleijn *et al.*, 2009). It only describes nitrogen pollution resulting from that activity; it does not account for other sources of nutrient input such urine and manure.

- A more comprehensive indicator is the nitrogen balance. It is computed as the ratio between N inputs and outputs and makes it possible to differentiate between different sources of N losses. It is one of the 26 indicators identified by the EEA (2007) as part of the the Streamlining European Biodiversity Indicators (SEBI) process.
- “Nutrients in transitional, coastal and marine water” is another EEA indicator. It measures nutrient pollution directly through nutrient concentration; however, this pollution can stem from various human activities and not just livestock production.

Data availability – Some data are available at global scale on N and P fertilization (FAOSTAT, 2013; LUGE, 2013). To be computed, nitrogen balances require information on various inputs, outputs and dynamic processes (e.g. N fixation, livestock manure production). This information is not available at global scale, although certain models could be used to estimate it (e.g. Global Livestock Environmental Assessment Model “GLEAM”, FAO, Gerber *et al.*, 2013).

4.2.2 Benefit: Nutrient cycling

Context – In grazed grasslands, animal excreta are an important part of nutrient cycling (Gibson, 2009), and nutrient loading in grasslands can benefit biodiversity. Diversity is higher in grazed fields, maintained partially by the concentration of nutrients from herbivore urine and dung (Karki *et al*., 2000; Augustine *et al*., 2003). Positive effect in plant growth is much stronger in mobile/migratory systems (Augustine & McNaughton 2006). Herbivores have also been shown to contribute to nutrient cycling in forested landscapes (Murray, 2014). Many livestock systems integrate agricultural and livestock activities because of their well-known mutually beneficial roles. While fallow land provides livestock with dry-season grazing, manure provides fertilization to the extent that farmers pay for this service (Powell, 1986). Additionally, dung beetles and other arthropods have a key role in nutrient cycling, and their presence is triggered by livestock activities (see section 4.4.5).

Scope – Haynes & Williams (1993) estimate that about 85 percent of the total above-ground herb biomass is consumed by livestock in grazed pastures. Although further bacterial activity is needed for full mineralization, humidity conditions of dung are key in that process and can be decisively assisted by arthropod burial (Slade *et al.,* 2016). Manure fertilization is known to be more effective in maintaining the right level of soil organic matter and labile organic matter fractions (e.g. in Yan *et al*., 2007).

Indicators – In the cited literature, calculations exist for the nutrient input from livestock into the ecosystem. Murray *et al*., (2014) offer a recent methodology. In the case of agro-pastoral systems, the services provided by livestock can be calculated in terms of the amount of fertilizer saved or by simple calculations of the excreta production of livestock.

Data – The benefits of nutrient cycling take place mainly in extensive systems and in developing countries (e.g. West Africa, India), where data availability is often lower. In these systems and countries, measuring nutrient cycling benefits can be a lower priority than measuring the effects of agricultural intensification projects. However, simple extrapolations can be made based on the number of livestock.

4.2.3 Pressure: Atmospheric nitrogen pollution

Context – Besides nutrient leaching and run-off in soil and water, livestock production is responsible for emissions of nitrogen gases into the atmosphere. These gases include nitrous oxide (N_2O), nitric oxide (NO) and ammonia (NH_3). Manure management and fertilization (from both synthetic N and manure) are associated with direct volatilization of NH_3 and they represent one of its most important sources (Mosier *et al.*, 1998). With time and aeration, combined nitrification-denitrification of ammoniacal N leads to production and emission of N_2O. Fertilization also stimulates soil microbial activity, which in turn increases nitrogen release (Vitousek & Aber, 1997). In addition, fossil energy use at all stages is associated with N_2O emissions.

Together with these emissions, atmospheric N deposition is also a very important driver of species change (Sala, 2000). N deposition fertilizes soil, which changes the community equilibrium in favour of species adapted to more fertile soil, which in turn result in a net loss of species (McClean *et al.*, 2011). N deposition can lead to eutrophication, or acidification in the presence of water, as well as through nitrification and other processes (Galloway *et al.*, 2004; Bobbink *et al.*, 2010). Eutrophication and acidification have been responsible for changes in community composition and plant species losses (Kleijn *et al.*, 2009; Van Landuyt *et al.*, 2008; Bobbink *et al.*, 2010).

Scope – N deposition tends to be more important in areas with high emissions (Galloway *et al.*, 2004). Production of N_2O from manure largely depends on the system and duration of waste management (Mosier *et al.*, 1998). However, as with diffuse pollution, nitrogen emissions rise as fertilization or manure production increases.

Indicators

- Emissions of nitrogen gases can be computed at different life cycle stages. They are closely related to livestock production activities.
- Indicators can also measure deposition. For instance, the EEA (2007) proposed the "critical load exceedance for nitrogen" as an indicator of pressure on biodiversity. It reflects how, beyond a critical load, nitrogen deposition leads to to harmful effects on biodiversity. As for diffuse nutrient pollution in water, deposition of atmospheric N does not only stem from livestock production.
- Acidification and eutrophication potential can be computed from the emission of the various N and P gases, and using specific factors such as those proposed by Guinée *et al.* (2002) (for more details, refer to Section 10.1).

Data availability – As a greenhouse gas, N_2O emissions are modelled in GHG LCAs, such as in the GLEAM model (Gerber *et al.*, 2013). However, ammonia (NH_3) emissions are not modelled while they predominantly contribute to nitrogen deposition (EEA, 2007) – although indirect N_2O emissions resulting from NH_3 are included in LCA studies based on IPCC guidelines. Other models have been developed to estimate N deposition at continental (Posch *et al.*, 2005) to global (Galloway *et al.*, 2004) scales.

4.2.4 Pressure: Pesticides and other products

Context – Livestock production is responsible for more pollution from synthetic products, which have a direct toxicity on organisms, than from nutrients. Agricultural intensification has been accompanied by an increased use of pesticides in crops, including those crops used as livestock feed (Tilman *et al.*, 2002; Steinfeld *et al.* 2006). In Europe, increase in pesticide use has been responsible for a significant decline in bird populations (Carson, 1962), especially before the ban of the most eco-toxic molecules. Such molecules may still be used in certain developing countries, however. More generally, hormonally active pesticides have adverse effects on a wide range of organisms (Colborn *et al.*, 1993). Other toxic products specifically associated with livestock production, such as veterinary products or hormones, also have direct impacts on biodiversity. A well-known example of such damege is the dramatic decline in vultures feeding on carcasses of livestock treated with the Diclofenac veterinary product in India (Baillie, 2004). Hormones used on livestock have been shown to contaminate water and cause endocrine disruption in fish (Soto *et al.*, 2004).

Scope – Intensive systems are often characterized by higher use of pesticides in feed crops, and veterinary products and hormones on animals. Ecotoxicity does not only depend on the amount of product used: less intensive systems in developing countries sometimes employ very environmentally harmful molecules (Ecobichon, 2001).

Indicators and data availability – The number of applications and the amount of pesticides or other products can be measured. However, few indicators exist regarding pesticides, veterinary products and hormones. No pressure indicators related to these components are part of the key biodiversity indicators of the EEA (2007) or CBD (2006). Data availability on the utilization of these products is low even at small scale. Research exist on the ecotoxicity of different substances (Section 10.4).

4.3 CLIMATE CHANGE

4.3.1 Pressure: GHG emissions

Context – The global contribution of the livestock sector to climate change impacts is significant. Gerber *et al.* (2013) estimated that global GHG emissions related to livestock production accounted for 14.5% of human-induced emissions. This estimate was based on a life cycle assessment and considers all the stages of production. The main sources of emissions are feed production and processing (45%), enteric fermentation from ruminants (39%) and manure storage and processing (10%).

As climate change tends to worsen (reflected by the last Stocker *et al.* report in 2013), the pressure on biodiversity will increase. Sala (2000) constructed biodiversity scenarios and predicted that climate change would probably become the second most important driver of biodiversity loss (after habitat change). Thomas *et al.* (2004) estimated that 15 to 37percent of the species in their global studied sample would be "committed to extinction" by 2050, due to climate change. Climate change also affects wild species by shifting and contracting their geographical range (Walther *et al.*, 2002). At the ecosystem level, climate change is a major threat to coral reef systems (Hoegh-Guldberg *et al.*, 2007). Regarding the effect on vegetation, rising atmospheric CO_2 concentrations may temporarily increase plant photosynthetic activity and net ecosystem productivity (NEP) but major uncertainties remain about the response of NEP to climate change (Cramer *et al.*, 2001; Long *et al.*, 2004).

Scope – GHG emissions of all livestock species are significant but they are dominated by emissions from beef cattle and dairy production, which respectively contribute 45 and 39 percent of the livestock sector's emissions (Gerber *et al.*, 2013). Intensive systems tend to have lower emissions *per* unit of product and lower CH_4 emissions from enteric fermentation in ruminants, because of higher productivity, feed quality and digestibility. The impact of GHG emissions on climate change is global even though global warming impacts are not evenly distributed across the world. They are stronger at higher latitudes, which put the species and ecosystems present there more at risk (Parmesan & Yohe, 2003; Deutsch & Tewksbury, 2008).

Indicators

- GHG emissions expressed in t CO2-eq (i.e. the amounts of each GHG causing the same global warming potential as 1 tonne of CO2) of the livestock sector have already been extensively computed in LCA studies (de Vries & de Boer, 2010; Thoma, *et al.*, 2010; Gerber *et al.*, 2013).
- Climate change (e.g. average change in temperature, average rainfall, frequency and/or intensity of extreme events) can also be computed. As for other categories of pollution, it is not possible to isolate the livestock sector from other areas of human activity.

Data availability – The GLEAM model provide robust estimates of the GHG emissions of livestock at global scale which can be disaggregated by species, regions, climate zones and production systems (Gerber *et al.*, 2013). The US National Aeronautics and Space Administration (NASA) provides global data on temperature (NASA, 2014a) and rainfall (NASA, 2014b). For the United Stataes, there is a GHG dairy industry LCA database available at the National Agriculture Library at the US Department of Agriculture (USDA, 2014).

4.3.2 Benefit: GHG sequestration

Context – a) Different types of livestock management are capable of improving or reducing the capacity of soils to retain carbon in grassland. Improved management has therefore been described as a powerful tool to increase the carbon storage ability of soils up to 3.04 Mg $C \cdot ha^{-1} yr^{-1}$, depending on the type of climate and ecosystem (Conant *et al.*, 2001).

b) A large share of GHG emissions, ranging up to 14.5 percent of the global total, has been attributed to livestock (Gerber *et al.*, 2013). However, such emissions largely come from high-fibre-diet, extensive systems and could be considered natural since the same amount could potentially be released by wild animals if livestock were withdrawn. This "replacement criterion" has been the object of criticism among the conclusions drawn from LCA studies.

Scope – a) The potential of different ecosystems to store carbon has been extensively reviewed. The relationship between land degradation and management practices is also well-known. The knowledge and characterization of grazing ecosystems as carbon sinks, as well as the restoration potential of degraded lands, show the potential of carbon storage in grazed soils (FAO, 2010b).

b) If the replacement criterion for GHG emissions is taken into account, livestock in purely extensive systems fed on fibrous diet could make a net contribution to GHG emission equal to zero. Conversely, evidence shows that extensive systems can use a negligible amount of fossil fuels to produce livestock (Casas Nogales & Manzano Baena 2010). Extensive systems could therefore have the capacity of mitigating carbon emissions by meeting part of the demand for livestock products with negligible net emissions of fossil fuel.

Indicators – a) Potential carbon storage for grazed ecosystems can serve as a good proxy for the amount of carbon that can be stored under carbon sink conditions, and these measures have already been used by the IPCC (2000). These same indicators can also show how much additional carbon can be stored in restored lands, with additional benefits for productivity and the livelihoods of local inhabitants.

b) In the case of extensive systems, and if the replacement criterion is considered, the use of fossil fuels may be a good proxy for estimating the impact of these systems on climate change (Casas Nogales & Manzano Baena 2010).

Data availability – a) Potential carbon storage can be deduced from global biome characterizations. Existing databases on land degradation also show the potential for carbon storage. Management practices will have a big impact on carbon storage capacity, but there is an issue here with the scale of measurement. See Data paragraph in Sections 4.1.3 and 4.1.4.

b) Life Cycle Analyses have provided substantial information on the use of fossil fuels for livestock production. Details on fossil fuel production can be easily obtained from these databases.

4.4 OTHER DRIVERS

4.4.1 Pressure: Invasive species

Context – 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 at global scale; the CBD requests the contracting parties to "prevent the introduction, control or eradicate those alien species which threaten ecosystems, habitats or species" (Glowka *et al.*, 1994). Several feral livestock species are classified as invasive alien species (Steinfeld *et al*. 2006). Along with other vertebrates, livestock contribute to the seed dispersal of invasive plant species (Rejmanek *et al.*, 2005). Among other human activities, livestock production contributed to the trans-Atlantic dispersal of invasive species. The disturbance associated with the introduction of livestock in natural grasslands of the New World (e.g. grazing, clear-cutting, fertilization, changes in fire regimes) favoured invasions by alien species (Seabloom *et al.*, 2003).

Scope – Invasions seem to be a minor problem in Europe (and China) where agriculture is older (Williamson, 1999). Mack (1996) mapped the areas where invasive plant species now dominate the landscape. The largest areas are found in America and Australia, as well as in some parts of Africa, India, and on various islands. Intensive systems may play a greater part in the dispersal of alien species as they rely more strongly on the trade of animal products, while extensive systems may be a more important source of feral livestock animals. All disturbance and degradation of natural and semi-natural systems increases the risk of invasion by alien species (MacDougall & Turkington, 2005); therefore, all the livestock species and production systems that generate such disturbances can contribute to invasive species pressure.

Indicators and data availability – 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 & Turkington, 2005; White *et al*., 2013)"mendeley": {"manualFormatting": "White *et al.*, 2013. Both the CBD (2006) and EEA (2007) define their pressure indicator on invasive species as a number of invasive species (cumulative number in Europe since 1900, completed by a list of the worst invasive alien species threatening biodiversity in Europe, for the EEA 2007). Isolating the burden associated with livestock production in terms of invasive species is very complex and no indicator exists to describe it. However, because there is a positive feedback loop between ecosystem degradation and alien invasive species, indicators of other categories of pressure can inform about the vulnerability to invasion.

4.4.2 Benefit: Invasive species control

Context – The nature of invasive species implies that they have certain traits that make them more successful at invasion, usually bearing a significant overlap with pioneer species (Manzano & Tallent in prep.) This means that, while livestock will contribute to disperse them preferentially (Manzano & Malo 2006), they will tolerate grazing less than species that appear in more advanced successional stages. Grazing can therefore be used to control invasive grass species (Germano *et al*., 2012).

Scope – In extensive systems, livestock can also exert a positive pressure on invasive species (reviewed in DiTomaso, 2000). Proper grazing management can control invasive species in several ways. Moderate grazing levels can minimize soil disturbance and effects on the plant community, preventing establishment or controlling spread of the invaders. If the invasive species is and edible plant chosen by the animals, intensive grazing can result in its control. Grazing multiple species of livestock can also control invasive species by avoiding dietary preferences and distributing the impact of grazing among desirable and undesirable species.

Indicators – Documenting management practices can be key in offering good predictors for invasive species control.

Data availability – See above for management practices and small-scale measurement.

4.4.3 Pressure: Overexploitation of wild populations

Context – Harvest and trade of species (e.g. for food, medicine, fuel, material use) is fundamental to economies and cultures everywhere, but can be pushed beyond sustainable levels. Overexploitation (not only related to livestock) is a major biodiversity threat which affects 33 percent of threatened species of mammals, 30 percent of threatened species of birds and 6 percent of the threatened species of amphibians (Baillie, 2004). The main contribution of livestock production to overexploitation is through overfishing for fish meals. According to Garcia & de Leiva Moreno (2005), 52 percent of fish stocks are fully exploited, 17 percent overexploited and 7 percent depleted. The indirect role of livestock in this exploitation is significant: Vannuccini (2004) estimated that in 2004, 24.2% of the world fishery production was used for fishmeal and fish oil for feed (see also Steinfeld *et al*., 2006). Most of these fishmeals are used in aquaculture but a significant share is still used for livestock production (62 percent and 38 percent, respectively, according to the Fishmeal Information Network 2008).

Scope – The use of fishmeal to feed livestock mainly concerns intensive systems, and more particularly pig and poultry production which respectively use 22 percent and 8 percent of all the fishmeal produced worldwide (Fishmeal Information Network, 2008).

Indicators and data availability – The fish catches and the status of various fish stocks are monitored globally by FAO. Fewer statistics are available on the share of these catches used for livestock rather than for aquaculture or human consumption. Some fish species are not suitable for human consumption (e.g. sand eel in Europe) or mainly used for fishmeal (e.g. sprat or capelin in Europe, anchovy in Peru) (Fishmeal Information Network, 2008). "European commercial fish stocks" is an EEA (2007) pressure indicator which describes the proportion of commercial fish stocks within safe biological limits.

4.4.4 Pressure: Competition with large mammals

Context – Competition between livestock production and wild mammals can occur in two ways: from direct interaction, or indirectly through resources (Steinfeld *et al*. 2006). In some regions, livestock losses from predation can be significant and long-term studies have indicated the importance of the prey base (Packer *et al*. 2005). In Kenya, these losses can amount to 2.6% of the annual economic value of the herd (Patterson *et al.*, 2004; Steinfeld *et al*. 2006). Locally, people do not often kill offending predators but commercial ranchers can decide to eliminate a predator after a livestock kill (Frank *et al.*, 2005). Conflicts with livestock production and killing of predators (e.g. wolf, bear) also exist in temperate regions such as Europe and North America (Treves & Karanth 2003; Musiani *et al*., 2003).

