A review of indicators and methods to assess biodiversity. Application to livestock production at global scale

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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 LEAP Technical Advisory Group (TAG) on biodiversity with a common ground of knowledge on the main biodiversity indicators and assessment methods that could be applied to livestock production. The objective of this group was to develop *Principles for the assessment of livestock impacts on biodiversity* (LEAP biodiversity principles, LEAP, in prep.). This document went indeed beyond its initiatial objective providing an important body of information on biodiversity assessment which could be useful to stakeholders of the livestock and related sectors with an interest in the topic.

This technical review was initially drafted by Félix Teillard (FAO) as preparatory document for the LEAP Technical Advisory Group (TAG) on biodiversity. Once shared the initial draft with TAG members, the following task force led by Félix Teillard (FAO, Italy; INRA, France) was formed for the finalization of this report: Assumpció Anton (IRTA, Spain), Bertrand Dumont (INRA, France), John Finn (Teagasc, Ireland), Beverly 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).

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The LEAP Steering Committee provided overall guidance for the activities of the Partnership and helped review and cleared this document for public release. During development of the guidelines the LEAP Steering Committee was composed of:

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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]
Hotspot analysis	Hot spot analysis aims to define areas of high occurrence versus areas of low occurrence of a feature of interest. Here, it refers to an assessment of the relative contribution of e.g. different pressures and threats, with the aim of identifying those that make the strongest contribution to biodiversity loss. [LEAP Biodiversity TAG]
Hotspot, biodiversity	A hotspot for biodiversity represents a geographical areas where there is a coincidence of high biodiversity and high level of biodiversity threats. [LEAP Biodiversity TAG]

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 mineralised. 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 to forest decline and lake acidification. [Adapted from Product Environmental Footprint Guide, European Commission, 2013]

Allocation Partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems [ISO 14044:2006, 3.17]

Background system The background system consists of processes on which no or, at best, indirect influence may be exercised by the decision-maker for which an LCA is carried out. Such processes are called "background processes." [UNEP/SETAC Life Cycle Initiative, 2011].

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]

Classification Assigning the material/energy inputs and outputs tabulated in the Life Cycle Inventory to impact categories according to each substance's potential to contribute to each of the impact categories considered.[Adapted from: Product Environmental Footprint Guide, European Commission, 2013]

Data quality Characteristics of data that relate to their ability to satisfy stated requirements [ISO 14044:2006, 3.19]

Dataset (both LCI dataset and LCIA dataset) A document or file with life cycle information of a specified product or other reference (e.g., site, process), covering descriptive metadata and quantitative life cycle inventory and/or life cycle impact assessment data, respectively. [ILCD Handbook, 2010]

Direct Land Use Change (dLUC) Change in human use or management of land within the product system being assessed [ISO/TS 14067:2013, 3.1.8.4]

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, which damage individual species and change the structure and function of the ecosystem. Eco-toxicity 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.
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 favors nitrophilous plant species and modifies the composition of the plant communities. [Adapted from: Product Environmental Footprint Guide, European Commission, 2013]
Foreground system	The foreground system consists of processes which are under the control of the decision-maker for which an LCA is carried out. They are called “foreground processes” [UNEP/SETAC Life Cycle Initiative, 2011]
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].
Indirect Land Use Change (iLUC)	Change in the use or management of land which is a consequence of direct land use change, but which occurs outside the product system being assessed [ISO/TS 14067:2013, 3.1.8.5].
Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned [ISO 14044:2006, 3.39].
Impact category indicator	Quantifiable representation of an impact category [ISO 14044:2006, 3.40].

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 crop land, grass land, 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 GHG emissions Sum of GHG emissions resulting from all stages of the life cycle of a product and within the specified system boundaries of the product.[PAS 2050:2011, 3.30]

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]

Life Cycle Interpretation Phase of life cycle assessment in which the findings of either the inventory analysis or the impact assessment, or both, are evaluated in relation to the defined goal and scope in order to reach conclusions and recommendations [ISO 14044:2006, 3.5]

Normalization After the characterisation step, normalisation is an optional step in which the impact assessment results are multiplied by normalisation factors that represent the overall inventory of a reference unit (e.g. a whole country or an average citizen). Normalised impact assessment results express the relative shares of the impacts of the analysed system in terms of the total contributions to each impact category per reference unit. When displaying the normalised impact assessment results of the different impact topics next to each other, it becomes evident which impact categories are affected most and least by the analysed system. Normalised impact assessment results reflect only the contribution of the analysed system to the total impact potential, not the severity/relevance of the respective total impact. Normalised results are dimensionless, but not additive. [Product Environmental Footprint Guide, European Commission, 2013]

Ozone depletion Impact category that accounts for the degradation of stratospheric ozone due to emissions of ozone-depleting substances, for example long-lived chlorine and bromine containing gases (e.g. CFCs, HCFCs, Halons). [Product Environmental Footprint Guide, European Commission, 2013]

Particular matter Impact category that accounts for the adverse health effects on human

	health caused by emissions of Particulate Matter (PM) and its precursors (NOx , SOx , NH3) [Product Environmental Footprint Guide, European Commission, 2013]
Photochemical ozone formation	Impact category that accounts for the formation of ozone at the ground level of the troposphere caused by photochemical oxidation of Volatile Organic Compounds (VOCs) and carbon monoxide (CO) in the presence of nitrogen oxides (NOx) and sunlight. High concentrations of ground-level tropospheric ozone damage vegetation, human respiratory tracts and manmade materials through reaction with organic materials.[Product Environmental Footprint Guide, European Commission, 2013]
Primary data	Quantified value of a unit process or an activity obtained from a direct measurement or a calculation based on direct measurements at its original source [ISO 14046:2014, 3.6.1]
Product(s)	Any goods or service [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]
Reference flow	Measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit [ISO 14044:2006, 3.29]
Releases	Emissions to air and discharges to water and soil [ISO 14044:2006, 3.30]
Reporting	Presenting data to internal management and external users such as regulators, shareholders, the general public or specific stakeholder groups [ENVIFOOD Protocol: 2013]
Resource depletion	Impact category that addresses use of natural resources either renewable or non-renewable, biotic or abiotic. [Product Environmental Footprint Guide, European Commission, 2013]
Secondary data	Data obtained from sources other than a direct measurement or a calculation based on direct measurements at the original source [ISO 14046:2014, 3.6.2]. Secondary data are used when primary data are not available or it is impractical to obtain primary data. Some emissions, such as methane from litter management, are calculated from a model, and are therefore considered secondary data.
Sensitivity analysis	Systematic procedures for estimating the effects of the choices made regarding methods and data on the outcome of a study [ISO 14044:2006, 3.31]
Soil Organic Matter (SOM)	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]
Uncertainty analysis	Systematic procedure to quantify the uncertainty introduced in the results of a life cycle inventory analysis due to the cumulative effects of model imprecision, input uncertainty and data variability [ISO 14044:2006, 3.33]
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 body	Entity of water with definite hydrological, hydrogeomorphological, physical, chemical and biological characteristics in a given geographical area Examples: lakes, rivers, groundwaters, seas, icebergs, glaciers and reservoirs. Note 1 to entry: In case of availability, the geographical resolution of a water body should be determined at the goal and scope stage: it may regroup different small water bodies. [ISO 14046:2014, 3.1.7]
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 consumption can be 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]
Water withdrawal	Anthropogenic removal of water from any water body or from any drainage basin, either permanently or temporarily [ISO 14046:2014, 3.2.2].
Weighting	Weighting is an additional, but not mandatory, step that may support the interpretation and communication of the results of the analysis. Impact assessment results are multiplied by a set of weighting factors, which reflect the perceived relative importance of the impact categories considered. Weighted impact assessment results can be directly compared across impact categories, and also summed across impact categories to obtain a single-value overall impact indicator. Weighting requires making value judgements as to the respective importance of the impact categories considered. These judgements may be based on expert opinion, social science methods, cultural/political viewpoints, or economic considerations. [Adapted from: Product Environmental

Footprint Guide, European Commission, 2013]

Introduction

1.1 The influences of livestock on biodiversity

Around 30% of the Earth's land surface is currently dedicated to livestock production (Monfreda *et al.*, 2008; Ramankutty *et al.*, 2008), through pastures (≈ 25%) and feed crops (≈ 5%). 30% of the terrestrial habitats are, therefore, directly modified by livestock, in various ways. At one extreme, undisturbed habitats can be destroyed, such as in conversions of primary forest to pastures or feed crops in the Brazilian Amazon (Lambin *et al.*, 2003; Wassenaar *et al.*, 2007; Nepstad *et al.*, 2009); although livestock is not the only driver and overall deforestation was 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 livestock. This relationship may be related to herbivory being a factor that shapes biodiversity in many ecosystems (Frank, 2005), where livestock has taken over the role of wild herbivores when under adequate management (Eriksson et al., 2002; Bond & Parr, 2010). In Europe, extensive livestock grazing is key to maintaining permanent grassland habitats that have 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 savannas, pastoralism is often compatible with wildlife and can enrich savanna landscapes and their biodiversity (Reid, 2012). Livestock producers can also help in preserving biodiversity through control of feral animals and weeds and managing the risk of damaging wildfires. Other types of habitat modifications 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 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 nitrogen and phosphorus diffuse pollution (Jongbloed & Lenis, 1998). Diffuse nutrient pollution has a great impact on aquatic ecosystems by 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 but 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 the livestock-related GHG emissions on biodiversity. Livestock can also have positive effect on biodiversity that is facing climate change. Klein *et al.* (2004) have shown in 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, the projected population growth and global increase of per capita income is predicted to lead to a dietary shift toward higher demand of livestock products and shall put more pressure on land and resources (McMichael *et al.*, 2007; Wirsenius *et al.*, 2010). For instance, meat consumption in China and Indonesia has increased significantly, and dairy consumption is increasing in India (FAOSTAT, 2014). With its burgeoning middle class, annual meat consumption in China has gone from being a third of that of the U.S. in 1978, to now more than double it (according to the Earth Policy Institute, 2012). Dietary shifts in emerging countries will have a major effect on the global demand for livestock products. Livestock production thus faces the challenge of satisfying an increasing food demand 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 conference of Rio de Janeiro (1992) on sustainable development. This conference opened the way toward the ratification of the Convention on Biological Diversity (CBD) in 2002, where 190 countries agreed to achieve a significant reduction in the rate of biodiversity loss. Biodiversity is an important item in the policy agenda, not only for its intrinsic value, but also because of its key role in supporting ecosystem services that benefit to human societies and economy. Biodiversity is essential to human wellbeing through different categories of ecosystem services – provisioning (food, water, wood, fuel, fiber, medecines, genetic resources), supporting (*e.g.*, water cycling, soil formation), regulating (*e.g.*, climate and erosion regulation) and cultural (*e.g.*, aesthetic, educational) (MEA, 2005). It supports resilience and function of ecosystem, *i.e.* capacity to sustain such services (Loreau *et al.*, 2001; Hooper *et al.*, 2005; Classen *et al.*, 2014). Regarding the contribution of biodiversity to economy, Costanza *et al.* (1997) estimated that the value of 17 selected ecosystem services was higher than the global gross national products.

In many ecosystems or biomes, biodiversity and livestock play a role in shaping up the landscape. The livestock sector is both a provider and a user of biodiversity and ecosystem services (Zhang *et al.*, 2007, Huntsinger & Oviedo 2014). As a human activity, the livestock sector is a user of ecosystem services. Key ecosystem services supporting livestock production include, amongst others, biomass production (provisioning service); micro-organism cycling of nutrients, soil formation, nitrogen fixation (supporting services); and pollination, pest control, climate regulation, 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 attain 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 nematods (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 for 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 to mitigate its negative impacts. Such frameworks could also foster synergies between the positive influences of livestock and the value of biodiversity for the sector. Assessment frameworks reveal the most efficient systems and those requiring improvements. They also help developing evidence-based environmental policies targeting the livestock sector (Gill *et al.,* 2010).

The large majority of existing assessments of livestock environmental performances have focused on GHG emissions. They used mostly a wide range of Life Cycle Assessment (LCA) approaches in order 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 criteria 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 advice on 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 to mitigate 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).

Intensification often results in lower biodiversity levels because of the associated habitat changes and negative effects of nutrient pollution and chemical inputs. Intensifying the production of already cultivated areas may spare land surface for biodiversity conservation although this view 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 are the primary method to convert plant biomass into food edible by humans in many marginal lands (Rodriguez, 2008) and they have a wider range of contribution to socio-economic activity and sustainable development than solely food production (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 do not consider the contribution of extensive systems to ecosystem services and the maintenance of biodiversity. If biodiversity and ecosystem services were considered as a product, emission intensity could 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 reveal what mitigation options will improve the overall sustainability of livestock production.

