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Ecosystem services assessment in livestock agroecosystems

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Foreword

This publication represents an important achievement for the FAO Livestock Environmental Assessment and Performance (LEAP) Partnership, a multistakeholder initiative that has evolved over more than a decade.

The livestock sector is a key component of sustainable food systems, contributing to food and nutrition security, economic development and employment. To date, however, there has been no comprehensive, science-based framework for assessing the environmental impacts of livestock – including both positive and negative dimensions. In particular, the ecosystem services provided through livestock-related activities have often been underrepresented in such assessments. The LEAP guidelines on ecosystem services in livestock agroecosystems seek to address this gap.

The FAO LEAP guidelines are grounded in scientific evidence and developed through a multistakeholder process that fosters inclusive dialogue and balanced outcomes. They benefit from contributions by internationally recognized academic experts, as well as engagement from public and private sector actors and civil society organizations. This collaborative model supports the development of outputs that are technically robust and globally relevant.

Since its inception in 2012, LEAP has produced 11 guidelines covering various environmental topics – from greenhouse gas emissions (GHG) to biodiversity – across the main livestock species. These guidelines promote methodological consistency while allowing for adaptation to specific production systems, species and regions. More than 400 academic contributors from around the world have supported this work, ensuring that the guidance reflects a diverse and extensive knowledge base.

I express my sincere appreciation for the achievements of this partnership. Particular recognition is due to the Food and Agriculture Organization of the United Nations (FAO) for its role as host, and to the dedicated staff whose professionalism and commitment have made this work possible.

It is a privilege to offer these reflections at this important juncture, and with the publication of this significant work. I remain confident that the best is yet to come.

Hsin Huang
FAO LEAP Chair (2016, 2022)

FAO Livestock Environmental Assessment and Performance Partnership

The FAO LEAP Partnership, launched in 2012, is a multistakeholder initiative dedicated to improving the environmental performance of livestock systems while ensuring their economic and social sustainability. The FAO LEAP Partnership develops science-based environmental assessment methodologies aligned with international standards to support evidence-based policy and action. LEAP technical guidelines, formulated by technical advisory groups (TAGs), undergo rigorous internal, technical and public reviews to ensure reliability and relevance. By engaging end users throughout the process, the partnership ensures its tools are applicable across diverse scales and geographical contexts. FAO LEAP aims to equip stakeholders with the tools and insights needed to drive positive change in the livestock sector through evidence-based policies and business strategies. It fosters science-based collaboration among governments, the private sector, academia, non-governmental organizations (NGOs) and civil society organizations.

FAO LEAP PARTNERSHIP STEERING COMMITTEE

The Steering Committee of the FAO LEAP Partnership provides overall leadership and approves the partnership's work programme. It comprises representatives from three key stakeholder groups: 1. Governments: Australia, Brazil, Canada, Costa Rica, France, Ireland, Kenya, the Kingdom of the Netherlands, New Zealand, Switzerland and Uruguay; 2. NGOs and civil society organizations: Compassion in World Farming, Environmental Defense Fund, International Union for Conservation of Nature, World Alliance of Mobile Indigenous Peoples and World Wildlife Fund; 3. Private sector: International Dairy Federation, International Egg Commission, International Feed Industry Federation, International Meat Secretariat, International Poultry Council and World Renderers Organization.

FAO LEAP PARTNERSHIP SECRETARIAT

The FAO LEAP Secretariat, hosted by the FAO Animal Production and Health Division, ensures that LEAP activities follow international best practices. As the partnership's central coordinating body, the Secretariat offers technical and administrative support to FAO LEAP partners and participants. It also supports TAGs in developing guidelines and ensures alignment with both FAO and partnership objectives. The FAO LEAP Secretariat includes Xiangyu Song, FAO LEAP Secretariat Manager; Paolo Medei, Sustainable Agriculture Specialist; Edoardo de Santis, Multistakeholder Partnerships Specialist; Julie Hanot, Partnership Intern; and Maud Lebeaupin, Partnership Intern.

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Collaborators

This publication is a product of the FAO LEAP Partnership.

ECOSYSTEM SERVICES APPROACHES TECHNICAL ADVISORY GROUP

The Ecosystem Services Technical Advisory Group (ES TAG) conducted the research and developed the technical content of the guidelines. The ES TAG was established in February 2023 and is composed of two TAG Leaders and 26 TAG Members (listed in alphabetical order by surname).

TAG leaders

- **Raimon Ripoll Bosch** (Wageningen University and Research, the Kingdom of the Netherlands)
- **Sophie Bertrand** (Centre National Interprofessionnel de l'Economie Laitière, France)

TAG members

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- **Jens Pistorius** (Julius Kühn-Institut, Germany)
- **Jeremy Pittman** (University of Waterloo, Canada)
- **Rachel Bardy Prado** (Brazilian Agricultural Research Corporation, Brazil)
- **John Walker Recha** (International Livestock Research Institute, Kenya)
- **Muhammad Riaz** (Government College University Faisalabad, Pakistan)
- **Gloria Claudia Rotolo** (Instituto Nacional de Tecnología Agropecuaria, Argentina)
- **Johannes Fritz Sauer** (Technische Universität München, Germany)
- **Gregory Sawyer** (The University of Sydney, Australia)
- **Dean Timothy Thomas** (Commonwealth Scientific and Industrial Research Organization, Australia)

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Multistep review process

This publication underwent multistep review processes: Internal (February 2024), technical (March and April 2024), FAO Animal Production and Health Editorial Committee review (April 2024), and public reviews (August 2024 and September 2024).

Step 1: Internal review

The FAO LEAP Secretariat and the FAO LEAP Partnership Steering Committee Members reviewed the initial draft provided by the ES TAG before the peer review.

Step 2: Technical review

The following technical reviewers and organizations contributed to the peer review of this report (listed in alphabetical order by surname):

- **Alexandre Berndt** (Brazilian Agricultural Research Corporation - EMBRAPA, Brazil);
- **Mariecia Fraser** (Aberystwyth University, United Kingdom of Great Britain and Northern Ireland);
- **Mutian Niu** (ETH Zurich, Switzerland);
- **José Paruelo** (National Institute of Agricultural Research - INIA, Colonia, Uruguay); and
- **Mieke Siebers** (Foundation for Sustainable Development, the Kingdom of the Netherlands)

Step 3: FAO Animal Production and Health Editorial Committee review

The document was reviewed by the FAO Animal Production and Health Editorial Committee members.

Step 4: Public review

The following scientists and organizations contributed to the public review of this report (listed in alphabetical order by surname):

- **Kara Barnes** (Canadian Roundtable for Sustainable Beef, Canada)
- **Sara Derighetti** (Federal Office for Agriculture, Switzerland)
- **Amanullah Khan** (The University of Agriculture Peshawar, Pakistan)
- **Jennifer Andrea Ricaurte** (WWF Colombia, Colombia)
- **Hans van Dam** (Nutreco, the Kingdom of the Netherlands)

The Public review version is available on the FAO LEAP website at the following link: https://openknowledge.fao.org/handle/20.500.14283/cd1949en

PERIOD OF VALIDITY

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

Abbreviations

ABM	Agent-based modelling
AES	Agri-environment schemes
ANPP	Aboveground net primary production
ASFs	Animal-sourced foods
BBNs	Bayesian belief networks
CAP	Common Agricultural Policy
CICES	Common International Classification of Ecosystem Services
EFTs	Ecosystems functional types
EMA	Emergy assessment
ES	Ecosystem services
EF	Ecological footprint
ESSI	Ecosystem Services Supply Index
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GHG	Greenhouse gas
GIS	Geographic information system
GNSS	Global Navigation Satellite Positioning Systems
GPS	Global Positioning System
HANPP	Human Appropriation of Net Primary Production
ICLF	Integrated crop–livestock–forestry system
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life cycle assessment
LEAP Partnership	Livestock Environmental Assessment and Performance Partnership
LUCI	Land utilization and capability indicator
LW	Liveweight
MA	Millennium Ecosystem Assessment
MFA	Material flow analysis
NASA	National Aeronautics and Space Administration, United States of America
NDCs	Nationally determined contributions
NDVI	Normalized Difference Vegetation Index
NPP	Net primary production
PES	Payments for ecosystem services
PPF	Production possibilities frontier
PPES	Programa de Pago de Servicios Ambientales
PS	Police sections
RMWOW	Remotely monitored walk-over weighing
SD	System dynamics
SDGs	United Nations Sustainable Development Goals
SV	Social value
SWAT	Soil and Water Assessment Tool
TAG	Technical advisory group
TEEB	The Economics of Ecosystems and Biodiversity
tESSI	Temporal Ecosystem Services Supply Index
UNEP	United Nations Environment Programme
UNSD	United Nations Statistics Division

Glossary

Bias

The deviation of results or inferences from the truth, or processes leading to such deviation. More specifically, the extent to which a statistical method fails to estimate the intended quantity or test the intended hypothesis.

Biodiversity

Variability among living organisms from all sources, including, inter alia, terrestrial, marine, and other aquatic systems and the ecological complexes that they are part of. This includes diversity within species, between species and of ecosystems (Article 2 of the Convention on Biological Diversity).

Biomass

Material of biological origin, excluding material embedded in geological formations and material transformed to fossilized material and excluding peat (ISO/TS 14067:2013, 3.1.8.1).

Ecosystem

A system in which the interaction between different organisms and their environment generates a cyclic interchange of materials and energy (OECD, 2001).

Ecosystem services

The direct and indirect contributions of (agro-)ecosystems to human well-being.

Emissions

Release of polluting substances to the atmosphere 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).

Extensive system

Low-input, low-output and consequently low-intensity system using minimal inputs of labour, fertilizers and capital relative to the land area being farmed. In less developed regions, these are often small-scale, mixed cropping subsistence farming systems. In highly developed regions, they are typically grassland-based farming systems, such as cattle and sheep grazing.

Feed (feeding stuff)

Any single or multiple materials, whether processed, semi-processed or raw, intended to be fed directly to animals (CODEX Alimentarius, 2004).

Footprint

Metrics used to report life cycle assessment (LCA) results that address a specific area of interest. They represent the sum of emissions and/or discharges caused by the production of one unit of the final product. LCA is based on international standards (ISO 14040:1997, ISO 14041:1999, ISO 14042:2000, ISO 14043:2000).

Grasslands

Synonymous with pastureland when referring to an imposed grazing-land ecosystem. The vegetation of grassland in this context is broadly interpreted to include grasses, legumes, other forbs and occasionally woody species.

Greenhouse gases (GHGs)

Gaseous constituents of the atmosphere, both natural and anthropogenic, that absorb and emit radiation at specific wavelengths within the spectrum of infrared radiation emitted by Earth's surface, the atmosphere and clouds (ISO 14064-1:2006, 2.1).

Habitat

The place or type of site where an organism or population naturally occurs (Article 2 of the Convention on Biological Diversity).

Input

A product, material, or energy flow that enters a unit process.

Life cycle assessment (LCA)

Compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle (ISO 14044:2006, 3.2).

Meat

Fresh, chilled, or frozen edible carcass, including offal, derived from food animals.

Nutrient

Substance required by an organism for growth and development. The primary crop nutrients are nitrogen, phosphorus and potassium.

Output

A product, material or energy flow that leaves a unit process. Products and materials include raw materials, intermediate products, co-products and releases.

Primary data

Quantified value of a unit process or activity obtained from a direct measurement, collected data or from a calculation based on direct measurements at its original source (ISO, 2014, 3.6.1).

Secondary data

Information obtained from sources other than direct measurement. This data has typically been collected by other parties, for a different purpose (and is used for new research or analysis) or obtained through modelling. Secondary data are used when primary data are unavailable or impractical to obtain.

Soil quality

Encompasses two distinct, but related parts: the innate elements and properties of soils (physical, chemical and biological conditions) and the capacity of soil to perform desired functions, such as biomass production or environmental services, considering its response to inputs and management. Soil quality varies, and soils respond differently depending on management inputs.

Soil health

The capacity of soil to function as a living system within ecosystem and land use boundaries, sustaining plant and animal productivity, maintaining or enhancing water and air quality, and promoting plant and animal health. Healthy soils maintain a diverse community of soil organisms that help control plant disease, insect and weed pests, form beneficial symbiotic associations with plant roots, recycle essential plant nutrients, and improve soil structure with positive repercussions for soil water and nutrient holding capacity, ultimately improving crop production (FAO, 2008). Two elements distinguish soil health from soil quality: (i) the inclusion of a time component (e.g. "the continued capacity of..."), and (ii) recognition of soil "as a vital living system".

Stock

Stocks represent real-world accumulations of materials. Each pool can store a quantity of nutrients, for example, as mineral or organic nitrogen in soils (such as in agricultural or semi-natural lands/ pools). This quantity is the nutrient stock. Nutrient stocks may be very large relative to nutrient flows (e.g. for soil pools) and are often difficult to quantify. However, the most relevant parameter for nutrient budgets is potential stock change – a variation over time in the respective accumulation – rather than the nitrogen stock itself. Nutrient stocks can be composed of nutrients in any form (adapted from UNECE, 2012).

Water use

Amount of water used (applied or consumed) for a specific purpose, such as irrigation in agriculture, industrial processes, cleaning or consumption (by drinking animals or domestic use).

Executive summary

Livestock agroecosystems drive much of rural economic activity and landscape management. Although they contribute significantly to food security and nutrition, public debate tends to polarize around their environmental, animal welfare and human health impacts. This focus on negative aspects often overshadows the beneficial services livestock systems provide to society – services that are seldom recognized or quantified.

Ecosystem services (ES) are the benefits people derive from ecosystems. For agroecosystems specifically, these services represent the direct and indirect contributions to human well-being. They fall into four categories: provisioning services like food and fibre production; regulating services including air quality control, climate regulation, water management, disease control, pollination and natural hazard mitigation; cultural services that provide recreational, aesthetic, educational, social and spiritual values; and supporting services such as soil formation, photosynthesis, and water and nutrient cycling.

To build strategies for truly sustainable livestock production systems, we need to understand and acknowledge how these systems rely on and support ecosystem services. A harmonized international approach is therefore needed to assess and evaluate ecosystem services in livestock production. This would ensure consistent methodologies and support sustainable management and policymaking in livestock agroecosystems. This document, which captures international experts' perspectives on ecosystem services assessment, represents an important first step towards developing more detailed guidance for assessing ecosystem services in livestock agroecosystems.

The guide explains the general concept of ecosystem services provided by livestock agroecosystems and explores various assessment methods to help readers effectively assess and promote these services. It covers biophysical, sociocultural, economic and modelling valuation methods, including their principles, advantages and limitations.

The guide makes two main recommendations:

- Use the Common International Classification of Ecosystem Services (CICES) as a standardized framework for identifying and categorizing ecosystem services.
- Apply the five-step roadmap to ensure robust valuation processes that produce reliable results for decision-making.

Part 1
Background

1.1 THE GENERAL CONCEPTS

1.1.1 Introduction: the history behind the concept of ecosystem services

The concept of ES is used to communicate the dependence of society on the biosphere (Daily, 1997; Groot *et al.*, 2002). It is attracting increasing attention in science (across disciplines), as well as in policymaking, the private sector and among practitioners.

The terms and concept of ecosystem services as a field of research, as we currently know it, emerged in the late 1970s and early 1980s (Gómez-Baggethun *et al.*, 2010). The work of Westman (1977) was one of the first attempts to measure the social benefits of ecosystem functioning. This led to the recognition that ecosystem functions can generate benefits to humans. These benefits were referred to as "services" and helped increase awareness and interest in nature conservation.

In the 1990s, the literature consolidated society's dependence on ecosystems through terminology such as "nature's services" or "ecosystem services" (Daily, 1997; Costanza *et al.*, 1997), and there was growing interest in valuing those services, particularly in monetary terms and natural capital accounting (Costanza *et al.*, 1997). In 1998, UNEP (United Nations Environment Programme), NASA (National Aeronautics and Space Administration, United States of America), and the World Bank published the report *Protecting Our Planet, Securing Our Future* (Watson *et al.*, 1998), which explored and explicitly recognized the inextricable linkages between environmental systems and basic human needs.

In the early 2000s, the United Nations launched the Millennium Ecosystem Assessment (MA), which was published in 2005. The MA was a landmark effort to assess human impacts on the environment and to report extensively on the contributions of nature and ecosystems to human well-being. The publication of the MA popularized the term "ecosystem services" and the benefits ecosystems provide to people. Moreover, the MA contributed to a surge in studies on ecosystem services (Fisher *et al.*, 2009; Rodríguez-Ortega *et al.*, 2014); supported adoption of the concept across scientific disciplines, including agroecosystems (Zhang *et al.*, 2007; Power, 2010) and livestock farming (Rodríguez-Ortega *et al.*, 2014); and helped bring the concept onto policy agendas (Gómez-Baggethun *et al.*, 2010).

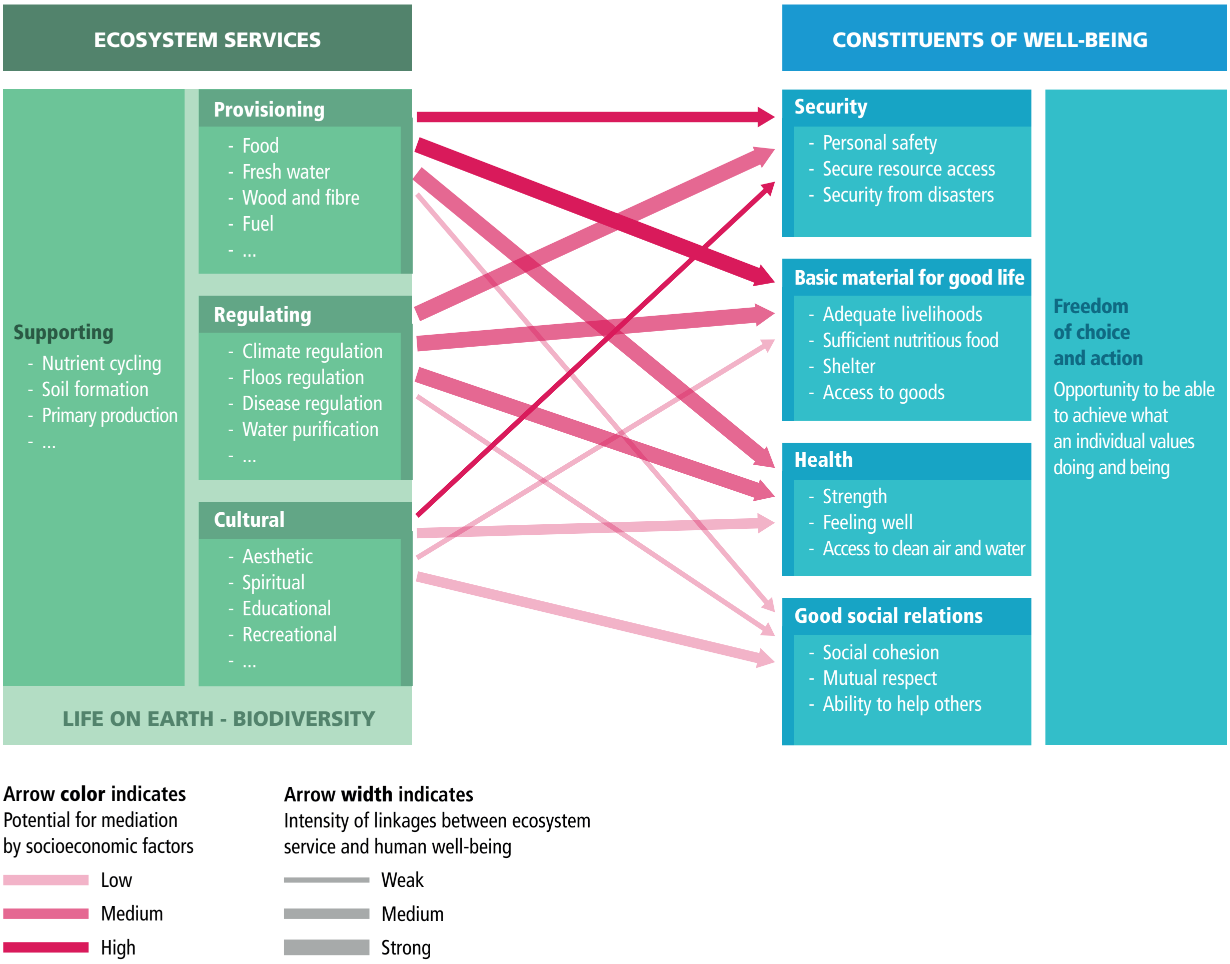
From the mid-2000s, another international initiative was launched, commissioned by the Government of Germany and the European Commission, to conduct a global analysis of the economic significance of biodiversity, the cost of biodiversity loss and the failure to take protective measures versus the costs of effective conservation (TEEB, 2010). This initiative culminated in the publication of *The Economics of Ecosystems and Biodiversity* (TEEB, 2010), which aimed to estimate the economic (monetary) value of ecosystem services across global biomes. It also delivered the TEEB dataset, which has evolved over the last decade into the Ecosystem Services Valuation Database (ESVD) (Brander *et al.*, 2024).

In 2010, the United Nations General Assembly appointed UNEP to convene a plenary meeting to establish the Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services (IPBES). IPBES is an intergovernmental body intended to strengthen the science–policy interface on biodiversity and ecosystem services, in a similar role to the Intergovernmental Panel on Climate Change (IPCC) in the field of climate science. IPBES released an initial conceptual framework in 2013 (IPBES, 2013) and published its global assessment report in 2019 (IPBES, 2019).

Today, the concept of ecosystem services continues to evolve, and research continues to expand across disciplines. Moreover, the use of the concept extends beyond academia into policy and government institutions, non-governmental organizations, and the private and financial sectors (Gómez-Baggethun *et al.*, 2010; Braat and de Groot, 2012; Jax *et al.*, 2018). For instance, there is an increasing development of economic incentives to steer decision-making toward conservation of biodiversity and ecosystem services – such as Payments for Ecosystem Services (PES) (Wunder, 2007; Engel *et al.*, 2008; Wunder, 2015). A more recent report has also highlighted growing evidence that biodiversity loss can have significant economic and financial implications, with declines in ecosystem services posing physical risks to economic actors who depend on them (NGFS–INSPIRE, 2021).

This increasing relevance also presents challenges for operationalizing the concept (Jax *et al.*, 2018). First, the concept of ecosystem services remains contested: i) it assigns a utilitarian function to nature, overlooking other dimensions such as intrinsic value and the right to exist; ii) the valuation and monetization of ecosystem benefits

FIGURE 1
The categories of ecosystem services and their link to constituents of human well-being



Source: Adapted from Millennium Ecosystem Assessment (Program) (Ed.). 2005. *Ecosystems and human well-being: Synthesis*. Island Press.

remains controversial (Westman, 1977) and has been perceived as commodification (Wilson, 2013; Silvertown, 2015); and iii) the term can be abstract and may not resonate with the general public (Díaz *et al.*, 2018; Bernués *et al.*, 2016).

Second, although widely used across scientific disciplines and increasingly adopted in policy and practice, the lack of harmonized terminology and conceptualization still makes implementation a challenge (Jax *et al.*, 2018).

Still, it is only possible to identify degradation or improvement in the environment and the services it provides when monitoring is in place. Monitoring, however, depends on shared definitions, clear metrics and established methodologies. The sections that follow in this chapter aim to guide the reader through the concepts related to ecosystem services, current frameworks, and their application in agroecosystems and livestock systems. Later sections will examine methodologies and metrics in more detail.

1.1.2 Ecosystem services

The definition of ecosystem services has been subject to debate in the scientific literature (Daily, 2007; Wallace, 2007; Fisher and Turner, 2008). Some authors (e.g. Costanza *et al.*, 1997; Daily *et al.*, 1997; Millennium Ecosystem Assessment [MA], 2005) defined ecosystem services as "the benefits people obtain from ecosystems". However, several aspects of this definition have been contested. For instance: i) whether "ecosystems" refers only to natural systems or also includes human-managed ecosystems. Authors note the increasingly blurred line between the two (Daily, 2007); ii) whether ecosystem services represent *direct* benefits obtained from ecosystems, with immediate impacts on human lives (e.g. Boyd and Banzhaf, 2007; Wallace, 2007), or whether *indirect* effects derived from ecological processes also constitute ecosystem services (as argued by Fisher and Turner, 2008); iii) whether "benefits" and "services" are essentially the same concept and can be used interchangeably (MA, 2005; Wallace, 2007), or whether they should be differentiated to facilitate integration and avoid double counting (Boyd and Banzhaf, 2007; Fisher and Turner, 2008); and iv) whether "benefits" or "services" are too technical and overly focused on utilitarian use, and whether "contributions" is a more appropriate and accessible term for a broader audience (Díaz *et al.*, 2018).

It is not the aim of this report to resolve these ongoing discussions. Instead, building on the existing literature, this report defines ecosystem services as "the direct and indirect contributions of (agro) ecosystems to human well-being".

As described in the Millennium Ecosystem Assessment (MA, 2005), ecosystem services are generally classified into four groups (see Figure 1):

1. **Provisioning** – Products obtained from ecosystems, such as food, fibre, timber, water, biochemicals or genetic resources.
2. **Regulating** – Benefits obtained from the regulation of ecosystem processes, such as climate regulation, pest control, water regulation, water purification or pollination.
3. **Cultural** – Non-material benefits obtained from ecosystems, such as cultural heritage, aesthetics or recreation.
4. **Supporting** – Necessary for the production of all ecosystem services, such as soil formation, nutrient cycling, water cycling or primary production.

The category of supporting ecosystem services is well defined (MA, 2005). However, there is ongoing debate as to whether "supporting ecosystem services" should be included as a distinct category when accounting for ecosystem services (see for example CICES, 2025), as they underpin the other categories of ecosystem services and do not constitute direct services to humans. Typically, they represent indirect services to human well-being only, and are delivered over long timeframes. In some cases, supporting ecosystem services are considered part of the regulating ecosystem services category. Often, the decision to include supporting ecosystem services in an assessment depends on the specific ecosystem, context and relevance (CICES, 2025). Further discussion on this issue is provided below (see Section 1.3), while guidance on whether to include this category in assessments is offered in Part 2 of this report.

Although the concept of ecosystem services is generally associated with those provided by natural ecosystems or nature, it also applies more broadly to forest ecosystems, grassland ecosystems, aquatic ecosystems and agroecosystems. Further information on agroecosystems and their role in providing ecosystem services is presented in Section 1.2.

1.1.3 The "cascade" principle of ecosystem services: intermediate versus final ecosystem services

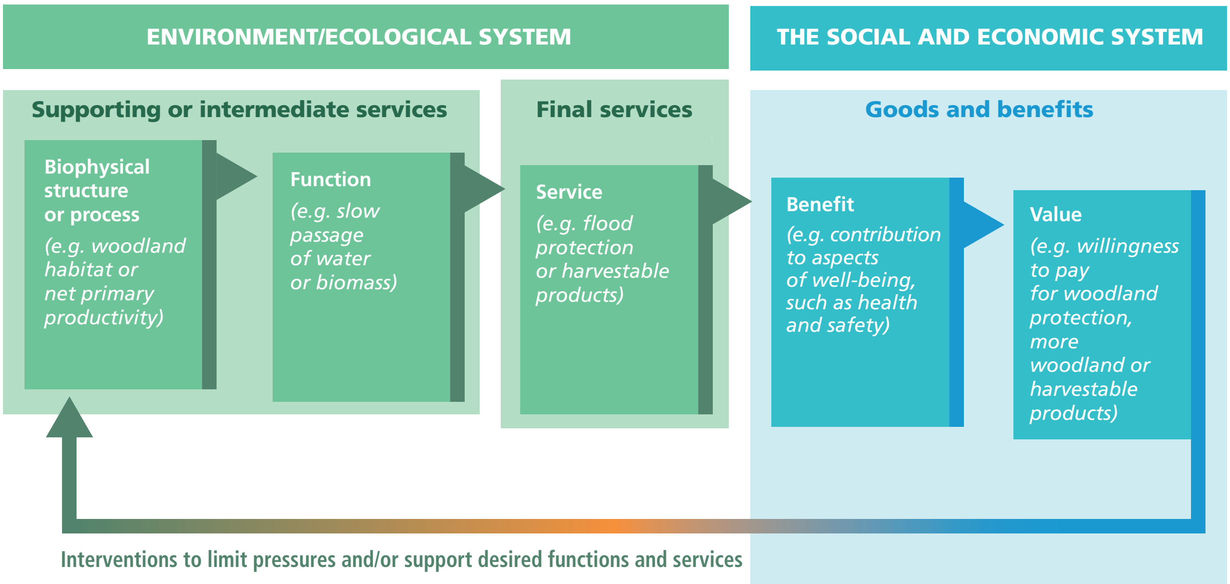
Across the literature, multiple definitions of ecosystem services exist. However, a shared underlying concept emerges: a "pathway" for delivering ecosystem services that begins with ecosystems – that is, their ecological structures and processes – and flows towards people, contributing to their well-being. This pathway has been described as a series of steps, or a cascade (see e.g. Fisher and Turner, 2008; Potschin and Haines-Young, 2011; Lamarque *et al.*, 2011; Maes *et al.*, 2012; Potschin and Haines-Young, 2016). Figure 2, adapted from Potschin and Haines-Young (2016), provides a visualization of the cascade.

The cascade begins by distinguishing between the biophysical system (related to ecosystems) and the social and economic system. These two systems are defined in various ways across the literature. For example, the biophysical system is also referred to as the biosphere, environment or nature, while the social and economic system is also termed the socioeconomic system, technosphere or anthroposphere. Each term carries slightly different meanings and nuances. This report does not promote one term over another but rather presents the biophysical components on the one hand, and the human components and their activities on the other – emphasizing their interconnectedness.

The cascade starts with supporting or intermediate services. These refer to the biophysical structures and processes in ecosystems (e.g. woodlands or net primary production), as well as their functions (e.g. the rate of water flow or biomass generation). These processes give rise to final services (still within the biophysical domain) that are ready to be harvested or experienced by humans. These final services relate to the formal categories of ecosystem services and include, for example, flood protection or harvestable products. While Figure 2 delineates a clear boundary between intermediate and final services, in practice, the distinction may be ambiguous. Intermediate services often stem from complex interactions between ecosystem structure and processes and can lead to final services (Fisher *et al.*, 2009). For instance, biodiversity or hydrological yield can be considered either final or intermediate services, depending on the context and the benefits of interest to stakeholders (see e.g. Fisher *et al.*, 2009).

Despite these nuances, the cascade principle assumes that final services are those that generate goods and benefits for humans. These services enter the social and economic system either as marketable products or as contributions to aspects of well-being. Ultimately, this links to the concept of the value of ecosystem services. "Value" refers to how humans perceive and appraise the services they receive, and should be understood in its broadest sense, encompassing value systems and worldviews (IPBES, 2022). Value systems

FIGURE 2
The cascade model of ecosystem services generation and flow



Note: CICES refers to "Common International Classification on Ecosystem Services", a systematic classification system to standardize ecosystem services for environmental accounting (CICES, 2025).

Source: Adapted from Potschin and Haines-Young (2011 and 2016). Potschin, M.B. & Haines-Young, R.H. 2011. Ecosystem services: exploring a geographical perspective. *Progress in Physical Geography: Earth and Environment*, 35(5): 575–594. https://doi.org/10.1177/0309133311423172; and Potschin, M. & Haines-Young, R. 2016. Defining and measuring ecosystem services. In: Potschin, M., Haines-Young, R., Fish, R. & Turner, R.K., eds. *Routledge handbook of ecosystem services*. Routledge.

differ among individuals, groups, and across spatial and temporal scales (Díaz *et al.*, 2015). The "valuation" of services can therefore be conducted using a range of methodologies – from economic valuation (e.g. beneficiaries' willingness to pay) to sociocultural valuation (e.g. stated preferences for certain services in specific contexts). These valuation methods are central to this report and are discussed in Part 3.

Ultimately, understanding values and valuation across time and space should inform the management of and mitigation of impacts on ecosystems. For example, Maes *et al.* (2012) and Wong *et al.* (2014) emphasize the need to distinguish between ecosystem characteristics and final ecosystem services, and to apply a cascade approach. This offers clarity and guidance for decision-making – both for those managing (agro)ecosystems and for those designing public policies to conserve or enhance the delivery of valued ecosystem services (Maes *et al.*, 2012; Wong *et al.*, 2014).

While the cascade has been widely accepted for the clarity it brings and its usefulness in communicating terminology, it has also faced criticism. One prominent critique is its portrayal of a linear relationship between ecological structures and processes on the one hand, and benefits and values on the other. In reality, these relationships are far more complex, challenging the oversimplified nature of the cascade model (Fischer *et al.*, 2009). Even within a single ecosystem, multiple linkages typically exist between ecological structures and processes, the functions they support, the other ecosystem services they interact with, and how these services are ultimately perceived and valued by their beneficiaries (Potschin and Haines-Young, 2016). Another criticism relates to definitions and terminology. For example, the concepts of "intermediate" ecosystem services or supporting ecosystem services originate from different disciplines and carry varying interpretations. As a result, the concept remains prone to confusion and controversy (Lamothe and Sutherland, 2018). Using the elements of the cascade to describe nature–human relationships, therefore, should not be seen as an end in itself. Rather, the cascade offers a framework and vocabulary for representing and better understanding the richness and complexity of these relationships (Potschin and Haines-Young, 2016).