Indirect competition where livestock consume the resources (e.g. food, water, habitat) of wild species occurs in a wide variety of rangeland contexts such as between kangaroos and sheep in Australia (Edwards *et al.*, 1996), between yak or elephants and various livestock species in India (Madhusudan, 2004; Mishra & Wieren, 2004), between elephants and cattle in Africa (Prins, 2000), or between cattle and elk in North America (Brewer *et al*., 2007). Indirect competition by cattle has stronger effects on wild ungulate species that are ecologically similar in terms of body mass and diet; for instance, African buffalo were shown to avoid cattle herd because of grass depletion, while the spatial distribution of browsing and mixed-feeding antelopes was less affected by cattle presence (Hibert *et al*., 2010).

Scope – The competition pressure concerns rangeland systems in regions where wild predators (for direct interactions) or herbivores (for indirect competition) are present. The proximity of national parks is also a factor enhancing competition.

Indicators and data availability – Competition between livestock and wild species of large mammals occurs in local, particular contexts and no relevant large-scale data are collected. Indirect competition for resources is difficult to quantify and its effect can often be confused with other pressures such as land use and intensity changes, or pollution. To our knowledge, no specific indicators target this particular issue. The number of direct kills by predators could be a straightforward measure of direct interactions. Indicators of the intensity (Section 4.1.5) combined with measures of the presence of wild large mammal, or proximity of national parks could be used to quantify indirect competition. Indirect interactions between livestock and wild mammals can be complex (Ogutu *et al*., 2014). For instance, extensive livestock grazing is often associated with an increase in permanent watering points, which can increase the density of wild mammals beyond previous carrying capacity. In these situations, increases in wild mammal populations is not necessarily a reflection of increased biodiversity.

4.4.5 Benefit: Food web maintenance

Context – The presence of domestic herds provides both obligate and facultative scavengers with key resources for their survival. Good examples can be found in Bamford *et al*. (2007), Marinković & Karadzić (1999), Olea & Mateo-Tomás (2009) or Xirouchakis & Nikolakakis (2002) for vultures, while the Deccan wolf is a good example of a mesopredator benefited by livestock (Ghotge & Ramdas, 2010). Livestock have also been shown to decisively influence nutrient dynamics (Section 5.2.1) and this can indirectly benefit other organisms such as reptiles (Donihue *et al.* 2013). The role of dung beetles is key in nutrient cycling and they provide very valuable services that can be affected by livestock practice (Wardhaugh & Ridsdill-Smith 1998; Beynon *et al.* 2012), but their conservation can be affected by the loss of extensive livestock management (Barbero *et al* 1999) which can in turn affect food webs more widely, especially birds. How livestock could provide additional food resource may have been overlooked in several organisms (e.g. ants Manzano *et al.*, 2010).

Scope – Grassland systems.

Indicators – Many biodiversity state indicators (Section 5), such as the abundance of endangered species, with special attention to those directly benefited by the action of livestock, can indicate whether the facilitation of food webs is occurring. Management type is also important, as mobility has been shown to incorporate an important sustainability factor (Olea & Mateo-Tomás, 2009).

Data availability – Similar sources to the endangered vegetation data.

4.4.6 Pressure: Disease emergence

Context – In the past, the introduction of disease by livestock had severe consequences for native populations of wild species. For instance, the rinderpest virus brought with cattle to Africa killed many resident wild species of ruminants such as buffalo, giraffe and eland (Reid *et al.* 2010). Today, emerging infectious diseases continue to move between domestic and wild animals. Perhaps the most striking example is highly pathogenic avian influenza (also called H5N1 bird flu). The first outbreak of bird flu probably occurred in livestock (domesticated geese) and the spread of the virus in Asia and Africa in 2005-2006 was partly due to introduction of both poultry and wild birds (Reid *et al.*, 2010). Fifty species of wild birds have been infected by the H5N1 virus and it is estimated that 84 percent of all bird species, 37.2% of red list carnivore mammals and 58.8% of primates could be at risk of fatal infections (Olsen *et al.*, 2006; Roberton *et al.*, 2006; Reid *et al.*, 2010).

Scope – In poultry, animals in backyard systems are often more resistant to diseases, while several factors favour disease emergence in intensive systems. High productivity comes with trade-offs that make animals more sensitive to disease. Fast population turnover means that populations are mostly naive. Finally, the strong concentration of animals favours contamination and emerging disease outbreaks. Some infectious agents such as influenza viruses can adapt to different species and increase their virulence after recombination; a virus may cross the host species barrier to humans either directly from birds or indirectly via an intermediate host such as domestic pigs (Kuiken *et al.*, 2011). Disease control is relatively efficient in developed countries. The risk of disease emergence is higher in transition regions (e.g. Southeast Asia), where intensive livestock farming is more recent, disease control not fully established and cohabitation with backyard systems and wild species more prevalent (Coker *et al.*, 2011).

Indicators and data availability – Jones *et al.* (2008) mapped emerging infectious disease events, originating in both wildlife and domestic animals, at global scale. Such events could be a biodiversity pressure indicator but they are difficult to predict and depend on many factors such as environmental, climatic and socio-economic pressures that are independent of livestock (Jones *et al.,* 2008). Pressure indicators could also reflect the factors favouring disease emergence associated with livestock farming (e.g. absence of disease control plans, animal concentration and non-therapeutic use of antimicrobial products (Gilchrist *et al.* 2007).

4.4.7 Benefit: Disease control

Context – Livestock-transmitted diseases are a threat to native biodiversity and vice-versa, for wild animals can constitute a reservoir for livestock diseases of economic importance. While malpractice can translate into dramatic effects for biodiversity, good practice can improve the sanitary situation in wild animals and reduce the threat of extinction.

Scope – The recent eradication of rinderpest, which constituted a threat to many endangered wild ruminants, shows how good practices in disease control can yield positive benefits for biodiversity. Moreover, livestock management practices can significantly reduce the spread of diseases shared by livestock and wildlife. These practices are often related to the animal density maintained in farms but also to cultural practices and innovative tools to cut back disease spread (Eisler *et al.*, 2014).

Indicators – Quality of veterinary services for the control of wildlife-threatening, livestock-borne diseases as well as good practices in livestock rearing can be used.

Data availability – Large-scale databases are lacking on good practices for disease controls that occur mainly on a small scale. However, the World Organisation for Animal Health (OIE) provides extensive information on vectorial and trans-species diseases and FAO provides global data to support veterinary services (FAO EMPRES, 2014).

5. State indicators

State indicators describe biodiversity, defined as the 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 ecosystems (Article 2 of the CBD). Therefore, biodiversity encompasses multiple levels of organization. The three fundamental levels – genes, species and ecosystems – are detailed below. Intermediate levels can also be described. A *population* is defined as a group of organisms from the same species that interbreed and live at the same place and time. A *community* is composed of several populations of different species, occupying a particular area and usually interacting with each other and their environment. A *landscape* can have many definitions, one being a heterogeneous land area composed of a cluster of interacting ecosystems (Forman & Godron, 1981).

Three dimensions of biodiversity – composition, structure and function – apply across these hierarchical levels (Figure 7). Biodiversity composition is an inventory of characteristics, such as species richness or abundance, the presence of threatened species or the extent of different habitats. The structure is the organization of biodiversity components. It can refer to the spatial pattern of populations, landscapes or ecosystems, or to other structure components (e.g. age classes, sex ratio of the populations, slope, density of the ecosystem). The function refers to processes that go across the biodiversity levels. Functional groups of species that share the same function have consequences at higher level in ecosystem processes (e.g. biomass production, organic matter decomposition, nitrogen mineralization).

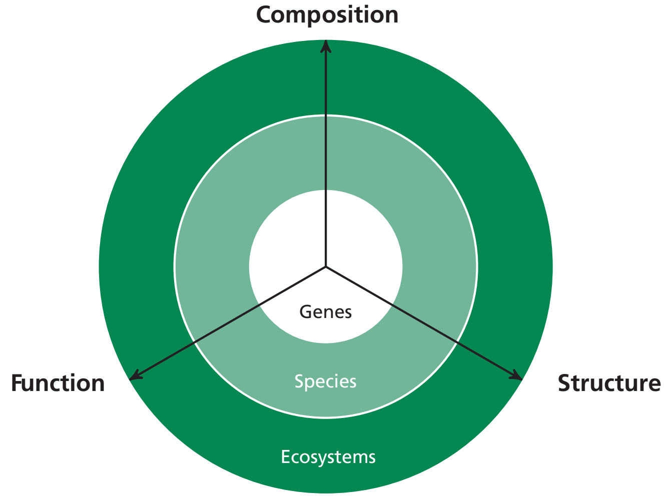
The next sections provide a description and examples of state indicators for the three dimensions of the species and ecosystem levels of biodiversity. The state indicators that we describe can be used to measure both the negative and positive influences that livestock have on biodiversity.

The genetic level of biodiversity is not in the scope of this report. In the context of livestock production, it relates to the diversity of livestock breeds and crop varieties, which respond to different pressures categories and mechanisms than wild biodiversity.

5.1 STATE INDICATORS AT THE SPECIES LEVEL

A species is usually defined as a group of organisms that is isolated reproductively from other such groups. It is difficult to apply to certain organisms such as many bacteria and plants with vegetative i.e. non-sexual) reproduction. Nevertheless, more than 1.7 million species have already been described and it is estimated that they represent only about 10 percent of the total number of existing species (CBD, 2001). Species biodiversity is not evenly distributed on the globe (Gaston, 2000). As a general pattern, species richness and endemism tend to increase toward the equator. Such increases correlate with solar energy and water availability, which enables higher net primary production by photosynthetic organisms and, in turn, more organic carbon for species at higher trophic level. Moist tropical forests are in general the most species-rich habitats. Coral reefs and Mediterranean climate habitats

Figure 7
The three levels and three dimensions of biodiversity.



Source: adapted from Noos (1999)

in various areas (including South Africa and Australia) are also particularly rich in species (Orme *et al.*, 2005).

The species extinction rate over the past 400 years has been 100 to 200 times higher than the background extinction rate (although estimating such figures is complex) (Hilton-Taylor & Mittermeier, 2000). In 2000, the International Union for Conservation of Nature (IUCN) estimated that 24 percent of all mammal species and 12 percent of all bird species were threatened with extinction. Although livestock production has positive effects on biodiversity, it contributes to the main global drivers of biodiversity loss (Section 4), and is partly responsible for the destruction of tropical forest habitats hosting rich biodiversity.

5.1.1 Composition dimension

Indicators at the species level and composition dimension reflect the identity and variety of species. They can be studied on two axes: species richness (number of species) and species abundance (number of individuals from several species or one particular species) (Figure 8). Measures of species diversity combine these two axes, *i.e.* richness and abundance. For instance, the Shannon index (*Shannon*), one of the most widely used diversity indicator, is computed as follows:

$$Shannon = -\sum_{i=1}^{R} p_i \ln p_i \qquad (1)$$

where p_i is the proportion of individuals belonging the species i, and R is the total number of species. The Shannon diversity index is high when there is a significant number of different species with individuals in similar abundance.

One limitation of the species richness or diversity indicators is that they do not inform about the identity of the species present. All species do not have the same conservation value. For instance, disturbance can lead to "biotic homogenization" where a few, common, generalist species replace several rare, specialist species (McKinney & Lockwood, 1999). Such biotic homogenization is not necessarily reflected by diversity indices. Indeed, specialist species are usually present in lower abundance and thus have less weight in the diversity indices. Several particular types of species can be interesting to focus on with specific abundance measures (Simberloff, 1998; Clergue *et al.*, 2005). Indicator species are species which can be used as an indirect measurement for the wider species richness or health of the ecosystem. Usually, they are also easy to identify and monitor (e.g. birds which, because of their high position in trophic chains, integrate variations from lower levels). Keystone species have a key role in sustaining populations of other species. They are often predators which allow for many species to coexist by selectively preying on species that would otherwise be competitively dominant (e.g. the starfish Paine 1966). Ecosystem engineers, i.e. species that creates or significantly modify habitats (like earthworms or beavers, Jones *et al.* 1994), are also considered as keystone species. Umbrella species need a wide range of resource and large tracts of habitat. Preserving them will thus automatically save many other species living in the same habitat. Threatened or rare species are species of particular conservation concern because they face a higher risk of extinction; the IUCN publishes a red list of such globally endangered species. Finally, flagship or patrimonial species are species with high cultural importance. Patrimonial species can be cultural symbols of specific areas.

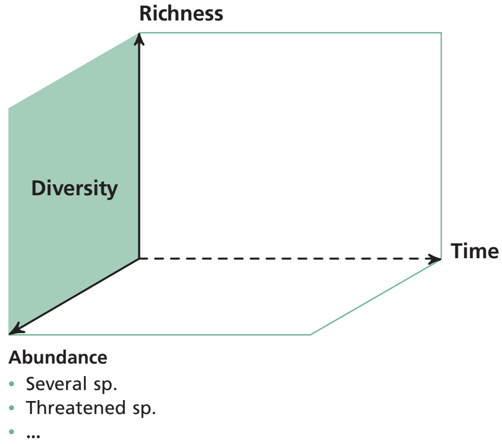
One way to compute indicators reflecting the composition of species while considering their conservation value is to compare the current situation with a reference situation (Nielsen *et al*., 2007; Vackar *et al*., 2012). This reference can be defined in various ways, e.g. as the situation in a specific year, in protected areas. The *intactness* index relies on empirical estimates to define this reference situation (Nielsen *et al.* 2007). It is an indicator of species composition that deals with the species conservation value challenge by weighting rarity and overabundance, and by incorporating both native and non-native species.

Indicators of species composition can be measured over time. This allows one to track whether a given livestock production system does, or does not cause decline in species richness, diversity, or in the abundance of certain species of particular interest. In the absence of time series, variation in species compositions can be calculated by comparing values to a reference which can be a given year, a baseline undisturbed habitat or an efficient livestock production system.

5.1.2 Structural dimension

Structure is the physical organization or pattern of a system. The structure of population in age classes can have an effect on demographics and on extinction risk (Boyce, 1992). At the species level, the most studied type of structure is the spatial structure of populations, in relation with the spatial patterns of the landscape. In certain landscapes, the species habitat is not continuous but split into patches. Species populations divided between such patches of habitat are called metapopulations (MacArthur, 1967; Levins, 1969). Along with the classic population dynamics occurring locally within each patch (survival, reproduction) there is a spatial, metapopulation dynamics of migration, colonization and local extinction between patches. The maintenance of the species requires healthy metapopulation dynamics, with patches large enough to sustain local populations, and patches close enough to sustain the spatial metapopulation dynamics (Hanski, 1994; Steffan-Dewenter,

Figure 8
Species composition can be described on two axes: richness (number of species) and abundance (of individuals in several or particular species). Diversity measures combine these two axes. State indicators of species richness, abundance or diversity can be computed in a static framework or over time.



2000). Conversely, fragmentation occurs when patches are too small and isolated. Fragmentation and the landscape structure can be described as a pressure indicator (see Sections 4.1.6 and 4.1.7) as it influences the metapopulation dynamics. Various state indicators can describe the metapopulation dynamics itself, e.g. the size of the whole metapopulation, the number of patches occupied, the ratio between local colonization and local extinction. Monitoring only local populations can be insufficient to describe the metapopulation dynamics. A source-sink dynamic often exists, where some patches (sources) host growing local populations and produce migrants that sustain declining populations in other patches (sinks).

5.1.3 Functional group dimension

The functional group dimension describes the diversity of functional groups of species rather than taxonomic groups, as in the composition dimension. There is a strong link between the diversity of functional groups at the species level and function at the ecosystem level (ecosystem processes and services, Tilman *et al.*, 2001; Loreau *et al.*, 2001).

Diversity at the species level and function dimension can be measured as the number of functional groups of species in a community (Hooper, 1998; Hector, 1999). Functional groups of species share a common function (functional trait), and can be defined in relation to their contribution to the function of the community or ecosystem, or in relation to how they respond to disturbances (Fonseca & Ganade, 2001). Several methods have been developed to define functional types in plants, in order to predict their distribution in different ecosystems or to analyze their influence on ecosystem function (Walker *et al.*, 1999). For instance, experiments have shown the crucial role of functional plant diversity for grassland productivity (Hector, 1999). For animals, a functional trait that has been extensively described with indicators is the trophic level (position occupied in a food chain). Pauly & Watson

(2005) developed the Marine Trophic Index, i.e. mean trophic level of fish communities. Authors showed that this index was very sensitive to disturbances; fish communities facing overfishing tend to have lower Marine Trophic Index values. Trophic indices have been extended to other communities, such as birds (Jiguet *et al.*, 2011). In areas strongly experiencing the effects of climate change, species composition is likely to change in response to this factor (e.g. shift or contraction of the range of certain species). Separating the effect of climate change from the effect of livestock production on species composition could be difficult. Using indicators of the functional group dimension could be a way to ensure that livestock contribute to maintain function at the ecosystem level despite the impact of climate change on species composition.

5.2 STATE INDICATORS AT THE ECOSYSTEM LEVEL

An ecosystem is a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit (Article 2 of the CBD, 2014). The Earth hosts a wide variety of aquatic and terrestrial ecosystems. Mangroves and coral reefs are well known examples of very particular aquatic ecosystems which not very widespread but host a rich biodiversity. Aquatic ecosystems also occur in inland waters and wetlands which were one of the first ecosystem targeted by an international treaty (Ramsar Convention, adopted in 1971). Terrestrial ecosystems are also very diverse, from forests (tropical moist and dry forests, temperate broadleaf and mixed forests, boreal needleleaf forests...), to drylands and agricultural ecosystems.