1.4 Objective

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

We conducted a systematic review of scientific articles (details in Section 6.1 of the Appendix) 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. These frameworks 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 Chapter 2, we describe biodiversity indicators and structure them using the Pressure-State-Response (PSR) framework (OECD, 1993). The PSR framework has been broadly used to structure biodiversity indicators and facilitate their interpretation. Chapter 2 also provides an overview of the different categories of pressures that livestock put on biodiversity (Section 2.2) and of the different levels and dimensions of biodiversity that can be described (Section 2.4). In Chapter 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 Discussion (Chapter 4) highlights complementarities between the PSR and LCA frameworks. 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, Section 6.3 of the Appendix 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 that we present apply well 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, *in prep.*)

This review focuses on wild biodiversity; the genetic, domestic diversity of livestock breed and crop varieties are not within the scope of this review.

2 The Pressure-State-Response indicator framework

2.1 The Pressure-State-Response framework

Indicators are a crucial tool to monitor biodiversity impacts or the improvement of the biodiversity performances. Indicators share certain properties: being rigorous, repeatable, widely accepted and easily understood (Balmford *et al.*, 2005). Making a selection among the many existing biodiversity indicators (EEA 2003 identified more than 600 of them at the European scale) should be based on logical frameworks (EEA, 2007). The pressure-state-response (PSR) framework (OECD, 1993) has been widely used to develop and structure biodiversity indicators. Several frameworks were derived from the original PSR: the driver-pressure-state-impact-response (Smeets *et al.* 1999) or the 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) of the stakeholders (political, socio-economic), undertaken to reach 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 2.1).

The PSR framework helps informing policy-makers by providing indicators that are structured and easy to interpret (Smeets *et al.*, 1999). The linear cause-effect relationships are a simplification that can be poorly adapted 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). The PSR framework has been used in global biodiversity assessments. 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 efforts of political responses. The CBD has encouraged the development of biodiversity indicators to monitor progress toward the target of reducing the rate of biodiversity loss.

At the global and European levels, the CBD (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 review existing indicators and identify gaps in indicator and data availability.

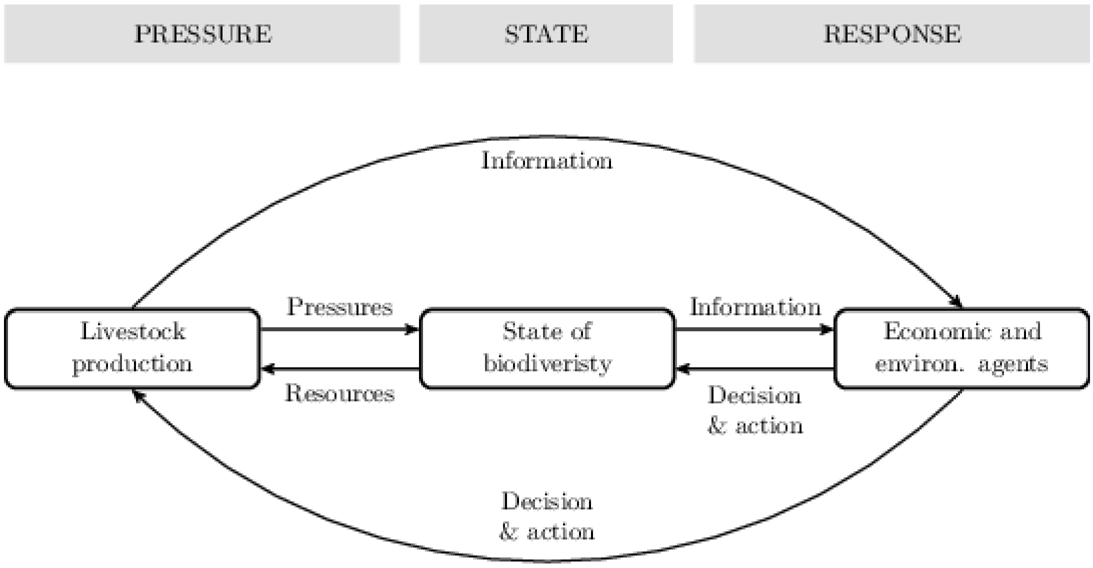

Figure 2.1: The Pressure-State-Response framework applied to livestock production and biodiversity (adapted from OECD 1993).

2.2 Pressure and benefits 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 being that are more specific to the context of livestock production (Figure 2.2).

The influence of livestock on biodiversity is not restricted to pressures, several types of benefits also exist. Pressure and benefits are often two sides of the same coin. For almost every category of biodiversity pressure identified in Figure 2.2, environmentally sound livestock production practices can lead to the opposite 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.

PRESSURE 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.

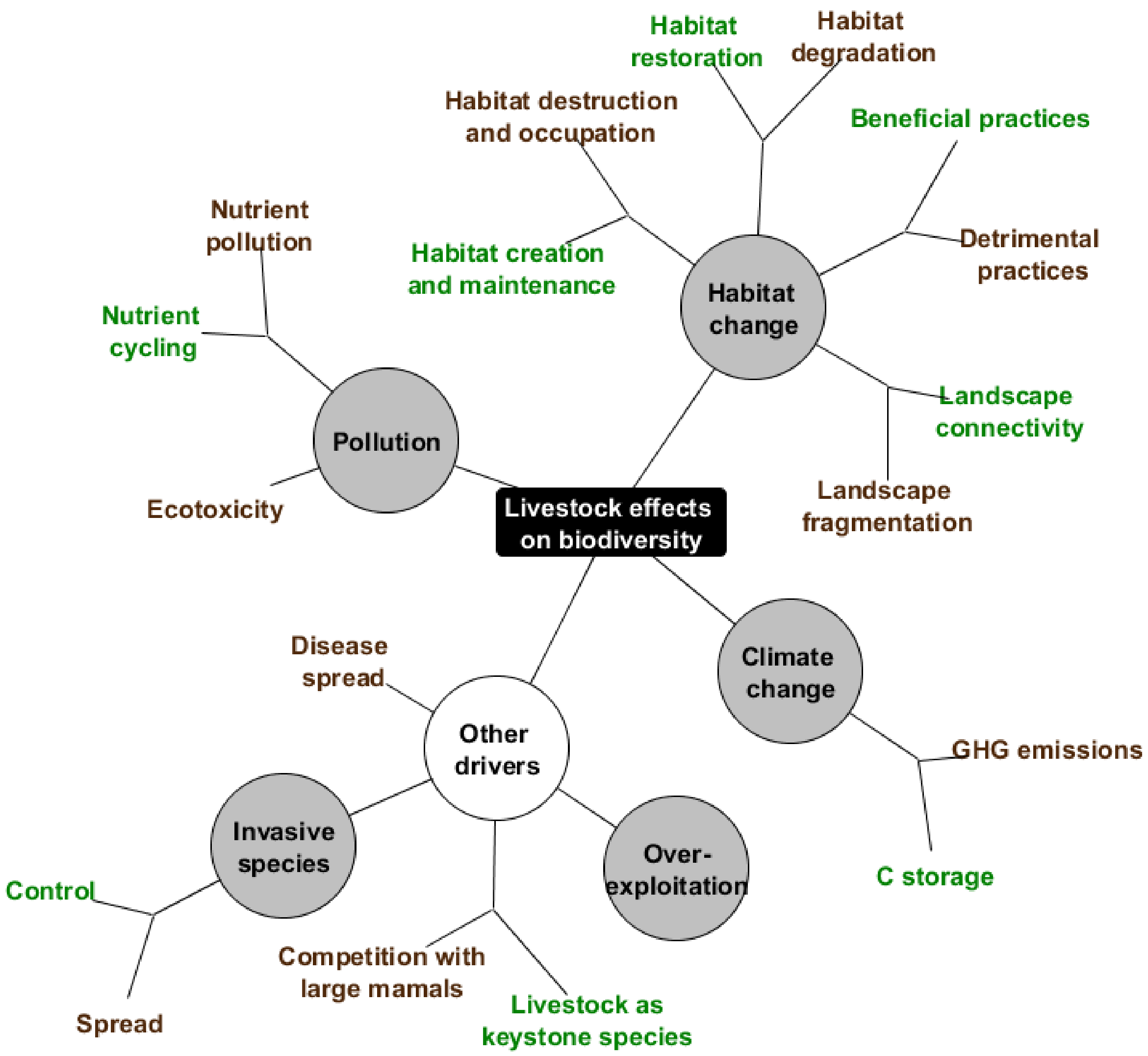

Figure 2.2: Overview of the categories of influences that livestock have on biodiversity. The five main drivers of biodiversity loss recognized by the Millennium Ecosystem Assessment (2005) appear in grey circles. However, for most of these drivers, livestock can either put pressure (brown) or provide benefits (green) to biodiversity.

2.2.1 Habitat change

2.2.1.1 Pressure: Habitat destruction and fragmentation

CONTEXT – Livestock production is responsible for a strong spatial dynamic of land cover and land use change, which is driven by both global (*e.g.*, demand, market opportunities, policy interventions, climate) and local factors (*e.g.*, resource scarcity, social organization) (Lambin *et al.*, 2003; Wassenaar *et al.*, 2007). This dynamic of land cover change can lead to the destruction or modifications of biodiversity. One striking example of such destruction 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% of all agricultural lands (Steinfeld et al., 2006). In total, the accumulated area of deforestation in Brazilian Amazon reached 58.7 million hectares in 2000 (Kaimowitz *et al.*, 2004). Soybean has also been a driver of deforestation, mainly due to an increased global demand. Land for soybean production, grew more than twofold between the years 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 have responsibilities.

There is often a strong, linear correlation between habitat destruction and biodiversity loss (**Error! Reference source not found.**). 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 restrained (Soares-Filho et al., 2006; Wright & Muller-Landau, 2006). In terms of ecosystem services, Amazonian rainforests account for about 15% of global terrestrial photosynthesis, they represent a considerable carbon sink, and the evaporation and condensation over Amazonia influence the 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). The negative effects of habitat loss on biodiversity are therefore worsened by fragmentation (**Error! Reference source not found.**). Fragmentation leads to smaller and more isolated patches of the original habitat, for which the island biogeography theory predicts reduced number of species (MacArthur, 1967; Levins, 1969). Small habitat size limits the number of species (as also shown by the species-area relationship, Connor & McCoy, 1979) and increase local species extinctions, while isolation is an obstacle to colonization. In practice, reduced biomass (Laurance, 1997) and species loss in fragmented habitats have been widely evidenced (review in Turner 1996).

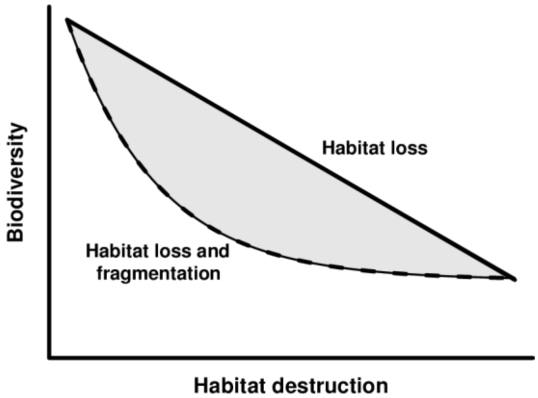

Figure 2.3: Theoretical effect of habitat loss and fragmentation on biodiversity (adapted from Andrén & Andren (1996). The negative effects of habitat loss on biodiversity are exacerbated by fragmentation, *i.e.* for the same area of habitat loss, negative effects on biodiversity are more important if the remaining habitat is fragmented.

Since 2004 in Brazil, enforcement of laws, interventions in soy and beef supply chains, expansion of indigenous reserves, protected and sustainable use areas have contributed to a 70% decline in the annual deforestation rate (Nepstad *et al.*, 2014). More than 85000 km^2 of Amazon rainforest were saved, which led to a strong reduction in GHG emissions (Boucher et al. 2014) while at the same time Brazil was able to increase 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 other countries that import soybean from this region. For instance, several European dairy company have committed to the Roundtable Responsible Soy certification (2014) which encourages future soybean is produced in a responsible manner to reduce social and environmental impacts while maintaining or improving the economic status for the producer.

SCOPE – Today, deforestation driven by livestock production mainly occurs as deforestation in humid regions, with some deforestation still in other regions including Australia (Bradshaw, 2012) and sub-Saharan Africa (Davidson *et al.*, 2003). The main drivers of deforestation differ among global regions (Boucher *et al.* 2011). Deforestation for pastures and soybean cultivation is the most important in Latin America, which is why the Brazilian Forest Code now includes interventions that specifically target soy and beef supply chains. Different drivers are predominant in other regions, *e.g.*, logging in Africa and Southeast Asia, palm plantations in Indonesia and Malaysia. Palm kernel cakes and other palm oil by-products are used as feed for livestock. Sustainability initiatives also target the palm oil sector, such as the Roundtable on Sustainable Palm Oil (2014) which developed a Certified Sustainable Palm Oil standard.