1.1.4 The existing frameworks for classifying ecosystem services

The fluid nature of the concept of ecosystem services can stimulate discussion, but it also creates challenges when attempting to measure these services or design a classification system for consistent reporting.

The identification and classification of ecosystem services has been the subject of extensive study and debate among scholars (Fischer and Turner, 2008). To address differences in terminology across studies, initiatives and regions, several influential publications and frameworks have been developed, proposed and adopted. Readers interested in publications that focus on the identification and classification of ecosystem services are referred to Daily (1997), Boyd and Banzhaf (2007), Wallace (2007) and Fischer *et al.* (2009).

For those seeking to understand which frameworks or classification methods are most commonly used in recent studies, which ecosystem services have been analysed, or how the different publications and frameworks compare, we recommend consulting Lamarque *et al.* (2011), Götzl *et al.* (2013), Keane (2016) and Merida *et al.* (2016).

In short, several prominent frameworks have been proposed to classify ecosystem services at both the international and (supra-)national levels. The most widely recognized include: the Millennium Ecosystem Assessment (MA, 2005); The Economics of Ecosystems and Biodiversity (TEEB) (TEEB, 2010); the UK National Ecosystem Services Assessment (UK NEA) (UK-NEA, 2014); the US National Ecosystem Services Classification System (NESCS) (US-EPA, 2015); the United Nations Statistical Division's System of Environmental-Economic Accounting (SEEA) (Edens *et al.*, 2022); the Common International Classification of Ecosystem Services (CICES, 2025), hosted by the European Environment Agency; and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, 2019).

Each of these publications and frameworks exhibits distinctive features. Some aim to provide a broad representation of ecosystem services that can be generalized across global contexts, supporting international standardization (e.g. MA, 2005; TEEB, 2010; CICES, 2025). Others are designed for specific applications, such as national-level accounting (e.g. UK-NEA, 2014; US-EPA, 2015). While terminology for individual ecosystem services may differ across classification systems, there is often scope to "translate" or align services across frameworks. It is therefore essential that practitioners indicate which framework or classification method they are applying when identifying and classifying ecosystem services.

Given the scope of this report and the need to ensure comparability and facilitate adoption of frameworks by users – particularly those conducting accounting based on administrative or geographic boundaries worldwide – it is recommended (though not exclusively) to adopt the CICES classification. Five reasons support this recommendation:

1. It aims to be comprehensive and was developed from the interaction between key partners in ecosystem service classification, including the European Environment Agency (EEA), US Environmental Protection Agency (US-EPA) and United Nations Statistical Division (UNSD) (CICES, 2025).
2. It is aligned with the System of Environmental-Economic Accounting (SEEA), currently led by UNSD, making it suitable for widespread adoption by countries and subnational authorities within their accounting and statistical systems.
3. It is widely referenced in academic literature – more than 1 000 scientific papers have cited the CICES classification since 2018 (CICES, 2025).
4. It is continuously revised and improved based on feedback from peer-reviewed publications.
5. It has proven useful and applicable to both scientific and policy communities.

1.1.4.1 The Common International Classification of Ecosystem Services (CICES)

According to CICES (2025), this classification method is not intended to replace any existing systems. Rather, it facilitates interoperability between different classification approaches and enhances clarity in the way ecosystem services are measured and analysed.

CICES adopts the "cascade" principle and, accordingly, focuses only on "final ecosystem services" within the categories of provisioning, regulating and cultural services. It does not include supporting services, as these are considered intermediate services that are part of the underlying ecological structures, processes and functions.

Final ecosystem services in CICES are organized into a five-level hierarchical structure: section, division, group, class and class type. Each level offers increased specificity and detail. This structure is aligned with UNSD best practice guidance and supports both information aggregation (e.g. when data are missing or reporting at different scales) and fine-grained analysis at local levels.

A complete list of the ecosystem services classified under CICES (see CICES v5.1), along with the five hierarchical aggregation levels, is available in Appendix 2. For further information and the latest updates, please consult the official website: https://cices.eu

1.1.5 Ecosystem services and ecosystem disservices

Although the ecosystem services framework was developed to illustrate the dependence of human well-being on nature through its goods and services, a substantial body of literature also recognizes the so-called disservices.

Disservices can be understood as ecosystem structures and processes that ultimately reduce human well-being by generating nuisances and disturbances. While these disservices may originate in nature and ecosystems in general, they are especially prominent in the literature concerning agroecosystems (Oostvogels *et al.*, 2024). Agroecosystems are often seen as both recipients of disservices (e.g. pests, weeds, diseases or predators) and sources of disservices (e.g. habitat reduction, increased erosion or pollution). These, in turn, generate negative feedback loops that reinforce disservices from ecosystems back to agroecosystems (Zhang *et al.*, 2007; Oostvogels *et al.*, 2024).

In the literature, disservices are referred to using various terms, including "disservices" (Zhang *et al.*, 2007), "negative effects" (Pascual *et al.*, 2023; Oostvogels *et al.*, 2024), "negative externalities" (Visser *et al.*, 2022), "undesired effects" (Bernués *et al.*, 2016) and "disvalues" (Lliso *et al.*, 2022; Oostvogels *et al.*, 2024).

Some studies address both ecosystem services and disservices in their scope and analysis (Zhang *et al.*, 2007; Merida *et al.*, 2022). However, a comprehensive and contrasted framework that encompasses both ecosystem services and disservices remains lacking.

Disservices are also linked to environmental impacts. Current food systems – including agriculture and livestock production – are recognized as major contributors to environmental impacts. Prominent environmental impacts associated with livestock production include greenhouse gas (GHG) emissions, nutrient losses and soil and water pollution, freshwater withdrawal, land use change, land degradation (e.g. due to overgrazing) and biodiversity loss. These impacts are well established and acknowledged in the scientific literature. The livestock sector, therefore, has a major responsibility to assess and mitigate such impacts.

In this regard, FAO LEAP has established technical advisory groups (TAGs) and published a suite of guidelines to assess a wide range of positive and negative impacts in livestock supply chains. These guidelines go beyond impact assessment and promote the adoption of good production practices that contribute to impact mitigation. For further information on the assessment and mitigation of such impacts, readers are referred to the relevant FAO LEAP guidelines on:

Greenhouse gas emissions and other environmental impacts of livestock species:

- Environmental performance of large ruminant supply chains
- GHG emissions and fossil energy demand from small ruminant supply chains
- GHG emissions and fossil energy demand from poultry supply chains
- Environmental performance of pig supply chains

Environmental impacts of feed and additive production:

- Environmental performance of animal feed supply chains
- Environmental performance of feed additives in livestock supply chains

Nutrient flows:

- Nutrient flows and associated environmental impacts in livestock supply chains

Water use:

- Water use in livestock production systems and supply chains

Biodiversity loss:

- Biodiversity and the livestock sector – Guidelines for quantitative assessment

The contributors to these guidelines are aware of and acknowledge the environmental impacts arising from livestock production at both local and global scales. However, the scope of the current guidelines is limited to the assessment of ecosystem services. Therefore, they do not address the impacts or disservices associated with livestock agroecosystems. Instead, they focus exclusively on assessing ecosystem services (and eventually limiting negative impacts and disservices).

1.2 THE CASE OF LIVESTOCK AGROECOSYSTEMS

Agroecosystems are "ecological systems modified by human beings to produce food, fibre or other agricultural products" (Conway, 1987), and they currently cover a significant percentage of the globe. Agriculture is the largest land use globally, covering almost 40 percent of available land (Power, 2010; Foley *et al.*, 2011). Livestock uses 50 percent of that total agricultural land globally, which accounts for the use of grasslands as well as arable land to produce feed (Mottet *et al.*, 2017). The largest share of that land used to feed livestock is made up of grasslands (total agricultural land currently used for livestock is 2.5 billion ha, with almost 2 billion ha being grasslands) (Mottet *et al.*, 2017). Some of the grasslands could be converted into arable land, but 65 percent of the currently existing pastures and rangelands can be considered non-convertible (Mottet *et al.*, 2017). Therefore, livestock agroecosystems play a key role in managing grasslands and maintaining the flow of ecosystem services to people within a globalized system.

Agroecosystems are very diverse and constitute an umbrella term for all forms of agricultural production, including various livestock production systems. These types of production exist across a gradient of human modification of natural systems, which have evolved. The classification of livestock systems may be based on animal species, breed, land use, agroecological zone, the integration of livestock with cropping systems, the intensity of production and the kind(s) of products produced (Steinfeld, Wassenaar and Jutzi, 2006). However, it tends to mainly rely on land use, agroecological zone and integration with cropping (Seré and Steinfeld, 1996). FAO and the International Livestock Research Institute (2011), based on Notenbaert *et al.* (2009), have developed a widely used classification system, which includes: (1) agro-pastoral, silvo-pastoral and other pastoral systems; (2) mixed crop–livestock systems in which natural resources are most likely to be extensively managed; (3) mixed crop–livestock systems in which natural resources can be managed to intensify the productivity of the system; and (4) intensive systems, which include an amalgamation of urban and landless systems (see Box 1 for other animals and farming systems not considered in this report, but of increasing relevance in the ecosystem services domain).

At one end of the spectrum, the agro-pastoral and pastoral systems and the most extensive forms of mixed crop–livestock can demonstrate coexistence and coevolution between humans, agricultural activities and natural habitats. These systems are bonded to the natural environment and must avoid excessive stocking and overgrazing to prevent degradation of natural resources. For example, in many parts of the world, traditional agroecosystems are complex social–ecological systems that result from the integration of domestic and wildlife biodiversity evolving together over millennia. In this context, farming can be considered an intermediary that modulates the flow of ecosystem services from nature to people (Tenza-Peral *et al.*, 2023). Traditional production systems have strong links with local agroecosystem characteristics and are closely connected with the maintenance of the health of the local environment (Hocquette *et al.*, 2018). These systems usually rely on minimal inputs, but their capacity to provide outputs (i.e. marketable products, such as food) is also limited.

At the other end of the spectrum are modern, intensive and specialized livestock production systems. These systems require high inputs (e.g. fertilizers, agrochemicals, feeds, labour, energy, capital) to sustain a higher delivery of outputs. The reliance on agricultural inputs has largely replaced the ecological processes and functions of the ecosystems that underpin agriculture. For instance, nutrient cycling and nitrogen fixation (by legumes and *Rhizobium*) have been substituted by nitrogen fertilizers; natural pest control has been replaced by pesticides and herbicides; and water cycles have been adapted to serve crop productivity through irrigation. While successful in boosting the production of affordable and safe food (despite existing challenges of accessibility and distribution of such food globally), this trend has also yielded adverse effects on ecosystems, leading to environmental impacts and limiting the provision of ecosystem services (Hazell and Wood, 2008).

Currently, there is a generalized trend of specialization and intensification in agriculture and livestock production with the primary aim of increasing food production. For agroecosystems, this means that there is a tendency to intensify or abandon more traditional ways of farming and shift toward more intensive forms of production. Striving for systems that rely on external inputs over the ecological processes underpinning agriculture will ultimately erode the wider provision of ecosystem services if no other preservation strategies are adopted alongside intensification. For instance, many studies relate to grassland ecosystems, which are influenced by livestock production and management. Grasslands are one of the most important and extensive agroecosystems and are key in delivering ecosystem services (Taube *et al.*, 2024; Lindborg *et al.*, 2023; Schils *et al.*, 2022). Nevertheless, the provision of ecosystem services depends on the management intensity and composition of the grasslands (Schils *et al.*, 2022; Lindborg *et al.*, 2023). Several studies report that there is evidence that these ecosystem services have been decreasing due to land use/land cover change and the degradation of native grasslands in favour of more intensive grassland management (Modernel *et al.*, 2016; Paruelo *et al.*, 2016; Staiano *et al.*, 2021; Lindborg *et al.*, 2023).

Agroecosystems and ecosystem services are intertwined and relate in different ways, including feedback loops (Zhang *et al.*, 2007; Tenza-Peral *et al.*, 2023) (Figure 3). Figure 3 illustrates how agroecosystems are positioned at the interface between the ecological system (expressed as ecosystem) and the socioeconomic system (expressed as society). Agroecosystems depend on a continued supply of ecosystem services (i.e. they are receivers of ecosystem services, namely provisioning and regulating) as well as providers of ecosystem services to society (namely provisioning, regulating and cultural). Agroecosystems, meanwhile, are both sources and recipients of disservices (Zhang *et al.*, 2007; Moonen and Barberi, 2008). As receivers, agroecosystems largely rely on ecosystem services such as pollination, nutrient cycling and pest control. As providers of ecosystem services to society, agroecosystems have been managed to increase the provisioning services of food and fibre. Nonetheless, they can also contribute to nutrient cycling, habitat provision or the maintenance of cultural landscapes, among others (Alders *et al.*, 2021). The disservices, which largely relate to environmental impacts from livestock agroecosystems or risks to production, are further elaborated in the previous section (Section 1.5 – Ecosystem services and disservices).

The feedback loops in Figure 3, which can be positive or negative, represent the capacity of agroecosystems to generate favourable or unfavourable conditions for the maintenance of the same agroecosystems and their functions. For instance, a positive feedback loop could be exemplified by an agroecosystem that creates habitats and conditions for pollinators (to pollinate crops or fruit trees). The presence of those conditions may increase the number of pollinators in the agroecosystem itself (hence not dependent on habitats adjacent to the agroecosystem), which may boost yield and perpetuate the conditions and presence of those desired pollinators. A negative feedback loop could be explained by an agroecosystem that, for instance, diminishes the presence of habitats and conditions for natural pest control. This can lead to fewer predators that control pests, and hence increase the incidence of pests. The subsequent management to further control the pests may continue to hinder the presence of the predators (and the favourable conditions for their presence) that would help suppress the pests.

Both the range of ecosystem services provided and the magnitude of those services depend on how people manage the agroecosystems (Power, 2010). Hence, the

FIGURE 3
Livestock agroecosystems (in the interface between the ecological and the social system) as receivers and producers of ecosystem (dis)services.

Note: The box (dashed line, in blue) depicts the primary scope of the guidelines.
Source: Authors' own elaboration.

provision of ecosystem services from agroecosystems is context- and farming system-specific. That is exactly the primary scope of these guidelines (Figure 3; box in dashed lines): to guide the assessment of ecosystem services provided by livestock agroecosystems.

Several publications have reviewed and reported ecosystem services provided by livestock agroecosystems and the diverse production systems (e.g. Rodriguez-Ortega *et al.*, 2014; Leroy *et al.*, 2018; Adlers *et al.*, 2021; Merida *et al.*, 2022). We briefly report on the ecosystem services from terrestrial livestock agroecosystems below, with a focus on the available literature.

Provisioning ecosystem services. A principal role of livestock keeping is to produce food (i.e. dairy, meat, eggs). Livestock contribute most to food security when transforming feed inedible by humans (e.g. utilizing grasslands, co-products or waste streams) into nutritious foods (Adlers *et al.*, 2021). However, the increased inclusion of high-quality feeds in livestock diets may entail feed–food competition and hinder food security (van Zanten *et al.*, 2018; Muscat *et al.*, 2020). Meanwhile, the ecosystem services framework falls short in capturing the quality aspects associated with the foods produced (see Chapter 14.3 of Bernués *et al.*, 2016;). In addition to food, livestock production generates

BOX 1

Consideration of other animals and production systems

Fisheries

Fish and bivalve aquaculture represent other types of agroecosystems of global importance with relevance to ecosystem services. Several studies have examined the ecosystem services provided by aquaculture (Alleway *et al.*, 2019; Gentry *et al.*, 2020; van der Schatte Olivier *et al.*, 2020; Weitzman, 2019). These systems have been shown to provide habitat (Gentry *et al.*, 2020), nutrient remediation (van der Schatte Olivier *et al.*, 2020) and various cultural services (Alleway *et al.*, 2019). However, the classification of fish and bivalves as livestock remains contested and uncertain. For instance, FAO does not typically categorize fish and bivalves as livestock. Nonetheless, the tools and methodologies proposed in these guidelines may also be relevant and applicable to fish and bivalve aquaculture systems. Furthermore, the latest revision of the CICES classification method includes increased attention to the ecosystem services provided by aquaculture and fisheries.

Beekeeping

Many insect species – including beetles, flies and others – contribute to pollination, but bees are considered the most important group of insect pollinators for crops and wild plants. Both wild bees and honeybees provide pollination services. Honeybees, which are managed by beekeepers in most parts of the world, are considered livestock and part of agriculture. Although beekeepers primarily manage honeybees to obtain bee products such as honey, pollen and wax, the pollination service provided is usually an inadvertent but essential and often financially uncompensated ecosystem service. While beekeepers may feed their bees during periods of insufficient forage, wild bees rely solely on floral resources in their surroundings and typically have smaller flight ranges than honeybees. Therefore, providing year-round forage for bees – in both agricultural and non-agricultural landscapes (including urban areas) – is essential to sustaining their pollination services. To promote and support beekeeping, FAO has published specific resources, such as *Good beekeeping practices for sustainable apiculture for the different bee species* (https://www.fao.org/3/cb5353en/cb5353en.pdf).

Insects

In recent years, the production of insects for food and feed has grown significantly, driven mainly by the demand for more sustainable protein sources (Larouche *et al.*, 2023). Compared to conventional meat production, rearing insects for food is generally less resource-intensive, and insects are widely discussed as a sustainable feed option for livestock (van Huis and Gasco, 2023). Although insects are animals and industrial insect farming shares similarities with livestock production, it is currently regarded more as an alternative to livestock than as a form of livestock production. Because industrial insect farming is typically conducted indoors, its direct impacts on ecosystem services are minimal. However, insect excreta (frass) are gaining attention as an organic fertilizer, with promising implications for sustainable agriculture (Poveda, 2021).

Source: Authors' own elaboration, based on cited sources.

a range of other products, such as wool, hides and leather, or feathers, and provides the basis for cosmetics, adhesives and pharmaceutical products, among others.

The production of manure by livestock is also a provisioning service. It provides nutrients and organic matter to soil and plants when applied in a balanced manner. Manure is important for supporting crop production in mixed crop–livestock systems and in grassland ecosystems, and it can improve soil health (Briones and Schmidt, 2017; Cozim-Melges *et al.*, 2024). On occasion, manure can be burned and used as a fuel source or as an energy carrier to produce energy in biodigesters (Muscat *et al.*, 2020). Nonetheless, excessive use and concentration of manure in particular geographical areas has become a widespread problem, creating disservices and resulting in negative environmental impacts (Sutton *et al.*, 2011).

Regulating ecosystem services. As shown in Figure 3 and extensively reported in the literature, agroecosystems largely depend on regulating ecosystem services. The management of intensive and extensive production systems, or the adoption of certain individual practices, influences the capacity of agroecosystems to deliver regulating ecosystem services. The regulating services are generally linked to grassland and pastoral systems and to extensively managed mixed systems (Cooper *et al.*, 2009), but smaller, marginal areas in intensive systems also provide regulating ecosystem services (e.g. riparian areas, buffer strips, field margins, wetlands). The services include the provision of habitats for a wide variety of plant, invertebrate and vertebrate species (Brandle *et al.*, 2004; Blumetto, 2022; de Santiago *et al.*, 2022; Kok *et al.*, 2020), as well as soil life and decomposition of organic matter (Merida *et al.*, 2022; Cozim-Melges *et al.*, 2024); contributing to seed dispersal of native plants (Manzano and Malo, 2006); biological control of insects and weeds (Gorosábel *et al.*, 2020); pollination (Kimoto *et al.*, 2012; Maccagnanie *et al.*, 2020); reducing the risk of wildfires by lowering fuel loads (Leroy *et al.*, 2018); or maintaining floodplains clear and contributing to water regulation (Merida *et al.*, 2022; Faccioni *et al.*, 2019).

The capacity to store and sequester carbon in soils, and the contribution to climate regulation, is also well documented (Rodriguez-Ortega *et al.*, 2014; Merida *et al.*, 2022; Wang *et al.*, 2024). This usually relates to land use and land use intensity, ranging from arable land to temporary and permanent grasslands and (semi-)natural grasslands. The relevance of this topic in livestock agroecosystems is prominent. For instance, FAO LEAP has already published the *Guidelines on measuring and modelling soil carbon stocks and stock changes in livestock production systems* (2019).

Cultural ecosystem services. Literature also explores the role of livestock agroecosystems in providing cultural ecosystem services (Ripoll-Bosch *et al.*, 2014; Rodriguez-Ortega *et al.*, 2014). In many regions, livestock have a strong influence on sociocultural systems, contributing to cultural identity and religious experiences (Adlers *et al.*, 2021; Merida *et al.*, 2022).

Livestock agroecosystems have a prominent role in shaping landscapes. Hence, they are acknowledged for their capacity to provide cultural landscapes and aesthetic experiences (Bernués *et al.*, 2016; Bernués *et al.*, 2019). Cultural ecosystem services from livestock agroecosystems are particularly prominent when livestock themselves and/or the historical structures associated with them (e.g. stone walls, barns and shelters, haystacks) are an identifiable part of the landscape.

Meanwhile, this promotes the generation of recreation and agritourism services (Faccioni *et al.*, 2019; Leroy *et al.*, 2018), stimulates artistic inspiration (Merida *et al.*, 2022) and contributes to cognitive development (Rodriguez-Ortega *et al.*, 2014; Merida *et al.*, 2022).

Part 2

Roadmap for the evaluation of ecosystem services from livestock agroecosystems and the conceptual framework of this report

2.1. ROADMAP FOR THE EVALUATION OF ECOSYSTEM SERVICES FROM LIVESTOCK AGROECOSYSTEMS

Given the wide range of methodological approaches and specific methods available for valuing ecosystem services provided by livestock agroecosystems, selecting appropriate methods requires consideration of: (i) their relevance to the purpose of the valuation exercise, (ii) their methodological reliability; and (iii) their feasibility within practical, contextual constraints. These guidelines propose a fit-for-purpose procedure for method selection, whereby chosen methods must meet user needs and be credible and feasible to apply (Hamilton *et al.*, 2022). The ideal valuation method lies at the optimal intersection of the following three dimensions (Figure 4):

- **Relevance:** This concerns how useful and informative the method is for end users, given the purpose of the valuation. Relevance extends beyond the technical quality or functionality of a method. Complex methods may provide more detailed outputs but may also be constrained by the availability of data and resources. Depending on the valuation's scope and objectives, it may not be necessary to model processes at high temporal or spatial resolution or to capture detailed inputs and outputs of livestock management systems. In such cases, simpler approaches may be more appropriate.
- **Reliability:** Reliability is contingent upon both credibility and legitimacy (Hamilton *et al.*, 2019; Hamilton *et al.*, 2022). *Credibility* refers to the technical and scientific validity and robustness of a method. Reliable methods are theoretically sound, transparently applied and well-documented throughout selection, implementation, analysis and interpretation. This is especially critical when valuations inform policy-level decisions. *Legitimacy* refers to fairness and equity, ensuring that selected methods adequately consider diverse ecosystem services and sociocultural values. Methods must also clearly acknowledge their scope and limitations.
- **Feasibility:** The feasibility of a valuation method depends on the specific context in which it is applied. The three most common constraints include financial, technical and human resource limitations. Time availability may also restrict the use of some methods. Suitability extends beyond how a method will be used to include the broader context in which it will be applied. These practical considerations are as important as scientific ones when selecting methods for ecosystem services valuation.

Fit-for-purpose method(s) are best identified through the involvement of multiple actors. Engaging both experts and end users in the selection process helps avoid choosing methods that are ill-suited to the valuation context. While stakeholder involvement may not resolve all challenges associated with method selection, it can help align priorities and enhance the overlap between utility, reliability and feasibility.

These guidelines propose five steps as a roadmap to support a robust and sound valuation process that yields meaningful results and provides quality input for decision-making:

1. Defining the purpose of the valuation
2. Framing the valuation
3. Selecting valuation approaches and methods
4. Implementing the valuation methods
5. Applying the outcomes of the valuation

These five steps (described below and illustrated in Figure 5) are adapted from previous frameworks (e.g. IPBES, 2022) for the valuation of ecosystem services. Each step begins with key questions that must be addressed, and all require decisions that affect the relevance, reliability and feasibility of the valuation process.

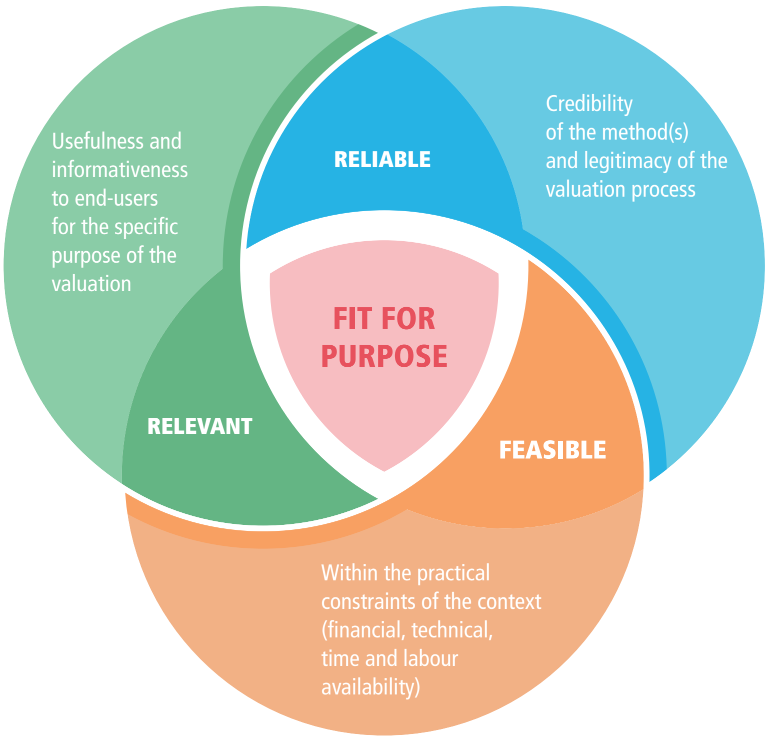
STEP 1. Defining the purpose of the valuation

Why is the valuation performed? Which outcomes are expected? Which decisions are aimed at?

The purpose of the assessment process is often clear from the context in which it takes place but fine-tuning and explicitly defining this purpose – particularly the expected outcomes – will aid in designing the assessment. A clear statement of why the valuation is being conducted and the type(s) of decision(s) it seeks to inform will help select the most appropriate method(s) (Barton and Harrison, 2017).

FIGURE 4

Fit-for-purpose framework for selection of methods to value ecosystem services provided by livestock agroecosystems



Source: Adapted from Hamilton, S. H., Pollino, C. A., Stratford, D. S., Fu, B., & Jakeman, A. J. (2022). Fit-for-purpose environmental modelling: Targeting the intersection of usability, reliability and feasibility. *Environmental Modelling & Software*, 148, 105278. https://doi.org/10.1016/j.envsoft.2021.105278

In general, seven main purposes for valuation processes can be distinguished:

i) Evaluate the impact of (changes in) livestock agroecosystem management; ii) inform policy development; iii) awareness raising; iv) understanding ecosystem service dynamics; v) advancing knowledge; vi) inferring people's preferences; and vii) benchmarking.

These seven purposes are not mutually exclusive and often complement and relate to each other.

i. **Evaluate the impact of (changes in) livestock agroecosystem management**
This includes assessing the impact of changes in socio-economic, environmental or policy domains, or more specifically, of changes in livestock farm management and practices, on the provision of ecosystem services from the agroecosystems in which the farms are embedded. Ideally, this assessment involves pre- and post-implementation valuation of ecosystem services, before and after a defined timeframe, changes at a regional scale, or shifts in farms and farming systems.

ii. **Inform policy development**
The assessment aims to inform the development of new policies at any scale of interest (local to international). It is essential for identifing the livestock agroecosystem(s) and the ecosystem services likely affected by the policies being developed. Benchmarking the current state of ecosystem services is central to assessing future policy implementation. In some cases, the purpose of the valuation process would be to anticipate the impacts of draft policies, which implies the use of methodologies that allow for future scenario analysis.

iii. **Awareness-raising**
Valuation processes often seek to raise awareness of the full range of ecosystem services provided by particular livestock agroecosystems, or of specific services undervalued or overlooked by society or certain actors (e.g. farmers, policymakers, consumers, environmental activists). Failure to account for these services typically leads to policies or farm management changes that ultimately diminish agroecosystem capacity to provide them. Overlooking the wide array of ecosystem services may result in prioritizing a reduced set of services rather than striving for an equitable balance between the benefits and drawbacks they entail.

FIGURE 5
Guiding framework for the assessment and valuation of ecosystem services in livestock agroecosystems

Source: Authors' own elaboration.

iv. **Understanding ecosystem service dynamics**
Valuation processes aim to understand the provision of ecosystem services by livestock agroecosystems and, in particular, the interactions (synergies and trade-offs) between them. Understanding the dynamics of ecosystem services could be a final purpose of the valuation per se, but also a first step needed to complement and inform other purposes, such as informing policy development.

v. **Advancing knowledge**
The main purpose here is to develop new scientific or technical knowledge – for example, valuing ecosystem services previously unvalued, developing new valuation methods or improving existing ones. This requires a sound, robust and transparent valuation process to ensure reliable and repeatable results. As in the previous case, advancing knowledge can be a final or intermediate purpose supporting other aims like awareness raising.

vi. **Inferring people's preferences**
Valuation may seek to understand the values, priorities and preferences of individuals or stakeholder groups regarding ecosystem services. This includes understanding human values, quantifying prefer-

ences (e.g. prioritization, ranking or monetary evaluation), and exploring trade-offs among different stakeholder groups' preferences.

vii. **Benchmarking**

Benchmarking involves comparing the performance of businesses (e.g. farms or companies), processes or products regarding their capacity to (co-)deliver ecosystem services. This may also include product certification or typification. However, further development and standardization of methodologies and metrics may still be necessary for robust benchmarking.

STEP 2. Framing of the valuation

Which particular livestock agroecosystem is being assessed? Which specific ecosystem services are important to consider for the valuation? Which resources (financial, technical, human and time) are available?

The framing of the valuation defines what livestock agroecosystem and which ecosystem services are being valued, as well as the financial, human and technical resources available for the valuation. The definition of ecosystem services provided by the agroecosystem has two components: (i) specifying the agroecosystem itself; and (ii) identifying the specific ecosystem services of interest. Therefore, Step 2 can be further divided into three substeps:

A. **Specify the livestock agroecosystem components and boundaries in space and time**

The specific livestock agroecosystem under consideration needs to be clearly defined in terms of the farming systems involved, as well as the social and environmental context in which it is embedded. Depending on the valuation scope, spatial boundaries may range from a single farm to a regional (e.g. Huntsinger *et al.*, 2014) or global scale. The components to be specified will largely depend on these spatial boundaries. Generally, key components of farming systems include livestock species, farm inputs and outputs and their origins and destinations, land use (and/or land cover) and land use intensity, the development of the farming system, and the stakeholders involved in managing the agroecosystem (i.e. producers who leverage ecosystem services) and the beneficiaries of those services.

For the wider context, key components include the type of biome or "natural" ecosystem boundaries (e.g. grain basin, watershed, plain, mountain) and climate or other factors determining the ecological functioning of the agroecosystem. When a dynamic approach to ecosystem service provision is required (i.e. when the evolution of ecosystem services provision needs to be assessed), the time scale of analysis should always be specified. Natural processes are continuously evolving, and natural resources do not have static values. The capacity of agroecosystems to deliver services is inherently time-dependent; therefore, including a time scale is essential for comprehensive valuation.

B. **Identify the ecosystem services provided by livestock agroecosystems**

The ecosystem services to be valued must be identified and described using standardized frameworks. Given the scope of this report (see Section 1.1), it is recommended to adopt the Common International Classification of Ecosystem Services (CICES) (see Appendix 2 on Ecosystem service classification following the CICES classification). It is important to clearly outline the connection between the identified ecosystem services and the particular agroecosystems examined, as this may influence the chosen methodological approaches. Special attention should be given to identifying all ecosystem services relevant to the valuation's purpose as defined in Step 1. A clear and standardized definition of ecosystem services to be valued will enable review and, if necessary, redefinition of the valuation purpose. In some cases, components and boundaries of the agroecosystem under study may also need to be redefined. Generally, all ecosystem services provided by the livestock agroecosystem should be explicitly stated, including which will be assessed or excluded, with appropriate justification. This ensures no ecosystem services are overlooked and that the valuation's scope and reach are acknowledged.

C. **Specify available resources and resources expected to be raised**

Finally, limitations related to financial, human and technical resources, as well as time constraints, must be evaluated. Existing resources available for the valuation should be explicitly stated, alongside potential additional resources to be raised. These available resources must be considered in the subsequent step when selecting valuation methods, and any mismatch between resources needed and those available should prompt replanning. In the long run, this will help prevent embarking on a valuation that is unfeasible to complete within the available resources. This iterative process may involve reconsideration of Steps 1, 2 and 3 or elements within them. Guidance on resource-related decision-making is provided in Section 2.3.

STEP 3. Selection of valuation approaches and methods

Which valuation approach, method or combination of approaches and methods is relevant to the decision? What is the balance between the usefulness and reliability of each methodological alternative? What expertise is needed to design and

implement the valuation method(s) and analyse their results? Which are affordable given financial, technical and time constraints?