As with species, ecosystem diversity is under threat. Aquatic ecosystems face direct impacts such as overfishing and damage from fishery operations. They also face indirect stresses from climate change and pollution. The latter may in fact lead to the loss of 60 percent of coral reefs by 2030 (Hughes *et al.*, 2003). Although a terrestrial activity, livestock production is involved in the pressures affecting aquatic ecosystems (over-fishing, pollution, climate change). On land, livestock production contributes to habitat change, which directly destroys or alters terrestrial ecosystems (e.g. deforestation, dryland degradation, agroecosystem intensification, Section 4.1). But livestock are also embedded in certain agroecosystems and have a key role in maintaining them (e.g. grasslands, rangelands, sylvopastoral systems (see Section 4.1.2).

5.2.1 Compositional dimension

As for species, the compositional dimension of ecosystem can be studied through richness, abundance and diversity (Figure 8). Maintaining a richness of ecosystems at global scale is important, even though the number of ecosystems is limited compared to the diversity of species. Diversity (evenness) may be less relevant at the ecosystem level and global scale because the distribution of ecosystems is conditioned by climatic conditions. In the context of livestock, the compositional dimension of ecosystems has mostly been described as abundance, more specifically as evolution of the ecosystem extent (area) over time.

The *trends in the extent of selected biomes, ecosystems and habitats* is a headline indicator of the CBD (2006). For instance, annual net change in forest area can be computed. In Latin America and Caribbean it amounted to a reduction of around 4 percent between 1990 and 2005 (CBD, 2006). The direct or indirect responsibility of livestock production in change of the state of forests ecosystems has also been computed (Section 4.1). This indicator can be considered as a state indicator at the ecosystem level because it directly describes the spatial extent of ecosystems, and a pressure indicator at the species level because it reflects the habitat loss pressure on biodiversity. Other pressure indicators correspond to state indicators at higher biodiversity levels: habitat degradation could be a state indicator at the ecosystem level and landscape structure could be a state indicator at the landscape level (Section 4.1).

5.2.2 Structural dimension

The structural dimensions are often linked at the ecosystem level. Structure can relate to vegetation structure, which is an indicator of ecosystem health in both grassland and forest ecosystems. In forest ecosystems, a variety of species depend on the architecture of certain species of trees, called foundation species (Ellison & Bank, 2005). The decline of such key foundation species due to various pressures (e.g. outbreaks of invasive species and pathogens) threatens the whole ecosystem structure and several processes are involved in its functioning. Conversely, CO_2 and climate change have an effect on the ecosystem function (net energy productivity) which translates into a change in the vegetation structure (Cramer *et al.*, 2001). In all terrestrial ecosystems, the soil structure also plays a very important role. Soil structural properties respond to abiotic factors (e.g. of the parent material, erosion, physical and chemical processes) and biotic ones (e.g. organic carbon, soil community of mycorrhizal fungi, micro and macro arthropods), (Bronick & Lal, 2005). Soil structure is often expressed as the degree of stability of aggregates. Both over-compaction and porosity represent alteration of the soil structure. The soil structure is closely related to its function (carbon sequestration in particular, Silver *et al.* 2000) but also to the above ground biomass and functioning of the whole ecosystem (Wardle *et al.*, 2004). In aquatic ecosystems, the food web and size structure are key properties for the stability of pelagic ecosystems (Verity & Smetacek, 1996).

5.2.3 Functional dimension

Function is a crucial dimension of ecosystems. By definition, ecosystems are networks and involve many processes, e.g. fluxes of energy and nutrients, biomass production and decomposition. From a human perspective, these processes are ecosystem services, i.e. they provide direct and indirect benefits to humans (Millennium Ecosystem Assessment, 2005). For example, biomass production in a grassland ecosystem represents forage production for cattle. Livestock are part of agroecosystems and they are both providers (e.g. production, encroachment control) and users (e.g. fertility, water cycling) of ecosystem services (Swinton *et al.*, 2007).

The functioning of ecosystems is associated with their health and resilience, and with high biodiversity levels (species diversity in particular (Hooper *et al.* 2005). Human pressure on biodiversity has led to ecosystem degradation (Vitousek *et al.*, 1997). There is much evidence that stressed and degraded ecosystems have become incapable of supplying services at the same level as in the past (Noble & Dirzo, 1997; Rapport *et al.*, 1998). Pressure from livestock has resulted in ecosystem degradation and diminution of the services they provided (Harrison *et al.*, 2010). Conversely, well-managed grassland production can maintain healthy ecosystems that provide a wide range of services, from carbon sequestration to water cycling and quality, to biodiversity conservation (Havstad *et al.*, 2007). Measuring ecosystem services could be a key to quantifying the benefits to biodiversity of livestock production.

Several approaches exist to quantify function at the ecosystem level. Historically, functional ecology described ecosystems and their functioning by quantifying fluxes of nutrients, water and carbon (Raich & Schlesinger, 1992; Baldocchi *et al.*, 2001). More recently, the main focus has been on ecosystem services. Ecosystem services particularly important to livestock production can be quantified with specific measures, such as carbon sequestration in $t\ C.ha^{-1}yr^{-1}$ (Soussana *et al.*, 2004) or biomass production in gm^{-2} (Hooper, 1998). Ecosystem services have also been monetized, i.e. calculated in terms of economic value (Costanza *et al.*, 1997; Balmford *et al.*, 2002). Frameworks have also been proposed to classify ecosystem services and link them to ecological and economic valuation methods (de Groot *et al.*, 2002) or to study synergies and trade-offs between them (Tallis & Kareiva, 2008).

5.3 STATE INDICATORS AT THE GENE LEVEL

Genetic diversity is the richness of gene variations within a species. It is a key mechanism in allowing species to evolve and adapt to changing environments. In the context of agriculture, biodiversity at the genetic level mainly relates to the diversity of crop varieties and livestock breeds, which is beyond the scope of this report. This "domestic biodiversity" is huge, as a result of thousands of years of artificial selection for various traits. Rischkowsky & Pilling (2007) estimate that more than 7 600 breeds of livestock have been developed. Livestock diversity plays a key role in both short- and long-term food production in diverse environments as well as in food security, nutrition and cultural identity. That diversity is under threat as intensive production using a limited number of breeds spreads while traditional production systems and the associated local breeds tend to become marginalized and disappear (Rischkowsky & Pilling, 2007). Out of the 7 600 reported livestock breeds, around 20 percent are classified as at risk of extinction. The vulnerability of domestic biodiversity also concerns crops. Although the number of accessions conserved *ex-situ* worldwide has significantly increased during the past decade, many countries report genetic erosion of crops (FAO, 2010a).

5.4 STATE INDICATORS AT REGIONAL TO GLOBAL SCALE

The diversity of state indicators can be almost as wide as biodiversity itself. State indicators can describe the three biodiversity levels and dimensions, and focus on specific species, taxa or ecosystems. At the local scale, many different state indicators have been used; however, very few state indicators are available at regional to global scale. In this section, four indicators of the state of biodiversity available at regional to global scale are described (Table 3), and their potential use to measure the impact of livestock production is discussed. Three of them (the Living Planet Index, the Farmland Bird Index and the Red List Indices) have been used in the CBD (2006) Global Biodiversity Outlook.

5.4.1 Remotely sensed vegetation indices

Remotely sensed vegetation indices are calculated from data collected by sensors on satellites measuring wavelengths of absorbed and reflected light at the surface of the earth. The most extensively studied measure is the Normalized Difference Vegetation Index (NDVI), derived from the red/near-red reflectance ratio:

Table 3: Four indicators of the state of biodiversity computable at large scale, along with their data collection method, potential utilization, and scale.

Computation method	State indicator	Utilization	
		Temporal trends	Spatial trends
Monitoring	Red List Indices	Global scale	
	Farmland bird indices	Continental scale	Continental scale
Meta-analysis	Living Planet Index	Global scale	
Remote sensing	Vegetation indices	Global scale	Global scale

$$NDVI = \frac{NIR - RED}{NIR + RED} \quad (2)$$

where NIR and RED are the amounts of near-infrared (0.75 to 1.5 μ*m*) and red (0.6 to 0.7 μ*m*) light respectively, reflected by the vegetation and captured by the sensor of the satellite.

NDVI is a measure of “greenness”, ranging from +1 to –1, with negative values corresponding to absence of vegetation. The NDVI value can be interpreted as vegetation density measure (Weiss *et al.*, 2001). There is a strong relationship between NDVI and vegetation productivity, as shown by its correlation with fAPAR (absorbed photosynthetic active radiation intercepted) which has been well documented both theoretically and empirically (Pettorelli *et al.*, 2005). It means that NDVI can be used as a state indicator for the ecosystem level and the function dimension, and also as an indicator of the habitat destruction and degradation pressures. NDVI has already been used to monitor vegetation response to environmental change at various scales (e.g. global, national, small regions), with relatively fine resolutions (e.g. less than 1 km^2). Kerr & Ostrovsky (2003) used NDVI to assess land cover changes and deforestation in particular. NDVI has also been used to study the extent of land degradation in various ecosystems (e.g. in the Sahel, Thiam, 2003; semi-arid ecosystems in South Africa) and also a measure of length of the growing season (Vrieling *et al*. 2013).

As a state indicator, it has key assets for studying the relationship between livestock and biodiversity. It is remotely sensed and could be available at global scale for time series. It reflects the state of vegetation and shows a strong link with grazing pressure and ecosystem degradation. The species-energy theory states that areas exhibiting higher energy are able to sustain more species (Wright, 1983); however, highly productive agricultural areas with a high human appropriation of net primary production (HANPP) do not conform to the theory. The global distribution of biodiversity in natural ecosystems matches the species-energy theory (Gaston, 2000) and evidence also exists at more local scales (Seto *et al.*, 2004). Therefore NDVI, which relates to vegetation productivity, is also correlated to overall species richness (Parviainen *et al.*, 2010). Several studies already used NDVI values at global scale to compare its variation with the occurrence of managed grazing (Asner *et al.*, 2004; Thompson *et al.*, 2009). Several variables derived from the NDVI can reflect several attributes of the vegetation (Table 4).

A limitation of NDVI is that the variables above require intra-annual time series to be computed. Moreover, the relationship between NDVI and biodiversity or ecosystem degradation is not always straightforward and linear (Thompson *et al.*, 2009). NDVI is influenced by factors unrelated to degradation, such as pedo-climatic factors.

Table 4: The different variables computable with NDVI and the vegetation properties that they reflect.

Variable	Measure
Mean NDVI	Vegetation productivity
Variance NDVI	Highest potential vegetation
Max NDVI	Heterogeneity, diverse vegetation

Strong inter-annual variations of these factors (e.g. rainfall) can lead to strong variations in biomass productivity, which can make it difficult to link NDVI to ecosystem degradation. Linking ecosystem degradation or functioning to NDVI patterns can require elaborated algorithm procedures (Alcaraz *et al.*, 2006; García *et al.*, 2008) or field measures for validation (Thompson *et al.*, 2009). NDVI-based indices of ecosystem degradation can also be developed by comparing the observed NDVI to the value expected in healthy ecosystems (Feng & Zhao 2011). Another approach is to assess degradation by relating net primary productivity to rainfall use efficiency (Prince *et al.*, 1998; Bai *et al.*, 2008).

5.4.2 Living Planet Index

The Living Planet Index was developed by the WWF in collaboration with scientific teams to measure the evolution of the biodiversity state at global scale (Loh *et al.*, 2005). A meta-analysis of published scientific literature and unpublished reports was conducted to feed a database of population time series. This dataset contains more than 4 000 population time series ranging from 1970 to 2003, and covering 1 411 vertebrate species (Collen *et al.*, 2009). It evidences biodiversity decline, which can be further examined by taxa or thematic area. Because of the coarse (regional) spatial resolution at which the Living Planet Index is computed, it would only be possible to correlate it to regional trends in the evolution of livestock production. Confusion with other factors affecting biodiversity (e.g. climate change or land use change unrelated to livestock production) would be difficult to control.

5.4.3 The IUCN Red Lists and Red List Indices

The Red List of Species – The IUCN publishes – and constantly updates - a global Red List (IUCN 2014) which is the most widely recognized objective system for classifying species according to their risk of extinction (Hambler & Canney, 2004). It provides comprehensive assessments for a number of taxon groups and regions. The explicit classification system can be applied at global and national scale and is already widely used by decision-makers. Butchart *et al.* (2004) used the Red List to build an indicator of trend in the status of biodiversity. Red List Indexes are calculated from the number of species in each conservation category and the number of species changing categories. Trends in Red List Indices have been computed for the 1988-2004 period for various taxa, regions and ecosystem (Butchart *et al.*, 2004; 2005). As with the Living Planet Index (Section 5.4.2), Red List Indices are computed at a coarse resolution which makes them difficult to compare with local evolutions or properties of the livestock sector. However, the IUCN Red List is a very useful tool to follow species composition with a focus on threatened species. It is applicable at different scales (e.g. continental, country) which makes it possible to monitor specifically threatened species and to investigate whether livestock production is associated with their preservation or decline.

THE RED LIST OF ECOSYSTEMS – The IUCN has recently created a global standard for assessing the status of ecosystems, the Red List of Ecosystems (RLE). It can identify which ecosystems are not currently facing risks of collapse, which ones are threatened at Vulnerable, Endangered, or Critically Endangered levels, and which ones have reached the final stage of degradation and are therefore in a state of Collapse. This is measured by assessing losses in area, degradation, conversion, and other major changes such as climate disruption (Keith *et al*., 2013). Further, IUCN is working to develop and apply a suite of knowledge products for more informed decision making about land/seascape planning and resource use, and to produce better outcomes for biodiversity conservation and human well-being. More information on the RLE and its applications are provided in Appendix 3.

5.4.4 Farmland bird indices

The European Farmland Bird Index (EFBI) is computed in several European countries as the geometric mean of the abundances of common farmland bird species. It describes farmland bird trends and can be used a proxy for wider biodiversity health in farmland. Conservation of farmland biodiversity is a central issue in Europe, where farmland species have suffered sharper decline than other species groups (e.g. for birds, Gregory *et al.*, 2005; Jiguet *et al.*, 2011). The EFBI has been adopted by the EU as a Structural and Sustainable Development indicator (Butler *et al.*, 2010).

The EFBI is computed from data collected under the Pan-European Common Bird Monitoring Scheme[2]. This scheme generates national population trend indices for 135 bird species (Gregory *et al.*, 2009). Within each country, the trends are computed from surveys at local sample points. In France for example, bird populations have been monitored in around 2 000 2x2km sample squares between 2001 and 2009 (Jiguet *et al.*, 2011). The high spatial resolution of these sample points would make it possible to compare distribution of bird populations with livestock farming properties at fine scale. The high sample size with a wide geographical coverage and several years allows one to conduct statistical analysis in order to isolate the effect of agriculture from other factors (Doxa *et al*. 2010; Doxa *et al*. 2012; Teillard *et al*. 2014).

2 http://www.ebcc.info/pecbm.html

6. Response indicators

Response indicators describe the decisions and actions that can be undertaken by stakeholders – including policy makers, the private sector and farmers – to mitigate impacts and improve the state of biodiversity. Decisions and actions cover laws, incentives, certifications, management plans or practices. An advantage of response indicators is that they can describe decisions and actions targeting improvement in specific pressures categories, biodiversity levels or dimensions. There is a wide variety of response indicators that reflects the diversity of possible stakeholders and actions. The following Sections (6.1 and 6.2) give a few examples of actions targeting biodiversity improvements that could be used to develop response indicators.

6.1 ACTIONS FROM THE PUBLIC SECTOR

In the EU, the Common Agricultural Policy (CAP) is a powerful set of rules, regulations and subsidies targeting the agricultural sector. It historically supported production; however, to meet the sustainability challenge, AESs were introduced to the CAP in 1992. European AESs, which are based on voluntary compliance, provide payments to farmers for adoption of management practices that reduce adverse environmental impacts, on biodiversity in particular. These practices target reduction in different categories of pressures and aim to strengthen benefits that farming can bring to biodiversity (examples of AESs are given in Table 5). The recent, 2013 CAP reform approved the mechanism of ecological cross compliance, i.e. that subsidies for production are conditioned by ecological criteria. These include a "greening" provision that farms should set aside at least 5 percent of their land as ecological focus areas (semi-natural habitats, including grasslands).

Equivalents of the AESs exist outside Europe. For example, the United States Department of Agriculture (USDA) has an Environmental Quality Incentives Program. Although biodiversity improvement is not directly the target, the programme addresses certain biodiversity pressures such as water quality or landscape. The concept of Good Agricultural Practices (GAP) was formalized by the FAO Commission on Agriculture and several countries (as well as farmers, the private sector and NGOs) have developed their own GAP code. Good Agricultural Practices are defined as those that address environmental, economic and social sustainability for on-farm processes, and result in safe and quality food (FAO GAP 2014). GAPs thus address biodiversity, among other environmental aspects and sustainability pillars.

Several countries have been developing public organic certification standards incorporating biodiversity aspects, such as the US National Organic Program (NOP), the EU organic farming regulations or the Chinese OFCD organic product certification standard. Maintaining and enhancing biodiversity is part of the principles of organic farming, whose rules include multi-annual crop rotations and limitation of livestock density, which corresponds to mitigation of biodiversity pressures.

In eastern and southern Africa, communities are creating conservation areas and are managing both wildlife and livestock numbers (Osano, 2013).