Pastures cover 70% of the global agricultural land uses (Steinfeld et al., 2006) and a large majority of the deforested areas. Among other species and production systems, grass-fed ruminant systems at the global scale are important users of land and potentially have 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 edible to humans. This ability makes it possible to take advantage of a wide rangeland areas that would be too dry or infertile to grow crops. However, the conversion of grass to beef is inefficient which means that pasture-based beef production requires more area than monogastric such as poultry or pork (Wirsenius et al., 2011). However, in some contexts, grass-fed ruminants are the only way to maintain semi-natural habitats against two opposite conversion pressures – conversion to arable land or abandonment leading to conversion to forests – which can both result in biodiversity loss (details in Section 2.3). Because soybean are also grown on deforested areas, more intensive systems and other species than cattle can also have a contribution to the deforestation pressure when they rely on this type of feed crop.

Deforestation is not the only type of habitat destruction driven by livestock production. In certain regions such as North America or Europe, grassland is being converted to arable land, which include feed crops (Gibson, 2009). Urban sprawl into agricultural land is another type of habitat destruction. Although it is not directly driven by livestock production, it can displace agricultural production into new land (in the same country or even in other global regions) and lead to further habitat destruction (Paül & Tonts, 2005; Chen, 2007).

PRESSURE INDICATORS

- Habitat destruction can be calculated as a rate of conversion over time. The rate of deforestation in the Amazon was 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).

DATA AVAILABILITY – Several free datasets of land cover are available at the global or continental level (**Error! Reference source not found.**). 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 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 fewer number of land cover classes that does not allow to distinguish between fine land use categories (*e.g.*, different grassland types or cropland intensities).

Table 2.1. Examples of global and continental land cover datasets.

Data	Based on	Year	Resolution	Nr. of classes
GLC	SPOT VEGETATION	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

2.2.1.2 Benefit: Extensive use, habitat creation and maintenance

CONTEXT – In Europe, there is a long history of farming which provided time for a large pool of species to adapt and specialize to agricultural land uses (Bignal & McCracken, 2000; Benton *et al.*, 2002). Today, extensively managed, permanent grasslands are among the habitats with highest biodiversity levels (Baldock *et al.*, 1993). Bignal & McCracken, (1996) estimated that more than 50% of Europe’s most highly valued biotopes for biodiversity occur in low intensity farmland. Without livestock, semi-natural grasslands would be lost through ecological succession habitats of lower conservation value; these open habitats would gradually “close” into shrubland and ultimately forest, habitats with lower conservation value. This habitat loss would involve the loss of many specialized species. In certain areas such as Eastern Europe, abandonment of agricultural activities can be as equally 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 species of open habitats (Cremene et al., 2005). In both Eastern and Western Europe, calcareous grasslands are extraordinarily species-rich habitats which are particularly threatened by the abandonment of grazing activities (Poschold & WallisDeVries, 2002). In the Mediterranean region, grassland-shrubland mosaics in the also suffer from the abandonment of traditional grazing activities which results in the loss of species diversity and endemicity (share of species which are 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 *et al.*, 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 their extensive use for livestock production (*e.g.*, 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). The effect of grazing on vegetation can increase species richness (WallisDeVries *et al.*, 2002). Livestock can use abundant and low-quality food: it 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 favors 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 traditional water infrastructure, promotes the habitat of organisms such as amphibians (Canals et al., 2011) that are of high conservation value and may otherwise disappear from the landscape.

The European Union (EU) recognizes well that certain farming systems - extensive livestock production in particular - have a high biodiversity value. The European Environment Agency has put efforts to characterize and map 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 play a key role in defining such HNV areas. At the policy level, the EU has developed Agri-Environment Schemes (AESs). AESs propose subsidies to farmers, based on voluntary compliance, for adoption of management practices that reduce environmental pollution, and preserve biodiversity and landscapes. These practices correspond to extensification at local and landscape scales. Many of them promote biodiversity in livestock farms, *e.g.*, maintenance of permanent grasslands, reduced fertilization or stocking rates (see **Error! Reference source not found.** also). More generally, AESs are part of the rural development goal of the EU Common Agricultural Policy. Other measures include subsidies to agricultural activity in “less favored area”; their abandonment would be a threat for both rural development and biodiversity.

The positive role of livestock for biodiversity has been thoroughly studied in Europe but it is not restricted to this region. In China, Akiyama & Kawamura (2007) also report situations where moderately grazed sites have higher species diversity than both 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, the more productive and resilient grassland is created and maintained by large herbivores that increase both the productivity and carrying capacity of the ecosystem (Van der Wal, 2006). In Australia, ecosystems have evolved with fire, they can benefit from livestock grazing. When used strategically, it maintains 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 US, Chaplin-Kramer *et al.* (2011) recently highlighted how crucial rangeland beef systems 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 the total pollination service value in the US, representing $3.07 billion (in 2003). Beside pollination, rangeland in the US and elsewhere provide many types of ecosystem services, e.g., safe food and clean water supply, carbon sequestration, cultural services (Havstad et al. 2007, see also Section 0). Livestock production managed in a sustainable manner is crucial to prevent encroachment, erosion and degradation of the rangelands. It maintains high biodiversity levels and ensures the healthy functioning of the rangeland ecosystems and their capacity to provide ecosystem services.

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 which let time have elapsed for species to adapt and grazing by livestock is the only way to maintain this habitat and its unique biodiversity. Natural grasslands also occur extensively around the world, as shown by the World Wildlife Fund (WWF) ecoregion map which characterizes potential vegetation. They typically occur in areas with long dry seasons (Watkinson & Ormerod, 2001), and where wild species of grazing herbivores are present, as in African savannas. In such areas, grazing livestock are not required to maintain natural grassland and sometimes compete with wild herbivores (Homewood *et al.,* 2001; Madhusudan 2004, see also Section 2.2.44). However, its careful management can also lead to neutral or positive effects on biodiversity (DuToit & Cumming, 1999). Management can be so important as to be determinant in whether livestock drives to oak savanna degeneration, in the case of sedentary herds (López-Sánchez et al 2014), or to oak savanna sustainability, in the case of seasonally mobile herds (Carmona et al 2013).

INDICATORS

- The 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.
- In the same line, the proportion of farm area with field margins (or other agro-ecological infrastructures), or on which biodiversity-friendly practices such as late grazing or exclusion of animals at flowering peak are being applied could be used as an indicator of the compliance of farm management with biodiversity conservation.

- Within semi-natural grassland, particular practices promote biodiversity and can also be measured: moderate livestock density, absence of fertilization. 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, allowing to deal with different types of information (qualitative or quantitative, more or less 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; Section 2.3.2.3.)
- In order to measure the positive effects of livestock on biodiversity, state indicators (e.g., grassland species richness, water provision service in a rangeland) will be very useful; and we describe them in Section 2.3.

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 **Error! Reference source not found.**. However, comparing these sources with data on livestock density (Section 2.2.1), fertilization (Section 2.2.2) or potential vegetation maps could make it possible to reveal the areas of semi-natural grassland and rangelands benefiting biodiversity.

2.2.1.3 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 factors, including overgrazing (*i.e.*, when livestock density exceeds the carrying capacity of the rangeland), climatic factors and changes in fire regime (**Error! Reference source not found.**). 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 favor 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 savannas (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 2.3). It maintains open grassland habitats and has an important role for 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 and the associated biodiversity loss is a concern in Africa but also in other arid regions of Australia and North America. About 2.1–2.6 million km^2 are 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 decreases the competition for 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 modifies 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).

Table 2.2: Pathways of habitat destruction and degradation across global bioclimatic conditions (adapted from Asner et al. 2004).

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

SCOPE – Among systems, those relying on grazing have the most important contribution to the habitat degradation pressure because overgrazing is one of its main causes.

Among geographical areas, all ecosystems do not have the same sensitivity to grazing pressure and degradation. This sensitivity depends on bioclimatic and edaphic conditions (**Error! Reference source not found.**). The grazing intensity at which the habitat becomes degraded is lower in arid and humid conditions, than in temperate conditions. Grazed areas are more vulnerable to degradation pressure at the extremes of the climatic gradient (Table 2.2). Milchunas & Lauenroth (1993) proposed that the relationship between grazing and biodiversity is also a function of the evolutionary grazing history of the 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.

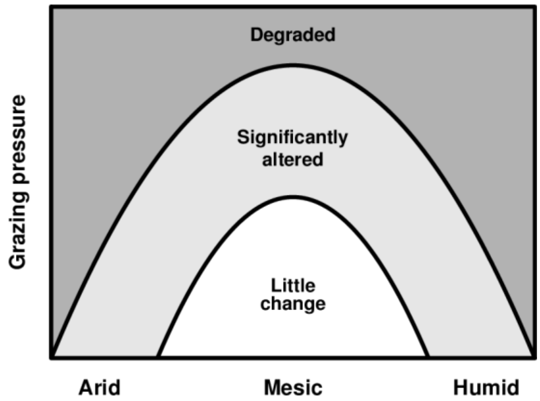

Figure 2.4: Effect of grazing pressure on habitat degradation across global bioclimatic conditions (adapted from Asner *et al.* 2004).

PRESSURE INDICATORS –

- The Normalized Difference Vegetation Index (NDVI) is a measure 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 the Section 2.3.4 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 the 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. OECD (2001) 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, 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 which 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 2.2.1 details data that could describe overgrazing and Section 2.2.4 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, that could be too large compared to the spatial scale required for an environmental assessment or for decision making.

2.2.1.4 Benefit: Habitat restoration

CONTEXT – In addition to clearing shrubs and trees, the re-introduction of livestock grazing can be used to restore abandoned grassland and the high biodiversity levels associated with these open habitats (see Section 2.2.1.2). There are examples of restoration through livestock grazing which resulted in increased species richness of vascular plants (Pykälä, 2003) or arthropods (Pöyry et al., 2004). The restoration of abandoned grassland is not always successful in recovering high species richness; therefore, 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 2.2.1.3). Conversely, if managed well, livestock can prevent degradation or even be a restoration tool. Rational livestock management or rotational grazing in the initial steps of grassland degradation can be a viable option (Yong-Zhong et al., 2005; Zhou et al., 2005). However, according to the context, removal of livestock for several years can be necessary for restoration.

SCOPE – Livestock is a relevant tool for restoration in all grassland and rangeland areas. In areas experiencing more strongly the effects of 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

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

provenance species are dying. Recent studies detailed how land managers could select species for re-vegetation 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.

INDICATOR AND DATA

- For the restoration of abandoned grassland state biodiversity indicators (Section 2.3) will be the most useful to ensure that the re-opening of the habitat do result in biodiversity gains. Indicators of the low management intensity (Sections 2.2.1.5, 2.2.1.2) 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 the grazing pressure and species distribution.
- At large scale, see Section 2.2.1.2 for indicator and data reflecting the degradation-restoration gradient.

2.2.1.5 Pressure: Detrimental practices and intensity

CONTEXT – Agricultural intensity can be defined as increased utilization or productivity of land (Netting, 1993); therefore, either output-oriented (production) or input-oriented (utilization) measures can 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 2.5). Increased yields have led to higher *per-capita* production, reduced hunger, and improved nutrition (Tilman *et al.*, 2002; Bank, 2007). They have been made possible by the adoption of highly productive breeds and by mechanization which increased work efficiency. They have also been accompanied by an increased input-intensity: greater 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).

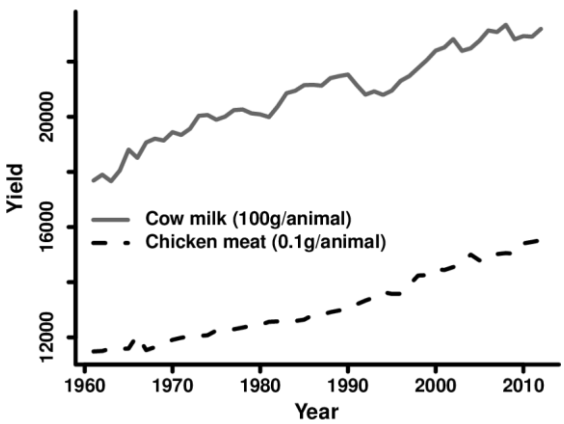

Figure 2.5. Evolution of the global cow milk and chicken meat yield (Data: FAOSTAT 2013).