Once the framing of the valuation has been clearly defined, the most appropriate methodological approach (i.e. modelling, biophysical, economic, sociocultural) or combination of approaches needs to be selected to achieve the valuation's purpose. The selected method should be appropriate to address the assessment's objective (Step 1). Several methods may fulfil the same purpose; therefore, understanding the context of different valuation methods is important. Section 2.3 provides an overview and comparison of methods based on resource and skill requirements to assist in method selection. Part 3 of this report offers detailed explanations of each method. The choice of approach and specific methods within each approach has a significant impact on the valuation results.

When selecting the most appropriate method(s), facilitators should consider the trade-offs between usefulness, reliability and feasibility among existing approaches (IPBES, 2022). It is rarely possible to conduct ecosystem services valuation using methods that simultaneously provide all relevant information from key stakeholders (relevance), robust data on all ecosystem services and their trade-offs (robustness), and require minimal financial, human and time resources (feasibility). These guidelines aim to provide a portfolio of available methods, clearly describing their strengths and limitations, to assess ecosystem services provided by livestock agroecosystems and facilitate method selection.

There is a growing consensus that no single valuation method can fully capture the performance of livestock agroecosystems, especially regarding ecosystem services provided to society. Combining methods broadens the scope and perspectives of the analysis, leading to a better understanding of ecosystem services and their diverse benefits to actors, stakeholders and society. Integrating diverse and complementary methodologies in valuation is recommended where possible (e.g. Gómez-Baggethun *et al.*, 2014; Scholte *et al.*, 2015; Dunford *et al.*, 2018; Rincón-Ruiz *et al.*, 2019). This is particularly critical for decision-making in practice, policy and research, since a narrow set of valuation methods risks overrepresenting the values of dominant or powerful groups in the valuation space (Jacobs *et al.*, 2023).

STEP 4. Implementation of valuation methods

What specific method or combination of methods will be implemented? What specific considerations or steps are further needed, depending on the method applied? What expertise is required to carry out the valuation? What practical issues must be considered when implementing the methods? What are the strengths and limitations of the selected methods?

This step addresses the implementation of the valuation method(s) selected in Step 3, whether biophysical, economic, sociocultural, modelling or a combination of the methods. Practitioners should recognize that implementing any valuation method requires different resources, skills and expertise (see Section 2.3). Therefore, having appropriate personnel to carry out the valuation is critical. Each methodological approach has specific requirements in this regard. Part 3 of this report details the particularities of each approach and valuation method, along with practical and theoretical considerations. Each method also has specific requirements that should be considered in Steps 2 and 3.

Nonetheless, this guideline should be considered an initial framework for selecting methods, and further literature and expertise will be necessary for the proper implementation of any valuation methods described herein.

STEP 5. Application of the outcomes of the valuation

How can the outcomes of the valuation be used? How should outcomes not be used? What uncertainties and risks are inherent in the methodology and the outcomes? What risks do these uncertainties entail?

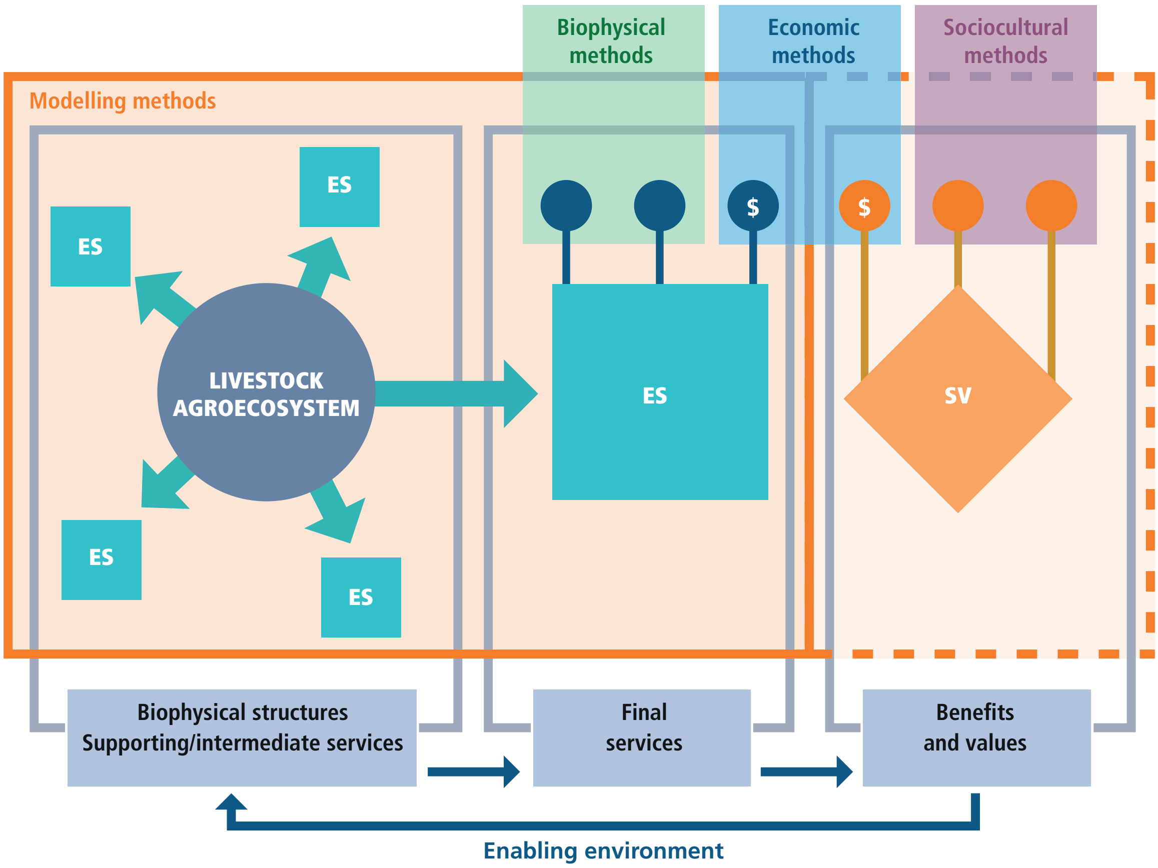
The final stage of the valuation process is to achieve the original purpose established in Step 1. This involves explaining how to implement the valuation results and transparently communicating the entire process that produced them, with particular emphasis on the role of livestock in delivering ecosystem services and the value of the services under consideration. For effective decision-making incorporating valuation findings, information must be communicated clearly and transparently.

Reporting should specify the livestock farming systems and ecosystem services examined in the valuation, as framed in Step 2, to highlight non-representative or specific aspects of the studied livestock agroecosystem, as well as those with potential for extrapolation. The process involves recognizing gaps and uncertainties in the valuation and acknowledging the potential effect of chosen methods on the relevance and reliability of the outcomes. These considerations were the focus of Step 3 and should be refined through practical implementation as described in Step 4.

Appendix 1 provides two examples of how ecosystem service valuation is performed in agroecosystems. These serve as illustrations of the application of Part 2 "Roadmap for the evaluation of ecosystem services from livestock agroecosystems," detailing the steps to design and conduct ecosystem service assessments in agroecosystems. They also demonstrate how individual methodologies described in Part 3 can be applied.

The two cases presented in Appendix 1 were developed by different authors. While both are in Latin America and

FIGURE 6
Conceptual framework of the guidelines, integrating the "cascade principle" with approaches of methods to assess ecosystem services in livestock agroecosystems



Note: The framework builds from the "cascade principle" described in section 1.3 and Figure 2. ES: Ecosystem services; SV: Social Values.
Source: Authors' own elaboration.

study pasture-based systems, they cover different regions, use different methodological approaches and address different steps in the valuation process. Case study 1 assesses pasture-based systems in Uruguay and illustrates the full assessment process encompassing the five steps defined in Section 2.1. Case study 2 assesses pasture-based systems in Brazil and demonstrates steps up to Step 3, focusing on the design and selection of methods (and indicators) to assess ecosystem services from agroecosystems.

2.2 CONCEPTUAL FRAMEWORK: LINKING CONCEPTS TO METHODS TO ASSESS ECOSYSTEM SERVICES

This chapter presents the overarching framework for these guidelines. The conceptual framework is graphically illustrated in Figure 6. Building on the concepts introduced in Part 1, the proposed framework aligns with the "cascade" model (Potschin and Haines-Young, 2011, 2016; see Section 1.3) but is simplified and adapted to the specific context of these guidelines. Furthermore, Figure 6 depicts how approaches and methods to assess ecosystem services in livestock agroecosystems relate to these concepts.

The starting point of the conceptual framework is a livestock agroecosystem (represented as a large circle in Figure 6). In the context of these guidelines, and as described in Part 1, the livestock agroecosystem is the ecosystem under study. In parallel with the "cascade principle" this is considered the environmental component, comprising biophysical structures, processes and functions. In summary, it corresponds to the supporting and intermediate services.

Key ecosystem services (indicated by rounded rectangles in Figure 6) originate from the management of the given livestock agroecosystem. These ecosystem services are regarded as final services, consistent with the "cascade principle" and correspond to the ecosystem services classified by frameworks such as CICES (CICES, 2025). The arrows represent the biophysical processes linking agroecosystem components and dynamics to the provision of these services. Up to this point, ecosystem services are delivered and can be measured in biophysical terms, using biophys-

TABLE 1
A key outlining the concepts and their symbols within the framework of ecosystem services assessment

Symbol	Concept, question, definition
LIVESTOCK AGROECOSYSTEM	Livestock agroecosystem *What system is under consideration?* Includes any type of farmed animal production agroecosystem, such as ruminants, monogastrics, aquaculture and insect rearing. Boundaries will vary depending on the point of application, e.g. livestock enterprise, farm, regional or national scale.
ES	Ecosystem service *What is the contribution of the livestock agroecosystem to human well-being?* Includes any contribution provided by the livestock agroecosystem to human well-being. These contributions should originate within the boundaries of the livestock agroecosystem, although they may also be perceived beyond those boundaries. Ecosystem services can be provisioning, regulating or cultural (see Background and Framework sections).
SV	Social values (society or individual) *What importance is assigned to a given ecosystem service by the whole society, a stakeholder group or an individual?* "...the moral principles and life goals held and expressed by individuals, groups and through the institutions (norms and rules) that guide people's interactions with nature and with each other…. [they] refer to how judgements regarding the importance of nature and its contributions to people are justified in 'specific' contexts" (Pascual et al. 2023). The social values are expressed with "value estimates" which stem from economic and/or sociocultural research methods.
	Ecosystem service indicator *Which variable will be used to quantify the ecosystem services?* An indicator of an ecosystem service is a variable that provides information on the provision of a specific ecosystem service. It can be measured, estimated or simulated using appropriate methods. Indicators may have quantitative or categorical values. Their values should be easily accessible through existing data, measurements, surveys or modelling simulations.
	Social value indicators *Which variables, themes or criteria quantify or qualitatively describe the social value assigned to a given ecosystem service by society, groups or individuals?* An indicator of social value is a variable, theme or criterion that can be measured, estimated, simulated or elicited. They can be assessed using existing data, surveys, interviews, focus groups, etc. Social value indicators can be qualitative or quantitative (including economic) in nature.

Source: Authors' own elaboration.

ical methods as well as modelling approaches. Modelling generally relates to understanding the mechanisms that generate and provide these services, but can also incorporate social values, including economic and sociocultural values.

Ecosystem services generated by agroecosystems then flow to society, encompassing public and private stakeholders with corresponding interests. According to the "cascade principle" ecosystem services enter the social and economic system and constitute benefits and values for society. In this framework, the flow of ecosystem services to society is represented as social values (SV), shown as a diamond in Figure 6. The concept of "social value" refers to "the values assigned to ecosystem services" by individuals, groups or society at large, which originate from moral principles and life goals (Pascual *et al.*, 2023). These social values are typically expressed in sociocultural and monetary terms and can also be modelled (as depicted in Figure 6).

It is important to note that "social values" do not refer exclusively to the common good for society; in livestock agroecosystems, there is substantial economic value for private stakeholders as well. However, public or private benefit should not be conflated with sociocultural and monetary valuation. Therefore, value assessments may need to address the distribution of different ecosystem services among diverse beneficiaries, such as private and public stakeholders.

Ecosystem services and social values require measurement through indicators (van Oudenhoven *et al.*, 2018). Indicators, represented by small circles originating from the ecosystem services (rounded rectangles) and social values (diamonds), denote measurable quantities that provide information about otherwise difficult-to-access aspects (Lebacq *et al.*, 2013); in this case, an ecosystem service or its associated social value. Defining or selecting suitable indicators is therefore fundamental. The suitability of an indicator depends on factors such as the study's purpose, data availability and resources (e.g. instruments, availability of expertise or stakeholder engagement willingness).

To properly frame research, readers are encouraged to consult Section 2.1, "Roadmap for the evaluation of ecosystem services from livestock agroecosystems"; for selecting methods based on requirements, Section 2.3, "Requirements to apply methods"; and for insight into individual methods, Part 3, "Methods."

Table 1 provides a schematic representation of the components (i.e. symbols) and concepts depicted in the framework proposed by these guidelines. It also includes guiding questions practitioners should consider when designing ecosystem service assessments in livestock agroecosystems.

Relating the framework to the methods for the evaluation of ecosystem services

As explained above, this conceptual framework is a simplified representation of the "cascade model" by Potschin and Haines-Young (2011, 2016) (see Section 1.1). The added value of our framework lies in the enhanced linkage between the generation of ecosystem services, the indicators and the main groups of methodologies used to evaluate ecosystem services.

Methods for ecosystem service evaluation correspond to different ways of assigning scores to ecosystem service indicators and/or the associated social values. Methods may differ in how they measure or model scores, their underlying hypotheses, the resources required and the extent to which they actively involve stakeholders.

Biophysical methods focus on estimating ecosystem functions in their natural units by assigning scores to ecosystem service indicators (Figure 7a). They accomplish this through diverse means such as direct measurement of indicators with appropriate instruments, elaboration of satellite data on vegetation and land cover, or remote and proximal sensing (see Section 3.1).

Sociocultural methods assign scores or qualitative descriptions to indicators of ecosystem service social values (Figure 7b). These methods are especially suited for intangible ecosystem services, such as cultural and aesthetic services, which are difficult to measure or model. This category encompasses a wide range of methods, many involving stakeholders individually (e.g. interviews, surveys) or in groups (e.g. focus groups). Others rely on quantitative or qualitative analysis of documentation or social media content (Section 3.2).

Economic methods address both ecosystem service and social value indicators by assigning appropriate monetary values (Figure 7c). This facilitates comparability across diverse ecosystem services. Economic valuation methods include changes in productivity, cost-based approaches, hedonic pricing, travel cost methods and contingent valuation (Section 3.3).

Modelling methods simulate scores of ecosystem service indicators (Figure 7d). These require mathematical functions linking driving variables within the agroecosystem to ecosystem service provision. Models can be conceptual, process-based, spatially implicit or explicit, or statistical (Section 3.4).

The following sections provide detailed descriptions of each family of methods, including specifications of individual methods.

2.3 COMPARISON OF METHODS BASED ON RESOURCE REQUIREMENTS

This section provides a straightforward comparison of the different approaches and individual methods addressed in this report (see Part 3). The approaches to assessing ecosystem services include biophysical, sociocultural, economic and modelling methods. Individual methods within each approach are summarized in Tables 2 and 3 below and are further developed and explained in their respective chapters in Part 3.

Other projects and initiatives have attempted to summarize similar information. A prominent example is the EU-funded OpenNESS project (Jax *et al.*, 2018), which proposed a decision-tree approach for selecting methods for ecosystem service assessment (see Harrison *et al.*, 2018).

However, the methods addressed in this report differ slightly, as do the considerations derived. The text below systematically provides insights for each approach to ecosystem service assessment, allowing comparison in terms of the spatial scales addressed, time and budget requirements, and necessary resources and skills. These considerations should be viewed as indicative. Depending on study design, context and existing resources, the applicability of these considerations may vary significantly.

For a more detailed comparison of individual methods regarding requirements necessary (marked with an X) or desirable (marked with an O) for method application, we provide overviews based on expert judgment in Table 2 (resources needed to apply the method) and Table 3 (skills needed to apply the method).

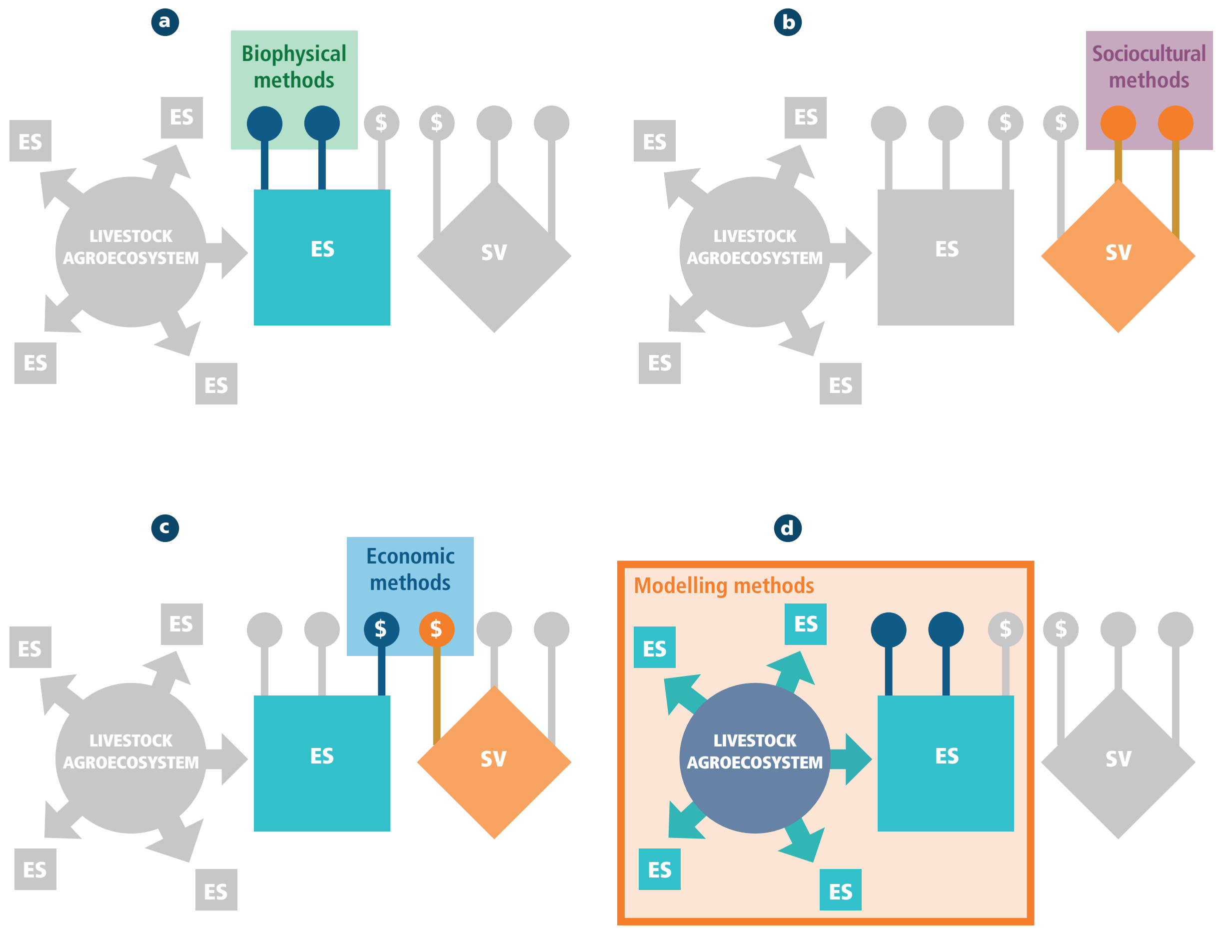
2.3.1 Considerations on biophysical methods

Spatial scales. Depending on the specific method, biophysical approaches cover different spatial scales. Direct measurement and indirect measurement based on proximal sensing are typically applied to relatively small areas (field to farm) with high spatial detail. In contrast, indirect measurements relying on benefit transfer and land cover proxies can be applied at all spatial scales, albeit with varying levels of uncertainty related to the spatial resolution of available data.

Time requirements. Time demands vary among biophysical methods. Direct measurements and proximal sensing methods can be time-consuming, often requiring on-site presence, equipment setup and data collection over multiple seasons or years. Conversely, methods using remote sensing or benefit transfer are generally less time-intensive, assuming relevant data are readily accessible. However, time requirements may increase if significant data processing, elaboration or curation – such as meta-analyses – are necessary.

Budget requirements. Budget considerations also differ. Direct measurements may incur costs related to personnel, equipment and data analysis. Benefit transfer methods may require specialized expertise, which can increase expenses. Remote sensing can be relatively cost-effective, but costs vary with data resolution and access fees. Proximal sensing, while becoming more affordable due to technological advances, still entails expenses, particularly for equipment acquisition.

FIGURE 7
Interrelations between the conceptual frameworks and the approaches of method



Notes: a) biophysical scores to ES indicators, b) scores or qualitative description to the ES SV indicators, c) assign economic value to both ES and SV indicators, d) model the scores of the ES, showing the performance of the agroecosystem analysed.

Source: Authors' own elaboration.

Necessary resources (see also Table 2). All biophysical methods require access to stakeholders and experts. Expertise is needed in all cases for operating equipment and for data acquisition and processing. Direct measurements and proximal sensing use or produce primary data, while remote sensing and benefit transfer rely on secondary data. Both remote and proximal sensing generally require specialized software or tools to process data and extrapolate the relevant ecosystem services. Equipment is essential for all indirect measurements.

Necessary skills (see also Table 3). Skills in quantitative statistics, data management and curation, coding, and trial and experimental design are highly desirable for implementing these methods. Specifically, quantitative statistics is essential for proximal sensing; data management and curation are required for remote and proximal sensing; coding is necessary for remote sensing; and trial and experimental design is required for direct measurements.

2.3.2 Considerations on sociocultural methods

Spatial scales. Sociocultural methods, by their nature of involving experts and stakeholders, are inherently well-suited for ecosystem service assessments at local scales, ranging from small territories to watersheds. These local scales are often well known by the stakeholders involved. However, these methods can also be applied at larger spatial scales, extending from regional to (inter)national levels. Techniques such as participatory mapping, social media analysis and Delphi methods enable engagement of stakeholders across broader geographic areas. It is important to note, however, that scaling up to larger territories may challenge the maintenance of the same level of accuracy.

Time requirements. The time needed to apply sociocultural methods depends on how easily stakeholders can be engaged and the mode of interaction. For individual stakeholder engagement (e.g. ranking, ratings, choice experiments, Q-methodology), arranging appointments can be relatively straightforward but requires significant time for preparation, conducting interviews and analysing qualitative data. Group discussion methods (e.g. deliberative multicriteria analysis, participatory mapping) pose scheduling challenges to maximize participation but reduce time for implementers by consolidating activities into fewer sessions. Timing between sessions is crucial, especially if multiple focus groups are conducted. Delphi methods, primarily online, require allowing experts ample time for thoughtful responses at each round. Planning for an appropriate response period is essential. Methods like content analysis and social media analysis do not require direct individual interaction. For content analysis, time depends on document accessibility and qualitative analysis requirements. Social media analysis is generally straightforward if data, software and tools are readily available.

Budget requirements. Sociocultural methods are generally considered cost-effective as they do not require expensive technology. Potential expenses include travel costs for researchers visiting various locations to engage with stakeholders or conduct fieldwork; venue rentals for focus groups or meetings (particularly in costly urban areas), participant remuneration for time invested, and software licensing or subscriptions for interview transcription, semi-qualitative analysis, or social media analysis. Budgeting for incidental expenses such as lunch or coffee breaks during extended stakeholder engagement sessions is recommended to foster a conducive discussion environment and acknowledge participants' time.

Necessary resources (see also Table 2). Access to stakeholders is fundamental for implementing sociocultural methods, except for social media analysis. Access to experts is often necessary, but less so for preference expression methods (rankings, rating, scoring, scaling, social media and content analysis). All methods generate or require primary data and are thus marked as necessary. Secondary data supports deliberative multicriteria analysis and Delphi methods and becomes essential for participatory mapping and scenario development. Specific software tools are generally not mandatory but may be required for rating, scoring, scaling, choice experiments, social media and content analysis (e.g. transcription of interviews, qualitative analysis, social media data extraction, or text analysis).

Necessary skills (see also Table 3). Required skills vary by method. Quantitative statistics are needed for rankings, rating, scoring, scaling, choice experiments, Q-methodology and deliberative democratic monetary evaluation, primarily for result analysis. Data management is useful for some methods. Coding is essential for social media and content analysis. Interview techniques and design are required for content analysis. Methods involving focus groups require group facilitation skills (e.g. deliberative multicriteria analysis, deliberative democratic monetary evaluation, participatory mapping, participatory scenario development). Questionnaire survey design is necessary for rankings, rating, scoring and scaling, choice experiments, Q-methodology and Delphi methods. Specific procedures and formats are important for each method.

2.3.3 Considerations on economic methods

Spatial scales. Economic methods can be a priori applied at any scale, provided that the necessary data are available and reliable. Methods requiring stakeholder interaction (e.g. contingent valuation, choice experiments) are generally better suited for territorial scales familiar to stakeholders but can also be applied at larger scales.

Time requirements. For methods not requiring stakeholder access, the time needed depends on data availability. When data are readily accessible, analyses can be completed relatively quickly if the method is well established. However, when data must be collected and compiled (maybe from multiple sources), time requirements are based on acquisition complexity. Meta-analyses require longer timeframes for both data collection and analysis due to the comprehensive research across sources and standardization of data formats. Methods involving stakeholder engagement (hedonic pricing, contingent valuation and choice experiments) require preparation time and individual interview sessions.

Budget requirements. Budget considerations are mostly influenced by data accessibility, as some sources may charge fees depending on institutional policies. Data analysis typically incurs minimal cost unless specific software licenses are needed. For methods involving stakeholders, travel expenses and provision of refreshments (e.g. lunch, coffee) during interviews should be budgeted as standard practice.

Necessary resources (see also Table 2). Access to stakeholders is essential for hedonic pricing, travel cost methods and contingent valuation. Expertise in economic theory, data analysis and result interpretation is crucial across all methods, necessitating the involvement of experts knowledgeable about the specific economic approaches. Primary data are required and produced by travel cost and contingent valuation methods, and may be relevant for production function, cost-based and hedonic pricing methods. Secondary data are essential for all methods except travel costs and contingent valuation. Familiarity with statistical techniques enables implementation on various open-source platforms, although specialized software may be advantageous.

Necessary skills (see also Table 3). Strong statistical proficiency is essential for all economic methods. Coding skills are beneficial, except for meta-analysis and value transfer methods. Data management and curation expertise are required for all approaches. Where stakeholder interaction occurs, interview techniques and survey design skills are necessary.

2.3.4 Considerations on modelling methods

Spatial scales. Modelling methods can be applied at any spatial scale depending on their formulation. For example, the Ecological Footprint method typically refers to scales ranging from country to global, but can also be applied to individuals or businesses.

Time requirements. Time needed for modelling involves model construction, model parameterization and validation, computational processing and result analysis. Model construction time is largely reduced when an established framework exists in the literature, such as emergy assessment (EMA), material flow analysis, ecological footprint, or life cycle assessment. Conversely, agent-based modelling, system dynamics modelling and telecoupling require conceptualization, construction and coding of models, including relevant rules and hypotheses, demanding more time and expert interaction. Parameterization is typically intensive across all modelling techniques, yet it also depends on data availability. Computational time varies with the number of model variables, simulation time horizon, and complexity of model rules. It can be particularly high for agent-based models with many simulated agents. Result analysis usually requires less time, though this depends on the model's use.

Budget requirements. Modelling methods are generally not considered expensive, except when specialized licensed software is required.

Necessary resources (see also Table 2). While access to stakeholders or experts is not strictly necessary for all modelling methods, maintaining communication with experts and stakeholders is important to ensure model accuracy and relevance. Expert input supports validation of assumptions, parameter value selection and scenario identification. Expert involvement is particularly critical for methods such as life cycle assessment, where specialist knowledge is essential for determining the parameter values. Primary data is not always mandatory, although it can often provide valuable insights if available. However, certain situations may require the collection of primary data to ensure accuracy and reliability. Telecoupling primarily relies on conceptual mind mapping and does not require extensive data inputs, whereas parameterization generally depends on secondary data to inform model parameters and ensure output accuracy. Certain modelling approaches – such as life cycle assessment, agent-based modelling and system dynamics modelling – require specialized software, while many others can be implemented with standard tools like Microsoft Excel or open-source coding platforms. No specialized equipment is typically necessary beyond a standard computer for modelling tasks.

Necessary skills (see also Table 3). Modelling requires proficiency in data curation and management, with strong statistical skills beneficial for result analysis and, in certain instances, inferring parameter values. Coding skills are advantageous, especially when models lack intuitive interfaces or require customization, and are essential for agent-based modelling due to model construction from scratch. Expertise in trial and experimental design is crucial for informed decision-making about model scenarios based on research questions. For telecoupling, which uses conceptual mind mapping rather than mathematical models, effective stakeholder engagement skills, including focus group facilitation, are important. Life cycle assessment may also require expert input for parameterization, making survey design skills beneficial.

2.3.5 Comparison of resources and skills required for each method

The two tables below provide an overall comparison of all methods in terms of the resources (Table 2) and skills (Table 3) necessary for implementation. For each method, an "X" indicates that the resource or skill is essential for implementation, while an "O" indicates that it is not necessary but still beneficial. Skills refer to those required by the people performing the method, e.g. as would be specified in a job announcement.

In Table 2, the following resources needed to implement each method are considered:

- **Access to stakeholders:** The possibility to engage relevant individuals or groups whose input is crucial for implementation. For example, in interviews assessing livestock ecosystem services, stakeholders may include farmers, landowners, conservationists and local community members.
- **Access to experts:** Availability of individuals with specialized knowledge or skills relevant to the method or system assessed. For instance, employing satellite imagery analysis requires access to experts in remote sensing or geographic information systems (GIS).
- **Primary data:** Original data collected specifically for the study. In livestock ecosystem service evaluation, this could include direct measurements of soil health, vegetation cover, animal populations or stakeholder preferences obtained via on-site surveys or field experiments.
- **Secondary data:** Existing datasets or information collected for other purposes but applicable to the current study. Examples include satellite imagery, agricultural records, previous research findings or environmental monitoring data.
- **Specialized software:** Software tools or programmes necessary for data analysis, modelling or visualization. Examples include open-source statistical packages like R, commercial software like SPSS or GIS software.
- **Specialized equipment:** Physical tools or instruments required for data collection or experiments. Examples include Global Positioning System (GPS) devices for geolocation, soil testing kits for assessing soil quality, proximity sensors or drones.

TABLE 2
Resources required (X) or recommended (O) for implementing each method

	Access to stakeholders	Access to experts on methods	Access to primary data	Access to secondary data	Software (specific tools)	Equipment
Biophysical methods						
Direct measurements, metrics and indicators using primary data	X[a]	X	X			X
Indirect measurements, benefit transfer using data and land cover proxies		X	O	X		
Indirect measurements, remote sensing	O[a]	X	O	O	X	X
Indirect measurements, proximal sensing	O[a]	X	O	O	X	X
Sociocultural methods						
Rankings	X	O	X			
Rating, scoring and scaling	X	O	X		O	
Choice-based approaches	X	X	X		O	
Social media analysis		O	X		O	
Content analysis	X	O	X		O	
Q-methodology	X	X	X			
Deliberative multicriteria analysis	X	X	X	O		
Deliberative democratic monetary evaluation	X	X	X			
Delphi methods	X	X	X	O		
Participatory mapping	X	X	X	X		
Participatory scenario development	X	X	X	X		
Economic methods						
Market price methods		X		X		
Production function methods		X	O	X	O	
Cost-based methods		X	O	X	O	
Hedonic pricing methods	X	X	O	X	O	
Travel cost methods	X	X	X	O	O	
Contingent valuation	X	X	X		O	
Choice experiment	X	X	X		O	
Meta-analysis		X		X		
Value transfer		X		X		
Transformation function method		X		X		
Modelling methods						
Emergy assessment	O	X	O	X	O	
Material flow analysis	O		O	X		
Ecological footprint	O	O	O	X		
Life cycle assessment	O	X	O	X	X	
Agent-based modelling	O	O	O	X	X	
System dynamics modelling	O	X	X	X	X	
Telecoupling	X	O	O	O		

[a] When dealing with livestock agroecosystems, some information will need to be collected or validated in private holdings. Hence, on occasion, it might be a must to count on stakeholders (i.e. land owners and/or managers). This may be particularly relevant when dealing with productivity parameters considered as provisioning ecosystem services.

Source: Authors' own elaboration.