Table 5: Examples of Agri-Environment Schemes targeting the improvement of benefits and mitigation of the agricultural pressures on biodiversity.

Targeted pressure category	Examples of AESs
Habitat loss	Converting arable land back to grassland Creation of set-aside areas for fauna or flora of interest Management and maintenance of existing High Nature Value habitats
Intensity	Extensive management of grassland
Pesticides pollution	Reduction of pesticide treatments Replacement of chemical treatment by mechanical treatment Replacement of chemical treatment by biological control
Nutrient pollution	Reduction of nitrogen input Partial replacement of mineral fertilization by organic fertilization Partial replacement of fertilizer input by including legumes in crop rotations Composting livestock manure
Soil degradation	Adoption of minimum tillage techniques Using an intermediate culture on bare ground in winter Increase of soil organic matter
Landscape structure	Creating and maintaining trees edges, clumps or isolated trees Creating and maintaining ponds or other water points Creating and maintaining grassy strips Diversifying crop rotations

6.2 ACTIONS FROM THE PRIVATE SECTOR

As environmental and social impacts of farming became an important concern for consumers, several standards were developed to certify the sustainability of agricultural products. Not all the existing standards address the environmental dimension or biodiversity. Many standards target specific products. The Sustainable Agriculture Standard originating from the Sustainable Agriculture Network and the Rainforest Alliance is general to agricultural products (Sustainable Agriculture Network, 2010). It addresses the three sustainability pillars with detailed criteria on biodiversity. These criteria could be used as response indicators targeting the state of biodiversity at both the ecosystem and species level. Indeed, they describe actions aiming at protecting and enhancing biodiversity at these two levels. For instance, criteria at the ecosystem level include:

- *Critical Criterion. All existing natural ecosystems, both aquatic and terrestrial, must be identified, protected and restored through a conservation programme. The programme must include the restoration of natural ecosystems or the reforestation of areas within the farm that are unsuitable for agriculture.*
- *Critical Criterion. From the date of application for certification onward, the farm must not destroy any natural ecosystem. (...)*
- *Production areas must not be located in places that could provoke negative effects on national parks, wildlife refuges, biological corridors, forestry reserves, buffer zones or other public or private biological conservation areas.*
- *There must be a minimum separation of production areas from natural terrestrial ecosystems where chemical products are not used. A vegetated protection zone must be established by planting or by natural regeneration between different permanent or semi-permanent crop production areas or systems.*

- *Aquatic ecosystems must be protected from erosion and agrochemical drift and runoff by establishing protected zones on the banks of rivers, permanent or temporary streams, creeks, springs, lakes, wetlands and around the edges of other natural water bodies. (...)*

and at the species level:

- *An inventory of wildlife and wildlife habitats found on the farm must be created and maintained.*
- *Ecosystems that provide habitats for wildlife living on the farm, or that pass through the farm during migration, must be protected and restored. The farm takes special measures to protect threatened or endangered species.*
- *Critical Criterion. Hunting, capturing, extracting and trafficking wild animals must be prohibited on the farm. (...)*

7. Strenghts and limitations of the PSR indicator framework

7.1 STRENGTHS

The PSR indicator framework provides a way to structure indicators which facilitates interpretation and decision making. Moreover, pressure, state and response indicators are complementary.

Table 6 compares pressure, state and response indicators according to three criteria: (i) whether they are directly related to biodiversity itself, (ii) whether they are directly related to management decisions and (iii) whether they can be computed with easily available information.

State indicators are those that are most closely related to biodiversity itself. They provide a direct measure of biodiversity that is not comprehensive (as it focuses on a specific level, dimension or taxon), but can be a proxy for wider biodiversity. However, state indicators are not directly related to management decisions. Different management decisions can lead to the same change in the biodiversity state, or conversely, the same management decision applied in different contexts can lead to different changes in the state of biodiversity (Whittingham *et al.*, 2007). Synergies and antagonisms can also exist between management decisions in how they influence biodiversity. Most importantly, the state of biodiversity is not solely influenced by the management decisions of a single sector like livestock. A wide number of factors influence biodiversity, ranging from short to long temporal scales, from local to global scales and from anthropic to natural and stochastic factors. When using state indicators, separating the impact of livestock production from the impact of these other (possibly interacting) factors, can be challenging. Multivariate statistical analyses can be a way of separating impacts but they require large sample sizes and information on both livestock production and the other factors. Finally, state indicators often require a large data collection effort to be computed. Such collections can be time-consuming, involve a high level of expertise, and require large sample sizes to be representative.

Pressure indicators stand at an intermediate level in the three criteria presented in Table 6. They do not provide a straight biodiversity measure but describe pressures for which a direct link with biodiversity has been widely evidenced in the literature. They also have a close relationship with management decisions. This relationship

Table 6: Comparison of the pressure, state and response indicators according to three criteria.

	Pressure indicators	State indicators	Response indicators
Direct link with the state of biodiversity	♠	♠	♠
Direct link to management decisions	♠	♠	♠
Computable with limited information	♠	♠	♠

can be straightforward, e.g. land use decisions influence the spatial heterogeneity of the landscape. This relationship can be more complicated but models often exist to understand the link between pressures and management decisions (e.g. LCA models addressing climate change). The close relationship between pressures and management decisions also means that the data required to compute pressure indicators are often relatively easily available.

Response indicators are less closely related to biodiversity itself for the several reasons cited above (confusion and interaction of effects). However, they describe management decisions. For this reason, they can often be easily computed from already available data.

Because of these complementarities, the PSR indicator framework can adapt to the goal and scope of a biodiversity assessment study, and to the level of information available. A simple analysis can be performed by monitoring only certain key response indicators. It is what several agri-environment policy and private certification standards require (Section 6). With a moderate amount of information, it would be feasible to use pressure indicators to perform a comprehensive assessment of all the pressures that livestock production exerts at various levels: farm, company, sector, country. Although addressing all the pressures, such an analysis would not consider the relative importance of the different pressures in influencing the state of biodiversity. For the most thorough analysis of livestock's impact and performance, state indicators need to be computed.

The Sustainability Assessment for Food and Agriculture (SAFA, 2013) provides an example of how the different categories of indicators can be used according to the level of information available. It includes practice-based and performance-based indicators which correspond to response and state indicators, respectively. There is a hierarchy between indicators, and performance-based indicators are considered the most relevant. When performance data is not available, the SAFA framework provides the option of using practice-based indicators. In computing the total sustainability score, practice-based indicators have a lower weight (they bring less "points") than performance-based indicators.

7.2 LIMITATIONS

One important limitation of the PSR indicator framework is that indicators are almost always computed for a single process of the supply chain and for a bounded area. Most pressure, state and response indicators are computed at the level of the farm (e.g. percentage of grassland, fertilization or species richness within the farm), sometimes including the surrounding landscape. LCA approaches show that significant environmental impacts related to a livestock product can occur outside the farm. In intensive production systems, an important part of the feed consumed by the animals in the farm can be bought from outside. Environmental impacts associated with this off-farm feed cultivation sometimes account for a major share of the total livestock product.

Restricting the computation of biodiversity indicators to the farm-level fails to account for such off-farm impacts. In addition, livestock supply chains are globalized. For instance, in 2011, a total of 86 countries exported soybean cakes and 114 countries imported them, for a total exchange volume of more than 58 million tons (FAOSTAT, 2014). In the case of biodiversity where land use is a key pressure category, land use impacts related to off-farm feed cultivation could potentially be very important.

As a simple example, the feed consumed in a given European dairy farm could originate from on-farm pasture (40%), on-farm maize cultivation (30%) and soybean cakes imported from South-America (30%). Computing a land use pressure indicator at the farm level would neglect the share of the total land use pressure occurring off-farm, i.e. 30 percent. Because the imported soybean cakes might originate from a deforested area that was previously a biodiversity hotspot, their relative impact on biodiversity could represent even more than 30 percent of the farm's total impact. An important challenge for taking into account the impact of feed cultivated off-farm is that, most often, farmers are not aware of the geographic origin of the feed they buy. The difficulty is even greater in the case of compound feeds that are blended from different raw materials from various origins.

Other limitations exist when restricting the computation of biodiversity indicators to the farm or to another bounded areas There can be a scale mismatch between this area and the ecological mechanism underlying the impact that is measured. For instance, landscape structure should not be measured only on-farm because the mechanisms of landscape structure on organisms involve a larger spatial scale. The state of biodiversity as measured within the farm is also influenced by pressures at larger scale, e.g. nutrient run-off from neighbour farms, structure of the surrounding landscape, global climate change.

In the next part, we detail several LCA approaches that address impacts on biodiversity. Compared to the PSR indicator framework, the strength of these approaches is that they account for the whole life cycle of the product.

PART 3

THE LIFE CYCLE ASSESSMENT FRAMEWORK

8. Overview

LCAs strive to be holistic assessments of the potential environmental impacts associated with a product, process or service, along some or all the production stages (all the way to consumption and product's end of life). It allows practitioners to quantify the burdens from cradle to grave (ISO, 2006a) and therefore exposes the overall environmental impacts and the relative contributions of the different stages of the supply chain. LCA studies also offer a way of identifying ways of reducing impacts and shed light on how other parts of the system may be affected (Garnett, 2009).

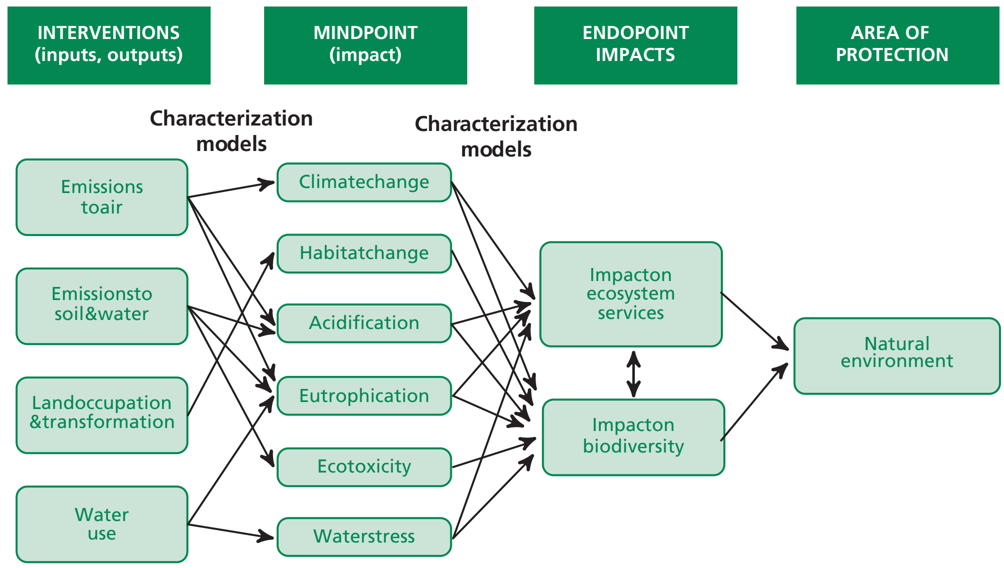
LCA studies comprise four steps according to ISO (ISO, 2006a): goal and scope definition, inventory analysis, impact assessment, and interpretation. The first step is the description of the goal and scope, which includes defining the objectives of the study and setting the systems boundaries. The scope should be sufficiently well defined to ensure that the breadth, depth and detail of the study are compatible and sufficient to address the stated goal. The second step, inventory analysis, involves data collection and calculation procedures to quantify relevant inputs and outputs for all processes along the product's life cycle (Life Cycle Inventory, LCI, Figure 9). In the third step, Life Cycle Impact Assessment (LCIA), LCI results are converted into potential impacts on the environment so conclusions can be drawn in the last step, Interpretation.

At the LCIA stage, characterization models reflect the environmental mechanism by describing the relationship between the LCI results and environmental impacts (Figure 9). Characterization models are used to derive the characterization factors, which are the values used to convert emissions and resources from inventory to common impact units to make them comparable. Impacts can be characterized anywhere along the environmental cause-effect chain, either at the midpoint or endpoint level. Midpoint impact category can be defined as a problem-oriented approach, translating impacts into environmental themes such as global warming, land occupation, acidification or human toxicity. Endpoint impact categories provide a damage-oriented approach (ISO, 2006b). Traditional characterization methods used midpoint modelling while nowadays there is an increasing acceptance that LCA results should reflect potential damage on endpoint impact categories (such as biodiversity loss) and areas of protections (human health, natural environment and natural resources, EC, 2010). The goal of this damage modelling is to aid in understanding and interpreting midpoints by computing endpoint categories corresponding to areas of protection that form the basis of decisions in policy and sustainable development.

8.1 LCAS OF LIVESTOCK PRODUCTION

LCA studies of livestock production systems mainly include impacts arising from feed production and associated input use, from animal husbandry itself and from downstream transport and processes until retail. Livestock production has contributed to numerous environmental impacts, such as climate change, land degradation and loss of biodiversity (Steinfeld *et al.*, 2006). De Vries & de Boer (2010) carried out a review of different environmental LCA studies of livestock products (e.g. milk, eggs, meat). Most LCA studies reviewed in this article are limited to estimating midpoint impacts, namely global warming, acidi-

Figure 9
Schematic representation of the environmental mechanism (cause-effect chain) pathway, from life cycle inventory (e.g. land occupation and transformation) to midpoint (e.g. habitat destruction) and endpoints (e.g. species loss or functional loss).



Source: adapted from Teillard *et al.* (2016)

fication and eutrophication. Little attention is paid to quantifying impacts on biodiversity. Land use impacts are mainly addressed in terms of land surface occupied, but no loss of biodiversity is assessed. Other LCA studies include biodiversity impacts relying on local biodiversity assessments, i.e. without taking into account the off-farm impact of any feed being imported into farms (e.g. Haas *et al*., 2001; Nemecek *et al*., 2011a;b; Jeanneret *et al.*, 2014). Jeanneret *et al.* developed an expert system for including biodiversity as an LCA impact category in agricultural production. The method is valid for grasslands, arable crops and semi-natural habitats of the farming landscape. A scoring system estimating the suitability of each farmland habitat as well as the reaction of 11 indicator-species groups to management options was developed. These methods cannot be extrapolated at regional or global scales, however. Guerci *et al.* (2013) relied on more generic characterization factors to compare the impacts of dairy farming on biodiversity through land use in several European countries. Recent LCA studies used a regionalized global approach to compute the impacts on biodiversity through land use, for livestock (Mueller *et al.*, 2014) or other food products (Coelho & Michelsen 2013; Milà i Canals *et al.*, 2013 Antón *et al* 2014; Milà i Canals & deBaan, in press). This approach makes it possible to account for off-farm impacts along the globally distributed life cycle of a product, while considering differences in biodiversity impacts among global regions. For milk, Mueller *et al.* (2014) found that the specific impact of different land use types was more important than the sole impact of the total area occupied. For margarine (independant of livestock), Milà i Canals *et al.* (2013) found that the impacts of land use dominated the impacts associated with processing. These findings justify the fact that most LCA studies addressing the impact of food products on biodiversity focus on land use.

8.2 LCA METHODOLOGIES ADDRESSING BIODIVERSITY ENDPOINTS

This section gives an overview of some of the existing Life Cycle Impact Assessment methodologies for computing endpoint impacts on biodiversity from one or several midpoint impact categories. Specific details of these methodologies are then described by midpoint impact categories in Section 9 (for biodiversity impact through land use) and 10 (for the biodiversity impact through other midpoint impact categories).

The ReCiPe methodology (Goedkoop *et al.*, 2012) includes several midpoint (land use, climate change, acidification, eutrophication and ecotoxicity) and endpoint impact categories, with LCIAs harmonized in terms of modelling principles. Biodiversity loss is one of the endpoint indicators covered. Models are provided to compute biodiversity indicators from the following midpoint impact categories: land use, climate change, acidification, eutrophication and ecotoxicity. The land use model, with a biodiversity indicator, is mainly based on the concept of Species-Area Relationship (SAR), as are most of the existing methods.

Mean Species Abundance (MSA) is a biodiversity indicator reflecting the mean abundance of current species relative to their abundance in undisturbed ecosystems (Alkemade *et al.*, 2009). The MSA was developed in the context of the GLOBIO3 modelling framework designed to assess scenarios of human-induced changes in biodiversity. The MSA has been used in the context of livestock production, for computing current and projected impacts under different scenarios (Alkemade *et al.*, 2012; Westhoek *et al.*, 2011). The MSA can also be used in LCAs as it corresponds to an impact factor, translating several midpoint impact categories into biodiversity values, and De Baan *et al.* (2013a) provide an application of the MSA in the LCA context. Characterization factors link MSA to land use, atmospheric N deposition, infrastructure development and climate change.

Most efforts to include biodiversity impacts in LCAs have focused on its link with a single midpoint impact category: land use. Research on biodiversity indicators for the assessment of land use impacts in LCA has been ongoing for more than 15 years (Souza *et al.* 2014), but no consensus has yet been reached on the use of a specific method. Weidema & Lindeijer (2001) developed global characterization factors for broad categories of land use, describing both the species richness and ecosystem productivity (NPP) components of biodiversity. Koellner & Scholz (2008) developed characterization factors for Europe, linking numerous classes of land use and intensity to biodiversity, expressed as an Ecological Damage Potential (EDP) indicator. This method has been used in the specific context of livestock (Guerci *et al.,* 2013). A number of methods contributed to the development of the current land use impact assessment conceptual framework, taking into account land use and land use change impacts: de Baan *et al*. 2013b; Geyer, 2010a,b; Michelsen, 2008, Schmidt, 2008; Souza *et al*., 2013. The UNEP-SETAC Life Cycle Initiative is driving global consensus on characterization factors and impact indicators for biodiversity in the context of LCA (Jolliet *et al.* 2014).