The practices and 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 ameliorated to some extent in grasslands by using grazing animals to crop fast-growing grasses, increasing ground-level light availability for other plant species (Borer *et al.*, 2014). Decrease of plant and arthropod abundance cause, in turn, a decline on species at higher trophic levels (*e.g.,* birds) which depend on arthropod and plants for food resources and habitat (Fuller & Gregory, 1995; Robinson & Sutherland, 2002). The large scale monitoring of bird populations strongly evidences this decline as it shows 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 the result of their selective grazing. Grazing impact varies according to livestock species and interacts with sward productivity and stocking rate (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 rate 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 2.3). Increasing grazing stocking rate in improved grassland also causes 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 2.2.1.1). We define intensity as relating to practices and input use within the same land use, without involving land use transformation. Unlike habitat destruction, the intensity pressure is reversible in a short time scale, 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 (Section **Error! Reference source not found.**), limited intensity increase in grazing systems such as grassland fertilization can lead to biodiversity damages (Vickery *et al*., 2001; Kleijn *et al.*, 2009). At the other extreme of the gradient, intensive landless production systems (*e.g.,* feedlots in the US, 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 more important in developed countries. In these countries, very few or no more pristine habitats remains and their conversion into agricultural land uses rarely occurs. Intensity has been a predominant pressure category compared to 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 quickly developing countries (*e.g.*, China, Brazil). Intensity may thus become an increasingly important pressure on biodiversity in these 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 that represents a pressure for 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 yield in already cultivated areas, sustainable intensification could reduce the pressure to increase production by converting new land to agriculture, and spare land for nature (Borlaug, 2007; Phalan et al. 2011). Whether the increase in yield does spare land is subject to debate (Perfecto et al. 2009; Ewers *et al*., 2009; Rudel et al. 2009). Moreover, it has been argued that the concept of sustainable intensification often focused on yield and technical solutions and that this narrow definition lacked engagement with the key principles of sustainability (Loos et al., 2014).

PRESSURE 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 unit – *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. The HANPP has been computed for terrestrial ecosystems at a global scale (Haberl *et al.* 2006).
- Livestock density (expressed as animals, live weight or livestock units *per* unit area) have 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 grassland 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, 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 measure 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 pedo-climatic 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 is quite rare, however, more datasets exist to describe output-oriented measures of intensity. FAO and other sources provide yield data at global (FAOSTAT, 2013; LUGE, 2013). Models have also been developed to estimate the 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 with a high resolution and it can be used to compute livestock densities (
Figure 2.6). As for habitat degradation, the land cover classes of global data do not make it possible to differentiate between several classes of intensity. Although data on inputs are rare, certain countries conduct agricultural census which include variable that can be used to compute input-oriented intensity measures. In Europe, it is the case of the Farm Accountancy Data Network (FADN), a network of farms that provide statistically representative data at infra national (NUTS 2) scale. The FADN has already been used to compute input-oriented intensity indicators in various European countries (Reidsma *et al.*, 2006; Teillard *et al.*, 2012).

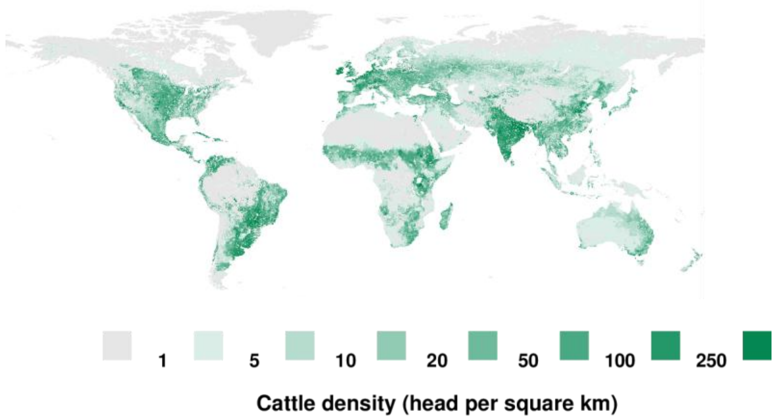

Figure 2.6: Cattle density across the globe (Data: GLW 2007, see also Robinson *et al.*, 2014)

2.2.1.6 Pressure: Landscape structure

CONTEXT –Important landscape homogenization has been associated with intensification, for increased efficiency of labor 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 a 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 *confgurational heterogeneity*. Among all land uses, semi-natural habitats (*e.g.*, grassland or grassy strips, tree or shrub edges, tree clumps) have a particular importance because they offer crucial resources to wild species. The high proportion of such semi-natural habitats compared to agricultural land uses, and their complex spatial arrangement in order to favor 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 are also important (Donald & Evans, 2006).

The negative correlation of landscape homogenization and biodiversity has been widely evidenced (Benton *et al.*, 2003). Both the composition and the structure 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, most studies have been conducted in East Africa savannas and in 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 to preserve 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) that aim to favor biodiversity in farmlands through measures enhancing 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. Striving for high landscape heterogeneity is not relevant in every context. Certain grassland or rangeland environments are historically homogeneous and heterogeneity is associated with disturbance. This disturbance can be associated with elevated biodiversity coming from new species which are not necessarily desirable (*e.g.*, invasive species, non-native species replacing endemic species, generalists replacing specialists). Conservation measures promoting heterogeneity in these historically homogeneous landscapes can even be associated with biodiversity losses (Batary *et al.*, 2007; Batáry *et al.*, 2011).

PRESSURE 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). Conversely, the proportion of arable land has also been used (Ekroos *et al.*, 2010; Filippi-Codaccioni *et al.*, 2010). In a few studies, multiple land uses (agricultural and semi natural) were 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 that addressed 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 even 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). Authors showed that their index correlated well with the occurrence of a bird of prey and with amphibian species richness in Japan. It also correlated to the pattern of traditional and high conservation value agricultural landscapes in other countries (*e.g.*, Iberian peninsula and Central America).

2.2.1.7 Benefit: Landscape connectivity

CONTEXT – Landscape heterogeneity is a key factor to maintain biodiversity in agricultural landscapes (Benton et al., 2003) and it could 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, 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 amongst 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 abilities 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 consist of a higher diversity of land uses which favor these species.

Landscape connectivity not only depends upon a favorable landscape structure, but also on the conservation of certain ecosystem functions associated with livestock mobility, fundamentally dispersal and landscape structures associated with mobility, *i.e.* drove roads. Dispersal is necessary for sessile organisms (plants) to respond to environmental changes which becomes more critical in the face of the present human-induced climate changes (Travis et al. 2013), Additionally, fragmentation and resulting inbreeding can drive to reproductive depression, further deterioration in dispersal capacity, and local extinction (Mix et al., 2006). Long-distance dispersal may be particularly affected and it is a key component of the issue (Ouborg et al., 2006). Mobile livestock has been identified as a key vector in long-distance dispersal, both for endozoochory (Manzano et al., 2005) and for epizoochory (Manzano & Malo, 2006). Drove roads are a general feature of landscapes where mobile livestock is 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 conservation of birds (Lentini et al., 2011) and arthropods (Azcárate et al., 2013b). Their structure is fractal, network-like (Manzano Baena & Casas, 2010; Lentini et al 2013), therefore fulfilling an important connectivity role that is, however, linked to sustained use.

European AESs promote extensification and habitat maintenance (Section 2.3.1.1), they also contain measures specifically targeting the enhancement of landscape heterogeneity (Table 2.5). Although the current effectiveness of AESs is still debated (Kleijn et al., 2001; 2006; Princé et al., 2012), they show that the EU recognizes that farmers can act to promote landscape heterogeneity, which can provides key benefits to biodiversity. Studies in Australia have found remnant vegetation and/or re-vegetated 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. This 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 by 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. The objectives included improvements to water quality, resilience to climate variability and change, as well as biodiversity conservation (Worboys & Mackey, 2013).

SCOPE – See Section 2.2.1.6. In addition, Couvreur et al., (2005) show that livestock has the capacity to disperse most of the plant community in a grassland, showing how important mobile livestock can be for keeping dispersal mechanisms that guarantee connectivity. Mobile livestock also adds the important role of maintaining drove roads. The degree of ecosystem functionality maintained by livestock will be given by the scale of movements done by the herd.

INDICATORS

- Indicators describing the composition and configuration components of landscape heterogeneity are provided in, see Section 2.2.1.6.
- At the landscape scale, an indicator can be the existence/enhancement of wildlife/biodiversity corridors linking fragments in the landscape. Livestock farmers in Australia are participating in Biodiversity Corridor schemes (2014). In Eastern Africa, there is concerted efforts to set up conservancies or wildlife management areas connected parks and dry season range for wild herbivore (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 – See Section 2.2.1.6

2.2.2 Pollution

2.2.2.1 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 as 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 amount 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, the decrease in species richness and abundance of these taxa influence other species groups such as birds that needs them as habitat and food resources (Vickery *et al.*, 2001). The negative effects of nutrient pollution are critical on aquatic biodiversity. There is a flow of nutrient from land based livestock production to lakes, rivers and coastal waters. The excess of nutrient loads in water results in an increased growth of nuisance species of algae and aquatic weeds. Those species compete and are harmful to other native species of algae. Their senescence and decomposition cause hypoxia (oxygen shortage) and they can release toxins, which is detrimental to various aquatic organisms (Carpenter *et al.*, 1998). Eutrophication has thus been shown to be a factor in the loss of aquatic biodiversity, even 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 according to the grazing management, slope, geology and soil cover (O'reagain et al. 2005). In intensive systems, animal 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.

For the habitat change driver, 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.

PRESSURE INDICATORS

- A simple indicator that has been 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 occurring as a result of this activity and does not account for other sources of nutrient inputs such as those from 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), from the Streamlining European Biodiversity Indicators.
- "Nutrients in transitional, coastal and marine water" is another indicator of the EEA (2007). It measures directly nutrient pollution through nutrient concentration; however, this pollution can originate from various anthropic 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 model Gerber *et al.* 2013).

2.2.2.2 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 2013). 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 its well-known mutual beneficial role. While fallow lands provide livestock with a dry-season grazing land, 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 practices (see section 3.3.2.3.).

SCOPE – Haynes & Williams (1993) estimate that about 85% 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 that can be decisively assisted by arthropod burial (Slade *et al.*, 2007). Manure fertilization is known to be more effective in keeping the 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 fertilizer that has been spared or by simple calculations of the excreta production of livestock.

DATA – The benefits on 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 the agricultural intensification projects. However, simple extrapolations can be made based on the number of livestock.

2.2.2.3 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.

Subsequently to these emissions, atmospheric N deposition is a very important driver of species change (Sala, 2000). N deposition fertilizes soil which changes the community equilibrium in favor of species adapted to more fertile soil, which result in a net loss of species (McClean *et al.*, 2011). N deposition can lead to eutrophication, or acidification in the presence of water and via 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 are more important as fertilization or manure production increase.

PRESSURE 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 nitrogen deposition is beyond a critical load leading to harmful effects to biodiversity. As for diffuse nutrient pollution in water, deposition of atmospheric N does not only originate 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 3.3).

DATA AVAILABILITY – As a greenhouse gas, N_2O emissions are modeled in GHG LCAs, such as in GLEAM model (Gerber *et al.*, 2013). However, ammonia (NH_3) emissions are not modeled 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.

2.2.2.4 Pressure: Pesticides and other products

CONTEXT – Livestock production is responsible for pollution by products than nutrients, which have a direct toxicity on organisms. 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, increases in pesticide use was responsible for an important decline of bird populations (Carson, 1962), especially before the ban of the most eco-toxic molecules. Such molecules can still be used in certain developing countries. More generally, hormonally active pesticides cause 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 is the dramatic decline of 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 fishes (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. The ecotoxicity effect does not only depend on the amount of product: less intensive systems in developing countries sometimes use 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, availability of existing indicators is low 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 3.3.4).

2.2.3 Climate change

2.2.3.1 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. These estimations were based on a life cycle assessment and they consider 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.* 2013 report), the pressure on biodiversity is will be accentuated. 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 37% 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 big threat to coral reef systems (Hoegh-Guldberg *et al.*, 2007). Regarding the effect on vegetation, the effects of rising atmospheric CO_2 concentrations may temporarily increase plant photosynthetic activity and net ecosystem productivity (NEP) although major uncertainties remains 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 contribute 45 and 39% of the livestock sector's emissions, respectively (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 globe. They are stronger at higher latitude which put the species and ecosystems 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 sectors 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 extrement events) can also be computed. As for other categories of pollution, it is not possible to isolate the livestock sector from other anthropic activities.

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) provide global data on temperature (NASA, 2014a) and rainfall (NASA, 2014b). For the US, there is GHG dairy industry LCA database available at National Agriculture Library from the US Department of Agriculture (USDA, 2014).

2.2.3.2 Benefit: GHG sequestration

CONTEXT –

a) Different types of livestock management have the capacity to improve or reduce the capacity of soils to retain carbon in grassland. Improved management has therefore been described as a powerful tool to increase the carbon storage capacity 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) Livestock has been attributed a high proportion of GHG emissions, ranging up to 14.5% of the total global emissions (Gerber et al., 2013). However, many of these emissions come from high-fiber-diet, extensive systems and could be considered natural, as they could potentially be equally released by wild animals if livestock were withdrawn. This “replacement criterion” has been a point of criticism of 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 have a net contribution to GHG emission equal to zero. Conversely, evidence shows that extensive systems can produce livestock using a negligible amount of fossil fuels (Casas Nogales & Manzano Baena 2010). Extensive systems could therefore have the capacity of mitigating carbon emissions by producing 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 on productivity and livelihoods of local inhabitants.

b) In the case of extensive system 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 –

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 issue here with the scale of measurement. See Data paragraph in Sections 2.3.2.3. and 2.3.2.4.
b) Life Cycle Analyses have provided substantial information on the use of fossil fuels for livestock production. From these databases, details on fossil fuel production can be easily obtained.