In Table 3, the following skills needed for method implementation are considered:

- **Statistical analysis.** This involves the ability to analyse and interpret data using statistical techniques to uncover patterns, trends and relationships. For example, statistical analysis skills might include calculating averages, correlations, regression analyses, or conducting hypothesis testing to determine the significance of results.
- **Data management.** Data management refers to the ability to organize, clean, store and manipulate data effectively. This includes tasks such as creating

TABLE 3
Skills required (X) or recommended (O) for implementing each method

	Quantitative statistics	Data management and curation	Coding	Trial and experimental design	Interview techniques and design	Group facilitation	Questionnaire survey design
Biophysical methods							
Direct measurements, metrics and indicators using primary data	O	O	O	X			
Indirect measurements, benefit transfer using data and land cover proxies	O	O	O	O			
Indirect measurements, remote sensing	O	X	X	O			
Indirect measurements, proximal sensing	X	X	O	O			
Sociocultural methods							
Rankings	X	O			O		X
Rating, scoring and scaling	X	O			O		X
Choice-based approaches	X	O			O		X
Social media analysis		O	X		O		
Content analysis	O	O	X		X		O
Q-methodology	X	O			O		X
Deliberative multicriteria analysis	O					X	
Deliberative democratic monetary evaluation	X	O				X	
Delphi methods					O	X	X
Participatory mapping						X	
Participatory scenario development						X	
Economic methods							
Market price methods	X	X	O				
Production function methods	X	X	O				
Cost-based methods	X	X	O				
Hedonic pricing methods	X	X	O				
Travel cost methods	X	X	O		X		X
Contingent valuation methods	X	X	O		X		X
Choice experiment	X	X	O		X		X
Meta-analysis		X					
Value transfer		X					
Transformation function method							
Modelling methods							
Emergy assessment	O	X	O	X			O
Material flow analysis	O	X	O	X			
Ecological footprint	O	X	O	X			
Life cycle assessment	O	X	O	X			O
Agent-based modelling	O	X	X	X			
System dynamics modelling	O	X	O	X			
Telecoupling						O	

Source: Authors' own elaboration.

databases, data entry, data validation and ensuring data integrity. For instance, skills in data management might involve using spreadsheet software or database management systems.

- **Coding.** Coding involves writing and executing computer programmes or scripts to automate tasks, perform data analysis or develop models. This could include programming languages such as Python, R, MATLAB, or specialized software languages for statistical analysis or modelling. However, for some sociocultural methods, coding may refer to text analysis (e.g. documents or interviews).
- **Experimental design.** Experimental design refers to the planning and organization of experiments or studies to effectively test hypotheses and gather meaningful data. This includes designing protocols,

selecting variables, determining sample sizes and controlling for potential confounding factors.

- Interview techniques. Interview techniques involve the ability to conduct structured, semi-structured, or unstructured interviews to gather information from stakeholders or participants. This includes skills such as active listening, asking open-ended questions and probing for additional information.
- Group facilitation. Group facilitation entails the ability to lead and manage group discussions, workshops, or focus groups effectively. This includes skills such as agenda setting, managing group dynamics, fostering participation and synthesizing group input.
- Survey design. Survey design involves creating and implementing surveys to collect data from a sample of individuals or populations. This includes designing survey questions (including their order and formulation), selecting survey methods, and ensuring survey validity and reliability.

Part 3
Methods for ecosystem services valuation in agroecosystems

3.1 BIOPHYSICAL ASSESSMENT METHODS

3.1.1 Conceptualization

Biophysical methods aim to assess ecosystem services derived from the functioning of agroecosystems and express the estimation of these services in their natural units. Biophysical methods conceptualize nature's variables as stocks and flows of materials, organisms or energy (Chapin *et al.*, 2011), as well as information (e.g. genetic information). This includes ecosystem components (e.g. species diversity, soil organic carbon stocks), processes (e.g. hydrological and nutrient cycles, state-and-transition models) and interactions (e.g. the structure of food and/or pollinator webs) (IPBES, 2022).

Livestock agroecosystems focus mainly on the appropriation of photosynthesis products by humans through domesticated animals – in other words, the conversion of mainly inedible feed for humans into highly nutritious foods (see Section 1.2; Mottet *et al.*, 2017). Furthermore, certain livestock systems, such as agro-pastoral or pastoral systems, are recognized as complex agroecosystems. Complexity refers to the number of components (e.g. animals, forages, soil, manure, landscape elements) and processes to be considered, as well as the numerous interactions and functions among these components (Ominski *et al.*, 2021). These measurements include intermediate ecosystem services, such as soil organic carbon gains, which are essential to deduce final ecosystem services like carbon sequestration.

Given the extensive land use by most livestock agroecosystems and the diverse types of animal management worldwide, many functions require an accurate spatial and temporal representation. The spatial and temporal representation of ecosystem services (Maes *et al.*, 2012) demands detailed information about the studied indicators. For many analyses (e.g. soil organic carbon sequestration, recharge of the water table), indicators must be estimated over long periods with fine spatial and temporal resolution. These requirements often make direct estimation unaffordable from a practical perspective.

To overcome this limitation, it is common to represent needed indicators indirectly. One approach involves using benefit transfer data (see Section 3.1.2.2) or space-for-time substitution. Another approach combines primary data and functions, represented by models (see modelling methods for further details in Section 3.4), to describe specific ecosystem services. A specific case of this is the use of sensor data, whether remote or proximal.

This chapter describes methods applicable to estimating ecosystem service indicators in their natural units. The focus is on livestock agroecosystem service estimation, whether direct or indirect, while considering advantages and limitations in practical application.

3.1.2 Methods

Biophysical methods can be divided into two main approaches to estimate the biophysical supply of ecosystem services (Wong *et al.*, 2015). These methods can broadly be classified as direct, involving the collection of primary data on ecosystem service indicators, and indirect, involving the use of mainly secondary data on ecosystem service indicators. Regarding the latter, three main indirect approaches are presented: first, benefit transfer using secondary data and land cover proxies; second, the use of sensors, either remote or proximal; and third, modelling (Vihervaara *et al.*, 2018; Richter *et al.*, 2021).

In the following sections, we describe the direct approach and two of the indirect approaches. The modelling approach is not described in detail here as it is covered separately in the Modelling Methods chapter (see Section 3.4).

3.1.2.1 Direct measurements: metrics and indicators using ground reference primary data

Direct measurement methods of ecosystem services quantify the state, value or process through direct observations that represent the entire study area in a representative manner. These measurements can be applied at different scales and express ecosystem service values in physical units corresponding to their indicators, ultimately measuring a stock or flow value.

While biophysical measurements can be very accurate, specific intra- and extrapolation protocols are required to generalize point data, which is especially critical in heterogeneous landscapes. This often involves carefully designed trials and field experiments, such as the long-term Park Grass experiment (Silvertown *et al.*, 2006), advanced and costly technologies like Eddy-Covariance towers for monitoring GHG emissions (Charteris *et al.*, 2021), expertise in handling specialized equipment (e.g. insect traps, soil bulk density samples), sample analysis (soil chemistry, biomass or water quality indicators), and skills for managing, analysing and storing large datasets (McAuliffe *et al.*, 2018). Periodic or continuous direct monitoring of ecosystem service indicators is both costly and time-consuming. Although biophysical measurements can be highly accurate, generalizing results and comparing ecosystem services involving different units remains challenging.

Direct measurements serve as foundational references for calibrating and validating all indirect measurements. However, translating direct observations into indirect estimates is not straightforward. For example, the primary provisioning service from livestock systems – high-quality food from animals – is rarely observed at meaningful spatial and temporal resolutions. Consequently, at the global scale, data spatial resolution limits detailed examination of the heterogeneity in the provisioning of this service within countries or across different biomes and land uses, with few detailed exceptions (Raynor *et al.*, 2024; Gutierrez *et al.*, 2020; Irisarri and Oesterheld, 2020; Irisarri *et al.*, 2014). Addressing this limitation is a key focus area for comprehensive consideration of ecosystem services in livestock systems.

Confidentiality legislation presents a challenge by limiting the disclosure of individual-level information. To overcome these issues, promoting data sharing and publicly accessible datasets is essential. Specific working groups, including local specialists, could facilitate access to data not only at administrative boundaries but also at biome and land-use scales.

A few publicly funded initiatives actively address these limitations. For instance, the North Wyke Farm Platform and the Central Plains Experimental Range have provided essential open-access data. This data is used to monitor biophysical aspects of ecosystem services and also serves as input to process-based models (Wu *et al.*, 2016) or to analyse long-term grazing system variability and market volatility (Irisarri *et al.*, 2019; Baldwin *et al.*, 2022).

3.1.2.2 Indirect measurement: benefit transfer using data and land cover proxies

Benefit transfer refers to applying measured values from one place and time to infer values at another place and time (Plummer, 2009). To enable benefit transfer that meets decision-makers' needs, a comprehensive database of primary data – derived from multiple primary studies – must be created to develop "general" ecological production functions. These general functions can be obtained through meta-analysis (Brander *et al.*, 2012).

However, the lack of data or limited data accessibility undermines the usefulness of benefit transfer for timely assessments of ecosystem services at meaningful scales (Wong *et al.*, 2015).

3.1.2.3 Indirect measurement: remote sensing

Conceptually, satellite Earth observation is attractive for measuring ecosystem services because it enables periodic observations of the entire planet and has the potential to provide long-term datasets. The number of observations depends primarily on three factors: the spatial resolution of each observation (pixel size), the frequency at which each observation is collected, and the capacity to store and process the acquired data. Over the last 40 years, satellite missions have significantly improved the first two aspects. For instance, today's observations have a spatial resolution of 100 m^2 and a revisit frequency of every five days (e.g. Sentinel-2 satellite mission by the European Space Agency).

The third factor – data storage and processing capacity – constitutes a bottleneck for democratizing (i.e. making publicly available) satellite information. However, the most impactful democratizing change occurred with the launch of the Google Earth Engine platform (Gorelick *et al.*, 2017). This platform provides access to the world's largest catalogue of satellite images at no cost and enables the processing of satellite observations within the platform.

A key ecosystem function currently monitored through remote sensing is the Aboveground Net Primary Production (ANPP). ANPP is an integrative variable of ecosystem functioning (McNaughton *et al.*, 1989) that also determines the input level of many ecosystem services (Costanza *et al.*, 1997). For pastoral systems, ANPP represents the main driver of ruminant spatial density variations (Oesterheld *et al.*, 1992).

ANPP is the organic matter accumulated by plants at a specific time and surface (aboveground). ANPP results from the transformation of incoming photosynthetic active radiation (PAR) into biomass, which is mediated by the proportion of incoming radiation absorbed by active photosynthetic tissues (**fPAR**), which yields the amount of incoming radiation absorbed through photosynthesis (**APAR**) and transforms to biomass through the Radiation Use Efficiency (**RUE**) (Monteith, 1972). Remote sensing estimates reflectance in the red and near-infrared portions of the spectrum, enabling calculation of the widely used Normalized Difference Vegetation Index (**NDVI**) (Tucker, 1979). **NDVI** shows a strong, mostly non-linear, positive association with **fPAR** (Zeng *et al.*, 2022). In the special case of **ANPP**, **RUE** varies according to photosynthetic pathways, life form, and environmental factors such as water and nutrient availability or temperature (Druille *et al.*, 2019).

National initiatives have described the state of ANPP for grazing areas within their countries for the last two decades (see Box 2). These initiatives rely not only on remotely estimated ANPP but also on land cover classifications derived from remote sensing data.

Grasslands, as open agroecosystems (Bond, 2019), supply both provisioning ecosystem services (e.g. meat,

BOX 2

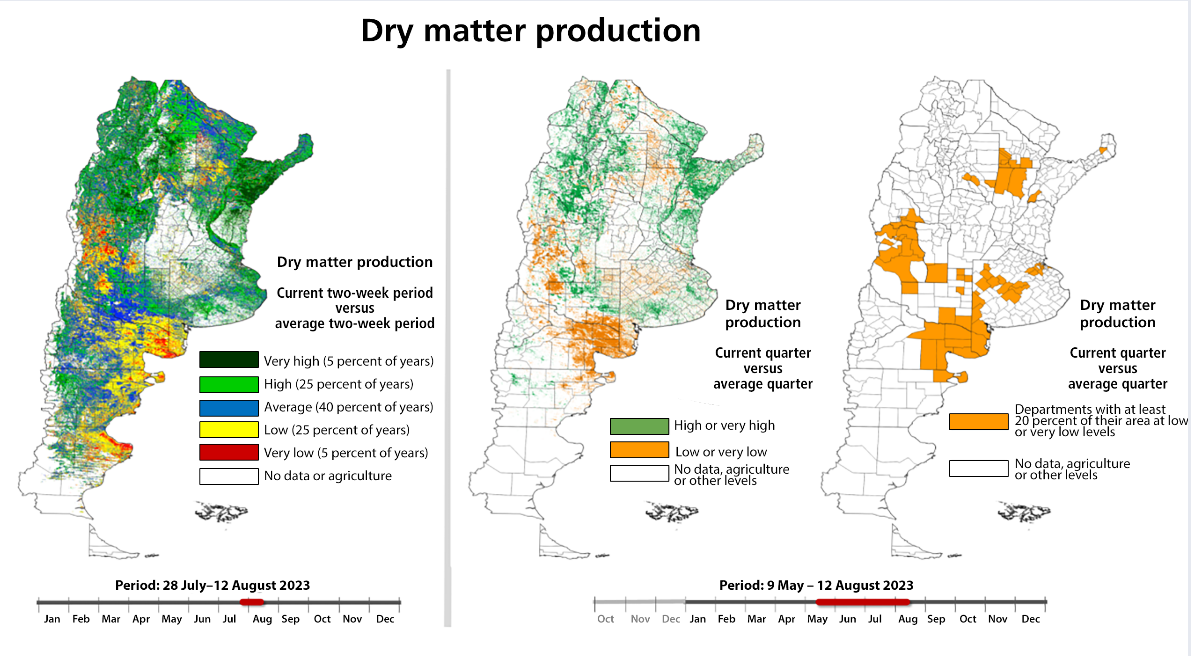
National forage production observatories

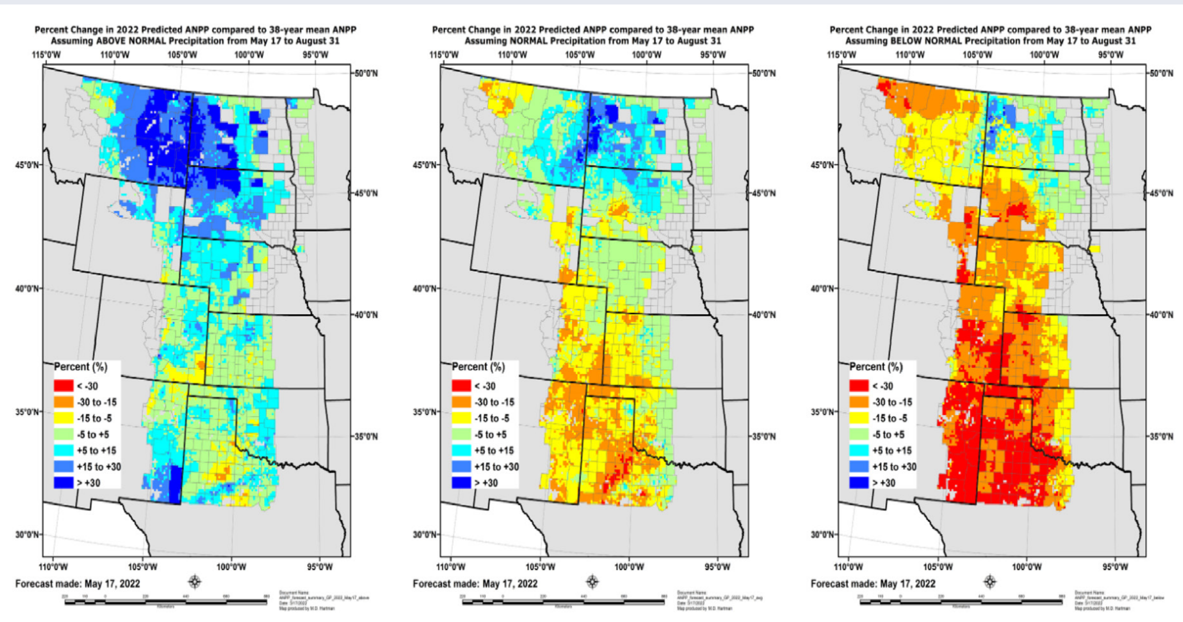
In Argentina (http://produccionforrajes.org.ar/), there is an initiative that reports the relative growth of vegetation in grazing areas every three months compared to the last 24 years. This information is derived from remote estimates of pasture growth.

In the United States of America (https://grasscast.unl.edu/), the scale of the initiative for the central plains surpasses that of the total area of Argentina and provides a forecast for the expected forage situation under different scenarios using a 36-year period. Three precipitation scenarios are considered (below average, average and above average) and describe the relative change in ANPP compared to the average for each scenario.

Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: Producción de Forrajes. (n.d.). Retrieved September 18, 2025, from http://produccionforrajes.org.ar/ and University of Nebraska–Lincoln. (n.d.). GrassCast: Grassland productivity forecast. Retrieved September 18, 2025, from https://grasscast.unl.edu/





BOX 3

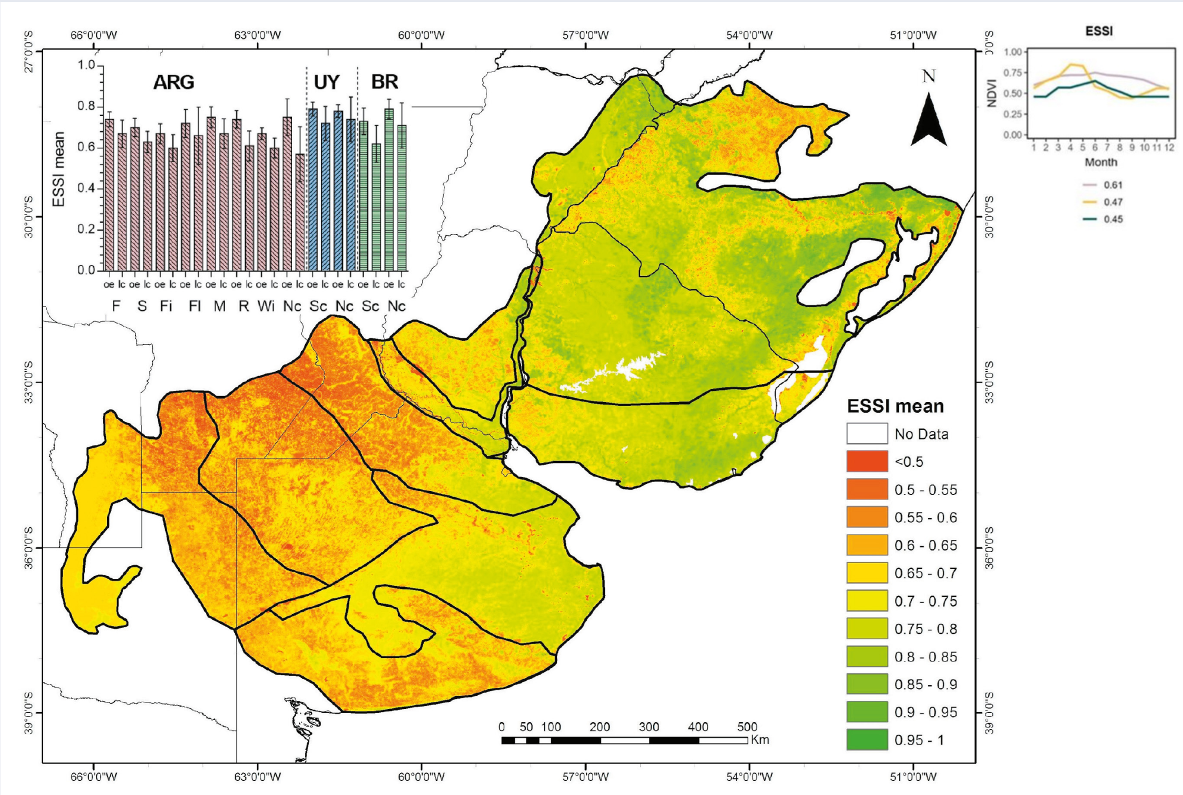
Remotely estimated synthetic ecosystem services indices. The case of the Ecosystem Services Supply Index (ESSI)

ESSI aims to integrate the seasonal phenological activity of actively growing vegetation through two components, NDVI's annual mean and its seasonal standard deviation (Paruelo *et al.* 2022). Consequently, as the annual mean reaches higher values and its seasonal variation lower ones, ESSI reaches its maximum.

ESSI = NDVIMEAN x (1 - NDVICV).

Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: Paruelo, J., Oesterheld, M., Altesor, A., Piñeiro, G., Rodríguez, C., Baldassini, P., … & Pillar, V. D. (2022). Grazers and fires: Their role in shaping the structure and functioning of the Río de la Plata Grasslands. Ecología Austral, 32(2bis), 599–820. https://doi.org/10.25260/EA.22.32.2.1.1880



wool or milk products) and regulating ecosystem services (e.g. carbon sequestration, water supply or habitats for biodiversity conservation). Given this dual role, there is a need for synthetic indicators that integrate these various services. Paruelo *et al.* (2016) proposed a synoptic indicator that groups supporting and regulating ecosystem services related to carbon and water dynamics: the ESSI (see Box 3 and the Case study 1 "Ecosystem services from pastoral livestock systems in Uruguay" in Appendix 1).

Originally, the ESSI was supported by positive relationships with four ecosystem services estimated from empirical data or mechanistic models: groundwater recharge and avian richness in Dry Chaco forests, and soil organic carbon in the Rio de la Plata Grassland (Paruelo *et al.*, 2016). ESSI represents a very attractive option to compare the supply of diverse ecosystem services across different land covers, over broad spatial areas and through long-term periods.

3.1.2.4 Indirect measurement: proximal sensing

The use of precision agricultural technologies, including proximal sensors for livestock, is rapidly developing and being deployed in both extensive and intensive agricultural ecosystems to provide key data in real time. Agriculture 4.0, characterized by the integration of digital technologies, is transforming the dynamics of near-time reporting of ecosystem services. Examples include livestock health and welfare monitoring through remotely monitored walk-over weighing (RMWOW) (Sawyer *et al.*, 2022), grazing pattern monitoring through global navigation satellite positioning systems (GNSS) (Bailey *et al.*, 2021), and nutritional observations via automatic feeders and greenhouse gas emission monitoring (Gonzalez *et al.*, 2018).

RMWOW allows measurement of liveweight (LW) and growth rate of animals in the paddock without the need for mustering (Sawyer *et al.*, 2022; Wishart, 2019;

González-García *et al.*, 2018; González *et al.*, 2014). The objective of RMWOW is to collect near-time data on animal weights while they drink or consume supplements, grain or preserved roughage. This is important for animal-related indicators associated with liveweight changes, including feed-to-meat conversion from LW gains, or female reproductive outcomes linked to significant LW loss. Liveweight monitoring can provide precise estimates of feed required by individual animals for growth – thereby improving production efficiency and reducing GHG emissions and emissions intensity (Gebbels *et al.*, 2022).

Global Navigation Satellite Positioning Systems (GNSS) have significantly enhanced farm data collection via electronic collars, ear tags and accelerometers. These technologies have fundamentally transformed farmers' and graziers' understanding of livestock interactions within their environments (Roberts, 2020). GNSS outputs enable livestock managers to locate animals in difficult terrain and weather conditions, track animal movement throughout the grazing season and improve handling practices (Bailey *et al.*, 2021). Specifically, GNSS sensor technology can map livestock distribution within a region, offering economic benefits by informing resource allocation, disease monitoring and land use planning (Manning *et al.*, 2017). It can assist in assessing grazing intensity, identifying overgrazed areas and guiding pasture management to prevent land degradation. Ultimately, this approach helps prevent ecosystem disservices and promotes delivery of desired ecosystem services, such as optimizing biomass utilization, enhancing carbon sequestration and storage, providing habitat or supporting nutrient cycling.

Automatic feeders and GHG emission monitoring allow better control of feed nutritional content and improve feed efficiency, ensuring animals receive appropriate nutrition to support growth and productivity (Gonzalez *et al.*, 2018). Efficient feeding can enhance digestion and reduce methane emissions from enteric fermentation (Gebbels *et al.*, 2022; Gonzalez *et al.*, 2023).

3.1.3 Advantages and limitations

The selection of the type and components of biophysical methods to quantify ecosystem services in livestock agroecosystems depends on the case, as highlighted in the roadmap for evaluation of ecosystem services (Part 2), and particularly the steps to undertake the evaluation (Section 2.1). The implementation of ecosystem services valuation methods is always limited by resource constraints. Consequently, the opportunity to reduce costs and provide continuous measurements across wide spatiotemporal areas has increased interest in applying remote and proximal sensing methods.

While these methods offer clear advantages, their development and application have not been without challenges. Here, we consider the pros and cons of various biophysical approaches to quantifying indicators, to support their selection and application. An overview of the main advantages and limitations of biophysical methods for ecosystem services valuation is presented in Table 4.

A key advantage of remote sensing, using archival data, is the possibility for time-series analysis. By monitoring spatiotemporal patterns, it is possible to evaluate the status, trends and impacts of ecosystem services (De Araujo Barbosa *et al.*, 2015). This information can be used at local to global scales to identify areas of change, hotspots of service provision, and potential risks or opportunities for management and conservation, thereby providing evidence-based information for decision-makers (Crossman and Bryan, 2009).

Time-series analysis also enables the detection and quantification of ecosystem services resilience. Temporal trends, spatial distribution and magnitude of ecosystem services provide key reference data. These data may be applied, for example, when assessing the consequences of specific policy implementations. De Araujo Barbosa *et al.* (2015) reported that 44 percent of studies using remote sensing for ecosystem service assessment had a temporal extent greater than ten years, and about half of these used observations extending over more than 20 years. As Earth observation data are collected from satellites typically operating for 20 to 30 years (Belward and Skøien, 2015), the ongoing provision of data services for estimating ecosystem service metrics needs to be considered.

As De Araujo Barbosa *et al.* (2015) point out, research efforts integrating Earth observation data into ecosystem services will need to ensure that temporal analyses and projected impacts are consistent. Furthermore, a sufficiently long time series is required to account for the effects of seasonal variability on ecosystem service metrics.

Measurement of ecosystem services is widely applied; however, these measurements are not always recognized under the term "ecosystem services" and span multiple scientific disciplines, including grassland sciences, animal sciences, soil sciences, biology and ecology, hydrology and climate sciences, among others. Consequently, maintaining a comprehensive overview of available data per ecosystem service is challenging. Moreover, such observations often become impractical and expensive beyond the site level. Nonetheless, these methods remain essential as inputs for biophysical mapping and modelling approaches or to validate certain mapping and assessment elements. In some cases, measurements are simply unavailable for all ecosystem services.

Remote sensing requires consideration of measurement error through standardized protocols, as well as adequate calibration and validation of ecosystem indicators. Sourcing required data is challenging for multiple reasons, such as i)

TABLE 4
Advantages and limitations of the different types of methods used for the biophysical assessment of ecosystem services

Method	Advantages	Limitations
Measurement	• High-quality data • Spatial extent closely managed • Gold standard for use to develop models	• Costly and time consuming • Spatial extent limited to local site • Limited opportunity for temporal analysis • Great variety of methods and indicators that might make comparability difficult
Indirect measurement. Benefit transfer using secondary data and land cover proxies	• Low-cost	• Assumes livestock agroecosystems behave in similar ways across space and time.
Indirect measurement Remote sensing	• Low-cost • In some cases, decades of historical data sets are available • Total global coverage	• Relies on third parties for access to data • Longevity of data streams determined by the life of the satellite • Atmospheric interference can limit the frequency of data availability • Needs broader skills to manage data
Indirect measurement Proximal Sensing	• Provides continuous data • End user has more control over the deployment and operation of the sensor • Hardware can be maintained or modified if required	• Limited coverage, only for the local area • Requires validation, possibly even for individual sensors in a sensor network due to variability in hardware • High-cost • Requires specific skills to manage the sensors and the data • Need for equipment maintenance

Source: Authors' own elaboration

estimating values beyond the temporal bounds of calibration data; ii) mismatches between plots and pixel scales; iii) available data not intended for remote sensing model calibration; iv) variable expert understanding of ground reference data; and v) limited ground reference data availability for validating estimated services across spatial and temporal resolutions (Senf, 2022). National initiatives, such as the North Wyke Farm Platform, the LTAR USDA-led network, and open long-term landscape-scale experiments, are valuable resources for ground referencing. However, their extrapolation is limited to the types of livestock systems in which they are implemented.

Ensuring data continuity, harmonization, and quality control across sensors and platforms remains a major challenge. Integrating data from multiple sensors and platforms to capture different ecosystem process scales is complex (Fritz *et al.*, 2019) – and harmonization efforts do not always improve product accuracy (Swinnen *et al.*, 2022). For example, merging data from Landsat, Sentinel-2 and MODIS sensors – each with different spatial resolutions (30 m, 10 m and 250 m, respectively) – requires resampling and interpolation, which may introduce uncertainties and distortions. Platforms such as Google Earth Engine provide tools to help overcome these challenges. Areas for improvement include atmospheric correction, which shows potential for enhancing data quality across sensors and spectral bands (Swinnen *et al.*, 2022).

The trade-off between cost and information accuracy in both assessing and economically valuing ecosystem services is evident and can threaten the robustness of policy advice and decision-making. Advances in data fusion techniques, machine learning algorithms, and cloud computing help mitigate these challenges and improve the accuracy and reliability of time series-derived ecosystem service assessments – thereby reducing the trade-offs between direct and indirect methods. We also recommend incorporating economic and cultural services valuation to generate more comprehensive outcomes.

Finally, two main points merit emphasis. First, the implementation of this or any other ecosystem services guideline will face budget constraints. Using open-source datasets and indirect indicators may help overcome such limitations, but these indirect indicators must be supported by extensive ground references for validation. Second, the main provisioning services of livestock systems – such as wool or highly nutritious food – cannot be estimated indirectly and will require other supply-chain-based data collection approaches. Furthermore, the spatial resolution, or "granularity" of available open information is too coarse for many key analyses. We strongly encourage the international community – including stakeholders interested in ecosystem service valuation and related domains such as food security and greenhouse gas emissions – to address this critical topic.

3.2 SOCIOCULTURAL VALUATION METHODS

3.2.1 Conceptualization

3.2.1.1 Sociocultural values and sociocultural valuation

Sociocultural valuation methods assess the values that people place on ecosystem services. Sociocultural values of ecosystem services can be defined as the importance that people (as individuals or groups) attach to ecosystem services (Iniesta-Arandia *et al*., 2014; Scholte *et al*., 2015; Walz *et al*., 2019). These values are based on a combination of instrumental, intrinsic and relational values that shape the way people relate to livestock agroecosystems, and, more specifically, to their economic, social and environmental components, and through them to the ecosystem services they provide (IPBES, 2022).

In the case of livestock agroecosystems, instrumental values refer to the material contribution of these agroecosystems to people (e.g. economic capital to farmers, food products to society, etc.). Intrinsic values refer to the values of livestock agroecosystems that are expressed independently of any reference to people and include entities such as habitats, species or livestock breeds that are worth protecting as ends in and of themselves. Relational values refer to the meaningfulness of interactions between people and livestock agroecosystems, and between people (including across generations) through these agroecosystems (e.g. sense of place, sense of belonging, spirituality, care).

It is important to clarify that sociocultural values are conceptually distinct from cultural ecosystem services. Cultural ecosystem services refer to "all the non-material and normally non-rival and non-consumptive outputs of ecosystems (biotic and abiotic) that affect physical and mental states of people" (Young *et al*., 2018), such as the aesthetic value of livestock agroecosystems as mountain pastures or the symbolic meaning that livestock have for pastoral communities throughout the world (e.g. Parel, 1969; Bettencourt *et al*., 2015; Quinlan *et al*., 2016). Sociocultural values refer to the importance people give to the benefits (both material and non-material) that derive from ecosystem services, which could be associated with the full set of provisioning, regulating and cultural ecosystem services.

In a broad sense, sociocultural valuation stems from social science and participatory methods, which are used for various purposes, including identifying and developing a ranking and/or rating of people's preferences for ecosystem services, analysing the underlying motivation of these preferences, or disentangling the relationships between different ecosystem services (e.g. Rodríguez-Ortega *et al*., 2014; Bernués *et al*., 2014; Garcia-Llorente *et al*., 2015; Faccioni *et al*., 2019; Gómez-Baggethun, 2016). The participation of the actual beneficiaries of the services – both local and distant – in their empirical measurement is a cornerstone of these methods (Scholte *et al*., 2015).

Sociocultural valuation is a heterogeneous collection of methods whose common feature is that they do not rely on biophysical or monetary metrics. These methods allow us to go beyond the simple association of ecosystem services values with money, markets and/or biophysical indicators, and provide an in-depth and comprehensive view of the plurality of perceptions and values that people attach to livestock ecosystem services. They provide contextualized quantitative and qualitative information on how different ecosystem services are interpreted and framed by individuals, stakeholders and communities, unravelling different perceptions between people and stakeholders with different values, interests, experiences and knowledge.

Sociocultural valuation methods are particularly suited to the valuation of intangible ecosystem services, allowing exploration of the contribution of livestock agroecosystems to society in terms of cultural, educational, spiritual or aesthetic values, among others. However, sociocultural valuation methods can be used to elicit any type of ecosystem services category. Indeed, one of their advantages is that they can assess the relative importance that people attach to different ecosystem services, thus providing robust information for decision-making processes to manage trade-offs between ecosystem services (Scholte *et al*., 2015).

Nevertheless, where biophysical quantification of ecosystem services (including non-provisioning) is feasible, it is usually more appropriate and accurate for measuring, benchmarking and monitoring the condition of livestock agroecosystems and ensuring their integrity (Bernués *et al*., 2014). However, in many cases (i.e. ecosystem service types and livestock production systems) and world regions, the data and sociotechnical and governance structures to generate biophysical or monetary measures are non-existent or very limited. In such cases, sociocultural valuations are the best option to fill the knowledge gap.

In many cases, different sociocultural valuation methods are combined with biophysical, monetary or modelling methods in valuation processes aiming at obtaining a comprehensive and complete view of the ecosystem services supply and/or demand, often with the ultimate aim of articulating ecosystem services values with decision-making.