9. The assessment of land use impacts on biodiversity in LCA

Livestock are a major user of land resources (Sections 1.1, 4.1), which makes land use one of the main drivers of livestock impact on biodiversity. Computing this impact in LCA while considering specificities of the livestock-biodiversity relationships (e.g. positive impacts, Section 4) is thus a key challenge.

In the last 15-20 years, many efforts have been carried out to address land use impacts on biodiversity in LCA (Michelsen, 2008; de Baan *et al.* 2013a;b; Souza *et al.* 2013; see also a review in Curran *et al*., 2011 and the other references cited in this section). A few reviews have also been carried out on the topic (Curran *et al.* 2011; Koellner *et al.* 2013; Milà i Canals and de Baan, *in press*; Souza *et al.* 2014, *in press*), discussing challenging gaps in modelling. However, no consensus exists on which methodology should be applied for current LCA studies. This is mainly due to the following: (i) biodiversity is a complex entity with multiple aspects that cannot be fully captured or represented by a single indicator; (ii) some assumptions of the land use model represent a linearization of dynamic processes in nature and lead to oversimplification (Souza *et al.* 2014, in press); (iii) LCA studies require the availability of global characterization factors, which can require large amounts of data if models are to be accurate.

In the following sections, we detail the general conceptual framework for land use impact assessment in LCA and review some of the existing methods addressing biodiversity impacts.

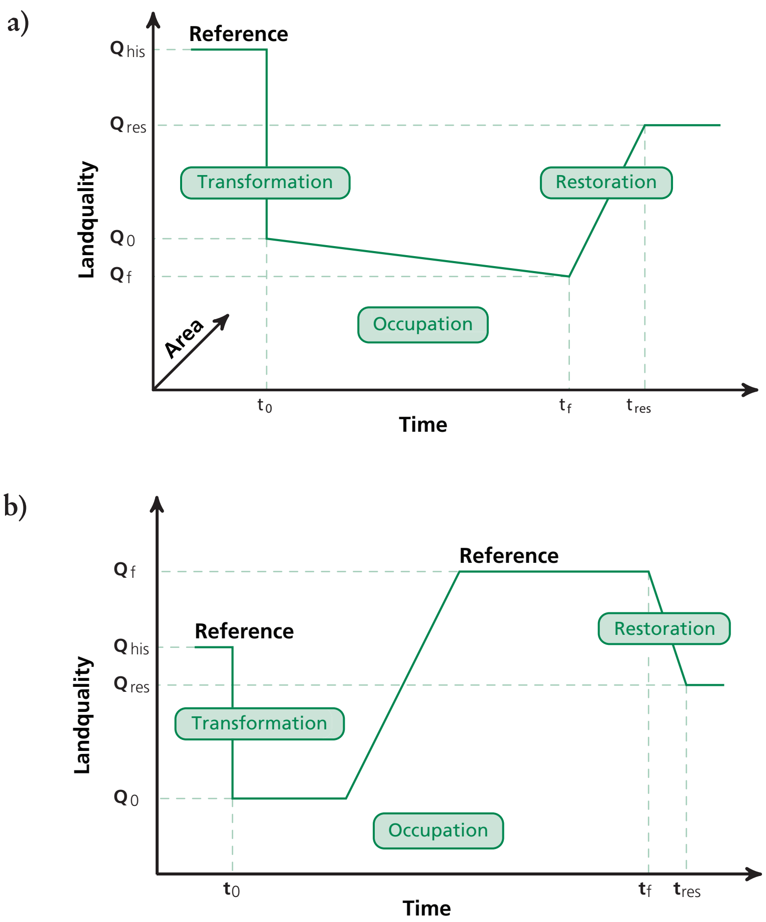
9.1 CONCEPTUAL FRAMEWORK

9.1.1 General situation

Milà i Canals *et al.* (2007) recognize two land use elementary flows (interventions) driving habitat and biodiversity changes: land transformation and land occupation (Figure 10). Land transformation is assumed to be a sudden process during which human activities convert the current land use/cover to make it suitable for a new use. Examples of land transformation include deforestation to establish a pasture, or conversion of natural grassland to cropland. Q_{his} stands for the historical land quality (before any land transformation). Land transformation leads to a change in land quality from Q_{his} to Q_0. Land occupation, during which the new land use takes place, starts at t_0 and lasts until t_f. During this time, land quality gradually evolves from Q_0, at the beginning of the occupation, to Q_{fin}, when land use ceases. In terms of biodiversity, it can involve a loss (or a gain) of species richness but also important changes in community or ecosystem composition. Figure 10a depicts a gradual drop in land quality during occupation. It could be caused by the use of fertilizers or pesticides, for example. More complex evolutions of land quality during occupation could occur, depending on land management practices. However, most existing frameworks are not able to take these alternative evolutions into account: they assume that land quality remains constant because only one land quality value is assigned to one land use. If the area is no longer used and land is set aside, land recovery processes – driven

Figure 10

Scheme of the conceptual framework for impact assessment in LCA, depicting two land interventions (occupation and transformation) and land recovery. Figure (a) describes a situation where natural land is transformed and used; (b) describes a situation that could apply to the livestock context: occupation lasts for a very long time and biodiversity adapts to it. his = historical, 0 = initial (after land transformation and at the beginning of land occupation), f = final (at the end of land occupation), res = restoration (at the end of restoration).



Source: adapted from Lindeijer (2000); Milà i Canals *et al.* (2007)

by natural ecological succession, or active restoration in the in case of human intervention – may take place. The duration of this process before reaching a new steady land quality Q_{res} (if the land remains undisturbed) can vary.

The existence of land transformation, occupation and restoration implies that they should ideally all be considered when computing the impacts of land use on biodiversity. However, some recent models just compute the impacts of occupation, since little or no information may exist on the impact of transformation and on natural recovery of land. Land use impacts should be computed as land quality multiplied by both time and area, which represent a third dimension in Figure 10a.

9.1.2 Specific situations in the context of livestock production

The example depicted in Figure 10a represents one possible situation. It can represent the conversion of a tropical forest to pastures (land transformation) then used for livestock production (land occupation). A different situation may arise when livestock production is abandoned after a very long tiem, as in many semi-natural grasslands in Europe and elsewhere (Figure 10b). Conversion from forest to pastures (land transformation) and the associated decrease in biodiversity took place hundreds of years ago. The very long duration and the extensive nature of the land occupation for livestock farming allowed a unique biodiversity to co-evolve with grazing (Bignal & McCracken, 1996; Poschlod & WallisDeVries, 2002; Section 4.1.2). Today, when livestock farming is abandoned in these semi-natural grasslands, the natural process of land recovery to original forest results in a loss of biodiversity (Verhulst *et al.*, 2004; Sebastià *et al.*, 2008). In this case, determining the land use and land quality value to be used as reference is not straightforward (Figure 10b). Most LCA studies have used potential vegetation and Q_{his} as a reference, but it is not always clear what should be the choice of vegetation/ecosystem type for use as the reference condition for natural and semi-natural livestock systems.

9.1.3 Using species-area relationships to compute characterization factors

A strong pattern in ecology is the Species-Area Relationship (SAR), which is to say that the number of species found in a region is a positive function of the area of the region (Arrhenius, 1921). Connor & McCoy (2001) described the different ecological mechanisms theorized as underlying SARs. The main ones are the habitat diversity hypothesis and the area *per se* hypothesis. The habitat diversity hypothesis proposes that larger areas have a greater variety of habitats, which in turn host a greater diversity of species. The area *per se* hypothesis is based on the assumption that larger areas allow for greater species population sizes, which have lower risk of extinction. SARs are used to infer biological processes and estimate biodiversity (Palmer & White, 1994). In the context of LCA, they are useful for computing characterization factors as they fulfil the same role: linking land use and area to a biodiversity value.

Fitted species-area curves are often nonlinear: as area increase, the number of species increases steeply at the beginning and gradually flattens out. Several nonlinear models have been used to fit a species-area relationship to a sample: the power (log-log) model (Arrhenius, 1921), the exponential model (Gleason, 1925) and the logistic model (Archibald, 1949). The most widespread is the power model (Arrhenius, 1921):

$$S = cA^z \quad (3)$$

where S stands for the species richness and A for the area. The parameters c and z correspond, respectively, to a multiplier (number of species in a unit area) dependent on taxa, and to the slope of the increasing number of species in relation to area (species accumulation rate) (Rybicki & Hanski, 2013). The transformed power model (log-log model):

$$\log \mathrm{S} = \log \mathrm{c} + \mathrm{z} \log \mathrm{A} \quad (4)$$

describes a linear relationship between $\log S$ and $\log A$ where $\log c$ is the intercept and z is the slope.

The species-area relationship has been used to compute characterization factors of land use impact on biodiversity (Koellner & Scholz, 2008; Goedkoop *et al.*, 2012). In a general procedure, the first step is to collect data on the species richness across various areas, in different categories of land use. These data can be collected through field surveys or by conducting a meta-analysis of already published surveys. For each land use, the linear log-log model (Eq. 4) is then fitted to the sample of species richness and areas, *i.e.* parameters c and z are calibrated. Figure 11 shows an example of these relationships fitted for a reference *ref* and an occupied land use type *occ*. When the species-area curves (Eq. 4) are known, it is possible to compute the characterization factor shown in Eq. 5 which corresponds to the biodiversity impact of the transformation of an area A_0 from the reference land use type r to an occupied land use type *occ* (see also Figure 11b).

$$CF = \frac{S_{ref}(A_0) - S_{occ}(A_0)}{S_{ref}(A_0)} = 1 - \frac{c_{occ}}{c_{ref}} A_0^{z_{occ}-z_r} \quad (5)$$

Different models have been developed around this general procedure using the SAR in order to compute characterization factors. We give some examples in the following Section. However, other ecological methods have also been used in modelling. De Baan *et al.* (2013b) used species distribution models, while Geyer *et al.* (2010a;b) applied habitat suitability models to calculate characterization factors. Souza *et al.* (2014, *in press*) discusses some of the limitations and drawbacks in using each of these approaches.

9.2 EXAMPLES OF LAND USE IMPACT ASSESSMENT MODELS

9.2.1 The Ecological Damage Potential

The Ecological Damage Potential (EDP) (Koellner & Scholz, 2008) is an impact factor based on the assessment of impacts of land use and intensity on species number.

The first step in calculating the EDP aims at eliminating the aspect of species number S attributable to area size. The species-area relationship is fitted according to Eq. 4 in order to compute a standardized species number in $1m^2$ of each land use type S_{1m^2}:

$$S_{1m^2} = S_{plot} - \Delta S \quad (6)$$

where S_{plot} is the species number measured by empirical studies on a plot of size A_{plot}. ΔS is the part of the species number which can be attributed to the area rather than to the type of land use. It is calculated as:

$$\Delta S = c(A_{plot}^z - A_{1m^2}^z) \quad (7)$$

using the coefficients c and z of the species area relationships fitted for each land use.

The EDP is then based on the ratio between the standardized species richness in an occupied land use (S_{occ}) and the average standardized species richness in the region S_r. The EDP can be computed either as a linear function of this ratio:

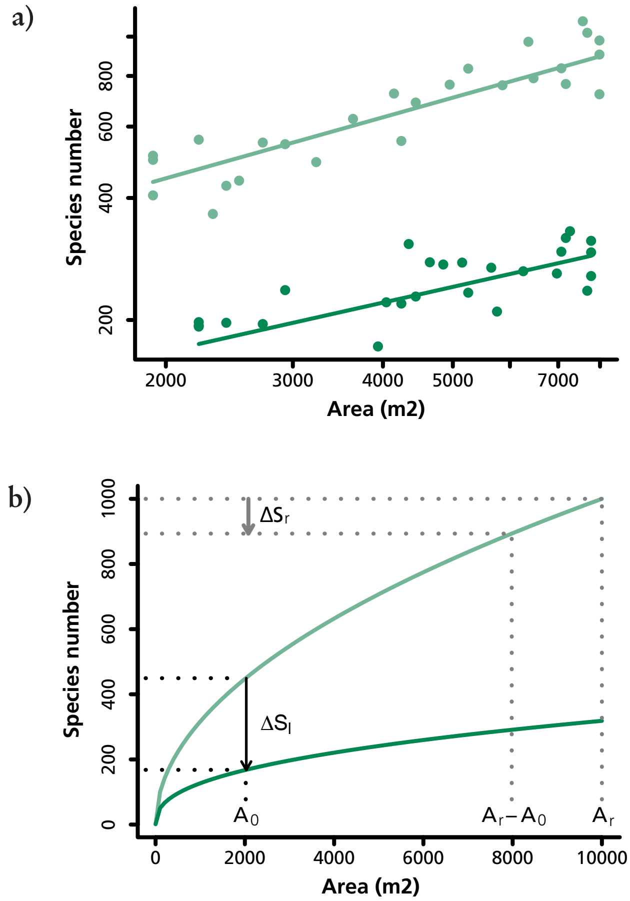
$$EDP_{linear} = 1 - \frac{S_{occ}}{S_r} \quad (8)$$

or as a nonlinear function:

$$EDP_{nonlinear} = 1 - a(\log \frac{S_{occ}}{S_{region}}) + b \quad (9)$$

Figure 11

Species area relationship in a reference (blue) and occupied (red) land use. (a) species area relationships are calibrated using the log-log model (Eq. 3, here log scales are used on both axes). (b) the species area relationships can be used to compute the loss of species richness resulting from the transformation of an area A0 of the reference land use into the new one (ΔSl), and from the loss of this area in the reference land use (ΔSr).



where the logarithmic function reflects the redundant species hypothesis which states that the addition of species in already rich ecosystems results in a lower marginal growth of utility in terms of ecosystem processes. The parameters *a* and *b* in Eq. 9 were quantified by Schlapfer *et al.* (1999), based on expert estimates.

The EDP impact factors translate 53 land use categories of the Corine Land Cover dataset and 6 intensity classes into species richness (Koellner & Scholz, 2008). The species richness of three taxa is considered: vascular plants, moss and mollusks. For vascular plants, a specific EDP was also computed for the number of threatened species. Authors relied on the Biodiversity Monitoring Switzerland scheme and on a meta-analysis to compute EDPs; therefore, EDPs are expected to be relevant for Central Europe. In addition, Schmidt (2008) computed EDPs for vascular plants in Southeast

Asia. The main source of uncertainty with the EDPs characterization factors are the stability of the results of the meta-analysis (where different methods have been used across studies) and the sample sizes. Fewer plots were investigated for moss and mollusks than for plant, which result in higher standard deviation around the mean EDP.

9.2.2 The ReCiPe methodology

Among other impact categories, the ReCiPe methodologies (Goedkoop *et al.*, 2012) provide characterization factors translating the impact of land use into Potentially Disappeared Fraction of species (PDF, i.e. of species richness).

The ReCiPe methodology considers damages on species richness at two scales. The local damage describes the change in species richness on the occupied area, in comparison with the reference land use. Eq. 5 is used to compute the characterization factor corresponding to this local damage CF_{loc} (corresponding to ΔS_l in Figure 11b).

Besides, the regional damage describes the marginal change in species richness outside the occupied area, caused by area reduction (corresponding to ΔS_r in Figure 11b). The regional change in species richness ΔS_r associated with the loss of an area A_0 is calculated as the first derivative of Eq. 3:

$$\Delta S_r = A_0 \times c_{ref} \times z_{ref} \times A_r^{z_{ref}-1} \quad (10)$$

where A_r is the area of the region (the other terms are detailed in Eq. 5).

The characterization factor CF_r corresponding to this regional damage is then calculated as the ratio between ΔS_r and S_r:

$$CF_l = 1 - \frac{A_0 \times c_{ref} \times z_{ref} \times A_r^{z_{ref}-1}}{c_{ref} A_r^{z_{ref}}} = \frac{A_0 z_{ref}}{A_r} \quad (11)$$

The ecological damage combining local and regional characterization factors is finally calculated as follows:

$$\begin{aligned} ED = &\left(1 - \frac{c_{occ}}{c_{ref}} A_0^{z_{occ}-z_{ref}}\right) \times t \times A_0 \\ &+ \left(\frac{A_0 z_{ref}}{A_r}\right) \times t \times A_r \\ = &\left(z_{ref} + 1 - \frac{c_{occ}}{c_{ref}} A_0^{z_{occ}-z_{ref}}\right) \times t \times A_0 \end{aligned} \quad (12)$$

where the local and regional characterization factors are first multiplied by the time t and area (A_0 and A_r, respectively) of occupation, and then summed together. This ecological damage is expressed as $PDF \cdot m^2 \cdot yr$.

Goedkoop *et al.* (2012) provide PDF impact factors at local and regional scale, and combined ecological damage levels are provided for 47 land use and intensity categories of the Ecoinvent database (Frischknecht *et al.*, 2005). PDF impact factors focus on the species richness of plants. Three sources of data were used to compute them, from the UK (Crawley & Harral 2001, Countryside Survey 2000) and Switzerland (Koellner, 2003). Based on these two countries, PDF impact factors are assumed to be relevant for Europe. No distinction is made between species with potentially different conservation values (e.g. common vs. red listed). The ReCiPe methodology takes into account the three stages of impact of land use, *i.e.*, transformation, occupation and restoration (Figure 10a). Restoration is considered from four different perspectives affecting restoration time and fragmentation (egalitarian, individualist and hierarchical).