2.2.4 Other drivers

2.2.4.1 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) favored 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 be more concerned by 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 disturbances and degradations of natural and semi-natural systems increase 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 the 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). 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 positive feedback loop between ecosystem degradation and alien invasive species, indicators of other categories of pressure can inform about the vulnerability to invasion.

2.2.4.2 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.

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, varied ways. Moderate grazing levels can minimize soil disturbance and effects on the plant community, preventing establishment or controlling spread of the invasive. If the invasive species is and edible plan chosen by the animals, intensive grazing can encourage use of the invasive species and subsequently 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 – See above for management practices and small-scale measurement.

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

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

2.2.4.4 Pressure: Overexploitation of wild populations

CONTEXT – Harvest and trade of species (*e.g.*, for food, medicine, fuel, material use) is fundamental to economy and culture at a global scale; however, this exploitation can be pushed beyond sustainable levels. Overexploitation (not only related to livestock) is a major biodiversity threat which affects 33% of the threatened species of mammals, 30% of the threatened species of birds and 6% of the threatened species of amphibians (Baillie, 2004). The main contribution of livestock production to overexploitation is through overfishing for fish meals. According to the Garcia & de Leiva Moreno (2005), 52% of fish stocks are fully exploited, 17% overexploited and 7% 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). A majority of these fishmeals are used in aquaculture but a significant share is still used for livestock production (62% and 38%, 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 use 22% and 8% 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 the 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). The “European commercial fish stocks” is a pressure indicator of the EEA (2007) that describes the proportion of commercial fish stocks within safe biological limits.

2.2.4.5 Pressure: Competition with large mammals

CONTEXT – Competition between livestock production and wild mammals can occur in two ways: from direct interactions, or indirectly through resources (Steinfeld et al. 2006). In some regions, livestock losses from predation can be significant and long-term studies have indicated importance of the prey base (Packer et al. 2005). In Kenya, these losses reach up 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 surrounding 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 large scale data are collected to monitor it. Indirect competition for resources is complicated to quantify and its effect often confound with other pressure 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 2.2.1) 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.

2.2.4.6 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 benefitted by livestock (Ghotge & Ramdas, 2010). 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 overseen in several organisms (e.g. for ants Manzano et al., 2010).

SCOPE – Grassland systems.

INDICATORS – Manly biodiversity state indicators (Section 2.4), such as the abundance of endangered species, with special attention to the ones that are directly benefited by the action of livestock, can indicate whether the facilitation of food webs is happening. Management type is also important, as mobility has been shown to incorporate an important sustainability factor (Olea & Mateo-Tomás, 2013; Section 2.3.1.1.)

DATA – Similar sources to the endangered vegetation data.

2.2.4.7 Pressure: Disease emergence

CONTEXT – In the past, the introduction of disease by livestock had severe consequences for naive population of wild species. For instance, the rinderpest virus brought with cattle to Africa killed many individuals of the continent's wild species of ruminants (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 the 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). 50 species of wild birds have been infected by the H5N1 virus and it is estimated that 84% 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. Several factors favor disease emergence in intensive systems. The high productivity comes with trade-offs and animals are more sensitive to disease. The important population turnover involve that populations are mostly naive. Finally, the strong concentration of animals favors 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.*, South-East Asia) where intensive livestock farming is more recent, disease control not fully established and cohabitation with backyard system and wild species more prevalent (Coker et al., 2011).

INDICATORS AND DATA AVAILABILITY – Jones *et al.* (2008) mapped emerging infectious disease events, originating from 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 factor that are independent from livestock (environmental, climatic, socio-economic, Jones *et al.* 2008). Pressure indicators could also reflect the factors favoring 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).

2.2.4.8 Benefit: Disease control

CONTEXT – Livestock-transmitted diseases are a threat to native biodiversity and vice-versa, 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 sanitary situation in wild animals and reduce the threat of extinction.

SCOPE – The recent eradication of rinderpest, which constituted a threat for 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 risk of diseases shared by livestock and wildlife. These practices are often related to the animal density maintained in farms but also to the cultural practices and innovative tools to cut 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 – Large scale database are lacking on good practecies for disease control that happen mainly at small scale. However, the World Organization 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).

2.3 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 encompass multiple levels of organization. The three fundamental levels – genes, species and ecosystems – are detailed hereafter. 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 in the same 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 (**Error! Reference source not found.**). 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 has 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.

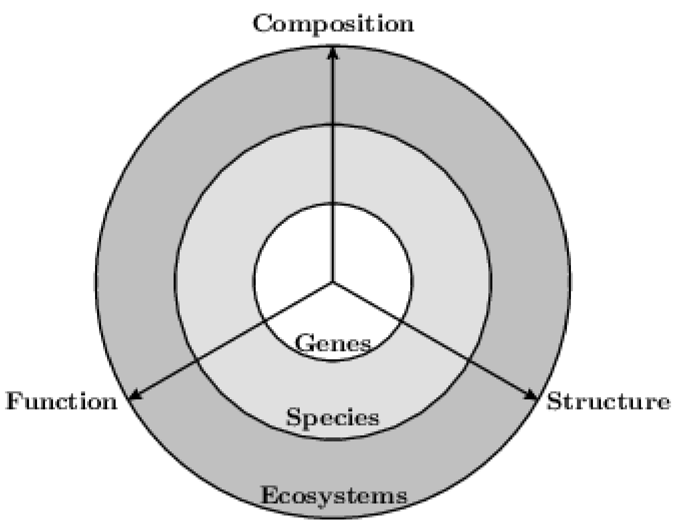


Figure 2.7: The three levels and three dimensions of biodiversity (adapted from Noss 1999).

2.3.1 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% 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 that makes it possible for 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 habitat. Coral reefs and Mediterranean climate habitats 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% of all mammal species and 12% of all bird species were threatened of extinction. Although livestock production has positive effects on biodiversity (Section 2.3), it contributes to the main global drivers of biodiversity loss (Section 2.2), and are partly responsible to the destruction of tropical forest habitats hosting rich biodiversity.

2.3.1.1 Composition dimension

Indicators at the species level and composition dimension reflect the identity and variety of species. They can be studied on two axes: the species richness (number of species) and the species abundance (number of individuals from several species or one particular species) (**Error! Reference source not found.**). Measures of the 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:

$$Shannon = -\sum_{i=1}^{R} p_i \ln p_i \qquad (2.1)$$

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 an important number of different species with individuals in similar abundances.

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 also are easy to identify and monitor (*e.g.*, birds; because of their high position in trophic chains, they 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 modifies 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 concerns because they face a higher risk of extinction. The IUCN publishes a red list of globally endangered species. Finally, flagship or patrimonial species are species with a 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 would allow one to track whether a given livestock production system 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.

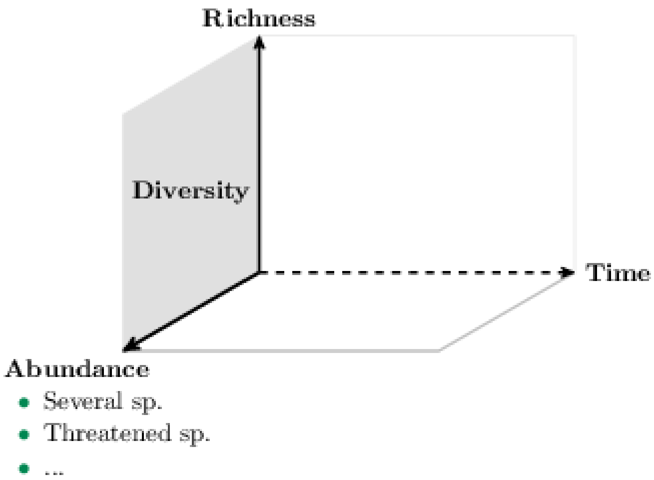


Figure 2.8: Species composition can be described on two axes: richness (number of species) and abundance (of individuals in several or particular species). Diversity measures combine these two axes. State indicators of the species richness, abundance or diversity can be computed in a static framework or over time.

2.3.1.2 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 spatial patterns of the landscape. In certain landscape, the species habitat is not continuous but split into patches. When the species population is also divided between such patches of habitat, it is called a metapopulation (MacArthur, 1967; Levins, 1969). Along the classic population dynamics occurring locally within each patch (survival, reproduction) there is a spatial, metapopulation dynamics of migration, colonization and local extinctions 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, 2000). Conversely, fragmentation occurs when patches are too small and isolated. Fragmentation and the landscape structure can be described as a pressure indicator (see Section 2.2.1) 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 patch occupied, the ratio between local colonization and local extinctions. 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).

2.3.1.3 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 the 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), they 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 difference 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. 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.

2.3.2 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 are relatively little widespread but host a rich biodiversity. Aquatic ecosystems also occurs in inland waters and wetlands that have been 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.

Like at the species level, ecosystem diversity is under threat. Aquatic ecosystems face direct impacts such as over-fishing and damages from fishery operations. They also face indirect stresses from climate change and pollution. Climate change and pollution may lead to the loss of 60% of the coral reefs by 2030 (Hughes *et al.*, 2003). Although it is a terrestrial activity, livestock production is involved in these pressures affecting aquatic ecosystems (over-fishing, pollution, climate change). On land, livestock production contributes to the habitat change driver, which directly destroys or alters terrestrial ecosystem (*e.g.*, deforestation, dryland degradation, agroecosystem intensification, Section 2.2.1). But livestock is also embedded in certain agroecosystem and has a key role in maintaining them (*e.g.*, grassland, rangeland, sylvopastoral systems, Section 2.3)

2.3.2.1 Compositional dimension

As for species, the compositional dimension of ecosystem can be studied through richness, abundance and diversity (**Error! Reference source not found.**). Maintaining a richness of ecosystem 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 ecosystem 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, the annual net change in forest area can be computed. In Latin America and Caribbean it amounted around 4% between 1990 and 2005 (CBD, 2006). The direct or indirect responsibility of livestock production in this change of the state of forests ecosystems has also been computed (Section 2.2.1). 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 (Section 2.2.1) could be a state indicator at the ecosystem level and landscape structure (Section 2.2.1) could be a state indicator at the landscape level.

2.3.2.2 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 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 (*e.g.*, of the parent material, erosion, physical and chemical processes) and biotic (*e.g.*, organic carbon, soil community of mycorrhizal fungi, micro and macro arthropods) factors (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).

2.3.2.3 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.* or indirect benefits of ecosystems for human (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 user (*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 pressures on biodiversity have led to ecosystem degradation (Vitousek *et al.*, 1997). There are many evidences that stressed and degraded ecosystems have become incapable of supplying services to the same level than in the past (Noble & Dirzo, 1997; Rapport *et al.*, 1998). The pressures that livestock have put on ecosystem have resulted in ecosystem degradation and diminution of the services provided by the ecosystems (Harrison *et al.*, 2010). Conversely, well managed grassland production systems can maintain healthy ecosystems that provide a wide range of services, from carbon sequestration to water cycling and quality, or biodiversity conservation (Havstad *et al.*, 2007). Measuring ecosystem services could be a key way to quantify the benefits of livestock on biodiversity.

Several approaches exist to quantify function at the ecosystem level. Historically, functional ecology described ecosystems and their functioning by quantifying fluxes of nutrient, 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.*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).

2.3.3 State indicators at the gene level

Genetic diversity is the richness of gene variations within a species. It is a key mechanism to allow 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 out of 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 7600 breeds of livestock have been developed. Livestock diversity plays a key role for both short- and long-term food production in diverse environment, food security, nutrition and cultural identity. It is under threat as intensive production using a limited number of breeds spreads while traditional production systems and the associated local breeds tend to be marginalized and disappear (Rischkowsky & Pilling, 2007). Out of the 7600 reported livestock breeds, around 20% are classified as at risk of extinction. The vulnerability of domestic biodiversity also concerns crops. Although the number of accession conserved *ex-situ* worldwide has significantly increased during the past decade, many countries report genetic erosion of crops (FAO, 2010a).

2.3.4 State indicators at regional to global scale

The diversity of state indicators could 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 2.3) 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.

Table 2.3. 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

2.3.4.1 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:

$$NDVI = \frac{NIR - RED}{NIR + RED} \qquad (2.2)$$

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 for an indicator of the habitat destruction and degradation pressures. The 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. The NDVI has also been used to study the extent of land degradation in various ecosystems (*e.g.* ecosystems 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 to study 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, the 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 2.4).

Table 2.4: 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

A limitation of the 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. 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 pattern can require elaborated algorithm procedures (Alcaraz *et al.*, 2006; García *et al.*, 2008) or complementary with field measures (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).

2.3.4.2 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 4000 population time series ranging from 1970 to 2003, and covering 1411 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.

2.3.4.3 The IUCN Red Lists and Red List Indices

THE RED LIST OF SPECIES – The IUCN published – 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 time period for various taxa, regions and ecosystem (Butchart *et al.*, 2004; 2005). As for the Living Planet Index (Section 2.4.4), 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 the species composition (see also Section 2.4.1) 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 contributes 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-wellbeing. More information on the RLE and its applications are provided in Section 7.4 of the Appendix.