3.2.1.2 Heterogeneity of sociocultural values and involvement of stakeholders

Due to the diversity of motivations, values and underlying worldviews, guiding principles and life goals that influence people's views and perceptions of ecosystem services, the value of ecosystem services provided by livestock agroecosystems may vary greatly between and within cultural traditions, socioeconomic systems, institutional settings, bio-geographical areas, interest groups, stakeholders and

BOX 4

Steps specific to sociocultural valuation processes

In addition to the general steps (see Roadmap section) for designing and implementing valuation processes in general, sociocultural methods have the following particularities, which should be integrated into the roadmap.

Defining the purpose of the valuation

In general, five main purposes are distinguished for ecosystem services valuation processes: i) evaluate (changes in) livestock agroecosystems management; ii) inform policy development; iii) assessment and awareness raising; iv) understanding ecosystem services dynamics; v) advance knowledge; and vi) inferring people's preferences.

In the specific case of sociocultural valuation, processes are to be established when there is particular interest in facilitating interaction and dialogue between stakeholders about conflicting issues regarding ecosystem services (IPBES, 2022), which could be critical for all purposes but particularly to evaluate policy implementation and to inform policy development.

Establishment of a legitimate valuation process

Given the participative and multistakeholder nature of sociocultural valuation, it is particularly important to build legitimate processes (IPBES, 2022). This will ensure transparency about the robustness of the valuation in terms of representativeness or participation. The first step to build legitimacy is to explicitly define the stakeholders involved in the process, taking into account who is affected by the ecosystem services provided by the studied livestock agroecosystem and who has the capacity to modify such agroecosystems and thus the provision of ecosystem services. Key stakeholders to be considered in most livestock agroecosystem valuation processes are farmers, residents of the region under study and social groups, such as tourists, consumers or environmental activists. It is also necessary to consider the degree of dependency of the stakeholder on these ecosystem services. Finally, it is crucial to make explicit the power relations between those involved in the process and how these might affect the outcomes of the process. Where power relations are considered to be relevant, policies and procedures should be in place to ensure that the process is inclusive.

Articulating the values in decision-making

In many cases, the final step of a sociocultural valuation process should be to inform the decision that originally led up to its implementation. The use of the process outcomes in decision making requires transparent communication of the results, in particular an acknowledgement of the limitations, omissions and uncertainties regarding their relevance and robustness, and how these might affect their applications. In addition to which specific ecosystem services have been targeted, it should be clear which stakeholders have been considered, and therefore which views are considered in the values elicited in the process.

Source: Authors' own elaboration.

individuals (e.g. Kaye-Zwiebel *et al*., 2014; Orenstein *et al*., 2014; Muñoz-Ulecia *et al*., 2022). Moreover, individual characteristics such as gender, age, education, environmental values and personal identity further influence how people within these contexts perceive and value ecosystem services (e.g. Zoderer *et al*., 2016; Mensah *et al*., 2017; Fortnam *et al*., 2019). As people's worldviews and experiences evolve, so too do the values they assign to ecosystem services, making valuation a dynamic and evolving process (e.g. Martín-López *et al*., 2009).

Despite this variability in ecosystem services values across people, when used in an appropriate context and within specified and clear limits, ecosystem services valuation can promote awareness of the dependence of society on ecosystem services and foster the deployment of political agendas or social movements for their protection and sustainable management (Gómez-Baggethun *et al*., 2016). This is particularly true in the case of livestock agroecosystems. The variability of values across stakeholders and temporal, geographical and sociocultural contexts poses some methodological challenges that are central to the use and interpretation of sociocultural method results. Addressing these challenges is critical to creating a robust and meaningful valuation process. The follow-up to the steps proposed below aims to address these challenges in a structured way.

Particularly, the type and quality of the information generated in the sociocultural valuation process will depend on the objectives and design of the process and, in particular, on the participants (or information considered in the case of media analysis) included in the valuation exercise. Participants involved in the analysis will define the stakeholders considered and, consequently, the extent to which the findings can be extrapolated to other geographical and sociocultural contexts. Participant sample representativeness is therefore central to achieving robust and reliable results in most sociocultural valuation exercises. Although participatory valuation rarely fulfils everyone's needs and points of view, it increases stakeholders' understanding of the positions and points of view of others. However, as Gómez-Baggethun *et al*. (2016) highlight, sociocultural valuation processes in general, and participatory methods in particular, when used out of context or with an approach that does not robustly capture how people value, or even perceive and define ecosystem services, stakeholders may bring their concerns into the valuation process. Hence, when applying sociocultural methods, it is recommended to take further consideration or steps than the ones defined in Section 2.1, Roadmap for the evaluation of ecosystem services (find the extra considerations/steps in Box 4).

Given the multiple stakeholders and interests that influence the condition, use and development of livestock agroecosystems, an essential outcome of sociocultural valuation processes is to identify shared group values and social values towards ecosystem services beyond the individual-specific values. While social values are understood in these guidelines as the set of values on which a particular society generally agrees, group values refer to the shared values within a particular group of people (usually stakeholders, such as livestock farmers) within that particular society. Sociocultural approaches can identify shared individual social values as well as group values. There are two main approaches to do this, depending on the nature of the information analysed (qualitative or quantitative). Firstly, individual values measured in qualitative terms can be aggregated into group values (i.e. stakeholders or interest groups), which in turn could be aggregated into social values, weighted according to some criteria defined for practical or representational purposes. Secondly, qualitative (or mixed) approaches can be used to identify collective and shared values following sociocultural valuation approaches such as deliberative methods.

3.2.2 Methods

There exists a myriad of sociocultural valuation methods and approaches, and consequently, several attempts have been made to sort and classify these methods into groups (e.g. Scholte *et al.*, 2015; Walz *et al.*, 2019; Barton and Harrison, 2017). In these practical guidelines, we address those methods that have been used extensively enough to value ecosystem services provided by agroecosystems and therefore have clear methodologies and sufficient examples to guide the reader in their design and implementation in livestock agroecosystems.

Methods are broadly grouped in these guidelines into: 3.2.2.1) Individual preferences assessments, which refer to quantitative structured processes and methods to elicit individual values from people expressing their preferences; 3.2.2.2) Narrative analysis, which gathers a range of methods that collect qualitative data from participants and are used to determine the core narratives regarding how they value ecosystem services; and 3.2.2.3) Deliberative processes, which bring together a diversity of stakeholders who engage in various modalities of collective discussions with the general purpose of reaching a common value.

3.2.2.1 Individual preference assessment

Individual preference assessment methods are widely used in many decision contexts beyond ecosystem services valuation (e.g. Burns *et al.*, 2022). This group of methods shares the common feature of employing structured processes to derive quantitative values reflecting people's perceptions and preferences towards ecosystem services. It is a useful approach for identifying relevant ecosystem services from various stakeholder perspectives (Garcia-Llorente *et al.*, 2012). This method can reveal differences and similarities in the values that different social groups place on ecosystem services provided by livestock agroecosystems.

Individual preference assessment methods can be categorized into ranking, rating, scoring and scaling, choice-based approaches and social media. The following subsections describe the main features and uses of each method.

Information on values is gathered by collecting people's statements through various tools, such as questionnaires and other data collection instruments (IPBES, 2022). Questionnaires are the most commonly used method for assessing individual preferences and can be administered face-to-face, by telephone, by email, by post or via the internet. At the core of the questionnaire is the assessment tool, which can be a ranking, rating, scoring, scaling exercise or a choice experiment, as described below. Questionnaires can be fully structured or semi-structured. The choice between these depends on the assessment objectives. Fully structured questionnaires offer limited flexibility and may not capture people's underlying motivations for valuing ecosystem services. Questionnaires can provide additional qualitative and quantitative information on factors of interest beyond the central assessment tool. This information can be statistically analysed and modelled to understand the drivers of people's values towards ecosystem services. However, it can be challenging to capture certain drivers of ecosystem services valuation – such as relational, instrumental or existing values – through questionnaires. Narrative analysis methods are better suited for analysing these aspects.

When designing questionnaires and survey processes to elicit sociocultural values attached to ecosystem services, it is essential to consider the technical aspects, challenges and limitations of questionnaire design to obtain reliable and meaningful results and to understand their interpretation and extrapolation. Special consideration should be given to the accuracy of statements used in conceptualizing ecosystem services, as well as the representativeness of respondents selected (IPBES, 2022).

To date, individual preference assessment in ecosystem services has mainly been used to raise awareness and, to a lesser extent, to set priorities and develop measurement tools (Walz *et al.*, 2019). There is a lack of standardized survey-based tools to measure how people value ecosystem services provided by agroecosystems, such as those existing in other disciplines (e.g. the New Environmental Paradigm Scale) (Dunlap *et al.*, 1978). The development of such a tool would promote comparability of ecosystem service values across agroecosystems, farming systems or geographical locations. In addition, it would provide benchmark information to monitor the evolution of ecosystem services provision and the impact of management and policy measures to improve it. However, creating such a tool poses significant challenges, primarily due to the difficulty of ensuring its relevance and applicability to each livestock agroecosystem of interest, while catering to the vast array of existing livestock agroecosystems and the particular combination of ecosystem services they might provide across different farming systems and geographical scales.

Rankings exercises

Ranking-based methodologies are commonly used to elicit ecosystem services values (Scholte *et al.*, 2015; Gómez-Baggethun, 2016; Walz *et al.*, 2019). Ranking exercises typically involve asking participants to order the importance they attach to a predefined list of ecosystem services. However, the use of simple rankings to generate robust quantitative values for ecosystem services is less advisable compared to other alternatives for assessing individual preferences, as the information produced is often too coarse. Specifically, rankings do not reveal the magnitude of the difference in importance between consecutive items (e.g. between 1st and 2nd, or 2nd and 3rd ranked ecosystem services). Additionally, if the list of items to be ranked is large, participants may find it challenging to provide an accurate and meaningful ranking of all items. Despite these limitations, rankings are extremely easy to design and implement. Therefore, they can be useful when combined with other methods. For example, rankings can serve as a preliminary step to select a subset of ecosystem services for more detailed evaluation – particularly when the subsequent methods, such as choice experiments, cannot consider the full range of ecosystem services provided by an agroecosystem.

Rating, scoring and scaling

Various methods for eliciting people's ratings, scores or scales of importance for ecosystem services are commonly used in ecosystem studies in general, and agroecosystems in particular. These methods provide more detailed information about the value of ecosystem services than simple rankings, as they generate scaled values for each specific service. The specific methodologies and scales employed are usually designed ad hoc for each valuation exercise and the particular ecosystem being assessed, which allows great flexibility to adapt to the context of interest. The drawback is that the comparability of values across different studies is often limited.

Choice-based approaches

Choice-based approaches involve multi-attribute valuation techniques that derive the relative utility of each attribute, based on Lancaster's theory of consumer demand (Lancaster, 1966). Similar to real-world decisions, respondents are asked to choose between a set of hypothetical alternatives defined by several variables at different levels. In ecosystem services valuation, the variables correspond to the different ecosystem services being evaluated, and the levels represent the degree of provision of these services. Respondents are expected to trade off the individual attributes (i.e. ecosystem services) of the alternatives and select the option providing the greatest utility (Horne *et al.*, 2005; Koemle *et al.*, 2020).

As with other individual preference assessment methods, choice-based approaches allow multiple ecosystem services of different types to be evaluated using the same unit of measure. However, the number of attributes assessed is generally limited to 5–7 to ensure the cognitive load on participants remains manageable (Caussade *et al.*, 2005; Hensher, 2006).

Although not strictly required, a monetary attribute is usually included to translate relative utility into monetary terms. This is particularly useful for valuing ecosystem services that lack a direct market value in monetary units. Since choice experiments are central to economic valuation methods, detailed methodological information is provided in the economic valuation chapter (Section 3.3).

Social media analysis

Social media is emerging as an important and useful tool and data source for the sociocultural valuation of ecosystem services (Velasco-Muñoz *et al.*, 2022; Hermes *et al.*, 2018). It uses individual data voluntarily provided on social media platforms to elicit values in the form of geolocated posts – that is, people's preferences can be inferred based on where they visit – and expressed ideas through textual, video, or photo content about key issues or events related to livestock. The opportunities and limitations associated with social media depend on the platforms used (Tian *et al.*, 2021).

Social media provides spatial data, if georeferenced, which can be used to map multiple dimensions of ecosystem services and preferences (Hermes *et al.*, 2018; Tian *et al.*, 2021). It can also assess cultural ecosystem services through sentiment analysis and other approaches (Retka *et al.*, 2019). Furthermore, social media offers a means to demonstrate revealed preferences, such as travel cost, which assist in valuation (Ghermandi, 2018; Tian *et al.*, 2021). It can be used to study the public's perceived value of various ecosystem services (Tian *et al.*, 2021).

Social media methods are useful for understanding ecosystem services provided by livestock agroecosystems, such as grasslands, particularly related to aesthetics and cultural ecosystem services, as well as flower presence and abundance, which in turn relate to various ecosystem services (Richter *et al.*, 2021). These methods generally involve working with geotagged textual or visual data (e.g. images) sourced from various platforms, enabling geographical analysis of ecosystem service provision. For example, Le Clec'h *et al.* (2019) use image data from Flickr to understand cultural ecosystem services in Switzerland's Canton of Solothurn grasslands. They use the number of images posted as an indicator of cultural importance (Le Clec'h *et al.*, 2019).

Potential sampling bias exists – for instance, data are limited to people using social media and the subset who choose to post – and there can be errors in interpretation. Enrichment by drawing on multiple data sources and analyses can help address such errors (Fox *et al.*, 2021). Social media data are also limited by the content provided online, in print, or through various other sources, which may or may not aim to document ecosystem services. Additionally, social media data availability may be geographically uneven, meaning these methods may not be applicable in all contexts.

3.2.2.2 Narrative analysis

Narrative methods encompass a range of social science techniques that collect qualitative data from participants to identify the core narratives around an issue of interest. In the field of ecosystem services, narrative analysis can be used to understand the meaning of livestock agroecosystems and their benefits to people (e.g. Openness Project Method Factsheet: Narrative Assessment of Ecosystem Services). These methods do not force participants to value ecosystem services within a predefined framework but allow them to freely articulate their values in their own words and worldviews (De Oliveira and Berkes, 2014; Satterfield, 2001).

Narrative analysis is usually employed to understand the underlying values and worldviews that shape people's perception of ecosystem services without requiring them to use the ecosystem services concept explicitly. Therefore, its application can improve understanding of why certain ecosystem services are important to people, reveal bundled cultural and social values associated with ecosystem services, and highlight hidden aspects of human–agroecosystem relationships that do not fit neatly within the ecosystem services framework (Klain *et al.*, 2014; Gould *et al.*, 2015). Narrative methods can be used to analyse any ecosystem service, but are particularly suited to studying cultural ecosystem services. They are generally applied to generate background and contextual information on stakeholder perceptions of ecosystem services to inform the development of more qualitative valuation or deliberative processes (see Section 3.2.2.3).

Narrative analysis employs tools from ethnographic, anthropological and social science research, such as in-depth and semi-structured interviews, voice and video recordings of events or artistic expression. In ecosystem services research, interviews are the most commonly used method to collect information about participants' experiences, views and beliefs. Sometimes interviews precede quantitative techniques and may generate individual information as part of individual preference assessment or inform deliberative processes.

Interviews tend to be time-consuming; however, new technologies such as mobile phones and online meeting platforms have greatly reduced time demands, particularly by eliminating travel time required for face-to-face interviews. While there are no specific guidelines for developing interviews focused on narrative analysis of ecosystem services, Young *et al.* (2018) developed guidelines centred on conservation research that can be useful for those interested in applying interviews in the analysis of ecosystem services provided by livestock agroecosystems.

Participants' responses to interviews can be analysed using structured semi-quantitative methodologies, with content analysis being among the most prominent. An alternative to structured interview-based narrative analysis is Q-methodology. Both content analysis and Q-methodology help identify existing narratives on ecosystem services and determine their relative importance (e.g. Pike *et al.*, 2015; Hermelingmeier *et al.*, 2017), or develop typologies of stakeholders according to their views on ecosystem services (e.g. Bredin *et al.*, 2015; Hubatova *et al.*, 2023), among other applications.

Content analysis

Content analysis is a social science research method to study any text (including recorded discourse) to identify patterns in ideas and narratives in a replicable and systematic manner. It entails reading texts systematically to pinpoint meaningful content units, which are then labelled or coded and analysed to reveal patterns of occurrence or relationships among codes, either quantitatively or qualitatively. This approach typically requires the transcription of interviews and is increasingly common because computer-assisted coding greatly reduces the workload. Various computer programmes automate text coding to varying extents and generally offer analytic methods for the resulting codes.

Content analysis has most often assessed stakeholder awareness of ecosystem services provided by livestock agroecosystems (Tauro *et al.* 2018) and their relation to agricultural practices (Bernués *et al.* 2016). Other applications include analysing narratives about ecosystem services in policy documents (e.g. Leduc *et al.* 2021; Maczka *et al.* 2016) to explore how concepts and, in particular, valuation methodologies have been embedded in policy-making and regulatory reforms. Similarly, analysing news media can capture the broad framing of ecosystem services within a society or location and typically yields spatial data useful for sociocultural valuation studies (Duffy *et al.* 2020). The most widespread new media analysis application aims to understand public perceptions and framing of ecosystem services (McLellan and Shackleton 2019; Weber *et al.* 2017; Martin and Doucet 2022; Lyytimäki 2014), which can inform valuation studies or infer different value orientations. However, so far, news media analysis has not been applied to the valuation of ecosystem services provided by agroecosystems. Finally, narrative analysis may be applied to social media data – for example, to assess users' experiences, preferences and values regarding different places or agroecosystems.

Q-methodology

Q-methodology provides a structured, replicable approach for analysing social perspectives on ecosystem services, thereby enhancing the reliability and credibility of narrative analyses. Originally developed in psychology and subsequently adopted across the social sciences, Q-methodology has also found application in environmental studies focusing on social perceptions.

This method involves the sorting of statements – reflecting individuals' perspectives and attitudes towards ecosystem services – according to a defined protocol; the resulting data are then subjected to factor analysis. The initial identification of narratives, which are transformed into the statements to be evaluated, typically draws on stakeholder surveys (e.g. Hubatova *et al.*, 2023), focus groups (e.g. Ciftcioglu, 2020) or literature reviews (Hermelingmeier *et al.*, 2017). A key strength of Q-methodology is its combination of quantitative and qualitative data and analytical techniques (Zabala *et al.*, 2018), offering a middle ground between the structure of surveys and the depth of interviews, and combining advantages of both.

Beyond the identification of ecosystem services, Q-methodology has been applied to practical contexts, such as determining preferences for farming practices and their relation to ecosystem services (Berg *et al.*, 2023), or disentangling value justifications and stakeholder narratives in cases of conflicting views on ecosystem services management to inform policy design (Maniatakou *et al.*, 2020). Zabala *et al.* (2018) offer comprehensive guidelines for conducting a Q-study, from design through to interpretation, with a focus on conservation studies that can be readily adapted to analyse ecosystem services in livestock agroecosystems.

3.2.2.3 Deliberative valuation processes

Single valuation methods are often insufficient to address the diverse and complex array of ecosystem services provided by livestock agroecosystems and to account for the needs and perspectives of all stakeholders involved – both in provision (e.g. land managers or livestock farmers) and demand (e.g. consumers of livestock products and services, locally and beyond). Deliberative valuation processes borrow methods from the political and social sciences and are founded on active stakeholder engagement, typically integrating knowledge systems, disciplines, diverse tools and data to elicit shared social values, views and opinions (Christie *et al.*, 2012; Spash, 2007). These processes provide a platform for stakeholders to gain insight into each other's worldviews and appreciate different perspectives, promoting social learning, conflict resolution and the development of socially robust policies (Murphy *et al.*, 2017; Huitema *et al.*, 2010). Through deliberation and discourse (Kenter *et al.*, 2011; Mavrommati *et al.*, 2017), stakeholders or experts actively engage to reach consensual decisions, facilitating the transition from individual to shared social values (Mavrommati *et al.*, 2021). Typically, deliberative methods yield group preferences that deviate consistently from those derived by simply aggregating individual preferences (Murphy *et al.*, 2017).

Deliberative valuation encompasses a suite of methods whose common feature is the engagement of stakeholders in open discussions to elucidate the values of ecosystem services. These processes are typically designed with the practical aim of informing decision-making related to the management of ecosystem services derived from livestock agroecosystems. Deliberative valuation employs different methods at distinct stages to guide the process toward its objectives, integrating a range of complementary valuation tools that often extend beyond traditional sociocultural approaches.

Typically, both the process design and selection of valuation methods are tailored to the specific objectives of each study. While well suited to identifying and valuing cultural ecosystem services, deliberative methods are equally effective for measuring stakeholder values for any other ecosystem service, enriching biophysical assessments and economic valuations (Slovák *et al.*, 2023). Method integration can range from simple combinations of two or three techniques to complex multilayered processes that consider varying geographical scales or future scenarios. The steps outlined in Box 4 are particularly relevant for structuring deliberative assessments. A common combination of methods consists of using some kind of narrative analysis – such as in-depth interviews with key stakeholders – to identify relevant ecosystem services, which are then used to inform the design of a statement-based assessment that generates qualitative value data (e.g. Bernués *et al.*, 2014). More comprehensive deliberative processes may involve one or multiple focus groups, enabling iterative discussion and consensus-building among participants.

Focus groups are a type of group interview that specifically incorporate group interaction as an integral aspect of the methodology. They are mainly structured in different moments with one or more facilitators. It is often necessary to involve stakeholders more than once along the deliberative process, focusing on different aspects and with the opportunity to elaborate on feedback from previous focus groups. During focus groups, participants have the opportunity to ask questions, share anecdotes and comment on each other's experiences and perspectives. Interaction may occur in plenary or subgroups, often supported by graphical or computer-based tools – such as mind maps or other graphic aids – which serve as intermediary objects to integrate diverse viewpoints through graphical representations. Graphical tools are particularly helpful in facilitating group discussions and bringing together stakeholder perspectives. For example, Ryschawy *et al.* (2019) introduced "the Barn", an integrated graphical tool that provides a visual representation of the ecological and socioeconomic aspects of livestock farming – highlighting multiple services and impacts across different systems. This tool showcases the diversity of livestock farming, fosters knowledge sharing and exchange of viewpoints, and aims to explore more sustainable options. Dernat *et al.* (2023) later transformed the Barn into a serious game. Although the tool was developed for broader livestock systems rather than exclusively for ecosystem services, it can easily be used to facilitate the deliberative valuation of ecosystem services provided by livestock agroecosystems.

There are a variety of ways in which different types of focus groups and other forms of interaction can be combined (Slovák *et al.*, 2023). To give an example, Shipley *et al.* (2020) organized a focus group after the implementation of a Delphi survey. How methods are combined to advance the deliberative process often depends on the particular context, the available resources, the local culture and other contingencies, such as the possibility of bringing stakeholders together.

A key challenge of deliberative methods – common to all group-based approaches (Schaafsma *et al.*, 2018) – lies in their complex organization. Securing the involvement of key representatives from all relevant stakeholder groups can be difficult. Moreover, because outcomes largely depend on who participates, a careful and context-sensitive selection of participants is essential. It is also important to acknowledge and address power relations that may exist between the participants before, during and after the deliberative process. In group discussions, dominant individuals or stakeholder groups can bias outcomes by marginalizing other voices (Dietz *et al.*, 2009). Therefore, process design should include strategies and tools to ensure that all participants' views are heard and considered (Barnaud and Van Paassen, 2013; Felt *et al.*, 2016; Turnhout *et al.*, 2020).

Furthermore, clear classification of deliberative methods is challenging given the diversity of techniques, variable reporting standards and highly context-specific designs. These guidelines do not aim to catalogue every method, but Table 5 presents six examples of widely used, relatively standardized approaches for ecosystem services valuation.

Deliberative multicriteria evaluation (DMCE)

Deliberative multicriteria evaluation integrates multi-criteria decision analysis with stakeholder deliberation. Multicriteria decision analysis explicitly considers multiple criteria to support complex decision-making. In the context of ecosystem services, this approach provides a framework for collectively assessing and valuing multiple, sometimes competing objectives – enabling participants to explore and represent trade-offs among ecosystem services (Mavrommati *et al.*, 2017; *et al.*, 2021). In the study by Mavrommati *et al.* (2021), participants were recruited and invited to a one-day workshop. During the workshop, scientists presented information on the scientific aspects of the ecosystem services under consideration. Participants then individually assigned preferences to predefined scenarios in which ecosystem services were delivered in different ways. In a subsequent phase, participants discussed their preferences together, guided by a facilitator and with the opportunity to seek additional information. The process concluded with the elicitation of individual values and the calculation of shared group values based on these inputs.

Deliberative democratic monetary evaluation

These evaluation methods aim to elicit social preferences and monetary valuations through structured deliberation. Deliberated preferences represent an advanced iteration of evaluation, they build on stated-preference techniques – such as contingent valuation and choice experiments – by embedding collective deliberation into the valuation process. The deliberative democratic monetary evaluation method (Orchard-Webb *et al*., 2016) provides a platform for participants to debate the benefits and costs of policy options related to ecosystem services, while also incorporating non-instrumental concerns, including deontological motivations – such as social norms, rights and duties – and virtues like fairness and responsibility, thus accommodating plurality and incommensurability of values.

Often, participants deliberate in small groups to discuss policy scenarios and allocate hypothetical budgets or payments across ecosystem services, guided by facilitators who ensure that non-economic values are voiced and considered. This approach yields both deliberated monetary values and a richer understanding of the ethical and social dimensions underpinning stakeholder preferences.

Delphi method

The Delphi technique is an iterative consultation method in which a group of experts or knowledgeable participants contribute information or judgements to build collective knowledge until convergence of opinions is reached. It may be employed alone or combined with other approaches within a broader deliberative process. Typically, individuals are asked in an iterative process to rate various aspects of interest. After each round, respondents receive an anonymized summary of the group's responses and the reasons underlying their judgements. Participants then revise their ratings, which may or may not change. Typically, respondents' answers converge during the iterative process. The process repeats until a predefined stopping criterion, such as a set number of rounds, consensus reached or stability of responses is met.

The Delphi method is widely used across disciplines, including medicine, sociology, policy analysis, ecology and conservation, and other environmental sciences (Mukherjee *et al*., 2015; Okoli *et al*., 2004). In ecosystem services research, it has been applied to elicit valuations in agroecosystems (Rositano and Ferraro, 2014; Shipley *et al*., 2020; Larsen *et al*., 2019). Within livestock agroecosystems specifically, Delphi approaches have assessed the effects of management practices on ecosystem service delivery in a variety of systems, such as mountain livestock agroecosystems and Mediterranean silvopastoral systems, Alpine permanent grasslands and semiarid rangelands (Lecegui *et al*., 2022; RodríguezOrtega *et al*., 2018; Mack *et al*., 2023; Azimi *et al*., 2020).

Participatory mapping

Deliberative or participatory mapping is an approach that integrates scientific information with stakeholder input to produce spatially explicit maps representing ecosystem services (Palomo *et al.*, 2013). This method is particularly useful for generating spatialized information on ecosystem services when objective spatial data are not fully available, especially for cultural ecosystem services. It can be used to map not only the supply of ecosystem services, but also their demand and associated social values. Stakeholders, selected to ensure diverse perspectives and based on their expertise and knowledge, engage in interactive workshops where, with appropriate support, they collectively produce maps by combining their knowledge, experience and preferences with scientific data. This process serves as a platform for capacity building and for incorporating experiential knowledge into the mapping process (Sieber, 2006). Traditionally, the mapping of ecosystem services has predominantly focused on the supply side, often neglecting societal demand and values. Using the deliberative mapping method, Palomo *et al.* (2013) aimed to explore the most significant ecosystem services associated with protected areas.

Participatory scenario development

Participatory scenario planning is a means of developing a range of plausible future trajectories of systems through the engagement of diverse actors within a given system (Oteros-Rozas *et al.*, 2015). Participatory scenarios are particularly useful for understanding processes of change, both historical and future, from the perspective of local people (Malinga *et al.*, 2013). They stimulate dialogue among participants about the nature of change in ecosystem services and provide insights into the multiple values that exist within a system (Oteros-Rozas *et al.*, 2015; Palomo *et al.*, 2011). Participatory scenario approaches can complement other scenario development methods, including those based on modelling or other analytical techniques (Oteros-Rozas *et al.*, 2015).

3.2.3 Advantages and limitations

In Table 5, the advantages and limitations of the sociocultural methods are presented. These are reported by groups of methods (i.e. individual preference assessment, narrative analysis or deliberative valuation methods). In general terms, the advantages and limitations apply to the methods described within each category. However, there may be additional considerations for a few specific individual methodologies.

3.3 ECONOMIC VALUATION METHODS

3.3.1 Conceptualization

Economic thinking about ecosystem services and their appropriate valuation follows a utilitarian framework. Accordingly, people assign value to ecosystem services based on direct use values through consumptive use (for example, consuming food products) or non-consumptive use (such as recreational use or landscape aesthetics), indirect use values (such as goods used as intermediates in production), or option values, by maintaining the possibility of using goods in the future (see also Section 1.1 of this report for further detail). Finally, people may also derive utility from non-use values, that is, from simply appreciating the fact that the ecosystem service exists, even if they never make use of it (Alcamo, 2003).

When applying these concepts to agroecosystems, farmers may derive some non-use values from ecosystem services arising during production (Hansson and Lagerkvist,

TABLE 5
Advantages and limitations of the different types of methods used for the sociocultural assessment of ecosystems services

METHODOLOGIES	Advantages	Limitations
INDIVIDUAL PREFERENCES ASSESSMENT	• Relatively simple setup with a list of distinct options • Easily understood by the participants and allows engagement with a large number of people • It can provide robust quantitative information (from a sample of people) • In the case of social media, it is easy to collect readily available data in some locations	• Highly demanding in personnel • They cannot capture underlying motivations for making certain decisions (sometimes, extra qualitative information is needed to understand the reasons behind the responses given) • Strong skills in questionnaire and survey design to avoid common limitations. Reliability of statements is a concern, as is the representativeness in the selection of respondents • Potential sampling bias
NARRATIVE ANALYSIS	• Does not force research participants to value ecosystem services within a given framework • It is suited for a large variety of stakeholders, regardless of their background and knowledge • Enables exploring bundles of ecosystem services	• Highly sensitive to interview biases • Implementing the field work and preparing, analysing and interpreting the data is very time-consuming and resource intensive. • Results are contextual; extrapolation outside the case study is limited
DELIBERATIVE VALUATION PROCESSES	• It can incorporate in the same analysis a diversity of ecosystem services, including those that are hard to measure quantitatively (notably, cultural ecosystem services) • A diverse group of stakeholders reaches a shared understanding of the system and gains insight into the worldviews and needs of others. • It can be tailored to specific contexts or situations.	• It is somehow difficult to involve all the relevant stakeholders and bring them together in the same place and at the same time • Power relations among participants can bias the outcomes of the process by excluding the perspectives of less dominant participants. • Difficult to replicate, with a high risk of inconsistency across different methods; outcomes often depend on contextual factors and the skills of the facilitator.

Source: Authors' own elaboration

2016). However, they arguably place greater weight on use values, particularly on marketable products, as these represent the ecosystem services that translate into farm revenue and, consequently, farmers' income. Several studies have explored how farmers and non-farmers (such as Bernués *et al.*, 2016), or local and distant citizens (such as Bernués *et al.*, 2014), recognize and assign value to ecosystem services, particularly in relation to their use and non-use values.

Moreover, many ecosystem services are considered public goods that are non-rivalrous in consumption (such as landscape aesthetics) and not traded in markets. As a result, these services tend to be under-provided, and the socially optimal quantity is not achieved (see also Lankoski, 2003). In summary, traditional agricultural markets experience market failure because only a portion of ecosystem services, those with direct consumptive use, are marketed and generate revenue. The others are typically neglected and undersupplied. This situation may be addressed if non-marketed ecosystem services are remunerated through, for example, payments for ecosystem services or agri-environmental schemes (see also Antle and Stoorvogel, 2006; and Section 4.2 of this report).