9.2.3 The Mean Species Abundance

Mean Species Abundance (MSA) is a biodiversity indicator reflecting the mean abundance of original species relative to their abundance in undisturbed ecosystems (Alkemade *et al.*, 2009). Unlike the EDP and PDF, the MSA is based on species abundance (number of individuals) rather than on species richness. The MSA was developed in the context of the GLOBIO3 model, which aims at assessing scenarios of human-induced changes in biodiversity. However, the MSA corresponds to an impact factor, translating several biodiversity pressures (midpoint impacts, including land use) into biodiversity values. De Baan *et al.* (2013) provide an application of the MSA in the LCA context.

In order to compute the MSA values of different land use categories, Alkemade *et al.* (2009) conducted a meta-analysis of papers that presented data on species composition in disturbed (occupied) *vs* (reference) land uses. All species (fauna and flora, without restrictions related to the taxa) were included in this meta-analysis. For each species k within each occupied land use occ, the ratio $R_{occ,k}$ was calculated as:

$$R_{occ,k} = \begin{cases} \frac{n_{occ,k}}{n_{ref,k}}, & \text{if } n_{occ,k} < n_{ref,k} \\ 1, & \text{otherwise} \end{cases} \quad (13)$$

where $n_{occ,k}$ is the abundance of the species k in the occupied land use and $n_{ref,k}$ its abundance in the reference land use.

The MSA of any occupied land use MSA_{occ} is then calculated by summing and weighting the ratios $R_{occ,k}$ of each species:

$$MSA_{occ} = \frac{\sum_k (R_{occ,k}/V_{occ,k})}{\sum_k 1/V_{occ,k}} \quad (14)$$

where $V_{k,e}$ is the variance of the ratios of species abundances for each study and copes for differences between studies.

The MSA values of the different land uses thus vary between 0 and 1. MSA=1 in undisturbed ecosystems where 100 percent of the original species abundances remains. Conversely, MSA=0 in a destroyed ecosystem with no original species left. Alkemade *et al.* (2009) and Alkemade *et al.* (2012) provide MSA values for 13 land use and intensity categories (Table A.1 in Appendix). Intensity gradients are described within three main land use classes (forest, grassland and cultivated land).

The MSA characterization factors for land use are relevant at global scale. No restriction related to the taxa was applied by Alkemade *et al.* (2009) when conducting the meta-analysis leading to the computation of the MSA values.

9.2.4 The UNEP-SETAC life cycle initiative

The UNEP-SETAC life cycle initiative aims to provide guidelines for taking into account impacts of land use on biodiversity in LCA, and to find consensus on impact indicators (Jolliet *et al*., 2014). In previous years the Life Cycle Initiative has pushed the methodological development of land use impact assessment in LCA: Milà i Canals *et al.* (2007) developed a framework for the LCIA of land use, which distinguished two main land use interventions: land transformation and land occupation (Section 9.1). This framework was later refined with guidance on different aspects of the land use impact assessment framework, such as irreversibility issues and spatial/temporal heterogeneity in the distribution of the impacts (Koellner *et al.* 2013b). This framework is described in the LEAP biodiversity principles (LEAP, 2016). Koellner *et al.* (2013a) proposed a harmonized global land use/cover classification for life cycle inventories and a method to regionalize land use elementary flows. Land use classes encompass four levels of detail ranging from coarse (n=11, e.g. agriculture, shrub land) to refined (n=74, e.g. arable irrigated intensive, pasture/meadow extensive). For regionalization, land occupation is described in $m^2 \times year$ of specific land use type in a defined region, and transformation is described in m^2 of land use type converted in another land use class, in defined region. As for land use classes, the regionalization system is multilevel, with five levels of details. de Baan *et al.* (2013a) relied on land use/cover classification and regionalization from Koellner *et al.* (2013a) to develop several characterization factors that quantify the land use impact on biodiversity across world regions, as species richness (Biodiversity Damage Potential, BDP), species abundance (MSA) and species diversity (Shannon and Fisher indices). The characterization model developed by Chaudhary *et al.* (2015) has recently been recommended by the UNEP-SETAC to estimate impacts on biodiversity related to land use in LCA.

10. Examples of impact assessment models for other midpoint categories linked to biodiversity

Land use is among the most important drivers of biodiversity loss, especially in the context of livestock production. Among pressures, it is the one most addressed in studies developing characterization factors to link midpoint impacts to biodiversity impacts. However, land use is not the only pressure that livestock production exerts on biodiversity at global scale (Section 4). The MSA and the ReCiPe methods (detailed for land use aspects in Section 9.2) also compute characterization factors linking other midpoint categories to biodiversity. In the following sections, we provide more detail on how characterization factors are computed to link changes in biodiversity to climate change, pollution and ecotoxicity endpoint impacts.

10.1 ACIDIFICATION AND EUTROPHICATION

Nutrient losses (N and P) can occur at two main stages of livestock production, from feed fertilization and from manure. Nitrogen cycling is dynamic and complex. Microbiological processes are responsible for mineralization, fixation and denitrification of soil nitrogen. Part of the N loss is emitted through direct and indirect volatilization such as nitrogen oxides (NO_x), nitrous oxide (N_2O) and ammonia (NH_3). These gases are transported and later deposited, which can lead to soil acidification. Another part of the N losses is transformed into soluble components such as ammonium (NH_4), nitrate (NO_3) and nitrite (NO_2). With rainfall and runoff, these soluble N components and various forms of P can determine both soil acidification and aquatic eutrophication.

10.1.1 Acidification

Several studies have described acidification at the midpoint level while accounting for the sensitivity of the receiving ecosystems. For instance, Potting & Schöpp (1998); Seppälä & Posch (2006); Posch *et al.* (2008) provide country-dependent acidification potentials by SO_2, NO_2 and NH_3 in Europe (in moles of H^+ equivalents).

Authors relied on models of the emission, dispersion and deposition of acidifying substances that integrated different critical load functions among ecosystems.

Other studies computed characterization factors linking acidification to endpoint biodiversity impacts, expressed as potentially disappeared fraction of species Van Zelm *et al.* (2007) or as net primary productivity Hayashi *et al.* (2004). The approach of Van Zelm *et al.* (2007) is used to address eutrophication in the ReCiPe methodology. The characterization factor of an acidifying substance x (CF_x, expressed in $m^2 \cdot yr \cdot kg^{-1}$) is calculated as:

$$CF_x = \sum_j \left(A_j \cdot \frac{dPNOF_j}{dM_x} \right) \quad (15)$$

where A_j is the size of a (European) forest area j. $dPNOF_j$ is the marginal change in potentially not occurring fraction of species due to a marginal change in emission of acidifying substance x (dM_x). Several steps lead to the calculation of this $dPNOF_j/dM_x$ ratio. The first step is to compute a fate factor from a model of the transfers of acidifying substances to atmosphere and soil. The second step is to compute an impact factor linking the PNOF to the elevated base saturation of the soil, computed through multiple regressions.

10.1.2 Eutrophication

Posch *et al*. (2008) also provide country-dependent values for eutrophication potential. The endpoint effect of freshwater eutrophication is included in the ReCiPe methodology, through the approach of Struijs *et al.* (2011). The characterization factor is expressed as PDF·m^3·day/kg P emission. As for acidification, it combines a fate factor and an effect factor. The fate factor is computed from a model linking the sources of phosphorus (manure and fertilizers, effluents from freshwater treatment plants) to its concentration in inland rivers. The impact factor is the disappeared fraction of macro-invertebrate genera, as a log function of phosphorus concentration. This impact factor used is calculated from a database in the Netherlands of more than one million records of different macro-invertebrate taxa (see Posch *et al*. 2008).

10.2 CLIMATE CHANGE

To our knowledge, the only operational assessment method for the impact of climate change on biodiversity was developed by Schryver *et al.* (2009) (it is included in the ReCiPe methodology). It focuses on the relationship between temperature increase and loss of terrestrial species, with an emphasis on plants and butterflies. Authors modelled a causal relationship between GHG emissions and global mean temperature increase. The final characterization factor links temperature change to biodiversity, and is expressed as km^2.PDF.°C^{-1}:

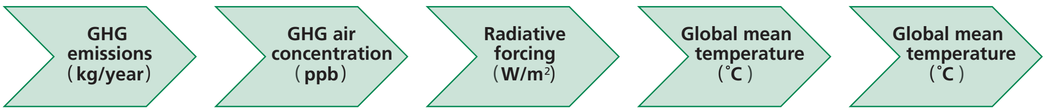
$$CF_{CC} = \frac{A.\Delta PDF}{\Delta TEMP} \quad (16)$$

where ΔPDF is the average change in potentially disappeared fraction of species due to a temperature change $\Delta TEMP$. A stands for the total surface of (semi-)natural terrestrial areas of the world, *i.e.* $10.8{\cdot}10^7 km^2$.

The characterization factor was based on the work of Thomas *et al.* (2004). In this study, data on 1 084 species across five regions (Europe, Mexico, Australia, South Africa, Brazil) were integrated into a climate envelope modelling approach in order to estimate range area and the associated extinction risk.

Livestock contribute to climate change through significant GHG emissions (Gerber *et al.* 2013). Although climate change is known as an increasingly important driver of biodiversity, isolating the contribution of livestock to this impact is complicated. The method developed by Schryver *et al.* (2009) models four consecutive steps linking GHG emissions to temperature increase and biodiversity damage (Figure 12). Characterization factors make it possible to predict biodiversity damage directly from GHG emissions, and therefore to isolate the impact of livestock.

Figure 12
Framework used by Schryver *et al.* (2009) to link GHG emissions to ecosystem damage.



10.3 WATER USE

The amount of water needed to produce animal products can be very high. For instance, Mekonnen & Hoekstra (2012) estimated that a global average of 15 415 litres of water is consumed along the supply chain to produce 1kg of beef (including water used for feed production, drinking water and service water used for animal husbandry). This total includes different categories of water – blue water (diverted from surface and groundwater), green water (rainwater evaporated from soil and plants) and grey water (needed to assimilate the load of pollutants). However, total water consumption does not necessarily reflect the water footprint. One litre of water consumed does not have the same impact in a temperate ecosystem as in an arid ecosystem where water availability influences vegetation diversity and ecosystem quality (Nilsson & Svedmark, 2002). Pfister *et al.* (2009) developed a method to assess the impacts of freshwater consumption on several endpoint categories including ecosystem quality, among global regions. Authors assessed water-shortage-related vegetation damage as Net Primary Productivity (NPP) and considered it as a proxy for PDF. They calculated a characterization factor for water consumption by weighting a water-limited NPP by the level of precipitations in spatially explicit grid cells of 0.5°. This characterization factor was calculated at global scale and expressed as $PDF{\cdot}m^2{\cdot}yr$ *per* m^3 of water consumed.

10.4 ECOTOXICITY

Many ecotoxicological studies have sought to establish relationships between Potentially Affected Fraction (PAF) of model species and the concentration of toxins (Larsen & Hauschild, 2007). PAF relationships lead to the calculation of effect concentration where 50% of test organisms are affected (EC_{50}). PAF and EC_{50} have then been used to compute effect factors for the biodiversity endpoint. Van Zelm *et al.* (2009) is the approach included in the ReCiPe methodology to calculate the ecotoxicological effect factor for biodiversity PDF_{tox}:

$$PDF_{tox} = 1 - \prod_k \left(1 - PAF_k^{EC50}\right) \tag{17}$$

where PAF_k is the EC_{50} derived from the PAF relationship. It is assumed that potentially affected fraction is equivalent to potentially disappeared fraction of species.

The UNEP-SETAC initiative provides characterization factors for freshwater ecotoxicity (Rosenbaum *et al.*, 2008). Authors compared several models in order to build the scientific consensus USEtox model. They used two databases with average

EC_{50} values for more than 3000 chemicals (van Zelm *et al.*, 2009; ECOTOX, 2002) and provide continental to global characterization factors.

In the context of livestock production the main categories of eco-toxicological components released in the environment are pesticides (used for feed production), veterinary products and hormones (Boxall *et al*., 2002).

11. Strengths and limitations of the LCA framework

11.1 STRENGTHS

As highlighted in the Part 2 (Section 7.2), the calculation of pressure, state and response indicators has almost always been limited to a single process of the supply chain and to a bounded area. On the contrary, the LCA framework examines the product's environmental impact at all stage of its life cycle. It includes impacts occurring off-farm and in potentially different global regions, which could be very significant for biodiversity.

LCA is an important tool for conducting environmental assessment. It is increasingly recognized by governments, the private sector and NGOs as guiding decisions towards better environmental performance (Tillman, 2000; Rebitzer *et al.*, 2004). The use of LCAs is already widespread in assessing certain environmental performances such as GHG emissions or fossil energy demand. Including biodiversity in the same LCA framework would ensure consistency with the assessment of other environmental criteria. A single impact assessment model could be used to compute midpoint impacts, and the calculation of endpoint biodiversity impacts would require additional modelling steps with specific characterization factors (Figure 9). Such consistency would allow comparability of results on the different environmental criteria and decision-making on a multi-criteria basis. In the case of consequential LCAs, it would shed light on how measures to reduce one environmental impact affect other criteria.

The UNEP-SETAC Life Cycle Initiative has launched a flagship project to run a global process aiming at global guidance and consensus building on a limited number of environmental indicators, including indicators for impacts from land use on biodiversity. A multi-year process engaging international experts and global stakeholders has been initiated to carry out this programme, with the intent of developing guidance on Environmental Life Cycle Impact Assessment Indicators based on a consistently applied set of selection criteria and rigorous analysis of different methods for assessing biodiversity damage produced by land use. Different methods in and out of the scope of LCA have been selected and evaluated according to criteria of completeness, scientific and environmental relevance and applicability. Table 7 summarizes the main characteristics of some selected LCIA methods for assessing the effects of land use on biodiversity.

11.2 LIMITATIONS

There is often a trade-off between the different performance criteria of the LCA approaches; it is challenging, for instance, to have both large geographic scope and taxonomic coverage while using detailed land use classes. Table 7 summarizes some of these performance criteria for several LCA approaches focusing on biodiversity impacts through land use. In the next subsections, we detail current limitations of the LCA approaches on these various performance criteria.

Table 7: Summary of main methodological approaches for LCA of the impact of land use on biodiversity.

Source	Indicator	Reference state	Land use classes	Geographic scope	Taxonomic coverage
Lindeijer (2000b)	Species richness of vascular plants	Most "undisturbed" vegetation in the present region	Specific activities: mining in South-America, sand extraction, industrial production, road traffic and landfill and forestry and hydropower in Europe	Palaeartic (Europe) and Neotropic (South America)	Vascular plants (herbaceous + woody)
Goedkoop *et al.* (2012)	Relative species richness change, represented as Potential Disappeared Fraction (PDF), values converted to species	PNV, Potential natural vegetation chosen for Europe as "broadleaf woodland"	Eighteen different, relevant land use types for occupation (including three types of intensity for arable areas) + four land conversions	Palaeartic (North West-Europe)	Plants
Weidema & Lindeijer (2001)	Species richness, ecosystem vulnerability, and ecosystem scarcity		Land use types corresponding to 12 types of biomes	Global	Plants
Schmidt (2008)	Absolute species richness change per 100 m2 of land use	Current land cover	17Seventeen: arable cereals/annuals, arable grasslands, agroforestry, managed forest, natural forest, natural heath and scrub, natural grasslands, natural bogs, and sealed land	Denmark and Indonesia/ Malaysia	Vascular plants
Geyer *et al.* (2010a,b)		Current land use maps 2010	1) Arable land, 2) Arable irrigated, 3) Grassland, 4) Pasture, 5) Forest, used, 6) Scrublands 7) Forest, 8) Forest, intensive	Specific location (S. California, four counties, 29 different habitat types, 11 crop production scenarios). (Neartic)	Terrestrial vertebrate species
Itsubo & Inaba (2012)	Expected Increase in Number of Extinct Species (EINES).	Land use (primary production) reference state is natural vegetation	Eighty land use types	Currently, only Japan. (Next version LIME 3 will be applicable to whole world).	Vascular plants
Souza *et al.* (2013)	Functional diversity	Natural or close-to-natural, assumed to represent PNV	Nineteen land use classes	Examples from Nearctic (North America) and Neotropic (South America)	Plants, mammals, birds
de Baan *et al.* (2013)	Absolute loss in regional species	The current, late-succession habitat stages as reference	Four major land use types: 1) Managed forest, 2) Agriculture, 3) Pasture, 4) Urban areas	CFs provided for 804 ecoregions, which can even be extrapolated to country level	Mammals, birds, plants, amphibians and reptiles.

11.2.1 Pressure and benefit categories

Many efforts to include biodiversity assessment in LCAs have focused on land use impacts. Several methodologies and characterization factors have been developed to convert the land use midpoint impact into a biodiversity endpoint impact. Four of them are detailed in Section 9.2 and additional methodologies exist (Weidema & Lindeijer, 2001; Geyer *et al*., 2010a;b). The UNEP-SETAC initiative is a step toward consensus on how to include biodiversity impacts from land use in LCAs. Knowledge is less advanced on other midpoint impact categories. For most midpoint impact categories (e.g. climate change, water use, eutrophication or acidification [Section 10]), only a few methods exist to compute biodiversity endpoint impacts. Methods to link midpoint impact categories to biodiversity often over-simplify the impact pathway. For example, land use characterization factors may not account for different levels of landscape structure or land degradation within the same land use, or climate change characterization factors may address average temperature increases without accounting for the higher frequency of extreme climatic events. For some categories mentioned in Section 4, no biodiversity characterization factors exist. It is the case for invasive species, over-exploitation, competition and disease emergence. Therefore, no widely accepted method exists to link biodiversity to all the categories of pressure related to livestock. For land use, methods are close to being more widely recognized and they could be adapted to the livestock sector, where land use represents a major pressure category.