2.3.4.4 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 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 monitoring scheme generates national population trend indices for 135 bird species (Gregory *et al.*, 2009). Within each country, these trends are computed from surveys at local sample points. In France for example, bird populations have been monitored in around 2000 2*2km 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 makes it possible to conduct statistical analysis for isolating the effect of agriculture from other factors (Doxa et al. 2010; Doxa et al. 2012; Teillard et al. 2014).

2.4 Response indicators

Response indicators describe the decisions and actions that can be undertaken by the stakeholders to mitigate impacts and improve the state of biodiversity. Stakeholders can be as various as policy makers, the private sector or farmers. 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 or biodiversity levels and dimensions. There is a wide variety of response indicators that reflects the variety of possible stakeholders and actions. The following Sections (2.4.1 and 2.4.2) give a few examples of actions targeting biodiversity improvements that could use to develop response indicators.

2.4.1 Actions from the public sector

In the EU, the Common Agricultural Policy (CAP) is a powerful set of 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 provide payments to farmers, based on voluntary compliance, for adoption of management practices that reduce adverse environmental impacts, on biodiversity in particular. These practices target the reduction in different categories of pressures and

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

the strengthening of the benefits that farming can bring to biodiversity (examples of AESs are given in Table 2.5). In the 2013 recent CAP reform, on proposition was the ecological cross compliance, i.e. that subsidies for production would be conditioned by ecological criteria. One of this criteria was for the farm to include at least 5% of ecological focus areas (semi-natural habitats, including grasslands).

Equivalents of the AES 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 program 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 practices 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 certifications incorporating biodiversity aspects, such as the US national organic program, the EU organic farming or the Chinese OFCD organic product certification standard. In organic farming for example, maintaining and enhancing biodiversity is part of the principles. Rules include multi-annual crop rotations and limitation of the livestock density which correspond to mitigation of biodiversity pressures.

In Eastern and Southern Africa, communities are designating conservation areas and are managing both the numbers of wildlife and livestock (Osano, 2013).

Table 2.5: 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 asides 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 the 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 the soil organic matter
Landscape structure	Creating and maintaining trees edges, clumps or isolated trees Creating and maintaining ponds or other water points Creation and maintaining grassy strips Diversifying crop rotations

2.4.2 Actions from the private sector

As environmental and social impacts of farming became an important concern for the 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 include at the ecosystem level:

- *Critical Criterion. All existing natural ecosystems, both aquatic and terrestrial, must be identified, protected and restored through a conservation program. The program 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. (...)*

2.5 Strengths and limitations of the indicator framework

2.5.1 Strengths

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

Table 2.6 compares pressure, state and response indicators on 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.

Table 2.6: Comparison of the pressure, state and response indicators on three criteria.

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

State indicators are those that are the most closely related to biodiversity itself. They provide a direct measure of biodiversity, which is not comprehensive (*e.g.*, 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 exists 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 factors (possibly interacting), can be challenging. Multivariate statistical analyses can be a way to separate 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. This data collection effort can be time-costly, 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 **Error! Reference source not found.**. They do not provide a straight biodiversity measure but they 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 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 relationships between pressures and management decisions also involves 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 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 2.5). 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 at various level: farm, company, sector, country. Although addressing all the pressures, such analysis would not consider the relative importance of the different pressures in influencing the state of biodiversity. For the most thorough analysis of the livestock impact and performances, state indicators need to be computed.

The Sustainability Assessment for Food and Agriculture (SAFA, 2013, see also Section 3.1) 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 to use practice-based indicators. For computing the total sustainability score, practice-based indicators have a lower weight (they bring less “points”) than performance-based indicators.

2.5.2 Limitations

One important limitation of the PSR indicator framework is that indicators were almost always computed for a single process of the supply chain and for a bounded area. Most pressure, state and response indicators were computed at the level of the farm (*e.g.*, % 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 represent an important 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, 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%. Because the imported soybean cakes could originate from deforested areas that was previously a biodiversity hotspot, their relative impact on biodiversity could represent even more than 30% 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 off the geographic origin of the feed that they buy. The difficulty is even greater in the case of compound feeds that are blended from different raw materials that can have various origins.

Other limitations exist when restricting the computation of biodiversity indicators to the farm or another bounded area. 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 in the farm because the mechanisms of the 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 neighbor farms, structure of the surrounding landscape, global climate change

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

3 The Life Cycle Assessment framework

3.1 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 quantifying the burdens from cradle to grave treatment[3] (ISO, 2006a) and therefore reveals the overall environmental impacts and the relative contributions of the different stages of the supply chain. LCA studies also offer a way to identify options to reduce impacts and shed light on how those may affect other parts of the system (Garnett, 2009).

LCA studies comprise four steps according to ISO (ISO, 2006a): the 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 3.2). 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 3.2). 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 comparables. 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 are examples of midpoint modeling while nowadays there is an increasing acceptance that results from inventory results should interpret into their 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 modeling is to aid in understanding and

[3] The term "cradle-to-grave" refers to the assessment of impacts from raw-materials extraction to end-of-life treatments, such as recycling or landfilling.

interpreting midpoints by computing endpoint categories corresponding to areas of protection that form the basis of decisions in policy and sustainable development.

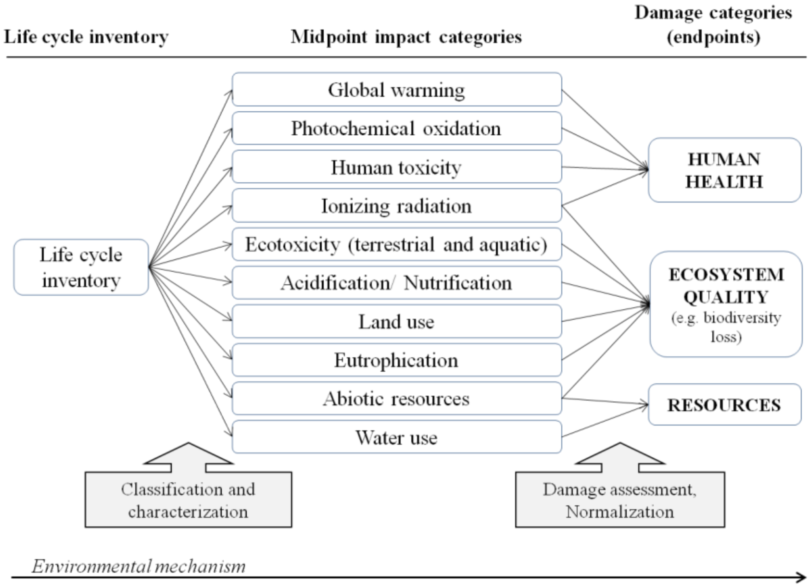


Figure 3.1: Schematic representation of the environmental mechanism (cause-effect chain) pathway, from life cycle inventory (e.g. land occupation and transformation) to midpoint (e.g. habitat destruction) and endpoints (e.g. species loss or functional loss).

3.1.1 LCAs of livestock production

LCA studies of livestock production systems mainly include impacts arising from feed production and associated input use, from the animal husbandry itself and from downstream transport and processes until retail (Figure 3.2). 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, acidification and eutrophication. Little attention is paid to quantify impacts on biodiversity. Impacts of land use are mainly addressed as a result of land surface occupied, but no loss of biodiversity was assessed. Other LCA studies include biodiversity impacts relying on local biodiversity assessments, *i.e.* without taking into account the off-farm impact of feed that is imported into a farm (*e.g.*, Haas *et al*., 2001; Nemecek *et al*., 2011a;b; Jeanneret *et al.*, 2014). Jeanneret *et al.* (2014) developed an expert system for including biodiversity as a 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 group to management options was developed. These methods cannot be extrapolated at regional or global scales. 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 a*l., 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 make 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 a*l. (2014) found that the specific impact of different land use types was more important that the sole impact of the total area occupied. For margarine (independant from livestock), Milà i Canals et al. (2013) found that the impacts of land use dominated the impacts associated with processing. These findings justify that most LCA studies addressing the impact of food products on biodiversity focused on land use.

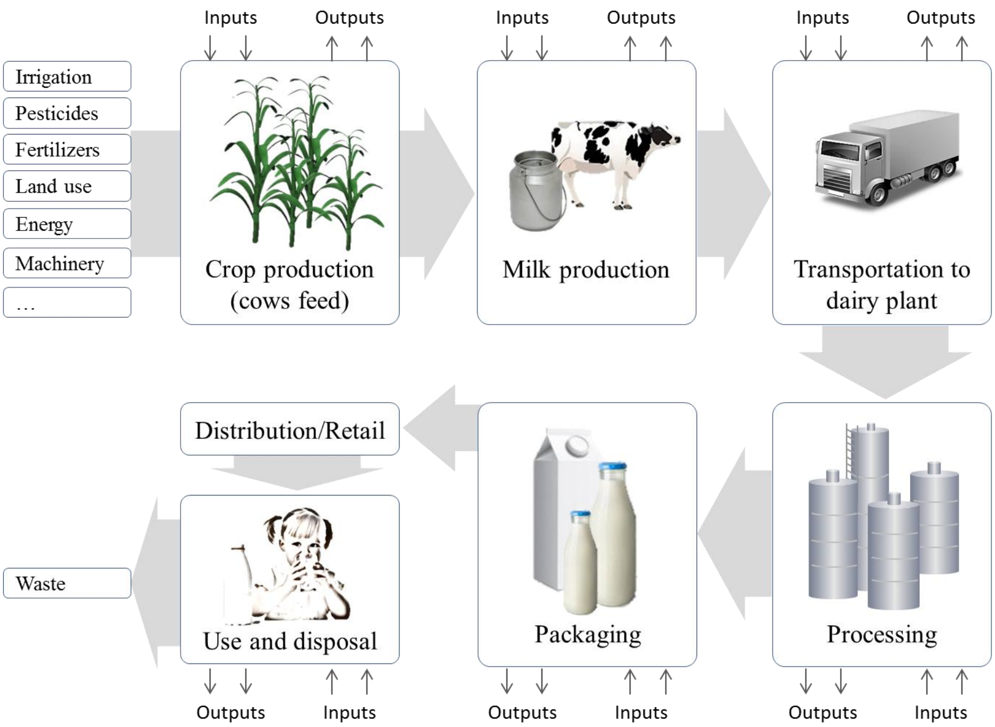


Figure 3.2: Example of summarized life cycle stages of milk production, from feed production and associated input use to end use and disposal.

3.1.2 LCA methodologies addressing biodiversity endpoints

This section gives an overview of some of the existing Life Cycle Impact Assessment (LCIA) 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 3.2 (for biodiversity impact through land use) and 3.3 (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 modeling principles. Biodiversity loss is one of the endpoint indicators that are 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 most of the existing methods.

The Mean Species Abundance (MSA) is a biodiversity indicator reflecting the mean abundance ofcurrent species relative to their abundance in undisturbed ecosystems (Alkemade *et al.*, 2009). The MSA was developed in the context of the GLOBIO3 model that aims at assessing 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; 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 factorx 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 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[4] is undertaking an effort to drive global consensus on characterization factors and impact indicators for biodiversity in the context of LCA (Jolliet *et al.* 2014).

3.2 The assessment of land use impacts on biodiversity in LCA

Livestock is a major user of land resources (Sections 1.1, 2.2.1), which makes land use one of the main drivers of the livestock impact on biodiversity. Computing this impact in LCA while considering

[4] http://www.lifecycleinitiative.org/

specificities of the livestock-biodiversity relationships (*e.g.*, positive impacts, Section 2.3) 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 secion). A few reviews have also been carried out in 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 modeling. However, no consensus exists on which methodology should be applied for current LCA studies. This is mainly true for several reasons. First of all, biodiversity is a complex entity with multiple aspects that cannot be fully captured or represented by one single indicator. Second, some assumptions of the land use model represent a linearization of dynamic processes in nature and lead to an oversimplification of the model (Souza *et al.* 2014, in press). Finally, LCA studies require the availability of global characterization factors, which can require large amounts of data if models are to be turns 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.