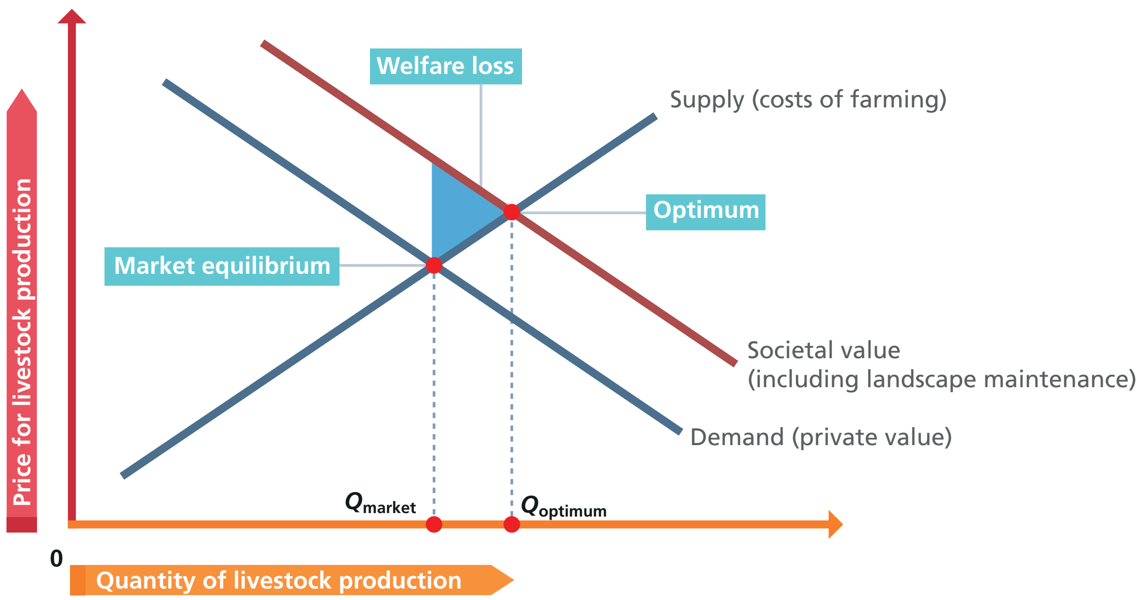
3.3.1.1 Ecosystem services as positive externalities

The failure of markets to support the provision of ecosystem services can be framed as a classical economic problem of positive externalities. This occurs whenever the actions of one economic agent (e.g. a farmer) directly affect another agent outside the market mechanism (e.g. society). A primary example in the context of livestock farming is the maintenance of natural or traditional landscapes through livestock grazing. In this case, the production efforts of the livestock farmer improve societal well-being, yet the farmer is not compensated by those who benefit. In other words, the societal value of grazing livestock exceeds the private value derived from livestock farming. As a result, the market mechanism leads to a lower-than-optimal provision of grazing livestock.

Such positive externalities are illustrated in Figure 8. In the figure, the blue lines define the traditional market equilibrium (Q_{market}), where the transaction of private goods (i.e. livestock products) occurs at a given production cost. However, this equilibrium does not account for the delivery of positive externalities, such as landscape maintenance. The optimal quantity of production – for example, through grazing livestock – would be higher.

These positive externalities are represented by the red line in Figure 8. The delivery of such externalities (i.e. landscape) would come at a higher cost of farming. Consequently, a new equilibrium ($Q_{optimum}$) would be established, where production costs would be higher, but the price of the products would also increase. Since traditional markets fail to reach $Q_{optimum}$, there is no incentive to provide positive externalities, which results in a welfare loss for society. This welfare loss is represented by the triangle between private demand and societal value in the figure.

FIGURE 8
Positive externalities from livestock farming, as defined by means of a utility function



Source: Authors' own elaboration.

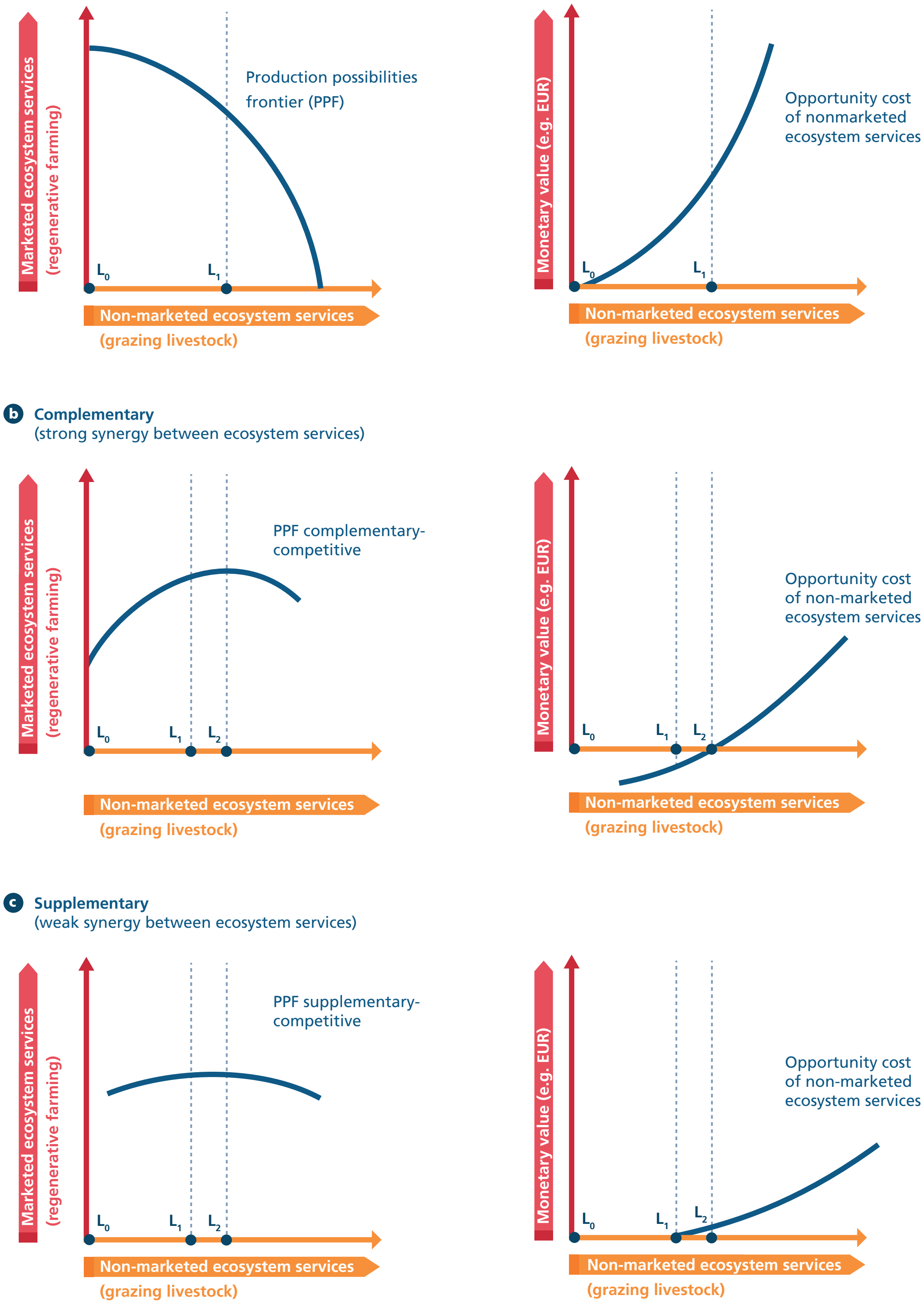
To achieve the socially optimal level of grazing livestock, the welfare loss would need to be compensated. This could be done, for example, by offering compensation payments to livestock farmers that incentivize them to provide a higher quantity of grazing services (see also Section 4.2 of this report for further detail).

Hence, a primary objective of economic methods is to demonstrate the importance of ecosystem services for society and human well-being. This generally implies determining the gap between the supply of ecosystem services (that is, those derived from market mechanisms) and the demand, or desired level of supply, of ecosystem services by individuals, groups of individuals or society at large. It also involves identifying the resulting loss of human welfare or well-being caused by the mismatch between the supply and demand of non-marketed ecosystem services, and deriving potential compensation mechanisms to incentivize their provision.

For illustration purposes, we introduced the concept using the classical example of grazing livestock and its contribution to human well-being, specifically through increased welfare resulting from the provision of aesthetically pleasing landscapes. However, the concept includes several important nuances that must be considered.

These include: i) determining which livestock farming systems contribute to human well-being through the generation of positive externalities, and identifying the specific externalities involved (see Part 2 of this report for further information). While the example focuses on landscape provision, the same concept can apply to other ecosystem services such as pollination, habitat provision, or carbon storage and sequestration; ii) determining the level of farming intensity that delivers the optimum amount of service. In livestock farming systems, both excessively high and overly low production intensity may degrade or undersupply the desired level of service. Therefore, to deliver positive externalities, incentives may need to support the maintenance of certain low-intensity production systems, for example, to prevent the abandonment or intensification of traditional systems, or to promote destocking to reduce production intensity; iii) Identifying the beneficiaries of those externalities. While beneficiaries are often referred to generically as "society", they may include a wide range of actors. These can include local or distant populations (Bernués *et al.*, 2016) or businesses. For instance, some farmers may create conditions that enhance pollinators, thereby generating benefits for other farmers who do not contribute to pollination services and who do not compensate those who do. Similarly, businesses may benefit from a well-maintained landscape by attracting clients, such as in the tourism sector or the hospitality, restaurant and catering industry, without contributing to the upkeep of the farming systems and agents, such as farmers, that shape those landscapes; and iv) designing appropriate compensation and incentive mechanisms to reward the delivery of positive externalities (see further insights in Section 4.2 of this report).

FIGURE 9
Potential production relationship between ecosystem services, portrayed as competitive



Notes: (a) Showing a trade-off; portrayed as complementary (b), Showing a synergy; or portrayed as supplementary (c), Showing a weak relationship. With PPF: production possibilities frontier, and L:ecosystem services from livestock farming.

Source: Adapted from Sauer, J., & Wossink, A. 2013. *Marketed outputs and non-marketed ecosystem services: The evaluation of marginal costs.* European Review of Agricultural Economics, 40(4): 573–603. https://doi.org/10.1093/erae/jbs040

3.3.1.2 Interrelationships between ecosystem services

Interrelationships among ecosystem services are discussed in general terms in Section 3.5.1, Assessing trade-offs and synergies in ecosystem services. However, this topic is also central to the field of economics and is therefore addressed here from an economic perspective. In economic science, welfare economics and production theory have shaped the way relationships between different ecosystem services are described and measured (see, for example, Wossink and Swinton, 2007; Smith *et al.*, 2013; or Bekele *et al.*, 2013).

The set of production possibilities refers to all combinations of ecosystem service levels that can be "produced" (that is, provided or accommodated) within a specific spatially defined area or system (for example, a landscape), given its natural resource endowments and the human and technological inputs available (such as labour, machinery and information technology). At the boundary of this production set, known as the production possibilities frontier (PPF), it is no longer possible to increase the provision of one ecosystem service without reducing the provision of another. This reflects a Pareto-optimal combination of ecosystem service levels.

Following this logic, economists conceptualize the provision of multiple services from the same ecosystem as a multiproduct scenario, in which trade-offs arise because different services compete for the same limited resources (for example, land). To measure such trade-offs empirically, economic approaches have explored the relationships between marketed and non-marketed ecosystem services. For instance, Sauer and Wossink (2013) distinguish between competitive, complementary and supplementary relationships in the joint production of marketed and non-marketed ecosystem services. As these relationships change, the marginal costs of providing the combined set of ecosystem services also change.

However, it is important to note that the concepts of jointness in production (in functional terms) and of complementarity, competitiveness or supplementarity (in terms of the production possibility set) are not mutually exclusive. For example, the provision of a specific service, such as landscape maintenance, may follow one type of product–product relationship, while each farm may still provide a bundle of services that cannot be produced separately.

In the context of livestock farming, significant biophysical trade-offs may occur. For example, there may be trade-offs between regenerative forms of farming that generate marketed outputs, or between carbon sequestration as a marketed service on the one hand, and the non-marketed service of landscape maintenance through grazing on the other. The economic implications of such biophysical trade-offs can be analysed and evaluated at the plot, farm or more aggregated level.

Figure 9 illustrates these relationships, including potential trade-offs and synergies between marketed and non-marketed ecosystem services in livestock farming. In panel (a), a marginal increase in landscape maintenance beyond level L_1 enhances output from regenerative farming. In panel (b), the reallocation of agricultural inputs to marginally increase landscape maintenance through grazing beyond L_1 has a direct negative effect on regenerative farming output, but a compensating positive indirect effect through the enhancement of regulating ecosystem services. Panel (c) presents the case where reallocation of productive inputs is not feasible without a net loss in marketed output from regenerative farming.

These cases of production relationships between ecosystem services can be empirically identified for a sample of livestock farmers by quantifying the opportunity costs of marginal changes in ecosystem service provision. Both farmers and policymakers are likely to be most interested in cases (b) and (c), where marginal increases in ecosystem service provision through livestock production occur without opportunity costs. In other words, the increased provision of non-marketed ecosystem services (for example, from level L_1 to L_2) is achieved without reducing the supply of marketed goods, and therefore, no monetary loss occurs.

This type of evidence can also be used to evaluate the cost-effectiveness of existing agri-environmental or other green payment schemes. A marginal increase in the provision of ecosystem services through livestock grazing is costly for the individual farmer if they are under-compensated for the income foregone from regenerative livestock farming. In contrast, livestock producers who are over-compensated may, at the margin, be able to provide more ecosystem services through grazing without incurring additional costs.

Based on this empirical classification of livestock farms, further investigation into the farm and production characteristics of each group can provide valuable insights to enhance the cost-effectiveness and economic efficiency of payment schemes. This may include offering regionally differentiated payment rates or tailored scheme options.

It is worth noting that the examples provided here (in relation to Figure 9) are simplified for illustrative purposes. For instance, the figure refers broadly to "grazing livestock" without identifying all the individual ecosystem service indicators associated with grazing. These relationships may vary depending on which ecosystem services are considered, whether individually or as part of a bundle. To better harness these interrelationships for financial incentives or improved management practices, it is important to integrate as many relevant ecosystem services as possible into the analysis.

3.3.2 Valuation methods

Ecosystem service valuation refers to the process of assigning appropriate monetary values to place ecosystem services of different natures and forms on a common scale. Depending on the extent to which use and non-use values are associated with an ecosystem service, suitable valuation methods must be selected. These methods vary in their underlying assumptions, data requirements and the reliability of the estimates produced (see also Alcamo, 2003).

Valuation methods that rely on observed market prices are suitable for ecosystem services associated with use values. These approaches are often referred to as "revealed preference" methods because valuation is derived from directly or indirectly observing how market participants value ecosystem services, either through real or surrogate markets. For example, prices for consumer goods are usually directly observable in existing markets. When the ecosystem service in question is the marketed good itself, its value is determined by its market price.

If an ecosystem service is only one component of a marketed product's total value, hedonic pricing models can be used to decompose the product's price into the values of individual attributes, including environmental characteristics. For ecosystem services that serve as inputs in the production of marketable goods, the production function approach can be applied. This method estimates the change in productivity resulting from the loss of the service as an input (Barbier, 2007).

Cost-based approaches estimate the expenses required to replace the ecosystem service using alternative technologies. For instance, these may include the cost of water treatment plants in areas where ecosystems no longer provide groundwater of sufficient quality, or the cost of flood protection infrastructure in the absence of natural barriers such as mangrove forests. Manual pollination (labour) in the absence of natural pollinators is another example (Barbier, 2007). The travel cost approach estimates the value of ecosystem services based on the expenditures individuals are willing to incur to enjoy them, such as visiting national parks for recreation.

Contingent valuation methods are used for valuing ecosystem services that reflect non-use values. Since conventional or surrogate markets do not exist for these services, hypothetical markets are created to assess the preferences of potential beneficiaries. One such method is the choice experiment, in which respondents are presented with a selection of hypothetical products that differ in their attributes. By analysing respondents' choices using regression-based techniques, it is possible to estimate their willingness to pay for specific attributes, including those related to ecosystem services.

3.3.2.1 Market price-based valuation

Market price-based valuation can be used for ecosystem services for which actual markets exist. These services are typically associated with direct use values and, more specifically, represent provisioning services. Examples include agricultural products that are traded on markets, or entrance fees to national parks that reflect their recreational value. According to market theory, well-functioning markets arrive at prices that reflect both marginal production costs and consumers' willingness to pay. This approach, therefore, makes use of the directly observable behaviour of buyers and sellers (Alcamo, 2003). The main limitation of this approach is that it cannot be applied to many ecosystem services that are not privately owned and for which no markets exist. In addition to the complete absence of markets, distortions such as subsidies, taxes or other imperfections may lead to mispricing. In such cases, adjustments to the estimated ecosystem service values may be required (Atkinson *et al.*, 2014).

3.3.2.2 Production function method

The production function method evaluates the value of an ecosystem service based on its role as a production input. This approach is particularly applicable to services that contribute positively to production without being directly purchased on input markets. Such services typically have indirect use values, for example, regulating services. In this sense, the production function method is a market-based approach, as it assesses the contribution of the ecosystem service to the production of a marketed good. The value of the ecosystem service is derived from the market value of that good. The service can be assessed by incorporating it into a production function estimation, examining its role in enhancing output – by increasing the productivity of other inputs – and valuing the resulting increase in output using prevailing market prices (Barbier, 2007). Conceptually, the value of the ecosystem service is reflected in higher productivity, which leads to lower average production costs. These cost savings are then passed on as increased surplus for either producers or consumers (Pascual *et al.*, 2010). One of the main drawbacks of this approach is its demanding data requirements. Sufficient data must be available to capture production outcomes under varying levels of ecosystem service provision to estimate its marginal contribution accurately.

3.3.2.3 Cost-based method

The cost-based method is another form of market-based valuation that uses market values to indirectly estimate the value of an ecosystem service. Specifically, it is examined what costs would be incurred if the ecosystem ceased to provide the service in question. The value of the ecosystem service can then be expressed as the direct costs resulting from the loss of the service, the costs required to replace the service through artificial means, or the costs necessary to restore the ecosystem's functionality after it has been impaired (Pascual *et al.*, 2010). Common examples include ecosystems that purify water for human consumption, compared with the cost of artificial water treatment if that natural service were no longer available (Alcamo, 2003). A central criticism of this method concerns its underlying assumption that costs are equal to benefits. This assumption may not be valid in all contexts, leading to potentially misleading valuations. As a result, researchers have recommended using this approach with caution (Barbier, 2007).

3.3.2.4 Hedonic pricing

The hedonic pricing method is primarily used to assess the value of ecosystem services that directly influence market prices. A common example is housing prices, which often reflect the value assigned to local environmental attributes. The method is based on the assumption that the price of a marketable good correlates with its characteristics, allowing the estimation of how much people are willing to pay as those characteristics vary. Implementing the method requires cross-sectional or time-series data on both the marketable good and the associated ecosystem service. Using regression analysis, it is then possible to estimate the extent to which changes in the ecosystem service influence the price of the good. Most applications of this method use residential housing prices to estimate the value of environmental amenities. In the agricultural sector, hedonic pricing is used to estimate how land prices change in relation to their production potential, reflecting environmental and agronomic factors that affect land productivity.

3.3.2.5 Travel cost method

The travel cost method is used to capture the use values people attribute to specific sites that provide recreational services, such as public parks or natural areas. The method is based on the assumption that the time and travel expenses people incur to visit a site reflect the recreational value they assign to it. Both secondary data on site visits and survey data from visitors can be used to derive a demand curve for the site. The quantity of the environmental good demanded is typically a function of variables such as price, visitors' income and socioeconomic characteristics. The price that respondents are willing to pay is usually calculated as the sum of the site's entry fee, the cost of travel and the opportunity cost of time spent reaching the location. The consumer surplus associated with the demand curve provides an estimate of the value of the ecosystem service offered by the recreational site under investigation.

3.3.2.6 Contingent valuation

The contingent valuation method is a stated preference approach used to estimate economic values for various ecosystem and environmental services, encompassing both use and non-use values. The method involves directly eliciting individuals' willingness to pay or willingness to accept compensation for an increase or decrease in some non-marketed good or service. In a typical contingent valuation survey, respondents are presented with a scenario describing a specific ecosystem good or service, and they are asked about their willingness to pay or accept to achieve the conditions described. The method's reliance on stated preferences, rather than revealed behaviour, has drawn some criticism due to its dependence on respondents' subjective opinions and their ability to accurately assess the hypothetical scenarios presented to them (often referred to as hypothetical bias).

3.3.2.7 Choice experiments

A choice experiment is a survey approach designed to elicit individual preferences within hypothetical markets. Respondents are presented with different bundles of goods, each described by its attributes or characteristics and the corresponding levels. One of these attributes typically includes either the price paid (to assess willingness to pay) or the payment received (to assess willingness to accept). In the context of environmental valuation, respondents are asked to choose among different options representing environmental services and goods. Grounded in the assumption that individuals will select the alternative offering the highest utility, the method enables the estimation of economic values for each characteristic of the goods, including those related to the ecosystem services under analysis. This experimental approach facilitates the identification of trade-offs that individuals make between different attributes and alternatives.

3.3.2.8 Meta-analysis

Meta-analysis is an established method for quantitatively analysing a large body of existing information to provide clear guidance and insights for decision-making and policy formulation. In the case of environmental valuation, a meta-analysis allows for a systematic review and aggregation of economic values for multiple ecosystem goods and services from a range of studies applying different valuation techniques. The construction of an econometric model enables the investigation of the effect of different context-specific independent variables on the economic value of ecosystem services. By analysing and combining data from multiple studies, a meta-analysis provides a more comprehensive and generalizable understanding of the economic values associated with environmental resources and ecosystem services.

3.3.2.9 Value transfer

Evaluating a policy or management change in terms of its impact on the aggregate value of ecosystem services presents the challenge of valuing multiple ecosystem services

TABLE 6
Skills required (X) or recommended (O) for implementing each method

	Direct market methods[a]			Indirect market methods[a]		Non-market methods[a]	
	MP	PF	CB	HP	TC	CV	CE
Provisioning services							
Food	X	X				X[b]	X[b]
Freshwater	X	X				X[b]	X[b]
Wood and fibre	X	X				X[b]	X[b]
Fuel	X	X				X[b]	X[b]
Regulating services							
Climate regulation			X			X[b]	X[b]
Flood regulation			X	X		X[b]	X[b]
Disease regulation	X		X			X[b]	X[b]
Water purification	X	X	X			X[b]	X[b]
Cultural services							
Aesthetics				X		X[c]	X[c]
Recreation				X	X	X[c]	X[c]
Education						X[c]	X[c]
Spiritual						X[c]	X[c]
Supporting services							
Nutrient cycling	X	X	X				
Soil formation	X	X	X				
Primary production	X	X	X				

[a] MP = market price method; PF = production function method; CB = cost-based methods; HP = hedonic pricing method; TC = travel cost method; CV = contingent valuation method; CE = choice experiment method;

[b] Although markets exist for most provisioning and regulating services, valuation of hypothetical changes and changes that have not yet taken place can only be done through non-market valuation methods;

[c] For most cultural services markets do not exist, implying that the valuation of both current and hypothetical situations and changes can only be done through non-market valuation methods.

Source: Adapted from Koetse, M.J., Brouwer, R. & Van Beukering, P.J.H. 2015. Economic valuation methods for ecosystem services. In J.A. Bouma & P.J.H. Van Beukering, eds. *Ecosystem services*, 1st ed., pp. 108–131. Cambridge University Press. https://doi.org/10.1017/CBO9781107477612.009

in a specific context. While not a stand-alone method, value (or benefit) transfer addresses this challenge by applying results from past studies to the context being researched. For example, the recreational value estimated for a national park (the "policy site") may be transferred to a similar national park under study (the "study site"). Although it may seem like a makeshift solution, value transfer can be cost-efficient and sometimes the only feasible option for many research projects, especially when a comprehensive assessment of multiple ecosystem services is required. There is a growing consensus that value transfers are valid if adjustments are carefully made between the original data source and the target context (Alcamo, 2003; Pascual *et al.*, 2010). Nonetheless, possible valuation errors may arise from differences between study contexts and insufficient adjustments. Sources of error include variable returns to scale when ecosystems of different sizes are compared, spatial decay when the distance between ecosystem services and beneficiaries varies, or differing income levels when socioeconomic contexts diverge (Pascual *et al.*, 2010).

3.3.2.10 Transformation function method

Finally, the transformation function method aims to evaluate product relationships, that is, trade-offs and synergies, between ecosystem services – both marketed and non-marketed. This method is similar to the production function method described above, as it evaluates the value of an ecosystem service in terms of its role as a production input, compensation payment received or marketed output. A transformation function represents the output that can be produced from a given input base under existing conditions, which also defines the feasible production set (see Sauer and Wossink, 2013, or Felthoven and Morrison-Paul, 2004). As such, this method is a market-based or policy-based approach and is data-intensive.

3.3.3 Advantages and limitations of economic valuation methods

As described above, various valuation methods exist and have been applied in numerous studies to estimate monetary values of different ecosystem services. These methods

TABLE 7
Advantages and limitations of the different types of methods used for the economic assessment of ecosystem services

	Advantages	Limitations
Market price method	• Easy to apply as it makes use of generally available information on prices, quantities and costs • Simple modelling and few assumptions • Uses data on actual consumer preferences, in contrast to non-market methods that use data on stated consumer preferences	• Many environmental services are not traded directly on markets • If markets for environmental services do exist but are highly distorted, the available price information will not reflect true social and economic values • Cannot be easily used to measure the value of larger-scale changes that are likely to affect the supply and demand for a good or service
Production function method	• In theory, well suited to value ecosystem services since it is based on the notion that ecosystem services and economic benefits are strongly linked	• Technically difficult to apply in practice • Substantial data requirements • Market value of other inputs in the production process needs to be considered • Limited to valuing those ecosystem services that can be used as inputs in production of marketed goods
Cost-based methods	• Relatively simple and inexpensive to apply (do not require the use of detailed surveys or complex analysis) • Provide surrogate measures of value that are as consistent as possible with the economic concept of use values, for services that may be difficult to value by other means • Integrate well with the types of economic analysis commonly used in practice	• Do not produce a strictly correct measure of economic value • Measures produced are not based on people's preferences for the ecosystem service, but on the assumption that if people pay to replace a lost ecosystem service or avoid damages, then that service must be worth at least the cost of replacement or damages avoided • Since the reduction measures will in most cases only reduce part of the negative spillovers experienced, the estimates produced by these methods likely understate the true values • Often difficult to find exact replacements for ecosystem services that provide an identical level of benefits • Subject to circularity: if a service is assumed to be unimportant and little or no replacement or avoidance costs are incurred, its estimated value will be low • Tend to yield inconsistent value estimates, as higher levels of ecosystem quantity or quality often entail increasing marginal costs for further improvements
Hedonic pricing method	• Estimation is based on data from an implicit but real market • Relatively inexpensive to apply if data is already available	• Estimates reflect willingness to pay for a non-marketed commodity specific to the site under investigation only • Possible omitted variable bias, especially in hedonic house price functions where it may be difficult to control for all factors influencing property prices • Multicollinearity – when independent variables in a valuation model are highly correlated, making it difficult to isolate the effect of each variable on the estimated value • Market segmentation – different population groups may assign different values to the same ecosystem service attributes, leading to varying coefficients and valuation outcomes across segments (e.g. based on income, preferences, or sociodemographic characteristics) • Difficult to apply for valuing ecosystem services in poorly documented areas due to large data requirements • Results depend heavily on model specification
Travel cost method	• Applies standard economic techniques for measuring value • Uses information on revealed rather than stated behaviour and preference • Based on the simple and well-founded assumption that travel costs reflect recreational value	• In order to be effective and reliable, it requires a relatively large data set and a large amount of information for each respondent, making it expensive and time-consuming • Complex statistical analysis and modelling are required to derive the required recreation values • Defining and measuring the opportunity cost of time, or the value of time spent travelling, is problematic • Simplest models assume that individuals take a trip for a single purpose • Availability of substitute sites will affect values • People who highly value certain sites may choose to live nearby, implying lower travel costs despite high appreciation
Contingent valuation method	• Does not rely on actual markets or observed behaviour, i.e. can be applied to any situation and any ecosystem good or service • Can estimate both use and non-use values	• Responses to willingness to pay questions are hypothetical and may not reflect true behaviour • Hypothetical scenarios described in contingent valuation questionnaires might be misunderstood or found to be unconvincing to respondents, leading to biased responses • Requires complex data collection and sophisticated statistical analysis and modelling • Large-scale surveys that are necessary for contingent valuation can be expensive to conduct

(Cont.)

TABLE 7 *(cont.)*
Advantages and limitations of the different types of methods used for the economic assessment of ecosystem services

	Advantages	Limitations
Choice experiment method	• Efficient means of collecting information • Economic values are not elicited directly but are inferred by the trade-offs respondents make between monetary and non-monetary attributes (bias less likely) • Research is not limited by pre-existing market conditions • Useful to use as a policy tool for exploring proposed or hypothetical futures or options • Allow individuals to evaluate non-market benefits described in an intuitive and meaningful way	• Choices made are hypothetical and may not reflect true preferences or behaviour • Requires complex data collection, sophisticated statistical analysis and choice modelling • Large-scale surveys that are necessary can be expensive
Meta-analysis	• Pooling the estimates from various studies may provide a preferable estimate of value • Meta-analysis generally provides greater possibilities for generalization than a single case study does • Development of Geographical Information Systems allows for gathering spatially specific case study data	• Time-consuming • Research is dependent on the availability of empirical evidence • Excludes case study-specific features
Value transfer	• Quick and affordable alternative to original valuation research	• Relatively little published evidence exists about its validity and reliability • Value predicted by a value transfer exercise, which can largely overstate or understate the true value • Conditions must be met if it is to provide reliable results, especially since local circumstances and conditions in the new decision-making context are closely related to those prevailing in the original research
Transformation Function Method	• Conceptually well developed • Assumes strong links between ecosystem services and economic benefits	• Technically difficult to apply in practice • Substantial data requirements at the farm level • Market value of other inputs in the production process needs to be considered • Limited to valuing those ecosystem services whose value can be proxied by market prices or or policy-based payments for ecosystem services

Source: Adapted from Koetse, M.J., Brouwer, R. & Van Beukering, P.J.H. 2015. Economic valuation methods for ecosystem services. In J.A. Bouma & P.J.H. Van Beukering, eds. *Ecosystem services*, 1st ed., pp. 108–131. Cambridge University Press. https://doi.org/10.1017/CBO9781107477612.009

– summarized in Table 6, which presents an overview of ecosystem services and the commonly used methods to value them – show how ecosystem services influence societal welfare either directly (e.g. as consumer goods) or indirectly (e.g. as intermediate goods). An important distinction exists between market-based and non-market-based methods. While market-based valuation methods rely on actual market behaviour and transactions to derive values for ecosystem services, indirect market valuation methods (revealed preference methods) are based on the real behaviour of decision makers and examine how non-marketed goods influence markets for other goods. Stated preference techniques, by contrast, are applied when market prices are unavailable, when revealed preference methods cannot be used, and when changes in ecosystem services are hypothetical (Koetse *et al.*, 2015).

Two additional methods, meta-analysis and value transfer, do not fit neatly into the categories above. In a narrow sense, these do not represent valuation methods themselves but rather tools for synthesizing or applying existing valuation data.

In the remainder of this section, we do not discuss each valuation method in detail. Instead, we present the general advantages and disadvantages of the methods most often used for ecosystem services valuation. This discussion is not intended to be exhaustive; readers interested in the underlying theory and practical application of these methods may consult Bateman *et al.* (2003), Kanninen (2007), or Freeman III (2014).

The advantages and limitations of the various valuation methods outlined above are summarised in greater detail in Table 7, followed by guidance on when to apply each method.

When choosing a valuation method, the first question to ask is which kind of ecosystem services are to be valued (see Table 6). If one is interested in provisioning services only, direct market methods can be applied. When potential future changes in the quantity or quality of these services are to be studied, non-market methods are probably more appropriate. A similar picture appears for most regulating services, which can generally be valued through direct or indirect market methods. For cultural services, markets

BOX 5

Definitions of a few common modelling terms used in this section

Several common ecosystem service models are described below. These models vary in their complexity and data requirements. Multiple descriptors may be used to describe a single model.

Conceptual

Conceptual models are simplified, theoretical representations of socioecological systems that illustrate major elements and the interactions between them. They are usually qualitative depictions of the complex interactions between livestock and components of the atmosphere, hydrosphere and biosphere, and they can also include dynamics between natural and human systems. Conceptual models assist with the visualization and communication of the multifaceted relationships between livestock, direct and indirect drivers of change, ecosystem functioning and biodiversity, and nature's benefits to people. They are complementary to quantitative models, as model development begins with conceptual models.

Example: The Barn (Duru *et al.*, 2018).

Empirical

Unlike conceptual models, empirical models, rely on observational and experimental data to identify relationships between an ecosystem service and other variables. While useful for predicting outcomes based on observations or experimentation, these models do not necessarily provide mechanistic understanding of ecosystem processes. Statistical models are likely the type of empirical model that biophysical scientists are most familiar with, as regression models are often used to analyse experimental data.

Example: Regression models (Nyberg *et al.*, 2020).

Mechanistic

Mechanistic models differ from empirical models in that instead of modelling simple input–output relationships, they explicitly account for the processes that transform inputs into outputs. For example, an empirical model could relate soil carbon accumulation to a single factor assigned to a management practice based on observational data. A mechanistic model would instead simulate the biogeochemical interactions between plant, soil and microbes in order to estimate soil carbon accrual.

Example: MEMS (Robertson *et al.*, 2019).

Static

Models differ in the ways that they account for time. Static models consider inputs and outputs from a single point in time, without regard for how things change over time.

Dynamic

In contrast to static models, dynamic models include some representation of time. These models are commonly composed of differential equations and simulate changes in the relationships between variables over time.

Deterministic

Deterministic models do not account for randomness. As such, they will return the same output for a single set of inputs.

Stochastic

Stochastic models account for randomness by assigning probability distributions to one or more variables in the model. Statistical models with error terms would be an example of a stochastic model. Monte Carlo simulations can be used to introduce stochasticity into static models. As model inputs vary randomly according to pre-assigned probability distributions, each run of a stochastic model will generally return a different result.

Process-based

These models are mechanistic and explicitly account for each step in a biological, physical or chemical process that governs ecosystem behaviour. Process-based models can also capture time-dependent interactions and feedback loops between livestock activities, ecological processes and the provisioning of ecosystem services. These models are often used to explore how ecosystems respond to different scenarios or perturbations. They can be useful for exploring how changes in one ecosystem component impact others.