An important limitation of most previous LCA studies of the land use impact on biodiversity is that they failed to consider beneficial biodiversity impacts, which can be important in the context of livestock production (e.g. semi-natural grasslands with high biodiversity value [Section 9.1.2]). Whether LCA methodologies are able to account for these beneficial impacts depend on the land use reference that is selected. This reference situation can either be potential natural vegetation (PNV), the (quasi-) natural land cover in each biome/ecoregion, or the current mix of land uses (Koellner *et al*. 2013). Many authors have used the potential natural vegetation as a reference (e.g. Alkemade *et al.*, 2009; 2012; Goedkoop *et al.*, 2012; de Baan *et al.*, 2013a). Selecting PNV as reference gives similar weight to land use impacts currently occurring (e.g. tropical deforestation) and land use impacts that occurred a long time ago (e.g. deforestation of European woodlands). With this methodology, species-rich, semi-natural grasslands in Europe are seen as deforested areas and their impact on biodiversity can only be negative. Alternatively, the selection of recent land use states as reference (e.g. land cover in year 2000) results in higher impact for current land use change processes, like deforestation occurring in tropical countries. In this case, although there cannot be a positive effect of land use that continues to support livestock production, it can be neutral if no land use change has occurred since the reference year. Furthermore, the effects of cessation of livestock production on species-rich grasslands and reversion to woodland will be represented as a negative effect on biodiversity. Koellner *et al.* (2008) used the regional average species richness as a reference, which did result in beneficial biodiversity impacts for extensively managed agricultural areas and semi-natural grasslands. According to Milà i Canals *et al*. (2007a), it is advisable to use (quasi-)natural land cover predominant in global biomes and ecoregions as a reference when assessing land use impact on a global scale. Nevertheless, defining a reference situation – an area for further exploration – is recognized as a value choice. Selection of a suitable reference should be a priority in order to make LCA methodologies relevant to the livestock sector. This reference should capture the difference between the presence and absence of various livestock production systems. For instance, for extensive grazing on species-rich grasslands, a decline in biodiversity is expected if grazing is removed while for intensive grazing on grass monocultures an increase in biodiversity is expected if grazing is removed.

11.2.2 Spatial coverage and resolution

The spatial coverage of methods and characterization factors described in Sections 9.2 and 10 differs considerably across different methods. It is global for the MSA land use characterization factor (Alkemade *et al.*, 2009), and the climate change (Schryver *et al.*, 2009) and water (Pfister *et al.*, 2009) methods. Other characterization factors are available at country scale to region scale. For instance, EDP land use characterization factors were computed for Central Europe (Koellner & Scholz, 2008) (Schmidt 2008 provided additional computations for Southeast Asia). The characterization factors described in Goedkoop *et al.* (2012) to account for eutrophication (Van Zelm *et al.*, 2007) and acidification (Struijs *et al.*, 2011) are available for Europe and the Netherlands, respectively.

A first aspect of the resolution of global characterization factors is the level at which they consider regional differences. Among the three global methods, the water use characterization factors consider differences in the damage on ecosystem quality at high resolution (*i.e.*, intra-country), based on a water stress index. Neither the climate change characterization factor (Schryver *et al.*, 2009), nor the MSA (Alkemade *et al.*, 2009) account for regional differences. It means that the biodiversity (MSA) value of undisturbed forest, or the biodiversity loss when these forests are converted to pasture is the same in Europe and Latin America (despite wide acceptance of greater levels of biodiversity in the latter). However, de Baan *et al.* (2013a) recently developed land use characterization factors based on MSA and accounting for regional differences (among 9 biomes). For land use characterization factors, the second aspect of resolution is the level of detail in the land use categories. The EDP characterization factors covers Central Europe and includes 53 land use and intensity classes. The MSA's coverage is global and includes 13 land use and intensity classes. These two examples illustrate the trade-off between spatial coverage and resolution. Although on a global scale, the MSA categories are coarse and do not permit considering biodiversity in specific and unique biodiversity habitats. This geographical differentiation of biodiversity is well known and although more work is required to achieve this greater specification, there have already been considerable advances that could be integrated into future approaches (e.g. Olson *et al*., 2001).

11.2.3 Biodiversity coverage

The characterization factors presented in Sections 9.2 and 10 mainly describe biodiversity through species richness (e.g. PDF) or species abundance (e.g. MSA). Historically, relative species richness, expressed as PDF.m^2.yr has been used as the unit to express damage at in the endpoint category. The PDF can be interpreted as the fraction of species that has a high probability of non-occurrence in a region due to unfavourable conditions. The PDF is based on the probability of occurrence (POO) and defined as 1-POO. This means the fraction of species that does not occur can also be described as the fraction of the species that has disappeared (Goedkoop & Spriensma

2001). Compared to the three levels and dimensions of biodiversity detailed in Figure 7, the PDF and MSA focus on the species level and the functional and structural attributes of biodiversity have been largely neglected. Souza *et al.* (2013) emphasized functional diversity (FD) as a more appropriate indicator of biodiversity loss in comparison to taxonomic indicators because of the association between species traits and ecosystem functions. The authors used an existing functional diversity index (Petchey & Gaston, 2002) for three different taxonomic groups (mammals, birds and plants) for occupation land use impacts for different eco-regions.

Some methods try to convert species richness into a final measure of damage to ecosystem quality. Goedkoop *et al.* (2012) undertook a rough estimate of a species density factor based on the total number of terrestrial, freshwater and marine registered species combined with the terrestrial area and the volume of fresh and marine waters. The EDP factor developed by Koellner & Scholz (2008) links species richness with ecological damage, using simple – linear and logarithmic – functions. This method could be adapted to develop other characterization factors based on species richness. It does not consider ecological complexities in the relationship between species richness and ecosystem functioning, such as thresholds or tipping points (Hooper *et al.*, 2005). The Pfister *et al.* (2009) characterization factor is based on NPP and therefore also addresses the ecosystem level and function dimension.

When considering the species level, the taxonomic coverage of the characterization factors is often limited. Many studies focus on vascular plants. More rarely, birds, mammals, amphibians or arthropods can also be addressed (see review in Curran *et al.* 2011). Yet, there is a weak correlation in the responses of different taxa to disturbance at global scale (Wolters *et al.*, 2006).

A disadvantage of using species richness as a proxy for biodiversity is that it only records the presence or absence of species within a sampling area and gives equal weight to all species/habitats recorded in a sample, without considering differences in conservation priorities. Moreover, characterization factors based on species abundance do not capture species extinction. There have been efforts to include differences in conservation value of species/habitats in LCA, and these represent important methodological advances. One of the first attempts of modelling included the threat status of species (Mueller-Wenk, 1998), which helps to recognize the conservation priority afforded to some species over others. Weidema & Lindeijer (2001) proposed a first approach for assessing broad categories of land use impacts including (in addition to species richness) values for ecosystem scarcity and ecosystem vulnerability in terms of ecosystem productivity (NPP). Koellner & Scholz (2008) developed characterization factors for Europe, linking numerous classes of land use and intensity to biodiversity, expressed as an Ecological Damage Potential (EDP) indicator. This method has been used in the specific context of livestock (Guerci *et al.*, 2013). Following those authors, Michelsen (2008) also developed a new method for assessing biodiversity indirectly by means of three factors: Ecosystem Scarcity (ES), Ecosystem Vulnerability (EV) and Conditions for Maintained Biodiversity (CMB). Mueller *et al.* (2014) calculated a Biodiversity Damage Potential of land use as the sum of land use occupation and transformation impacts. They adapted the relative species richness method proposed by de Baan *et al.* (2013a) by applying a biodiversity weighting factor based on absolute species richness, irreplaceability and vulnerability. Souza *et al.* (2014) discussed some of the limitations and drawbacks in using each of these approaches.

PART 4

CONCLUSION

12. Complementarities between frameworks

The LEAP biodiversity principles (LEAP, 2016) provide principles applying to both the PSR indicator framework and the LCA framework. They highlight several ways of taking advantage of the complementarities existing between the two frameworks and of discussing future directions to bridge the gap between them. As a first step, they recommend adopting a life cycle perspective when computing PSR indicators.

12.1 COMPLEMENTARITIES OF SCOPE

The methods that are currently available to characterize biodiversity in LCA are reliant on relatively coarse spatial scales and capture only part of the links between livestock and biodiversity. For instance:

- They rely on broad land use classes.
- They have a low level of biogeographical differentiation.
- They include a limited number of midpoint impact categories, and;
- They focus on the species level of biodiversity and on certain taxa.

Given this current state of knowledge, LCA approaches are not well-suited to answering some questions. This is especially the case for questions such as 'is livestock production practice A better than practice B for its effect on biodiversity?' when both production practices occur within one of the broad land use classes of the current LCA approaches. Such approaches, when based on large geographical scales, are much more suited to assessing land use changes impacts across bioregions, and not suited to assessing other more qualitative changes (such as the impacts of over- or under-grazing) within a bioregion. However, LCA is a very useful tool for conducting broad assessment of impacts on biodiversity at large spatial scales and finding hotspots of impact along the supply chain or among spatial entities. LCA could be used to reveal supply chain or spatial hotspots for further investigation with more detailed assessment methods. PSR indicators are part of these more detailed assessment methods and they could be used to differentiate the effect of different practices or expand the analysis to other pressures and biodiversity levels and taxa.

12.2 COMPLEMENTARITIES OF PERSPECTIVE

LCAs address the environmental impact of a product and take into account all stages of production along its life cycle. In contrast, most PSR indicators have focused on environmental impact within a bounded spatial area such as a farm, a landscape or a region.

A life cycle perspective should be adopted when computing PSR indicators. This life cycle perspective could include the impact of feed cultivated off farm, as well as other production stages. Conversely, the spatial perspective of PSR indicators demonstrates the ecological importance of certain scales that are not necessarily those of the production units, such as the impact of landscape-scale processes on biodiversity. Adopting the spatial and landscape perspective could be an important step in improving the ecological relevance of LCA approaches that can otherwise be insensitive to these issues.

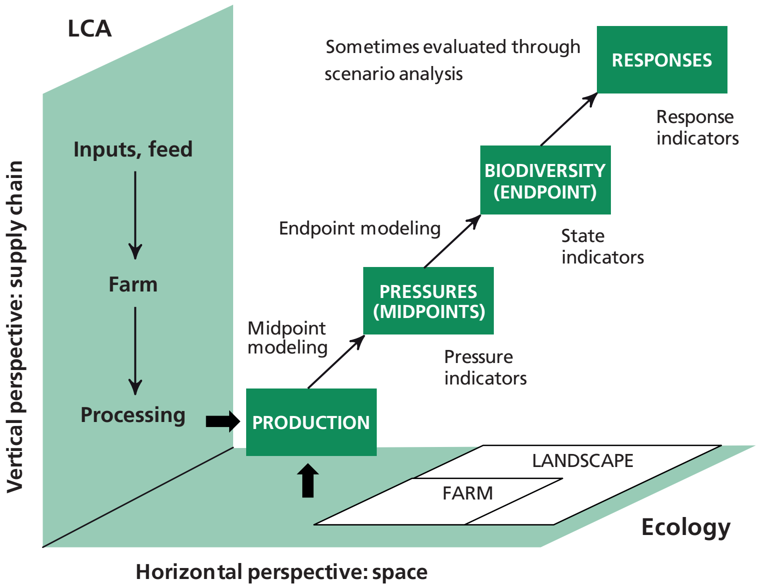
As LCA focuses on products, impacts are often calculated on a 'per unit of production' basis. This approach could also be relevant to PSR indicators in order to tackle the issue of minimizing biodiversity impact while producing a certain amount of food. PSR indicators from the field of ecology and agricultural or animal sciences also show that livestock systems provide a much wider range of ecosystem services than just food production. Agricultural and livestock systems also provide environmental, social and economic services. A future challenge will be to incorporate this wider contribution in LCA studies of livestock systems.

12.3 COMPLEMENTARITIES ALONG THE ENVIRONMENTAL CAUSE-EFFECT CHAIN

The PSR indicator framework and LCA were presented separately; however, they have complementarities and could be combined. They both follow the same environmental cause-effect chain (Figure 13). The main difference is that the PSR framework describes the different points of the environmental cause-effect chain with indicators while the LCA models the link between them. At the different points of the environmental cause-effect chain, complementarities could exist between the PSR and LCA framework.

1. The principle of LCA is to account for the whole life cycle of the products. Pressure, state and response indicators have mainly been computed for livestock production in a bounded area and a single step of the supply chain (e.g. the farm). However, these indicators could also be computed in a life cycle perspective. For instance, pressure indicators reflecting the land use pressure category could be computed along the whole life cycle, by considering the feed cultivated on-farm and feed cultivated off-farm and imported. Similarly, state indicators (e.g. species richness) could be computed in the area used for feed crops both on-farm and off-farm.
2. The first step of LCIA models is to compute midpoint impact categories. Many of these midpoint impact categories (e.g. GHG emissions, land use, eutrophication) correspond to biodiversity pressures (European Commission, 2010). LCIA models could therefore be used to compute pressure indicators that would account for whole life cycle of the livestock product. In order to cover all pressure categories comprehensively, pressures modelled from LCIA could be combined with indicators computed through other methods.
3. A limitation of the LCA framework for computing biodiversity impacts is that no methods exist to account for all the categories of pressure and for the different levels and dimensions of biodiversity. If state indicators were computed in addition to the LCA biodiversity impact, it would be possible to (i) compare the LCA and indicators results to validate the LCA model or to (ii) address biodiversity levels and dimensions that are not covered by existing LCA methods.
4. Response indicators are closely linked to management decision but their relationship with the state of biodiversity is indirect. Some LCA models (consequential LCA in particular) make it possible to explore different scenarios or mitigation options and their effect on midpoint and endpoint impacts. Such LCA models could thus be used to estimate the effect of various response indicators and to select the most relevant.

Figure 13
Complementarities between the Pressure-State-Response indicator framework and the LCA framework along the same causality chain. Complementarities at the different steps are discussed in the main text.



13. Concluding remarks

Measuring the impact of livestock production on biodiversity poses important methodological challenges. These challenges include: the need to address both the positive and negative influences of livestock production on biodiversity; improving the link between local and large scales; and the consideration of a wide range of mechanisms. Across contexts, biodiversity and the factors influencing it vary greatly. Because of this, an absolute and equivalent value of biodiversity does not exist. This makes it difficult for generic assessment frameworks to be relevant. All indicators and assessment frameworks presented in this review have limitations; however, they also have complementarities and there are opportunities for elements to be combined.

Because of these methodological challenges, developing guidance for the quantitative assessment of biodiversity in livestock and other sectors is an emerging area of work. The LEAP partnership set up an international group of expert from various backgrounds – ecologists, LCA experts, members of NGOs and the private sector – to share views on biodiversity assessments and develop *Principles for the assessment of livestock impacts on biodiversity* (LEAP, 2016). The objective of that document is to develop principles applicable to different assessment method in order to guarantee a minimum level of soundness, transparency, scientific relevance, and completeness. This initial step will foster discussions on biodiversity assessment and open the way towards recommending specific methodologies and providing the associated, detailed guidelines.

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APPENDICES

Appendix 1

Systematic review method

The systematic review was conducted by using the combination of the following key words in the Web of Science database.

(biodiversity OR “ecosystem* quality” OR “ecosystem* service*” OR “ecosystem* degradation” OR “ecosystem* function*” OR naturalness)
AND
(livestock OR agricultur* OR farming OR graz* OR pasture* OR cattle OR sheep* OR goat* OR pig* OR poultry)
AND
(indicator* OR assessment* OR effect* OR impact* OR pressure* OR footprint*)

This research yielded 874 articles. After a first selection solely based on titles and abstracts, 137 articles were retained. Examination of the full texts of the articles led to a final pool of 64 articles, which are included in this review.

Appendix 2
Mean Species Abundance values of different land uses

Table A1: Mean Species Abundance values of several land use and intensity classes, as computed by Alkemade *et al.* (2009).

Land use category	Description	MSA	SE
Snow and Ice	Areas permanently covered with snow or ice considered as undisturbed areas	1	<0.01
Bare areas	Areas permanently without vegetation (for example, deserts, high alpine areas)	1	<0.01
Forests			
Primary vegetation	Minimal disturbance, where flora and fauna species abundance are near pristine	1	<0.01
Lightly used natural forests	Forests with extractive use and associated disturbance like hunting and selective logging, where timber extraction is followed by a long period of re-growth with naturally occurring tree species	0.7	0.07
Secondary forests	Areas originally covered with forest or woodlands, where vegetation has been removed, forest is re-growing or has a different cover and is no longer in use	0.5	0.03
Forest plantation	Planted forest often with exotic species	0.2	0.04
Scrublands and grasslands			
Primary vegetation	Grassland or scrubland-dominated vegetation (for example, steppe, tundra, or savannah)	1	<0.01
Livestock grazing	Grasslands where wildlife is replaced by grazing livestock	0.7	0.05
Mad-made pastures	Forests and woodlands that have been converted to grasslands for livestock grazing	0.1	0.07
Agroforestry	Agricultural production intercropped with (native) trees. Trees are kept for shade or as wind shelter	0.5	0.06
Cultivated areas			
Low-input agriculture	Subsistence and traditional farming, extensive farming, and low external input agriculture	0.3	0.12
Intensive agriculture	High external input agriculture, conventional agriculture, mostly with a degree of regional specialization, irrigation-based agriculture, drainage-based agriculture	0.1	0.08
Artificial surfaces	Areas more than 80% built up	1	<0.01

Appendix 3
Other frameworks

FROM ACADEMIC RESEARCH

In the scientific literature, a wide variety of biodiversity indicators, as well as assessment and footprinting methods, have been proposed. The hemoroby concept is a measure of the human influence on ecosystems (Brentrup *et al.*, 2002). It is inversely correlated with naturalness and it closely relates to land use intensity. Brentrup *et al.* (2002) used hemeroby values expressed as Naturalness Degradation Potential to conduct a life cycle impact assessment of land use. Similar to the hemeroby concept, Reidsma *et al.*, (2006) carried out a literature review to determine an ecosystem quality value of several intensity classes of cropland and grassland. Unlike MSA, EDP and PDF characterization factors, which are quantitatively calibrated on species richness/abundance data, the hemeroby values are assigned to classes on a land use intensity gradient from qualitative comparisons.