3.2.1 Conceptual framework

3.2.1.1 General situation

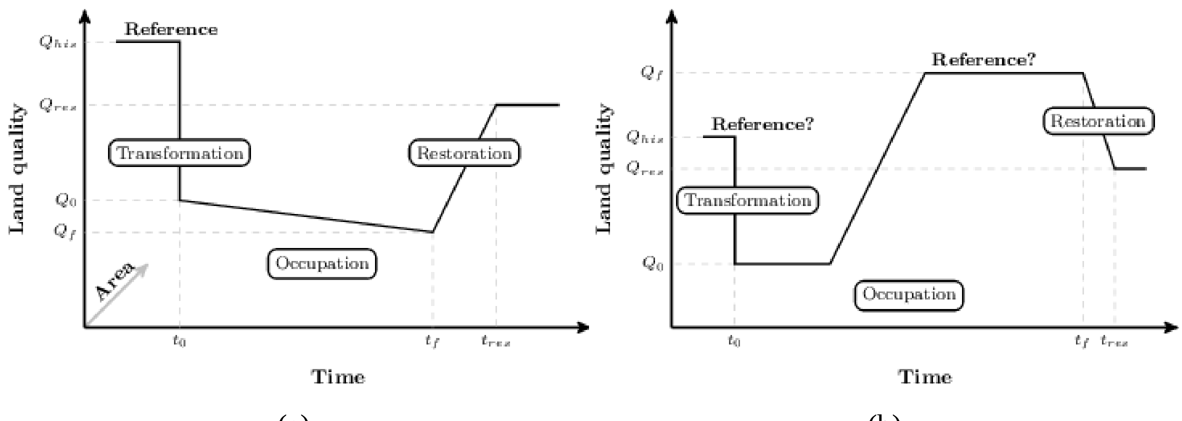


Figure 3.2: Scheme of the conceptual framework for impact assessment in LCA, depicting two land interventions (occupation and transformation) and land recovery. (a) describes a situation where natural land is transformed and used. (b) describes a situation that could apply to the livestock context: occupation lasts for a very long time and biodiversity adapts to it. Adapted from Lindeijer (2000); Milà i Canals *et al.* (2007). *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).

Milà i Canals *et al.* (2007) recognize two land use elementary flows (interventions) which consist of pressures driving habitat and biodiversity changes: land transformation and land occupation (Fig. 3.2). 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. Fig. 3.2a 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 framework are not able to take these alternative evolutions into account; they assume that land quality remain 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, driven by natural ecological succession, or active restoration, in case of human intervention, processes 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 Fig. 3.2a.

3.2.1.2 Specific situations in the context of livestock production

The example depicted in Fig. 3.2a represents one possible situation. It can illustrate the case where a tropical forest is converted (land transformation) to pastures, and used for livestock production (land occupation). A contrasting situation can exist in the context of livestock production. Many semi-natural grasslands in Europe (and elsewhere) illustrate such a situation, idealized in Fig. 3.2b. 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 has allowed time for a unique biodiversity to co-evolve with grazing (Bignal & McCracken, 1996; Poschlod & WallisDeVries, 2002; Section 1.1, 2.3). Today, when livestock farming is abandoned in these semi-natural grassland areas, the natural process of land recovery to original forest results in a loss of biodiversity (Verhulst *et al*., 2004; Sebastià *et al*., 2008; Section 2.3). In this case, determining the land use and land quality value to be used as reference is not straightforward. 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.

3.2.1.3 Using species-area relationships to compute characterization factors

A strong pattern in ecology is the Species-Area Relationship (SAR), *i.e.* 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 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 size, 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 to compute characterization factors as they fulfill 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 increase steeply at the beginning and gradually becomes flat. 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) or the logistic model (Archibald, 1949). The most widespread is the power model of Arrhenius (1921):

$$S = cA^z \qquad (3.2)$$

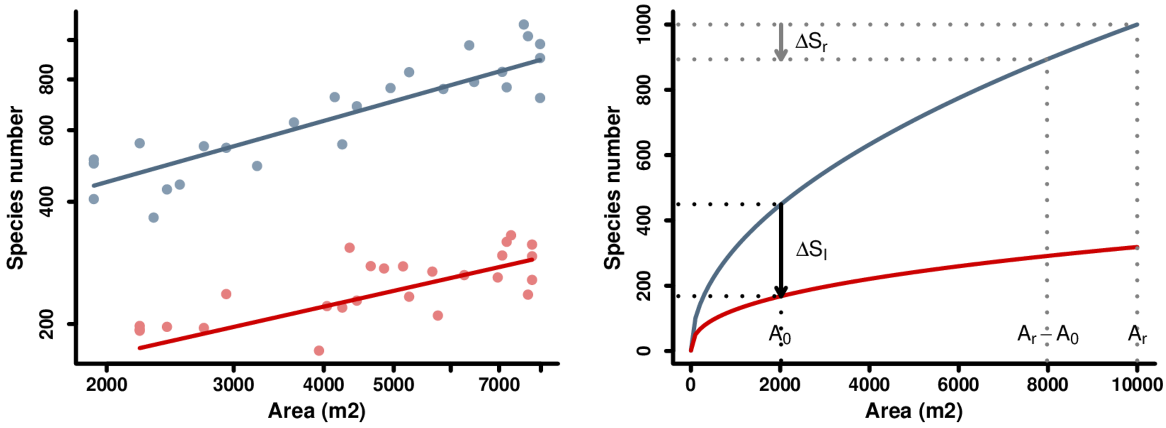
where *S* stands for the species richness and *A* for the area. The parameters *c* and *z* correspond, respectively, to a multiplier (number os species in a unit area) dependent on taxa, and to the slope of the increasing number of species in relation to area (species accumulation rate) (Rybicki & Hanski,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

$$\log \mathrm{S} = \log \mathrm{c} + \mathrm{z} \log \mathrm{A} \qquad (3.3)$$

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 the 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 uses. 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. 3.3) is then fitted to the sample of species richness and areas, *i.e.* parameters *c* and *z* are calibrated. Fig. 3.3 shows an example of these relationships fitted for a reference *ref* and an occupied land use type *occ*. When the species-area curves (Eq. 3.3) are known, it is possible to compute the characterization factor shown in Eq. 3.4 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 Fig. 3.3b).

$$CF = \frac{S_{ref}(A_0) - S_{occ}(A_0)}{S_{ref}(A_0)} = 1 - \frac{c_{occ}}{c_{ref}} A_0^{z_{occ}-z_r} \qquad (3.4)$$



(a) (b)

Figure 3.3: Species area relationship in a reference (blue) and occupied (red) land use. (a) species area relationships are calibrated using the log-log model (Eq. 3.2, 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 A_0 of the reference land use into the new one (ΔS_l), and from the loss of this area in the reference land use (ΔS_r).

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 modeling. 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.

3.2.2 Examples of land use impact assessment models

3.2.2.1 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. 3.3 in order to compute a standardized species number in $1m^2$ of each land use type S_{1m^2}:

$$S_{1m^2} = S_{plot} - \Delta S \qquad (3.5)$$

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:

$$\Delta S = c(A_{plot}^{z} - A_{1m^2}^{z}) \qquad (3.6)$$

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:

$$EDP_{linear} = 1 - \frac{S_{occ}}{S_r} \qquad (3.7)$$

or as a nonlinear function:

$$EDP_{nonlinear} = 1 - a(\log \frac{S_{occ}}{S_{region}}) + b \qquad (3.8)$$

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. 3.8 were quantified by Schlapfer *et al.* (1999), based on expert's 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 South-East Asia. The main source of uncertainty of 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 mollusk than for plant, which result in higher standard deviation around the mean EDP.

3.2.2.2 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.

The Eq. 3.4 is used to compute the characterization factor corresponding to this local damage CF_{loc} (corresponding to ΔS_l in Fig. 3.3b).

Besides, the regional damage describes the marginal change in species richness outside the occupied area, caused by area reduction (corresponding to ΔS_r in Fig. 3.3b). 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. 3.2:

$$\Delta S_r = A_0 \times c_{ref} \times z_{ref} \times A_r^{z_{ref}-1} \qquad (3.9)$$

where A_r is the area of the region (the other terms are detailed in Eq. 3.4).

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

$$CF_l = 1 - \frac{A_0 \times c_{ref} \times z_{ref} \times A_r^{z_{ref}-1}}{c_{ref} A_r^{z_{ref}}} = \frac{A_0 z_{ref}}{A_r} \qquad (3.10)$$

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

$$\begin{aligned} ED &= \left(1 - \frac{c_{occ}}{c_{ref}} A_0^{z_{occ}-z_{ref}}\right) \times t \times A_0 \\ &\quad + \left(\frac{A_0 z_{ref}}{A_r}\right) \times t \times A_r \\ &= \left(z_{ref} + 1 - \frac{c_{occ}}{c_{ref}} A_0^{z_{occ}-z_{ref}}\right) \times t \times A_0 \end{aligned} \qquad (3.11)$$

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.*, occupation and restoration (Fig. 3.2a). Restoration is considered under four different perspectives affecting restoration time and fragmentation (egalitarian, individualist and hierarchism).

3.2.2.3 The Mean Species Abundance

The 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 that 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:

$$R_{occ,k} = \begin{cases} \frac{n_{occ,k}}{n_{ref,k}}, & \text{if } n_{occ,k} < n_{ref,k} \\ 1, & \text{otherwise} \end{cases} \qquad (3.12)$$

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:

$$MSA_{occ} = \frac{\sum_k (R_{occ,k}/V_{occ,k})}{\sum_k 1/V_{occ,k}} \qquad (3.13)$$

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% 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.2 in Appendix). Intensity gradients are described within three main land use classes (forest, grassland and cultivated land). The MSA characterization factors for land use (Table A.2 in Appendix) are relevant at global scale.

The MSA characterization factors for land use (Table A.2 in Appendix) 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.

3.2.2.4 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 distinguisheed two main land use interventions: land transformation and land occupation (Section 3.2.1). 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 extensively described in the LEAP biodiversity principles (LEAP, in prep.). 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 5 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). Work is currently on-going with a focus on building global consensus to identify indicators capturing biodiversity impacts in LCA, with expected results in 2015[5].

3.3 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. It is the pressure that has been 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 puts on biodiversity at global scale (Section 2.2). The MSA and the ReCiPe methods (detailed for land use aspects in Section 3.1) also compute characterization factors linking other midpoint categories to biodiversity. In the following sections, we provide more details on how characterization factors are computed to link changes in biodiversity to climate change, pollution and ecotoxicity endpoint impacts.

[5] For regular updates, see www.lifecycleinitiative.org

3.3.1 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 are transformed into soluble components such as ammonium (NH_4), nitrate (NO_3) and nitrite (NO_2). With the action of rainfall and runoff, these soluble N components and various forms of P can be responsible for both soil acidification and aquatic eutrophication.

3.3.1.1 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:

$$CF_x = \sum_j \left(A_j \cdot \frac{dPNOF_j}{dM_x} \right) \quad (3.15)$$

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 factors 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.

3.3.1.2 Eutrophication

Similarly to acidification (Section **Error! Reference source not found.**) 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 $PDF \cdot m^3 \cdot day/kg\ P\ emission$. 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).

3.3.2 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 modeled a causal relationship between GHG emissions and global mean temperature increase. The final characterization factor links temperature change to biodiversity, and is expressed as $km^2 \cdot PDF \cdot °C^{-1}$:

$$CF_{CC} = \frac{APDF}{\Delta TEMP} \quad (3.16)$$

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 1084 species across five regions (Europe, Mexico, Australia, South Africa, Brazil) were integrated into a climate envelope modeling 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 damages (Figure 3.4). Characterization factors make it possible predict biodiversity damages directly from GHG emissions, and therefore to isolate the impact of livestock.

Figure 3.4: Framework used by Schryver *et al.* (2009) to link GHG emissions to ecosystem damage.

3.3.3 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 15415 liters of water were consumed along the supply chain leading to the production of 1kg of beef meat (including water used for feed production, drinking water and service water used for animal husbandry). This total include 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 liter of water consumed has not the same impact in a temperate ecosystem and in an arid ecosystem where plant growth can be limited by water availability, which influences vegetation diversity and the whole 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 damages 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 calculated at global scale and expressed as $PDF \cdot m^2 \cdot yr$ *per* m^3 of water consumed.

3.3.4 Ecotoxicity

Many ecotoxicological studies have sought to establish relationships between Potentially Affected Fraction (PAF) of model species 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}:

$$PDF_{tox} = 1 - \prod_k \left(1 - PAF_k^{EC50}\right) \quad (3.17)$$

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

The UNEP-SETAC 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).

3.4 Strengths and limitations of the LCA framework

3.4.1 Strengths

As highlighted in the Chapter 2 (Section 2.5.2), 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 to conduct environmental assessment. It is increasingly recognized by governments, private sector and NGOs to guide decision toward better environmental performances (Tillman, 2000; Rebitzer *et al.*, 2004). The use of LCAs is already widespread to assess 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 modeling steps with specific characterization factors (Fig. 3.1). Such consistency would allow comparability of the 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 program, with the intent to develop guidance on Environmental Life Cycle Impact Assessment Indicators based on a consistently applied set of selection criteria and rigorous analysis of different methods to assess biodiversity damage produced by land use. Different methods in and out the scope of LCA have been selected an evaluated according to criteria of completeness, scientific and environmental relevance as well as its applicability. Table 3.1 summarizes the main characteristics of some selected LCIA methods to assess effects of land use on biodiversity

3.4.2 Limitations

There is often a trade-off between the different performance criteria of the LCA approaches, for instance it is challenging to have both a large geographic scope and taxonomic coverage while using detailed land use classes. Table 3.1 summarizes some of these performance criteria for several LCA approaches focusing on biodiversity impacts through land use. In the next sub sections, we detail current limitations of the LCA approaches on these various performance criteria.