Example: LPJ-GUESS (Willcock *et al.*, 2019).

Spatially explicit

Models of this type incorporate spatial information, allowing users to explore how ecosystem dynamics vary across different geographic areas. Spatially explicit models integrate geographical and ecological data to capture the spatial patterns and interactions involved in ecosystem services related to livestock. These models are valuable for assessing the localized and landscape-level impacts of livestock activities and for guiding spatially informed management decisions.

Example: InVEST (Crossman *et al.*, 2013).

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Hybrid

Hybrid or coupled models are usually novel modelling frameworks that incorporate two or more models in an ecosystem services assessment. They are particularly useful when assessing multiple systems in a landscape, e.g. cropland and rangeland, natural and social. There are often existing models available to assess dynamics within one of these systems, but not both. The outputs of one model are generally used to initialize another. Coupling models with different capabilities can allow users to bypass the need to develop an entirely new model.

Example: Alternative crop–livestock management in South Africa (APSIM + aDGVM2; Pfeiffer *et al.*, 2022).

Grey box

Process-based and spatially explicit ecosystem service models described above are "white box" models developed with theoretical knowledge and mathematical equations. Grey box models combine prior knowledge with data-driven approaches like artificial intelligence. This is different from machine learning methods (below), as the structure of the grey box model is partly based on expert knowledge, allowing for some interpretation into mechanisms.

Example: Ag-EcoSOpt Tool (Nguyen *et al.*, 2019).

Data-driven

Data-driven models are useful when there is limited mechanistic understanding of a system or systems. They can also be applied in situations where there are large volumes of data with high dimensionality. These models are developed with machine learning approaches, which can be supervised (e.g. regression trees) or unsupervised (e.g. neural networks). Machine learning models are often considered to be "black box" because they do not incorporate causal relationships or theoretical understanding (Scowen *et al.*, 2021).

Example: Random forest for carbon–diversity hotspots in Brazil (de Oliveira Silveira *et al.*, 2019).

Source: Authors' own elaboration.

typically do not exist, which is why direct market methods are not applicable. Hedonic pricing can, and often is, used to assign value to recreation and aesthetic services, while travel cost methods are generally used for recreation services only. In general, non-market methods are applied for valuing cultural services. Supporting services are typically valued through direct market methods, although the possibilities may vary strongly depending on the specific service (Koetse *et al.*, 2015).

Besides considering the type of ecosystem service to be valued, some general points need to be considered – namely, the type of economic value to be estimated, the purpose of the valuation, data and information availability and the required accuracy of the results. Finally, potential trade-offs and synergies between marketed and non-marketed ecosystem services can be addressed by the transformation function method, if data availability permits.

3.4 MODELLING VALUATION METHODS

3.4.1 Conceptualization

Models are powerful tools for answering questions about the past, better understanding the present and exploring potential futures regarding ecosystem services provided by agroecosystems. As simplified representations of biological and social systems, models not only represent our understanding of ecological processes and their interactions but also reflect different and sometimes competing theoretical perspectives. Ecosystem services models couple socioeconomic systems, particularly actions or interventions of individuals or governments, to ecological processes whereby ecosystem services or disservices emerge from the interactions and feedback between these systems. Models also fulfil an important role in ecosystem service evaluation because they enable the assessment of services that are difficult to measure directly, such as pollination or erosion, using indicators that combine earth observation or other big data products with conceptual understanding (Ramirez-Reyes *et al.*, 2019).

For decision-makers, models are useful for quantifying outcomes of previous policies and exploring hypothetical what-ifs using scenarios (see further information on scenario analysis in Section 3.5.3). Ecosystem service models enable users to model the impacts of livestock management on ecosystem services and other global change drivers, as well as to translate policy questions into scientifically informed actions that can support progress towards food system transformation. While ecosystem services models generally only provide a snapshot in time about the status of ecosystem services, growing data availability and computing capacity will enable dynamic, real-time assessment and the feedback between policies, management and agroecosystems (Ramirez-Reyes *et al.*, 2019). A brief discussion of model descriptors is presented in Box 5.

This chapter does not aim to provide an exhaustive overview of how to select appropriate models for specific decision-making contexts. Rather, it reviews modelling approaches and common ecosystem service models applied to livestock in agroecosystems. The focus is further narrowed to modelling approaches and models capable of evaluating the relationships between livestock management decision-making (i.e. indirect drivers), ecosystem changes (i.e. direct drivers), ecosystem function and the delivery of ecosystem services to society.

3.4.2 Methods

Quantitative ecosystem services models are numerical or statistical representations of relationships between living organisms (plants, animals and microorganisms) and their physical environment (air, water and soil), and how they respond to various environmental changes (direct drivers) and human decision-making (indirect drivers) within agroecosystems. Ecosystem service models are used to understand and predict the behaviour, interactions and dynamics of the system of interest.

Models vary in their approach to representing these relationships. For example, some models are conceptual and designed to help organize mental models (e.g. telecoupling), whereas others are biophysical and designed to calculate mass and energy flows through ecosystems (e.g. material flow analysis [MFA]). Some are product-centric (e.g. life cycle assessment), while others are natural resource-centric (e.g. ecological footprint [EF]). These models are developed according to several methods, described in this section. Some methods, such as EMA, LCA, EF and MFA, among others, can be combined in a multicriteria analysis since all of them share most of the same database and their indicators can broaden the scope of the analysis by providing a better understanding of the system behaviour (Patterson *et al.*, 2017; Yu *et al.*, 2019). Examples of models developed using each of the methods discussed in this section can be found in Annex 3.

3.4.2.1 Emergy assessment

Emergy assessment is a biocentric method based on thermodynamics (Odum, 1983, 1996). Emergy is the available energy (exergy) of a single class (e.g. solar energy) that is used directly and indirectly in a process to obtain a product or service, or to sustain a determined flow (Odum, 1983, 1996). It accounts for the total amount of energy provided by nature and society to create or sustain a product, service or flow through its full life cycle. As EMA is based on the laws of thermodynamics, ecosystem services are measured in terms of energy output (Zhang *et al.*, 2024; Wang *et al.*, 2020; Brown and Ulgiati, 2016; Coscieme *et al.*, 2014; Rugani and Bennetto, 2012).

This method assumes that an entity's value is determined according to what was invested to obtain it ("value from the giving side", or donor value), whereas exergetic analysis and economic evaluations maintain that a product, service or flow's worth is its utility ("receiver value"). Thus, emergy has been proposed as a measure of environmental support for a multitude of processes in the biosphere and for economies (Ulgiati *et al.*, 2010). It has been applied to the evaluation of ecosystem services in several contexts (Zhang *et al.*, 2024; Nadalini *et al.*, 2021; Wang *et al.*, 2019; Franzese *et al.*, 2017; Rótolo *et al.*, 2015; Zhao and Wu, 2015; Campbell and Brown, 2012). An example of its application to livestock systems can be found in Rótolo *et al.* (2007), which used EMA to assess the environmental impacts of livestock grazing in the Argentine pampas.

Several indicators can be obtained from EMA, including the Emergy Yield Ratio (EYR) – also referred to as the emergy appropriation ratio – which indicates the ability of the analysed system to use and make available local natural resources (renewable and slowly renewable, such as soil) through human intervention. Another indicator is the Emergy Loading Ratio (ELR), which reflects the pressure of the system on natural resources and may be considered a measure of ecosystem stress. EMA is among the most commonly used methods in natural capital assessments (Pulselli *et al.*, 2011).

3.4.2.2 Material flow analysis

Whereas EMA is used to evaluate energy flow, material flow analysis tracks the flow of materials diverted or extracted from their natural ecosystem and used in the production of a good, from extraction through recycling and disposal (Bringezu and Moriguchi, 2018). MFA, often referred to as an "ecological rucksack" (Lettenmeier *et al.*, 2009), evaluates the environmental disturbance induced by the withdrawal or use of a material in the ecosystem (Ulgiati *et al.*, 2006; Schmidt-Bleek, 1993). Indicators used in MFA include direct material inputs, domestic material consumption and others (Cárdenas-Mamani *et al.*, 2022; Kovanda, 2021; EUROSTAT, 2018).

As an example, MFA has been applied as a tool to support policy decision-making in the field of resource and environmental management in Vienna and the Swiss lowlands, for early recognition and priority setting, and to analyse and improve sustainability (Hendriks *et al.*, 2000).

3.4.2.3 Ecological footprint

The ecological footprint is a biophysical accounting approach for provisioning and regulating ecosystem services. It is a tool for measuring and visualizing the land and water area required to support the flow of matter and energy from nature to economies and vice versa (Rees and Wackernagel, 2023; Wackernagel and Rees, 1996). EF tracks the use of productive surface areas such as cropland, grazing land, fishing grounds, built-up land, forest area and carbon demand on land. The unit of measurement is the global hectare (gha). By relating "biocapacity" – the productivity of an ecosystem defined by area (Mancini *et al.*, 2018) – to consumption patterns, EF estimates whether a society is operating within the carrying capacity of an ecosystem. When the ecological footprint exceeds the carrying capacity, resource use is considered unsustainable (Zhao *et al.*, 2005).

For example, in 2022, the global ecological footprint was 2.6 gha per person, while available biocapacity was only 1.5 gha per person – indicating a 70 percent overshoot of Earth's regenerative capacity at the time (Rees and Wackernagel, 2023). Ecological footprints can be reported at scales ranging from country to field level. They can also be calculated from both consumption (e.g. the biocapacity required to maintain household consumption) and production (e.g. the resource needs driven by income required to support household consumption) perspectives.

3.4.2.4 Life cycle assessment

Similar to EMA and MFA, life cycle assessment evaluates the environmental impacts of goods and services over the entirety of their life cycle. LCA is carried out in four stages: (1) goal and scope definition; (2) data collection for a detailed life cycle inventory of all inputs and outputs, including emissions to the environment; (3) impact assessment by aggregating the inventory data into impact categories such as climate change, ozone depletion potential or natural resource depletion; and (4) interpretation of the results. LCA is governed by ISO 14040.

As LCA is product-centric and primarily designed to evaluate negative environmental impacts, ecosystem services are often excluded from analysis. A framework was recently developed to guide the inclusion of ecosystem services in LCA (Alejandre *et al.*, 2019). A few studies have proposed approaches for incorporating ecosystem services into livestock system analyses, though the approaches vary, as do the number and types of services considered. These approaches are typically applied either at the allocation stage (during goal and scope definition) or at the impact assessment stage of LCA.

Examples of allocation-based approaches include the use of conservation payments as a proxy for non-provisioning ecosystem services, as in the case of grazing sheep in the Andes for biodiversity conservation and landscape preservation (Ripoll-Bosch *et al.*, 2013). Another example is the use of stakeholder-ranked functions to develop a "livelihood allocation" to account for cultural ecosystem services in an assessment of milk and meat from smallholder Kenyan dairy farms (Weiler *et al.*, 2014).

Alternatively, characterization factors have been developed to account for ecosystem services at the impact assessment stage. The FAO LEAP Guidelines on Biodiversity Assessment propose characterization factors for biodiversity impacts based on land use intensity (FAO, 2020). At the time of this report, the approach has been piloted in a comparison of livestock grazing on intensively and extensively managed pastures (McClelland *et al.*, 2023). Finally, based on estimates of relative pollinator abundance, Alejandre *et al.* (2023) developed a set of global-scale characterization factors to translate land use into relative impacts on wild pollinator abundance. These are proposed as a first step toward systematically incorporating pollinator impacts in LCA studies.

3.4.2.5 Agent-based modelling

This bottom-up approach allows for interaction among agents with the capacity to update norms and preferences over time (Sun *et al.*, 2013; Chen *et al.*, 2012). Agent-based models (ABMs) are based on their ability to represent individual behaviours; they typically integrate participatory data and quantitative survey data with traditional equation-based ecological models to simulate decision-making processes and ecological impacts (Sun *et al.*, 2013; Chen *et al.*, 2012; Helbing, 2012).

Agent-based modelling is a method for representing complex decision-making by individuals or multiple stakeholders (i.e. agents) in order to assess social and environmental outcomes (Murray-Rust *et al.*, 2011). Among other uses, ABM has been applied as a dynamic simulation method to explore feedback loops between animals, land and people (Boone *et al.*, 2011, 2014). In contrast with top-down modelling approaches, where populations may be represented as aggregate entities or "stocks", agent-based modelling adopts a bottom-up approach by modelling population members as autonomous individuals interacting with one another.

ABM has been widely used to model links between decision-making and ecosystem services in pastoral and grassland-based systems (Moritz *et al.*, 2023; Bateki *et al.*, 2019). Unlike the previously discussed methods, ABM is neither product-centric nor a direct environmental impact assessment tool. Rather, it is used to model how decision-making affects ecosystem services or how agent decisions evolve in response to changes in ecosystem services. ABM is facilitated by software platforms such as AgentScript, GAMA and NetLogo.

3.4.2.6 System dynamics modelling

System dynamics (SD) is a modelling approach commonly applied to complex systems with nonlinear relationships that evolve over time (Sterman, 2000). SD models are composed of partial differential equations and structured using stocks and flows. Unlike some other approaches used for modelling ecosystem services, SD enables bi-directional interactions and feedback between systems. Causal relationships can be represented as time-dependent or nonlinear (Berrio-Giraldo *et al.*, 2021). SD is particularly well suited to represent complex interactions and feedback within and between coupled systems across time and space (Aspinall and Staiano, 2019; Uehara *et al.*, 2016).

SD and ABM share many similarities but differ in their approach. Like ABM, SD facilitates dynamic representations of interactions among animals, land and people, and their

TABLE 8
Examples of ecosystem service models applied to the assessment of livestock and agroecosystems

Model	Classification	Description	References
DayCent (Daily Century)[a] and other ecosystem models	Process-based, spatially-explicit	Models simulate the dynamics and movement of carbon, nitrogen, phosphorus and/or sulphur in vegetation, soil and the atmosphere at daily and sub-daily time scales.	Shepherd, A., Hartman, M. D., Fitton, N., Horrocks, C. A., Dunn, R. M., Hastings, A., & Cardenas, L. M. 2019. *Metrics of biomass, live-weight gain and nitrogen loss of ryegrass sheep pasture in the 21st century*. Science of The Total Environment, 685: 428–441. https://doi.org/10.1016/j.scitotenv.2019.05.038; see Ma, L., Derner, J.D., Harmel, R.D., Tatarko, J., Moore, A.D., Rotz, C.A., Augustine, D.J., Boone, R.B., Coughenour, M.B., Beukes, P.C., Van Wijk, M.T., Bellocchi, G., Cullen, B.R. & Wilmer, H. 2019. *Application of grazing land models in ecosystem management: Current status and next frontiers. In: Advances in Agronomy*, Vol. 158, pp. 173–215. Elsevier. https://doi.org/10.1016/bs.agron.2019.07.003 for additional examples
GloBIO[b] (Global biodiversity model for policy support)	Empirical, spatially-explicit	Simple cause–effect model representing the relationships between mean species abundance (MSA) and human-driven environmental pressures, such as land use.	Alkemade, R., Reid, R. S., Van Den Berg, M., De Leeuw, J. & Jeuken, M. 2013. Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. *Proceedings of the National Academy of Sciences*, 110(52): 20900–20905.
LUCI[c] (Land utilization and capability indicator)	Process-based, spatially-explicit	Based on the Polyscape GIS framework, LUCI quantifies ecosystem services from land cover at the landscape scale, including water, carbon and agricultural production.	Pereira, F.C., Charters, S., Smith, C.M.S., Maxwell, T.M.R., & Gregorini, P. 2023. *A Geospatial Modelling Approach to Assess the Capability of High-Country Stations in Delivering Ecosystem Services*. Land, 12(6): 1243. https://doi.org/10.3390/land12061243; Emmett, B.A., Cooper, D., Smart, S., Jackson, B., Thomas, A., Cosby, B., Evans, C., Glanville, H., McDonald, J.E., Malham, S.K., Marshall, M., Jarvis, S., Rajko-Nenow, P., Webb, G.P., Ward, S., Rowe, E., Jones, L., Vanbergen, A.J., Keith, A., Jones, D.L. 2016. Spatial patterns and environmental constraints on ecosystem services at a catchment scale. *Science of The Total Environment*, 572: 1586–1600. https://doi.org/10.1016/j.scitotenv.2016.04.004; Jackson, B., Pagella, T., Sinclair, F., Orellana, B., Henshaw, A., Reynolds, B., McIntyre, N., Wheater, H. & Eycott, A. 2013. Polyscape: A GIS mapping framework providing efficient and spatially explicit landscape-scale valuation of multiple ecosystem services. *Landscape and Urban Planning*, 112: 74–88. https://doi.org/10.1016/j.landurbplan.2012.12.014
SWAT[d] (Soil and water assessment tool)	Process-based, spatially-explicit	Hydrological model operating at the watershed scale, simulating the effects of management and land use on water quantity, water quality and sediment movement.	Kim, J., Ale, S., Kreuter, U.P. & Teague, W.R. 2023. Grazing management impacts on ecosystem services under contrasting climatic conditions in Texas and North Dakota. *Journal of Environmental Management*, 347: 119213. https://doi.org/10.1016/j.jenvman.2023.119213; see Francesconi, W., Srinivasan, R., Pérez-Miñana, E., Willcock, S.P. & Quintero, M. 2016. Using the Soil and Water Assessment Tool (SWAT) to model ecosystem services: A systematic review. *Journal of Hydrology*, 535: 625–636. https://doi.org/10.1016/j.jhydrol.2016.01.034

[a] Natural Resource Ecology Laboratory (NREL). n.d. *DayCent: Daily Century Model*. Colorado State University. https://www.nrel.colostate.edu/projects/daycent;

[b] GLOBIO. n.d. *Why use GLOBIO?* [Cited 31 July 2025.] https://www.globio.info/why-use-globio;

[c] UK Centre for Ecology & Hydrology (UKCEH). n.d. LCM2021 *Land Cover Map of Great Britain*. [Cited 31 July 2025.] https://catalogue.ceh.ac.uk/documents/adead10b-a6a0-45fa-b350-acd16b23c3fe; d Texas A&M AgriLife Research. n.d. *Soil and Water Assessment Tool (SWAT)*. [Cited 31 July 2025.] https://swat.tamu.edu/

Source: Authors' own elaboration.

influence on ecosystem services or other key outcomes of interest. However, while ABM models individuals with agency, SD typically adopts a more top-down approach, representing populations as single groups with defined behaviours (albeit capable of including variability within those behaviours), rather than modelling individuals whose interactions produce emergent behaviours (Boone *et al.*, 2014).

SD has been used to assess whether payments for environmental services lead to improved restoration, preservation and conservation outcomes in grazed highlands in Colombia (Benavides *et al.*, 2023); to link the management of agricultural systems with provisioning and other ecosystem service outcomes in Scotland (Aspinall *et al.*, 2019); and to evaluate the effectiveness of common agricultural and natural resource management practices in achieving improved natural resource outcomes (Turner *et al.*, 2016). SD modelling is supported by software such as Stella and Vensim.

3.4.2.7 Telecoupling

Telecoupling is a conceptual framework for examining complex relationships between social and environmental systems across multiple spatial and institutional scales (Liu *et al.*, 2019). Each location in a telecoupling framework is represented as a coupled socioecological system, with flows between locations driven by agents (Liu *et al.*, 2016; Liu *et al.*, 2013; Liu *et al.*, 2007). This framework facilitates the assessment of feedback between systems, including the long-distance exchange of ecosystem services (Liu *et al.*, 2016).

Telecoupling enables the analysis of causes and effects arising from the interactions among producers, consumers and spillover systems. It captures the complexity of increas-

TABLE 9
Examples of common ecosystem services modelling platforms previously used in livestock or agroecosystem assessments

Platform	Classification	Description	References
Artificial Intelligence for Environment and Sustainability **(ARIES)**[a]	Hybrid, data-driven	Provides an open-source, user-friendly interface to support non-technical users in assessing ecosystem services.	Notte, A., Marongiu, S., Masiero, M., Molfetta, P., Molignoni, R., & Cesaro, L. 2015. *Livestock and Ecosystem Services: An Exploratory Approach to Assess Agri-Environment-Climate Payments of RDP in Trentino*. Land, 4(3): 688–710. https://doi.org/10.3390/land4030688 Bagstad, K. J., Semmens, D. J., Waage, S. & Winthrop, R. 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosystem Services*, 5: 27–39. https://doi.org/10.1016/j.ecoser.2013.07.004
CO$TING-Nature[b]	Hybrid, spatially explicit	Offers a web-based, open-source dashboard for both national-level assessments and user-defined case studies. Model metrics are globally indexed and mapped.	Mulligan, M. 2015. *Trading off agriculture with nature's other benefits, spatially. In: Impact of Climate Change on Water Resources in Agriculture*, pp. 184–204. CRC Press. https://doi.org/10.1201/b18652 Mulligan, M., Guerry, A., Katy, A., Bagstad, K., Ferdinando, V. & Silvestri, S. 2010. *Capturing and quantifying the flow of ecosystem services*, In: Silvestri, S. & Kershaw, F. (Eds.), *Framing the Flow: Innovative Approaches to Understand, Protect and Value Ecosystem Services Across Linked Habitat*, UNEP World Conservation Monitoring Centre, 62 p. ISBN 978-92-807-3065-4.
CLIMSAVE Impact Assessment Framework[c]	Hybrid, spatially explicit, data-driven	Links ecosystem services models via a web-based interface to assess climate change impacts across multiple economic sectors in Europe.	Veerkamp, C. J., Dunford, R. W., Harrison, P. A., Mandryk, M., Priess, J. A., Schipper, A. M., Stehfest, E., & Alkemade, R. 2020. *Future projections of biodiversity and ecosystem services in Europe with two integrated assessment models*. Regional Environmental Change, 20(3), 103. https://doi.org/10.1007/s10113-020-01685-8 Willcock, S., Hooftman, D. A. P., Balbi, S., Blanchard, R., Dawson, T. P., O'Farrell, P. J., Hickler, T., Hudson, M. D., Lindeskog, M., Martinez-Lopez, J., Mulligan, M., Reyers, B., Shackleton, C., Sitas, N., Villa, F., Watts, S. M., Eigenbrod, F., & Bullock, J. M. 2019. *A Continental-Scale Validation of Ecosystem Service Models*. Ecosystems, 22(8), 1902–1917. https://doi.org/10.1007/s10021-019-00380-y
Ecosystem Services Inventory and Identification **(ESII)**[d] and Ecosystem Intelligence **(EI)**[e]	Empirical	Uses a web-based interface to calculate an ecosystem integrity score across multiple ecosystem services at one or more locations.	Brownson, K., Cox, C. & Padgett-Vasquez, S. 2021. The impacts of agricultural windbreaks on avian communities and ecosystem services provisioning in the Bellbird Biological Corridor, Costa Rica. *Agroecology and Sustainable Food Systems*, 45(4): 592–629. https://doi.org/10.1080/21683565.2020.1838029
Ecosystem Services Valuation Database **(ESVD)**	Empirical	Supplies an open-source, publicly available database and tool offering standardized monetary values for ecosystem services across all biomes and continents.	Brander, L.M., De Groot, R., Schägner, J.P., Guisado-Goñi, V., Van 't Hoff, V., Solomonides, S., McVittie, A., Eppink, F., Sposato, M., Do, L., Ghermandi, A., Sinclair, M. & Thomas, R. 2024. Economic values for ecosystem services: a global synthesis and way forward. *Ecosystem Services*, 66: 101606. https://doi.org/10.1016/j.ecoser.2024.101606
Integrated Valuation of Ecosystem Services and Tradeoffs **(InVEST)**[f]	Empirical, spatially explicit	Provides an open-source tool for quantifying and mapping the flow of ecosystem services (biophysical or economic) in terrestrial, freshwater, marine and coastal ecosystems.	Enahoro, D., Kozicka, M., Pfeifer, C., Jones, S.K., Tran, N., Chan, C.Y., Sulser, T.B., Gotor, E. & Rich, K.M. 2023. Linking ecosystem services provisioning with demand for animal-sourced food: An integrated modelling study for Tanzania. *Regional Environmental Change*, 23(1): 48. https://doi.org/10.1007/s10113-023-02038-x Posner, S., Verutes, G., Koh, I., Denu, D., & Ricketts, T. 2016. *Global use of ecosystem service models*. Ecosystem Services, 17: 131–141. https://doi.org/10.1016/j.ecoser.2015.12.003 Goldstein, J.H., Caldarone, G., Duarte, T.K., Ennaanay, D., Hannahs, N., Mendoza, G., Polasky, S., Wolny, S. & Daily, G.C. 2012. Integrating ecosystem-service tradeoffs into land-use decisions. *Proceedings of the National Academy of Sciences*, 109(19): 7565–7570. https://doi.org/10.1073/pnas.1201040109
Social Values for Ecosystem Services **(SolVES)**[g]	Empirical, spatially explicit, data-driven	Offers an open-source tool for quantifying and mapping social values of ecosystem services among different stakeholder groups.	Makovníková, J., Kobza, J., Pálka, B., Mališ, J., Kanianska, R. & Kizeková, M. 2016. *An approach to mapping the potential of cultural agroecosystem services*. Soil and Water Research, 11(1): 44–52. https://doi.org/10.17221/109/2015-SWR

[a] Integrated Modelling Partnership. n.d. ARIES – *Artificial Intelligence for Environment & Sustainability*. [Cited 31 July 2025.] https://aries.integratedmodelling.org/;

[b] PolicySupport.org. n.d. *Costing Nature*. [Cited 31 July 2025.] https://www.policysupport.org/costingnature;

[c] CLIMSAVE. n.d. *Climate change integrated assessment methodology*. [Cited 31 July 2025.] https://www.climsave.eu/;

[d] ESII Tool. n.d. *Ecosystem Services Identification & Inventory Tool*. [Cited 31 July 2025.] https://www.esiitool.com/;

[e] Ecosystem Intelligence. n.d. *Ecosystem Intelligence Platform*. [Cited 31 July 2025.] https://www.ecosystemintelligence.com/;

[f] Natural Capital Project. n.d. InVEST – *Integrated Valuation of Ecosystem Services and Tradeoffs*. [Cited 31 July 2025.] https://naturalcapitalproject.stanford.edu/software/invest;

[g] United States Geological Survey (USGS). n.d. *Social Values for Ecosystem Services (SolVES)*. [Cited 31 July 2025.] https://www.usgs.gov/centers/geosciences-and-environmental-change-science-center/science/social-values-ecosystem

Source: Authors' own elaboration.

ingly distant environmental and socioeconomic interconnections and their diverse drivers. Applications of the telecoupling framework include: 1. Evaluating drivers of ecosystem service specialization by smallholder farms in Ethiopia (Brück *et al.*, 2023); 2. Understanding the impacts of global trade policies and product attribute incentive programmes on smallholder farms in the Plurinational State of Bolivia, Colombia, the United States and Morocco (Zimmerer *et al.*, 2018); and 3. Exploring the potential impact of bioengineering and novel protein production technologies on agricultural practices and feedstock demand (Newman *et al.*, 2022).

3.4.2.8 Existing ecosystem service models

Over the past two decades, the availability of ecosystem service models and platforms has expanded considerably, particularly with the development of decision-support tools designed for users with limited technical modelling expertise. This section briefly presents several of the most widely used models and platforms in previous livestock system assessments. The review is not exhaustive but aims to illustrate the range of available ecosystem service models. Additional guidance for practitioners on resource requirements and skills considerations when selecting between methods is provided in Section 2.3.

Several open-access ecosystem service models (Table 8) and modelling platforms (Table 9) are publicly available, with varying degrees of relevance to livestock systems. These tools primarily represent the biophysical components of socioecological systems and provide estimates of provisioning, regulating and, to a limited extent, supporting services. While most were initially developed for other modelling purposes, many have since been adapted for ecosystem service assessments.

These models can also be coupled with socioeconomic models to support integrated, scenario-based analyses of human–environment interactions. Further information on models applicable to extensive livestock production systems can be found in Ma *et al.* (2019); for SWAT model applications, see Francesconi *et al.* (2016); and for general summaries and applications of ecosystem service models, see Weiskopf *et al.* (2022) and Rieb *et al.* (2017).

Dedicated ecosystem services modelling platforms are also available to support assessments and scenario analyses (Table 9). A key advantage of these platforms is their capacity to estimate a broader range of ecosystem services than individual models, which can be particularly useful for initial scoping analyses or assessments at broader spatial scales. However, these platforms are generally less customizable without additional technical expertise, and not all are open-source or freely available.

Beyond the modelling platforms listed in Table 9, several others – including Envision, EcoAIM (Ecological Asset and Inventory Management), ESValue, NAIS (Natural Assets Information System) and TESSA (Toolkit for Ecosystem Service Site-based Assessment) – may also be suitable for assessing ecosystem services in livestock agroecosystems (Booth *et al.*, 2014; Nemec and Raudsepp-Hearne, 2013; Peh *et al.*, 2013).

3.4.3 Advantages and limitations of modelling

Modelling offers several advantages. It enables insights into complex interactions between management practices and ecological processes that influence ecosystem services, particularly where direct observation is not feasible. Models can also predict, with varying degrees of uncertainty, how ecosystems may respond to changes such as shifts in climate, land use or pollution. Furthermore, they can support decision-making in natural resource management, conservation and environmental policy by allowing users to explore scenarios and test hypotheses.

However, models are simplifications of reality. No model can fully capture the complexity of ecosystems or the intricate interconnections that govern their functioning. Therefore, results must be interpreted in light of modelling assumptions and inherent uncertainties. For ecosystem service modelling, limitations include uncertainty in both input data and model structure, oversimplified representations of ecological processes, unnecessarily complex models where simpler ones may suffice, and a lack of technical expertise or data availability (Taoumi and Lahrech, 2023).

An overview of the advantages and limitations of the modelling approaches discussed in this chapter is provided in Table 10.

3.5 CROSS-METHODS AND APPROACHES TO ASSESS ECOSYSTEM SERVICES

Part 3 of this report has presented an overview of individual methods, grouped according to biophysical, sociocultural, economic and modelling approaches to ecosystem service assessment. These categories reflect distinct disciplinary lenses through which ecosystem services are typically analysed. However, several methodologies and considerations emerge that cut across these disciplinary boundaries. In practice, some assessments do not fall exclusively within one of the above categories, but instead combine elements from multiple approaches. These cross-cutting methods include the assessment of trade-offs and synergies (Section 3.5.1), the use of meta-analysis (Section 3.5.2) and the application of scenario analysis (Section 3.5.3).

3.5.1 Assessing relationships (trade-offs and synergies) between ecosystem services

Trade-off and synergy analysis aims to identify relationships among ecosystem services that co-occur in a given context. These relationships can be antagonistic (trade-offs), reinforcing (synergies), or neutral.

TABLE 10
Advantages and limitations of the different modelling methods are revised in this chapter

Method	Advantage	Limitations
Emergy assessment	• Suitable for analysing complex systems, including the interactions between human economic activities and the natural environment • Reflects a donor-side perspective by accounting for the environmental support provided by ecosystems – often overlooked in conventional assessments – both at the system level and for each component • Enables conversion of diverse resource types into a single common unit, solar emergy • Applicable to both large- and small-scale systems • Accounts for the use and flow of natural and societal resources • Shares a common database with life cycle assessment and material flow analysis, allowing for integrated assessments • Supports the formulation of evidence-based policies	• Cannot account for direct emissions • Requires regular updating of national environmental databases • Depends on clear and consistent definition of system boundaries • Highly dependent on raw data, which must be accurate and of high quality (both technical and procedural measures)
Material flow analysis	• It uses input/output methodologies • Accounts for material use • It shares the database with emergy analysis and LCA	• Highly dependent on raw data (they need to be accurate), data quality (technical and procedural measures of raw data) • Model assumptions
Ecological footprint	• Simple to calculate • Describe the demand and supply of resources providing a measure of resource consumption and waste generation	• Data limitations may lead to overestimation or underestimation • Does not consider underground resources • Uses hypothetical land • Risk of double counting • Does not drive policy recommendations
Life cycle assessment	• User-side perspective • Accounts for direct and indirect emissions for the system analysed and for each component • Applicable to both large and small systems • Shares databases with emergy analysis and MFA • Can inform policy development	• Partially quantifies the impact of renewable resources • Requires clear definition of system boundaries • Highly dependent on raw data accuracy and quality (technical and procedural) • In some cases, local emission factors or locally adjusted data are lacking
Agent-based modelling	• Models heterogeneity in behaviours within and between actors in a system • Captures effects of feedback, causality, delays and change over time on system behaviour • Useful for scenario modelling	• Significant data requirements • Can be difficult to validate against real-world data due to complexity of agent interactions • Computationally intensive
System dynamics	• Captures effects of feedback, causality, delays and change over time on system behaviour • Uses standardized language for describing systems and system behaviour • Facilitates shared understanding of system structure in multi- or transdisciplinary environments	• Significant data requirements • Can be cumbersome to represent geographical variation • Can obscure heterogeneity due to representation of system elements in aggregate

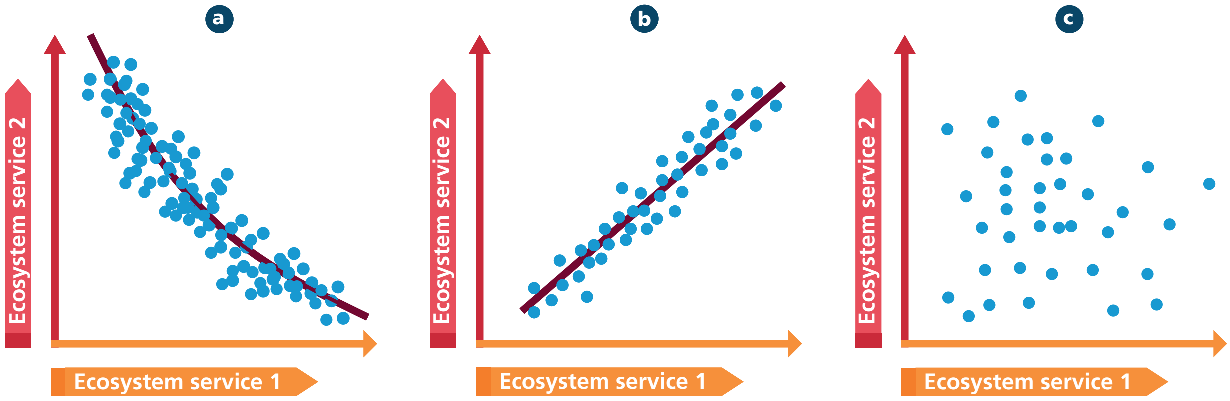
Source: Authors' own elaboration.