Vačkář (2012) compared several indicators of biophysical sustainability:

1. The Ecological Footprint is a measure of the demand that human activity places on ecosystems (Global Footprint Network, 2010). Its computation is based on a ratio between demand and yield of a product (e.g. cropland, forest, grazing land, fishing grounds). Authors provide an atlas of the Ecological footprint at global scale and country resolution.
2. Biocapacity is linked to the Ecological Footprint: it reflects what people are able to harness from ecosystems.
3. The HANPP measures how human activities influence net primary production through land conversion and ecosystem use.
4. The Environmental Performance Index (Esty *et al.*, 2008) aggregates and weights 25 indicators related to core policy targets (biodiversity being one of them).
5. The Ecosystem Wellbeing Index measures if the ecosystem is capable of maintaining its diversity, quality, and capacity to support people and wildlife (Prescott-Allen, 2001). Biodiversity, one of the five dimensions of the index, is represented by the percentage of threatened species and protected areas.

Vačkář (2012) conducted a cross-national comparison and showed strong relationships between these indicators. Ecological Footprint and Biocapacity were closely related to the HANPP, and negatively correlated to measures of ecosystem health such as the Ecosystem Wellbeing Index. Most of these indicators were developed to assess biodiversity and sustainability at farm or landscape scale rather than for a specific sector (e.g. livestock production). However, it could be interesting to apply the Ecological Footprint, Biocapacity and HANPP concepts to livestock production in order to transform absolute measures of biodiversity performance into an assessment of the balance between demand and yield of ecosystem services.

The Agri-Environmental Footprint Index (AFI) is a methodology for assessing changes in the environmental impacts at farm scale, and the effects of European AESs (Louwagie *et al.*, 2012). The AFI include a multi-criteria analysis, with several agri-environmental indicators integrated in a context-specific, customisable index. Within the scope of the evaluation set by the evaluators, stakeholders participate in identifying environmental issues and management options, weighting the environmental issues and identifying appropriate farm-level indicators (Mauchline *et al.*, 2012). The context-specific AFI is computed from farm-level data, and indicators are converted to scores and weighted. The sensitivity of the results to changes in scores and weights is also computed. The AFI does not provide a standardized framework allowing comparison of biodiversity performances between different systems and regions; however, it provides a context-specific assessment of changes in relevant environmental impacts.

FROM INTERGOVERNMENTAL ORGANIZATIONS

The FAO Sustainability Assessment for Food and Agriculture (SAFA)

The FAO SAFA (2013) guidelines describe a holistic framework for assessing the sustainability of agriculture, forestry and fisheries value chains. The framework covers the environmental, economic, social and governance dimensions. It is structured by themes (21 themes are covered, including biodiversity), sub-themes and indicators (116 in total). Three types of indicators exist, based on performance (i.e. the state of biodiversity), practice (i.e. response indicators) and target (policies or monitoring plans with targets and ratings based on steps toward implementing them). Biodiversity is addressed at three levels: genes, species and ecosystems. At the ecosystem level, indicators focus on the share, diversity and connectivity of natural and semi-natural habitats. At the species level, they focus on the diversity and abundance of threatened or vulnerable wild species.

Link to more information:
http://www.fao.org/nr/sustainability/sustainability-assessments-safa/it/

The WRI Corporate Ecosystem Service Review

The World Resources Institute has developed a methodology called corporate Ecosystem Service Review (ESR). It offers a framework for companies to proactively develop strategies to manage risks and opportunities arising from their dependence and their impact on ecosystem services. It is particularly relevant to the agriculture sector because of its close interaction with ecosystems. The ESR consist of five steps: select the scope, identify priority ecosystem services, analyze trends in priority services, identify business risks and opportunities and develop strategies. The ESR is mainly based on qualitative questions and criteria and would therefore not be suitable for conducting quantitative assessments of the biodiversity performance of livestock production. However, it could provide interesting elements to integrate with biodiversity indicators targeting the ecosystem level.

Link to more information:
http://www.wri.org/publication/corporate-ecosystem-services-review

FROM NON-GOVERNMENTAL ORGANIZATIONS

High Conservation Value (HCV) approach

The World Wide Fund for Nature has initiated or actively participated in the development of voluntary sustainability standards schemes, of which several use the High Conservation Value approach. Maintaining HCVs is a keystone principle of major sustainability and certification standards in forestry, palm oil, sugar cane and soy production, as well as in biofuels and bioenergy standards, ecosystem carbon management and aquaculture production.

The six HCVs are:

HCV1 – Concentrations of biological diversity including endemic species, and rare, threatened or endangered species, that are significant at global, regional or national levels: e.g. the presence of several globally threatened bird species.

HCV2 – Large landscape-level ecosystems and ecosystem mosaics that are significant at global, regional or national levels, and that contain viable populations of the great majority of the naturally occurring species in natural patterns of distribution and abundance: e.g. a large tract of Mesoamerican flooded grasslands and gallery forests with healthy populations of Hyacinth Macaw, Jaguar, Maned Wolf, and Giant Otter, as well as most smaller species.

HCV3 – Rare, threatened, or endangered ecosystems, habitats or refugia: e.g. patches of a regionally rare type of freshwater swamp.

HCV4 – Basic ecosystem services in critical situations, including protection of water catchments and control of erosion of vulnerable soils and slopes: e.g. forest on steep slopes with avalanche risk above a town.

HCV5 – Sites and resources fundamental for satisfying the basic necessities of local communities or indigenous peoples (for livelihoods, health, nutrition, water, etc.), identified through engagement with these communities or indigenous peoples: e.g. key hunting areas for communities living at subsistence level.

HCV6 – Sites, resources, habitats and landscapes of global or national cultural, archaeological or historical significance, and/or of critical cultural, ecological, economic or religious/sacred importance for the traditional cultures of local communities or indigenous peoples, identified through engagement with these local communities or indigenous peoples: e.g. sacred burial grounds within a forest management area or new agricultural plantation.

The HCV process is the following:

Identify	• **What** are the potential HCVs in this region? • Which values occur in the area? • **Where** are these values located?
↓	
Manage	• What are the existing **threats** to the values? • Habitat **area** needed to maintain the values? • How should forest be **managed**?
↓	
Monitor	• **What** needs to be monitored? • **How** will monitoring be done? • How will the results of monitoring be used?

Link to more information:
http://www.hcvnetwork.org/about-hcvf/the-six-high-conservation-values

The IUCN Red List of Ecosystems

Ecosystem services are increasingly important in the international discourse, e.g. through IPBES (the Intergovernmental Panel on Biodiversity and Ecosystem Services). The Red List of Ecosystems (RLE) risk assessments is one way to enhance capacity in order to strengthen countries' contributions to IPBES, as well as to report on the Aichi targets. From the perspective of RLE, the key areas relate to knowledge of the status of ecosystems and their capacity to deliver services; the risk of losing ecosystem services through reduced extent and condition of ecosystems; and identifying the most important drivers and impact factors working to reduce ecosystems services. Such RLE risk assessments are valuable and effective in different sectors relevant to sustainable development, including:

- Global environmental reporting: they help partner countries provide better information on progress towards the Aichi targets under the Convention on Biological Diversity.
- Conservation: they help prioritize investments in ecosystem management and restoration, reforms of resource use, and are a means of rewarding good ecosystem management.
- Land use planning: they help highlight the risks faced by ecosystems under current and potential land use scenarios (e.g. land conversion, degradation), and the potential effects on services such as clean water, maintenance of soil fertility, pollination, the availability of natural products, and, ultimately, the likely impact on food security.
- Macro-economic planning:they assist in providing a globally accepted standard helping planners (at different levels, but in particular at the national level) to evaluate the risks of ecosystem collapse and the related economic costs of reduced ecosystem services or, conversely, the potential economic benefits of improved management.
- Improvement of governance and livelihoods: they help inform the development of governance systems in ways that improve ecosystem management, livelihood security and social outcomes (gender and equity).
- Private sector: they serve as a means of assessing potential environmental and social benefits and the costs of alternative designs of future development projects as well as monitoring/reporting on environmental impacts.

Standardized Red List criteria allow risks of ecosystem collapse to be assessed objectively, transparently and repeatedly, and highlight losses of ecosystem functionality and services – invaluable information for effective planning and development. At the global level, IUCN will assess the conservation status of the world's terrestrial, freshwater, marine and subterranean ecosystems, aiming to achieve complete coverage by 2025. National and regional assessments are being carried now. Criteria for determining threat categories are based on ecosystem extent, and declines in ecosystem distribution and function over historical, present-day and future time frames.

The results of RLE can support management (conservation, land/water use, agriculture, climate change adaptation, restoration, and food security) decisions with the best available information (spatial, condition, and drivers) on ecosystem degradation and the subsequent loss of services. The RLE, as well as being spatially underpinned, will highlight underlying causes of ecosystem changes (positive or negative). This forms an entry point for action, e.g. restoration, management, governance, gender, tenure. Based on the underlying causes of ecosystem changes, early applications might include: a) improving environmental safeguards by highlighting problems concerning certain interventions, e.g. mining; b) improving landscape management and human well-being, including restoration (e.g. ecosystem, forest); c) highlighting the risks to key ecosystems services (e.g. water, products), which often underpin sustainable development; d) demonstrating how ecosystems are already changing as a result of climate change and highlighting the need for adaptation; e) helping provide advance warning of natural disasters, especially slow-onset ones (drought, sea level rise) and helping mitigate them; assisting longer-term monitoring.

Although Environmental Impact Assessments are often carried out, they are usually performed at the level of sites (district, catchment) or sectors (health, water) rather than at national level. Strategic Environment Assessments may also be carried out, but these tend to be sectoral (e.g. the forest sector). Simple and repeatable national (or sub-national) assessments based on internationally accepted criteria and categories are rarely used or carried out, although the Red List of Threatened Species does this for species. The RLE has the potential to fill that gap at the national and regional levels, and could assist in monitoring the effectiveness of overall national policies on land, water and environmental use, as it would:

- provide a standard and repeatable way of understanding the impacts (positive and negative) of certain approaches, policies, and management practices on ecosystems and the environment;
- inform national government and development partners on the risks of ecosystem collapse, on how it could happen, and on how it could best be mitigated; and
- serve an indicator for assessing the impacts which development cooperation and national policy could have on the well-being of ecosystems.

The InVEST model

The InVEST model is developed by the Woods institute for the environment (Stanford University), WWF, the Nature Conservancy and the institute on the environment of the University of Minnesota. It can be used to quantify, map and value the services provided by ecosystems. Biodiversity is one of its components. *"Patterns in biodiversity are inherently spatial, and as such, can be estimated by analyzing maps of land use and land cover (LULC) in conjunction with threats. InVEST models habitat quality and rarity as proxies for biodiversity, ultimately estimating the extent of habitat and vegetation types across a landscape, and their state of degradation. Habitat quality and rarity are a function of four factors: each threat's relative impact, the relative sensitivity of each habitat type to each threat, the distance between habitats and sources of threats, and the degree to which the land is legally protected. Required inputs include a LULC map, the sensitivity of LULC types to each threat, spatial data on the distribution and intensity of each threat and the location of protected areas."*

http://ncp-dev.stanford.edu/~dataportal/invest-releases/documentation/2_6_0/habitat_quality.html.

FROM THE PRIVATE SECTOR

Sustainable Agriculture Initiative Platform (SAI Platform)

The SAI platform provides a Farm Sustainability Assessment (v2.0), which is available as an Excel tool and an online tool. The online version provides some guidance about using the tool. *"FSA 2.0 is a simple tool to assess farm sustainability, fully in line with the Principles and Practices for sustainable agriculture as they are developed by SAI Platform. FSA 2.0 covers environmental, social and economic aspects.*

An easy scoring mechanism provides farmers with an overview of their farm's sustainability.

The purpose of FSA 2.0 is to:

provide a way to assess farmer sustainability and a basis for improvement plans;

create a single benchmark for certification schemes and proprietary codes;

remove the need for company-specific sustainable agriculture codes."

Together with the Rainforest Alliance, the SAI platform has developed a Sustainable Agriculture Standard (Sustainable Agriculture Network, 2010). This standard includes biodiversity criteria at both species and ecosystem levels (see also Section 6.2)

The SAI platform is developing Biodiversity guidelines to assist corporate members undertaking biodiversity projects in a diversity of regions and countries and using a range of commodities. The guidelines became available in 2015.

http://www.saiplatform.org/

http://www.standardsmap.org/fsa

http://san.ag/web/

International Dairy Federation

The International Dairy Federation is developing a guidance document for biodiversity assessments on dairy farms, with international applicability.

Global roundtable for sustainable beef

The Global Roundtable for Sustainable Beef (GRSB) is a global, multi-stakeholder initiative developed to advance continuous improvement in the sustainability of the global beef value chain through leadership, science and multi-stakeholder engagement and collaboration. The GRSB envisions a world in which all aspects of the beef value chain are environmentally sound, socially responsible and economically viable.

http://grsbeef.org/

Field to Market: The Alliance for Sustainable Agriculture

Field to Market is developing a Habitat Potential Index (HPI) to assess biodiversity on US farms that grow crops such as corn, soy and wheat that can be used to feed livestock.

http://www.fieldtomarket.org/fieldprint-calculator/

Innovation Center for U.S. Dairy

The Innovation Center is drafting indicators and metrics to be included in the Stewardship and Sustainability Guide for U.S. Dairy. These indicators can be used by dairy farmers to measure and report on their biodiversity plans, management and outcomes.

http://www.usdairy.com/sustainability/reporting

The Dairy Sustainability Framework

The Global Dairy Agenda for Action is working together with the International Dairy Federation and others in the dairy industry to implement the Dairy Sustainability Framework (DSF) globally. The DSF includes a biodiversity category which dairy organizations are encouraged to manage, monitor and report publicly: "Direct and indirect biodiversity risks and opportunities are understood, and strategies to maintain or enhance it are established."

http://www.usdairy.com/sustainability/reporting

Nestlé Commitment on Natural Capital

The Nestlé company commits to several principles on Natural Capital, which includes biodiversity, ecosystem services and natural resources. One of these principles is to publicly report on risks (externalities) and responses.

http://www.nestle.com/asset-library/documents/library/documents/corporate_social_responsibility/commitment-on-natural-capital-2013.pdf

Unilever

The Unilever approach to sustainability includes a biodiversity component. In particular, there are several requirements for the sustainable sourcing of agricultural raw materials.

http://www.unilever.com/sustainable-living-2014/reducing-environmental-impact/sustainable-sourcing/protecting-biodiversity/

RESTORATION AND REVEGETATION INITIATIVES

Landcare International

Landcare is a community-driven programme that encourages activities which integrate management of environmental assets with agricultural production. The Landcare model is based on self-forming landholder groups that work on natural resource management issues of interest to them. Landcare started in Australia in 1986 and has since spread to 12 other countries including South Africa, USA, Kenya and New Zealand (http://www.worldagroforestry.org/projects/landcare/). A keystone Landcare activity is revegetation to buffer existing native vegetation remnants, enhance connectivity between remnants, create windbreaks for livestock protection, protect waterways and aquatic biodiversity and provide habitat for locally endangered species. Evaluation of Australian and New Zealand Landcare projects found that revegetation significantly increased species richness (Blackwell *et al*, 2008; Lindenmayer *et al*, 2012a; Munro *et al*, 2007; Vesk *et al*, 2008).

Between 2000 and 2012, more than half (58 percent) of Australian dairy farmers undertook revegetation programmes on their properties and 47 percent of farmers protected areas of remnant vegetation (Watson & Watson 2012). Many of these revegetation programmes were supported by Landcare. Revegetation plans are informed by the Australian dairy industry environmental assessment tool, DairySAT. http://www.dairysat.com.au/

COMDEKS - Community Development and Knowledge Management for the Satoyama Initiative

COMDEKS provides small grants to local community organizations in 20 countries to develop sound biodiversity management and sustainable livelihood activities. Target landscapes include pastoral systems. Restoration practices include revegetation, establishment of connectivity corridors and agro-forestry. The project collects and distributes knowledge and experiences from successful on-the-ground actions for replication and upscaling in other parts of the world. Additionally, as part of ongoing collaboration with UNU-IAS and Biodiversity International, COMDEKS is piloting a set of socio-ecological production landscape indicators to help tracking, measurement and understanding of the resilience of target landscapes. http://comdeksproject.com/

USDA Restore Conserve Program

The Conservation Reserve Program (CRP) is a land conservation programme administered by the Farm Service Agency (FSA). In exchange for a yearly rental payment, farmers enrolled in the programme agree to remove environmentally sensitive land from agricultural production, and plant species that will improve environmental health and quality. Contracts for land enrolled in CRP are for 10-15 years. The long-term goal of the programme is to re-establish valuable land cover to help improve water quality, prevent soil erosion, and reduce loss of wildlife habitat.

http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=crp

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http://www.fao.org/partnerships/leap