Table 3.1: 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-American, 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”	18 different, relevant land use types for occupation (including three types of intensity for arable areas) + four land conversions	Palaearctic (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	17: 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	80 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	19 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	4 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	Mammals, birds, plants, amphibians and

		level	reptiles.

3.4.2.1. 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. Three of them are detailed in Section 3.2.2 and additional methodologies exist (Weidema & Lindeijer, 2001; Geyer et al., 2010a;b). The UNEP-SETAC initiative is a step toward consensus on the 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 3.3), only a few or a single method 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 2.2, 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 that addressed the land use impact on biodiversity is that they faild to consider beneficial biodiversity impacts, which can be important in the context of livestock production (*e.g.*, semi natural grasslands with high biodiversity value, Sections **Error! Reference source not found.**, 3.2.1.2). 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 weights 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 process, 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 would be recommended 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 is an area for further exploration 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.

3.4.2.1 Spatial coverage and resolution

The spatial coverage of methods and characterization factors described in Sections 3.1.2 and 3.2 differs considerably across different methods. It is global for the MSA land use characterization factor (Alkemade *et al.*, 2009), the climate change (Schryver *et al.*, 2009) and water (Pfister *et al.*, 2009) methods. Other characterization factors are available at country- to region-scale. For instance, EDP land use characterization factors were computed for Central Europe (Koellner & Scholz, 2008) (Schmidt 2008 provided additional computations for South-East 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 details in the land use categories. The EDP characterization factors covers Central Europe and includes 53 land use and intensity classes. The MSA covers a global scale and includes 13 land use and intensity classes. These two examples illustrate the trade-off between spatial coverage and resolution. Although covering a global scale, the MSA categories are coarse and do not allow for 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).

3.4.2.2 Biodiversity coverage

The characterization factors presented in Sections 3.1.2 and 3.2 mainly describe biodiversity through species richness (*e.g.*, PDF) or species abundance (*e.g.*, MSA). Historically, relative species richness, expressed as m2.yr of Potentially Disappeared Fraction (PDF) of species 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 no occurrence in a region due to unfavorable 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 Fig. 2.7, 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 damage. Goedkoop et al. (2012) did a rough estimation 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 modeling 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 to assess 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 to assess 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.

4 Conclusion

4.1 Complementarities between frameworks

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

4.1.1 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.

With this current state of knowledge, LCA approaches are not well suited to answer 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 that are based on large geographical scales are much more suited to assessing land use changes impacts across bioregions, and not suited to assessing other more qualitative changes (such as the impacts of over- or under-grazing) within a bioregion. However, LCA is a very useful tool to conduct broad assessment of impacts on biodiversity at large spatial scales and to find 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 as they could be used to differentiate the effect of different practices or expand the analysis to other pressures and biodiversity levels and taxa.

4.1.2 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 (Section 13).

4.1.3 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 (Fig. 4.1). 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 those 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 comprehensively all pressure categories, pressures modeled from LCIA could be combined with indicators computed through other methods.

3. A limitation of the LCA framework for computing biodiversity impacts is that methods do not 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 could allow 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.

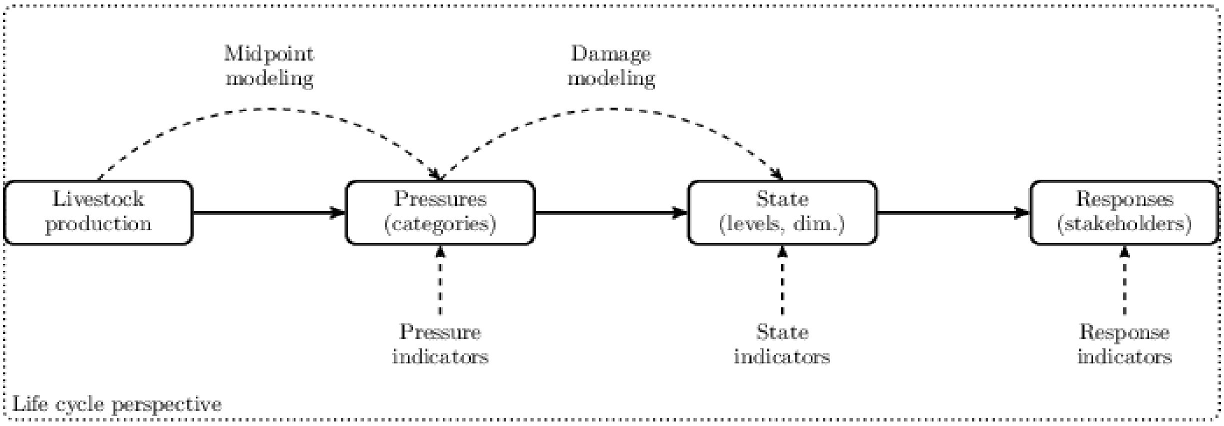


Figure 4.1: Complementarities between the Pressure-State-Response indicator framework and the LCA framework along the same causality chain. Complementarities at the different steps are discussed in the main text.

4.2 Concluding remarks

Measuring the impact of livestock production on biodiversity poses important methodological challenges. These challenges include: the need to address both 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 of them 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 with 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, in prep.). The objective of this 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 an open the way towards recommending recommend specific methodologies nor provide the associated, detailed guidelines.

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5 Appendix

5.1 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.

5.2 Mean Species Abundance values of different land uses

Land use category	Description	MSA	SE
Snow and Ice	Areas permanently covered with snow or ice considered as undisturbed areas	1.0	< 0.01
Bare areas	Areas permanently without vegetation (for example, deserts, high alpine areas)	1.0	< 0.01
Forests			
Primary vegetation	Minimal disturbance, where flora and fauna species abundance are near pristine	1.0	< 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	< 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	< 0.01

Table 6.1: Mean Species Abundance values of several land use and intensity classes, as computed by. Alkemade *et al.* (2009).

5.3 Other frameworks

5.3.1 From the academia

In the scientific literature, a wide variety of biodiversity indicators, assessment and footprinting methods has 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 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 puts 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. The 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 (see also Section 2.2.1).
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 in a condition where it maintains its diversity, quality, and capacity to support people and wildlife (Prescott-Allen, 2001). Biodiversity is one of the five dimensions of the index, it 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. The 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 the 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, several agri-environmental indicators are integrated in a context-specific, customisable index. Within the scope of the evaluation set by the evaluators, stakeholders participate to 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, 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 that would allow the comparison of biodiversity performances between different systems and regions; however, it provide a context-specific assessment of the change in relevant environmental impacts.

5.3.2 From inter governmental organizations

- **The FAO Sustainability Assessment for Food and Agriculture (SAFA)**

The FAO SAFA (2013) guidelines describe a holistic framework to assess the sustainability of agriculture, forestry and fisheries value chains. The framework covers the environment, economy, 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, they are 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 the 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 developed a methodology called corporate Ecosystem Service Review (ESR). It is a framework for companies to proactively develop strategies to manage risks and opportunities arising from their dependence and impact on ecosystem services. It is particularly relevant to the sector of agriculture which interact closely 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; therefore it would not be suitable to conduct a quantitative assessment of the biodiversity performance of livestock production. However, it could provide interesting elements to integrate in response biodiversity indicators targeting the ecosystem level.

LINK TO MORE INFORMATION:

http://www.wri.org/publication/corporate-ecosystem-services-review

5.3.3 From non-governmental organizations

- **High Conservation Value (HCV) approach**

World Wildlife Fund 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, sugarcane and soy production, as well as in biofuels and bioenergy standards, ecosystem carbon management and aquacultural 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.

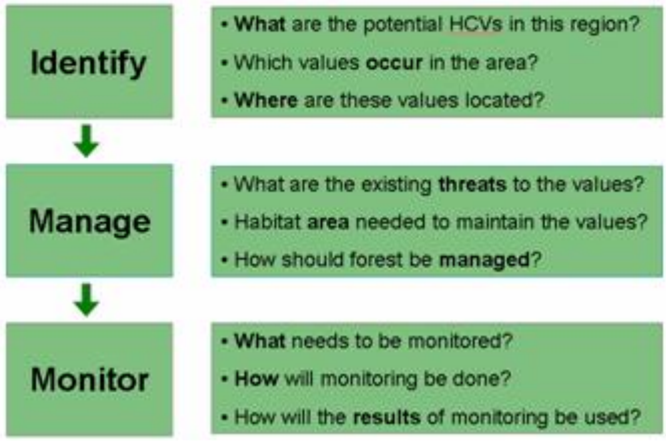
HCV3 Rare, threatened, or endangered ecosystems, habitats or refugia. E.g. patches of a regionally rare type of freshwater swamp.

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:



LINK TO MORE INFORMATION:
http://www.hcvnetwork.org/about-hcvf/the-six-high-conservation-values

- **The IUCN Red List of Ecosystems**

Ecosystems 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 under this proposal are one mean to enhance capacity that strengthens countries' contributions to IPBES, as well as reporting 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 reducing ecosystems services. Such RLE risk assessments are valuable and effective tools for different sectors relevant to sustainable development, including those working on:

- Global environmental reporting: partner countries would be in a better position to inform on progress towards the Aichi targets under the Convention on Biological Diversity.
- Conservation: to help prioritize investments in ecosystem management and restoration, reforms of resource use practices, and as a means for rewarding good ecosystem management.
- Land use planning: to highlight the risks faced by ecosystems under current and potential land use scenarios (e.g. land conversion, degradation), and the effects this might have on services such as clean water, maintenance of soil fertility, pollination, and the availability of natural products, and so the potential impacts on food security;
- Macro-economic planning: to provide a globally accepted standard that will support 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 and, conversely, the potential economic benefits of improved management;

- Improvement of governance and livelihoods: to inform the development of governance systems in ways that improve ecosystem management, livelihood security and social outcomes (gender and equity); and
- Private sector: a means of assessing potential environmental and social benefits and costs of alternative designs of future development projects as well as for 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 actions e.g. restoration, management, governance, gender, tenure. Based on the underlying causes of ecosystem changes, early application could include: a) the role of environmental safeguards where ecosystem risk assessments can highlight problems concerning certain interventions, e.g. mining; b) be one basis for improved landscape management and human wellbeing including restoration (e.g. ecosystem, forest); c) highlight 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 highlight the need for adaptation; e) help provide advance warning of natural disasters, especially slow onset ones (drought, sea level rise) and can be one means to help in their mitigation; and be on means for longer term monitoring.

Although Environmental Impact Assessments are often carried out, they are usually at the level of the site (district, catchment) or sector (health, water) and not more broadly at the national level. Strategic Environment Assessments may also be carried out, but these tend to be sectoral (e.g. forest sector). Simple and repeatable national (or sub-national) assessments (and ones which are based on internationally accepted criteria and categories) are rarely used or carried out, though the Red List of Threatened Species does this for species. The RLE has the potential to meet this demand at the national and regional levels, and could assist in monitoring the effectiveness of overall national level policies on land, water and environmental use, as it would:

- Provide a standard and repeatable way of understanding the impacts (positive or negative) certain approaches, policies, and management practices might have on ecosystems and the environment;

- Inform national government and development partners on how the risk of ecosystem collapse will impact, how this might happen, and how it could be best mitigated; and
- Be one indicator for assessing the impacts of development cooperation and national policy might have on ecosystem well-being.

- **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. I 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.

5.4 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 tool provides some guidance about use of 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 developed a Sustainable Agriculture Standard (Sustainable Agriculture Network, 2010). This standard include biodiversity criteria at both species and ecosystem level (see also Section 2.4.2)

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 will be available in 2015.

http://www.saiplatform.org/farmerselfassessment/introduction

http://www.standardsmap.org/fsa

http://sanstandards.org/sitio/

- **International Dairy Federation**

The International Dairy Federation is developing a guidance document for biodiversity assessments on dairy farms, with international applicability. This document will be available in early 2015.

- **Global roundtable for sustainable beef**

The Global Roundtable for Sustainable Beef (GRSB) is a global, multi-stakeholder initiative developed to advance continuous improvement in 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, 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. They are expected to be released in summer of 2015.

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/

5.5 Restoration and revegetation initiatives

- **Landcare International.**

Landcare is a community driven program 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 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 58% of Australian dairy farmers undertook revegetation programs on their properties and 47% of farmers protected areas of remnant vegetation (Watson & Watson 2012). Many of these revegetation programs were supported by Landcare. Revegetation plans are informed by the Australian dairy industry environmental assessment tool, DairySAT. http://www.dairysat.com.au/

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Watson, P & Watson, D; (2012) Dairying for Tomorrow Survey of NRM Practices of Australian Dairy Farmers http://www.dairyingfortomorrow.com/index.php?id=27

- **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 up scaling in other parts of the world. Additionally, as part of on-going 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 program administered by the Farm Service Agency (FSA). In exchange for a yearly rental payment, farmers enrolled in the program 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 10-15 years in length. The long-term goal of the program 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

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