Trade-offs arise when a system cannot simultaneously optimize the provision of multiple ecosystem services, leading to the enhancement of one service at the expense of another (Rodriguez *et al.*, 2006; Bennett *et al.*, 2009). Such trade-offs are often observed in land use and land cover dynamics, where the expansion of one land cover type leads to the reduction of another, along with its associated ecosystem services (Accatino *et al.*, 2019). In agroecosystems, for instance, the increased provision of food (a provisioning service) is frequently associated with reductions in regulating or cultural services.

In contrast, synergies occur when an increase in one ecosystem service coincides with or promotes an increase in another. Since ecosystem services are interrelated, an intervention intended to enhance one service may unintentionally affect another – positively or negatively (Bennett *et al.*, 2009). It is therefore essential to assess these interconnections when planning or managing interventions.

Bennett *et al.* (2009) identify two main types of interactions that can cause trade-offs or synergies. First, a common external driver may simultaneously influence multiple ecosystem services either positively or negatively. For example, maintaining semi-natural grasslands (by avoiding intensification or abandonment) can benefit both livestock production and carbon sequestration. Second, ecosystem services may directly influence one another. For example,

FIGURE 10
Examples of scatter plots showing different relationships among ecosystem services: decreasing or trade-off (a), increasing or synergistic (b), or non-existent (c)



Note: Each point represents a combination of values of ecosystem service 1 and ecosystem service 2 in a spatial unit. Panel a): representation of a decreasing relationship, suggesting a trade-off. Panel b): representation of an increasing relationship, suggesting a synergy. Panel c): representation of a "cloud" of points suggesting the absence of a relationship between the two ecosystem services.
Source: Authors' own elaboration.

the provision of animal products may improve landscape appeal, which in turn helps sustain demand for local products. These relationships may also be more complex, involving multiple drivers or feedback loops between services.

Several methods exist for assessing trade-offs and synergies. In the ecosystem services literature, such relationships are often explored through correlations between service indicators (as illustrated, for example, in Figure 10a). However, it is important to recognize that correlation does not imply causation (Vallet *et al.*, 2018). While some argue that correlation analysis should not be considered a robust method for assessing ecosystem services interactions, it remains widely used in the literature as a useful exploratory tool for identifying patterns and generating hypotheses about service relationships.

3.5.1.1 Trade-offs and synergies assessed using measured or simulated data

Observation time variation of ecosystem service indicators

One method for understanding trade-offs and synergies among ecosystem services is to analyse time series of biophysical indicators associated with these services. This type of analysis helps trace the historical trajectory of a system. Specifically, changes observed in response to management actions can reveal relationships among ecosystem services, based on how indicators co-evolve over time. When indicators shift in opposite directions, this suggests a trade-off; when they move in the same direction, it suggests synergy.

For example, Martín-López *et al.* (2014) reviewed historical documents and scientific literature containing data on ecosystem service indicators to reconstruct the historical trajectory of service provision. Tomscha and Gergel (2016) used aerial photographs covering six decades in urbanizing floodplains to assess long-term changes. Geng *et al.* (2022) examined changes in ecosystem service indicators over time in the Yellow River area and identified trade-offs where ecosystem services shifted inconsistently. Dynamic simulation models can also generate time series trajectories that can be analysed for correlations.

Correlations among ecosystem services in different spatial units

Pairwise comparisons of ecosystem service indicators across a spatial dataset – where each pair represents the levels of two ecosystem services within a spatial unit – can provide insights into potential relationships or the absence thereof (Figure 10). A preliminary step involves visual inspection of scatter plots for each pair of services. A negative trend in the distribution of points (Figure 10a) suggests a trade-off, while a positive trend (Figure 10b) indicates a synergy. A dispersed cloud of points without a clear pattern (Figure 10c) suggests no significant relationship between the two ecosystem services.

To facilitate visual interpretation, bagplots – introduced by Jopke *et al.* (2015) – are effective tools. A bagplot is a bivariate extension of the boxplot concept proposed by Tukey (1977). It consists of a polygon that depicts, across two dimensions, the median and a broader region that excludes outliers.

For a more precise quantification of relationships between ecosystem services, several correlation coefficients can be used, including Kendall's tau (Kendall, 1938), the

Pearson coefficient, and the Spearman coefficient. These coefficients range from -1 to 1, with -1 indicating a complete negative correlation and 1 indicating a complete positive correlation. In all cases, it is essential to verify the statistical significance of the coefficients. Following the categorization by Raudsepp-Hearne *et al.* (2010), Pearson's r values can be interpreted as highly correlated when $|r| > 0.5$, moderately correlated when $0.3 < |r| < 0.5$, and weakly correlated when $|r| < 0.3$. Positive values suggest synergies, whereas negative values indicate trade-offs.

It is important to emphasize that correlation among ecosystem service indicators does not imply causation. If the data are derived from models, the model structure may shed light on mechanistic relationships underlying observed trade-offs or synergies. However, if the data are based on measurements, causality can only be hypothesized and interpreted with caution. For a comprehensive overview of statistical tests and metrics used to analyse correlations among ecosystem service indicators, see Mouchet *et al.* (2014).

Assessment of bundles of ecosystem services (with cluster analysis)

Bundles refer to integrated sets of ecosystem services that tend to co-occur in specific contexts (Raudsepp-Hearne *et al.*, 2010; Queiroz *et al.*, 2015). In the analysis of multidimensional datasets, these bundles can be identified using methods such as principal component analysis followed by cluster analysis. A range of clustering techniques has been applied for this purpose, including self-organizing maps (Crouzat *et al.*, 2015; Mouchet *et al.*, 2017) and hierarchical ascendant clustering (Ryschawy *et al.*, 2017). The consistent co-occurrence of ecosystem services within a bundle indicates a synergy, whereas the lack of co-occurrence points to a trade-off. For example, Ryschawy *et al.* (2017) conducted a study of ecosystem service bundles associated with different livestock typologies across various departments in France.

3.5.1.2 Trade-offs and synergies with models

Models are valuable tools for evaluating and quantifying trade-offs and synergies among ecosystem services. They link driving variables with objective variables through statistical or mechanistic relationships. Driving variables are those whose changes represent specific management actions, such as land cover types or herd size. For instance, adjusting the number of animals allocated to a particular pasture may influence the amount of grass grazed or the level of carbon sequestered.

Objective variables are the model outputs and typically represent specific targets – particularly ecosystem services – to be maximized or minimized (Accatino *et al.*, 2019). Once models have been developed, they can be used in two primary ways to assess trade-offs and synergies among ecosystem services: (i) by exploring a set of scenarios, or (ii) by formulating and solving optimization problems.

Scenario exploration with models

Scenario exploration involves defining a set of scenarios characterized by different combinations of values for the driving variables – and, where relevant, different parameter values. By comparing the outputs generated under these different scenarios, it becomes possible to draw conclusions about the relationships among modelled ecosystem services, as discussed in Section 3.4. For example, Nelson *et al.* (2009) applied the InVEST model to a river basin and defined various spatial configurations for the driving variables across their scenarios. Comparing the resulting outputs allowed them to identify trade-offs and synergies among ecosystem services. Such relationships become evident when certain ecosystem service pairs change in the same direction across scenarios (indicating synergies), while others change in opposite directions (indicating trade-offs).

Multi-objective optimization

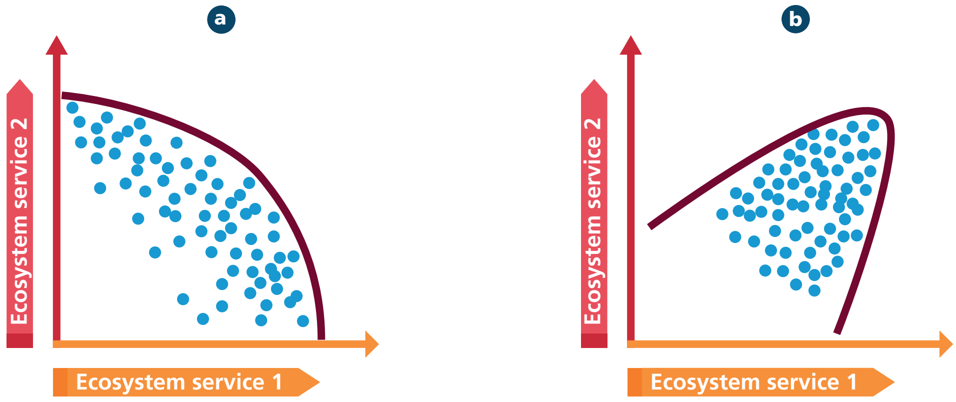
Formulating an optimization problem involves two key steps: (1) establishing the objectives to be minimized or maximized, and (2) defining constraints on the driving variables or the objectives themselves. For example, Accatino *et al.* (2019) set livestock production as a maximization objective, while placing constraints on other objectives to prevent net losses in critical ecosystem services such as crop production, carbon sequestration and timber growth. Additional constraints were applied to land-cover variables, with boundaries defined by geographical region.

Once formulated, the optimization process systematically explores combinations of driving variable values – within the defined constraints – to identify output configurations that are considered optimal.

In the case of multi-objective optimization, optimality is achieved through the calculation of the Pareto frontier (Groot and Rossing, 2011), also known as the production frontier (Vallet *et al.*, 2018). The Pareto frontier represents a set of alternative configurations where no objective can be improved without compromising another (Castelletti *et al.*, 2010). This approach is commonly used in economics to analyse trade-offs among objectives (see Figure 12; further details in Section 3.3.1.2, Interrelationships between ecosystem services). The Pareto frontier has as many dimensions as there are objectives. When there are only two objectives, the frontier can be visualized as a line; with more than two objectives, it becomes useful to examine projections of the frontier onto two-dimensional planes to evaluate pairwise relationships.

FIGURE 11

Representation of the projection of a multi-dimensional Pareto frontier on the plan dedicated to ecosystem service 1 and ecosystem service 2, representing a trade-off (a) or a synergy (b)



Note: Panel a): representation of a trade-off among the two ecosystem services, as the points of the Pareto frontier show a decreasing relationship. Panel b): representation of a synergy among the two ecosystem services, as the points of the Pareto frontier show an increasing relationship. Note that the configuration depicted in Panel b) is possible only if the Pareto frontier is computed considering at least three ecosystem services.

Source: Authors' own elaboration.

The arrangement of points on the projection of the Pareto frontier for two ecosystem services provides insight into the relationship between them. A downward trend in the projection indicates a trade-off (Figure 11a), while an upward trend suggests a synergy (Figure 11b). The optimization process – systematically exploring combinations of driving variable values to maximize or minimize objectives – can be conducted using a range of techniques. If the model is relatively simple, mathematical programming may be appropriate. However, when the model includes many variables, is spatially explicit, or features complex constraints, evolutionary algorithms may be better suited (Shi *et al.*, 2021; Groot and Rossing, 2011).

When comparing scenario exploration with multi-objective optimization, the latter offers more comprehensive information on the relationships among ecosystem services. Optimization allows for the exploration of a vast number of combinations of driving variables, while scenario analysis is typically limited to a smaller predefined set (Seppelt *et al.*, 2013). Moreover, examining the structure of the model used in the optimization process can provide valuable insights into the causal mechanisms that lead to trade-offs and synergies among ecosystem services.

3.5.1.3 Trade-offs and synergies among ecosystem services with semi-qualitative and participatory methods

Participatory and modelling approaches are valuable tools for analysing trade-offs and synergies among ecosystem services. Some methods, in particular, support the participatory construction of causal-effect networks of interactions (see Section 3.3). One such method is the "influence network framework", introduced by Crouzat *et al.* (2016). This method reveals the connections between external variables, biodiversity and various aspects of ecosystem services. The network is represented schematically, making cause-and-effect interactions explicit at a qualitative level. Bayesian belief networks (BBNs) are another type of causal network. They use probability distributions to quantify relationships among different components of a system. In BBNs, nodes represent system variables, while arrows indicate causal links. Each node is defined by a finite, discrete set of possible values, and causal relationships are described using conditional probabilities. While there is no standardized procedure for building and calibrating BBNs, they can be developed using quantitative data and expert or stakeholder knowledge, as demonstrated by Forio *et al.* (2020). By making relationships among system components explicit, these frameworks help foster shared understanding and dialogue among stakeholders. They can also support the development of innovative strategies to manage trade-offs among ecosystem services.

3.5.1.4 Trade-offs and synergies among ecosystem services with economic valuation methods

The analysis and quantification of relationships between different ecosystem services is also a focus in economic science (see e.g. Wossink and Swinton 2007; Smith *et al.* 2012; Bekele *et al.* 2013). A key method for exploring these relationships is the Pareto frontier. The fundamentals of the Pareto frontier have already been introduced in Section 3.5.1.2. However, the conceptual framing of ecosystem service relationships from an economic perspective, and the specific application of the Pareto frontier in economic valuation studies, is elaborated in Section 3.3, particularly in Section 3.3.1.2 on interrelationships between ecosystem services.

3.5.2 Meta-analysis (and literature review)

Meta-analysis aims to systematically assess the results of previous research to conclude a broader body of evidence. It most commonly involves a quantitative approach to combining findings from earlier studies to produce more generalizable results. As such, meta-analyses are often used to synthesize biophysical measurements and direct quantification of ecosystem services, as well as economic and monetary valuations. Given the significance of meta-analysis in the field of economics, a dedicated discussion is provided in Section 3.3, particularly in Section 3.3.2.8. Readers interested in meta-analysis from an economic valuation perspective are encouraged to consult that section. Meta-analysis also serves as a valuable input for modelling approaches. By aggregating findings across multiple studies, the resulting data are often more robust and widely applicable, making them especially useful for informing models.

In addition to quantitative applications, meta-analysis can also be used with more qualitative data. This includes generalizing results from studies that use sociocultural assessment methods, either to identify shared patterns within a specific context using various methodologies, or to compare findings across different contexts using the same method. Literature reviews can also be considered a type of meta-analysis when they systematically synthesize state-of-the-art knowledge on ecosystem services. These reviews may focus on individual methods or assessment approaches, or they may compare multiple approaches, as outlined in Part 3. Given the rapid growth of the ecosystem services literature, such reviews are increasingly common.

3.5.3 Scenario analysis

Scenario analysis is used to explore potential alternative future situations. These scenarios, or "alternative futures", can serve various purposes – from anticipating possible developments to challenging assumptions and broadening perspectives on current dilemmas.

Scenario analysis typically addresses long-term horizons and high levels of uncertainty. Its primary aim is not to predict precise system behaviour, but rather to describe a set of plausible events that may lead to different future outcomes. It seeks to explore structurally distinct and internally consistent futures rather than provide exact forecasts.

Scenario analysis can integrate diverse methods and approaches, including sociocultural techniques and modelling tools. For further details on the sociocultural approach to scenario development, see Section 3.2.2.3, particularly the subsection on participatory scenario development.

Broadly, scenario analysis can be structured in five steps:

1. **Define the agroecosystem under study and the parameters of interest**, such as ecosystem services or socioeconomic variables.
2. **Identify the changes affecting the current agroecosystem,** including socioeconomic, political or ecological drivers. These may include shifts in market demand, policy changes, or climate-related developments.
3. **Assess how the identified changes could influence the parameters of interest.** This step can involve either quantitative tools (e.g. modelling or value transfer methods) or qualitative approaches (e.g. sociocultural methods).
4. **Formulate plausible scenarios.** Most studies present four to six scenarios to ensure clarity and ease of communication. However, more extensive scenario sets may be used depending on the research scope.
5. **Evaluate scenario performance** in terms of relevant indicators, such as ecosystem service provision, environmental impact or farmers' livelihoods.

These steps serve as a general guide and may be adapted, reordered or merged depending on the specific objectives of the study. For instance, some studies may begin by identifying potential scenarios (step 4) and then define the agroecosystem and changes implied. Similarly, in step 2, some studies focus on regional drivers such as local policies, labour shortages or natural resource constraints, while others emphasize global trends like international trade agreements, climate change or shifts in global demand.

Finally, although scenario analysis may be rooted in a single disciplinary approach (such as modelling or sociocultural analysis), combining methods is often advantageous. A mixed-methods approach can yield a more comprehensive understanding and support more robust decision-making.

Part 4
Enabling environment and other considerations

4.1 FARMING APPROACHES WITH POTENTIAL TO PROMOTE ECOSYSTEM SERVICES

As explained in the background section (Part 1), livestock agroecosystems are very diverse due to the ecological and socioeconomic context in which they are in, as well as the degree of specialization and the intensity of management of those livestock agroecosystems. Intensity of management generally refers to the increased use of resources and inputs (i.e. agrochemicals, energy, machinery) to maximize production, which tends to substitute the ecological process (and the natural resources) that underpin agriculture.

The main purpose of agriculture is to secure the provision of food to people. In the current context, where the global population is growing and the population increasingly live in urban environments (hence, not contributing to food production), the intensification pathway has been a mainstream development in agriculture. Intensification, however, has had a toll on the provision of ecosystem services. Generally, the most intensive systems aim to maximize the provision of food (a provisioning service) at the expense of a wide range of other (usually regulating and cultural) ecosystem services. Meanwhile, more extensive systems that rely on ecological processes tend to provide a wider range of ecosystem services. Yet, on many occasions, it is acknowledged that the aim to optimize a wider range of ecosystem services compromises the maximization of food production.

To counter the trend of specialization and intensification, and address the problems associated with it, a range of farming approaches has emerged, and continues to emerge. These approaches promote the adoption of ecological principles to underpin production rather than relying on external inputs. These approaches receive different names, such as regenerative (Newton *et al.*, 2020), circular (Boer and van Ittersum, 2018; Muscat *et al.*, 2021), organic (Kareem *et al.*, 2022; IFOAM, 2020), low input practices (Sarkar *et al.*, 2020), agroforestry (Nerlich *et al.*, 2013), nature-inclusive (Runhaar, 2017; Vermunt *et al.*, 2022), agroecology (Wezel *et al.*, 2018), permaculture (Didarali and Gambiza, 2019), conservation agriculture or industrial ecology (Dumont *et al.*, 2013), among others.

Substantial fundamental differences may be found between those concepts. For instance, some focus more on agricultural practices (e.g. regenerative, organic or conservation agriculture), others take a food systems approach (e.g. circular agriculture), while others include social elements and fairness (e.g. agroecology). It is not the intention of this report to dwell on the specificities of each system and explore differences and overlaps. Other publications have attempted that before (see for example Vermunt *et al.*, 2022). Nonetheless, and in brief, the largest communality across these proposed systems is a larger reliance on ecological principles and multifunctionality. Hence, in principle, these systems foster higher potential to balance the delivery of ecosystem services (Foley *et al.*, 2005), particularly mitigating the trade-off between provisioning and the other ecosystem services, namely regulating and cultural. These proposed alternative approaches also share similar criteria of management practices by considering the ecosystem functions and services. The main systems and practices used, along with the key aspects, are summarized in Table 11. According to the focus of interest, all approaches can be applied at different scales (Xu and Mage, 2001). However, certain practices can be more suitable for a specific scale, and others can be combined and/or applied at multiple scales. In general, all these systems strive for attributes, such as diversity of crops, integration of crops and livestock, multifunctionality and resilience. Attributes such as resilience are not under the scope of this report. Hence, these are only briefly addressed in Section 4.3 (Other sustainability considerations).

4.2 ENABLING ENVIRONMENT FOR ECOSYSTEM SERVICES IN LIVESTOCK SYSTEMS

Policies, economic incentives (both public and private) and access to finance are powerful drivers shaping agricultural systems. In recent decades, these drivers have largely promoted the expansion and intensification of agricultural activity. As a result, alternative farming approaches with the potential to deliver a wider range of ecosystem services have often been overlooked or underfunded (Ripoll-Bosch and Scheonmaker, 2020). As discussed in Part 1 of this report, this has led to increased food production, but frequently at the cost of other ecosystem services and biodiversity.

At the same time, policies, economic incentives and financing mechanisms can be directed to support alternative livestock farming systems that contribute more broadly to ecosystem services. A wide range of actions and incen-

TABLE 11
Sustainable management approaches in agriculture that enhance ecosystem services

<table>
<tr><th>Approach[a]</th><th>Main systems used</th><th>Some of the principles or key aspects shared that favour ecosystem function and services</th></tr>
<tr><td>Agroecology
(e.g. Altieri, 1999 and 2002; FAO Agroecology)</td><td>Different kinds of crop, vegetable and grazing animal systems that focus on overall system health, avoiding the use of chemical products, and grounded in cultural knowledge and active human integration.</td><td rowspan="7">• Farming systems care for nature and biodiversity
• Farming systems focus on sustainable use of functional biodiversity
• Farming systems use of agroecological innovation
• Farming systems integrate crops and animals within the rotation
• Farming systems include grazing animals[b] in the system
• Farming systems increase soil cover
• Farming systems improve structure
• Farming systems reduce mechanical disturbance of soils
• Farming systems reduce fertilization of agrochemical use
• Farming systems increase perennial elements (herbs, shrubs, trees)
• Farming systems increase plant diversity
• Farming systems use grassland with plants of different functional traits, such as grasses, herbs and forbs
• Farming systems increase in animal species diversity to exploit complementarities
• Farming systems reduce weed and insect impact through crop rotation
• Farming systems enhance nutrient cycling, mainly by utilizing system components
• Farming systems use systematic approach
• Farming systems include design or planning of components and management
• Farming systems emphasize multifunctionality</td></tr>
<tr><td>Organic
(e.g. Kareem et al., 2021; IFOAM, 2008 and 2020)</td><td>Different kinds of crop, vegetable and grazing animal systems that prioritize soil health, without using agrochemical products.</td></tr>
<tr><td>Regenerative
(e.g. Newton et al., 2020; Schreefel et al., 2020)</td><td>Mixed crop and vegetable systems with grazing animal species[b] and pastures that build the foundation of the entire system.</td></tr>
<tr><td>Circular
(e.g. Boer and van Ittersum, 2018)</td><td>Other agricultural systems that prioritize rotation, nutrient cycling within the system and the reuse, recycling and reduction of residues.</td></tr>
<tr><td>Agroforestry
(e.g. Rosati et al., 2020; Nerlich et al., 2013; FAO Agroforestry)</td><td>Some agricultural systems that integrate trees (e.g. forest, arable and pasture), silvopasture (crops and trees) and agrosilvopastoral systems (crops, trees and animals).</td></tr>
<tr><td>Permaculture
(e.g. Didarali and Gambiza, 2019; Permaculture Research Institute)</td><td>Farming systems aim to allow agricultural systems to express and evolve naturally.</td></tr>
<tr><td>High Nature Value farmland
(e.g. Lomba et al., 2020)</td><td>Farming systems recognize that many European habitats and landscapes important for biodiversity conservation depend on the continuation of specific farming systems.</td></tr>
</table>

[a] This is not an exhaustive list of approaches. Also, approaches are not completely different from one another, on many occasions, overlap in approach, concepts and practices (Vermunt *et al.*, 2022);

[b] Grazing animals: refers to cattle, sheep, pigs, poultry and others.

Source: Authors' own elaboration.

tives are available to assist farmers and farming systems in conserving or adopting practices that enhance biodiversity and ecosystem service provision. While this report does not aim to provide an exhaustive list of such mechanisms, more detailed information can be found on the FAO working group on Incentives for Ecosystem Services (FAO, n.d.). The platform outlines a set of measures designed to help farmers adopt sustainable agricultural practices that benefit the environment – particularly ecosystem services – while improving long-term food security.

This section focuses on approaches to economically incentivize or compensate farmers and other land users for maintaining and developing livestock systems that deliver additional ecosystem services beyond provisioning. Incentives have received considerable attention in the literature and exist in many forms and designs. For this reason, the principles behind the commonly referenced concept of "payments for ecosystem services" are outlined here. While this concept is well established, in the European context it often takes the form of agri-environmental schemes, which follow a similar rationale. Both approaches are covered in this section.

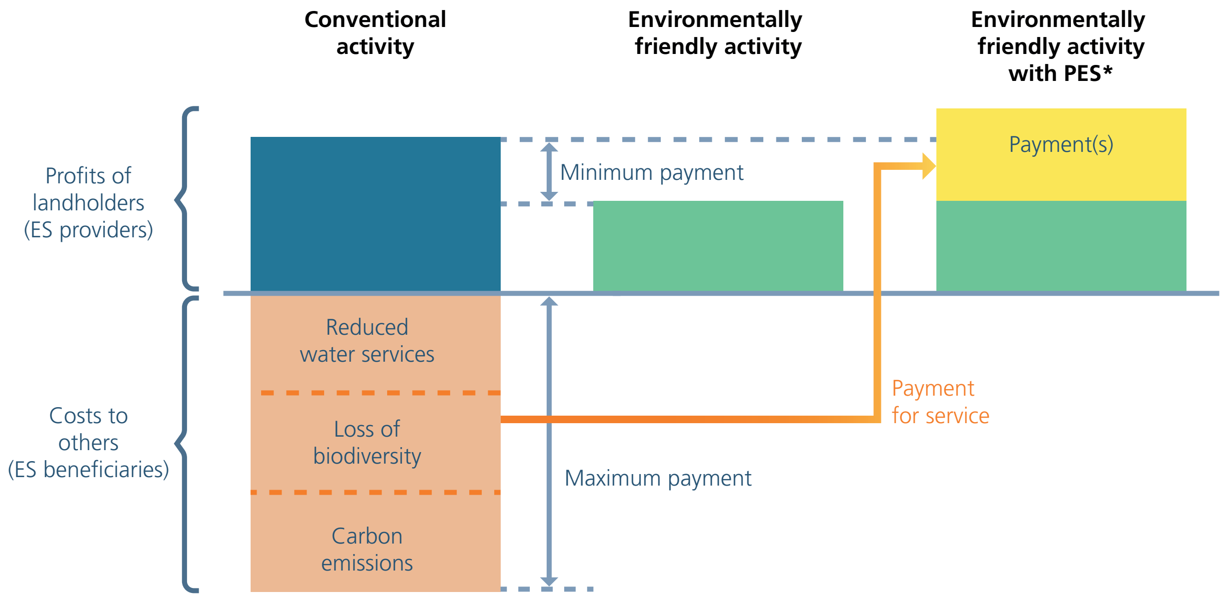
The following sections address: the concept of ecosystem services and associated agri-environmental measures (Section 4.2.1); other enabling elements such as risk mitigation, legal frameworks and knowledge transfer (Section 4.2.2); and the widely recognized case of Costa Rica's payment for ecosystem services programme (Section 4.2.3).

4.2.1 Payments for ecosystem services

There is a mismatch between the costs of ecosystem (dis) services linked to agriculture and their benefits due to market failure (see Section 3.3); the social optimum is therefore not achieved. Several tools and measures exist to correct this market failure, mainly by incentivizing landholders and other resource stewards to adopt environmentally friendly practices for protection or restoration.

Among these measures are payments for ecosystem or environmental services. As (re)defined by Wunder (2015), PES are voluntary transactions between service users and service providers, conditional on agreed rules of natural resource management for generating off-site services. Broader and more comprehensive definitions also exist, extending in some cases to include all economic incentives.

FIGURE 12
Basic idea of payment for ecosystem services



Note: * Payments for ecosystem or environmental services (PES).

Source: Engel, S. 2016. The Devil in the Detail: A practical guide on designing payments for environmental services. *International Review of Environmental and Resource Economics*, 9(1–2): 131–177. https://doi.org/10.1561/101.00000076

PES are grounded in the seminal work of Coase (1960), and – by establishing direct payments from ecosystem service users to providers – may be more cost-efficient than indirect approaches (Ferraro and Simpson, 2002; Pagiola and Platais, 2002). In contrast to the "polluter-pays principle" of environmental taxation, PES follow the "provider-gets principle." This characteristic means that many long-term environmental subsidy programmes in agriculture share core PES features, such as the US Conservation Reserve Program (Claassen *et al.*, 2008) and the agri-environment measures under the second pillar of the common agricultural policy (CAP) (Engel, 2016).

Generally, PES can be classified into three types: **Coasean PES**, which involve direct negotiation between ecosystem service beneficiaries and providers; **Pigouvian PES**, which are environmental subsidies provided by a government agency using user fees or general taxation; and **Hybrid PES**, which combine elements of both approaches (Engel, 2016).

Despite some well-documented limitations – including self-selection of participants, lack of additionality, poor administrative targeting, noncompliance, leakage, rebound effects, motivation crowding and the risk of paying for inadequate proxies rather than actual ecosystem service delivery (Wunder *et al.*, 2020) – PES remain widely used as agri-environmental policy instruments. Their continued popularity may lie in their intuitive appeal.

This logic is illustrated in Figure 12. The starting point is a situation in which a given land use (activity A – conventional) reduces ecosystem service provision. In contrast, an alternative land use (activity B – environmentally friendly) avoids such losses but entails reduced profits for farmers, who are the ecosystem service providers. If the societal benefits of switching to activity B exceed the farmers' profit losses, this shift becomes socially desirable. A portion of these societal benefits can be redistributed to compensate farmers for their loss through payments, thus ensuring their total profit exceeds that of conventional activity (Engel, 2016).

A comprehensive overview of payment for ecosystem services can be found in FAO (2011).

A well-known example of payments for ecosystem services (PES) is the programme developed and implemented in Costa Rica, which is further described in Section 4.2.3. Initiated in 1997, the programme aims to protect and restore forested areas – including those within agroforestry systems – and the ecosystem services they provide. Although the programme has a broader scope than the focus of this report on livestock agroecosystems, it offers valuable insights into how to operationalize PES through long-term commitments (in this case, more than 20 years) and has shown demonstrable success. While livestock farmers are not the programme's primary target, they are eligible to participate. Future efforts may increasingly seek to engage livestock farmers, particularly in the context of climate change mitigation and adaptation.

Agri-environmental schemes

In practice, the link between agricultural practices and the provision of ecosystem services is not always fully understood, nor is the value of different ecosystem services across varying contexts always known (see Section 2.1). For this reason, classical agri-environmental schemes (AES), arguably the most common type of PES, are typically action-based. This means that farmers are compensated for pre-defined management practices rather than for the actual delivery of ecosystem services. Although this approach does not fully align with the PES definition outlined earlier and is considered less efficient than paying for ecological outcomes (result-based AES) (Mennig and Sauer, 2019; White and Hanley, 2016), it remains the dominant form of AES due to advantages in design, programming and administration.

In general, AES – more recently also referred to as agri-environmental-climate schemes (AECS) – have been mainstreamed in agri-environmental policies globally as a means to financially incentivize farmers to undertake nature-protecting activities and to mitigate environmental harm (Batáry *et al.*, 2015). At their core, action-based schemes aim to compensate land managers for additional costs and income foregone from adopting farming practices with higher environmental and ecological quality standards. In most parts of the world, the conservation of critical natural capital and the development of specific programmes and schemes to that end are considered a legitimate task of government.

The forerunner of modern AES began in the United States of America in the 1930s with a programme aimed at protecting soil and reducing the production of surplus crops (Baylis *et al.*, 2008). In Europe – apart from some northwestern countries that implemented agri-environment programmes before any European Union (EU) regulations – most AES originated in the Agricultural Structures Regulation of 1985 (European Union Regulation 797/85), which established the legal basis for compensating farmers for income losses from adopting less intensive practices. Five years later, AES became mandatory for all EU Member States (EU Regulation 2078/92). To this day, AES remain a central element of the Rural Development pillar of the CAP, and have also been integrated into the first pillar under the CAP reform 2023, where they are referred to as eco-schemes.

Payments for ecosystem services in perspective

Payments for ecosystem services – and by extension, agri-environmental schemes – have been widely discussed in the literature as mechanisms to incentivize ecosystem services, or the management practices that enhance their provision (FAO, 2011; Wunder, 2005; Engel *et al.*, 2008; Silvestri *et al.*, 2012). Practical guidance for the design and implementation of PES is available in the work of Smith *et al.* (2013) and Engel (2016).

However, PES schemes entail several shortcomings (see Engel *et al.*, 2008, for a more detailed discussion). Two key challenges are briefly highlighted here. First, designing a scheme that accurately compensates the true value of the ecosystem services being promoted or conserved is extremely difficult. This is often linked to previously discussed issues, such as differing values placed on ecosystem services by different stakeholder groups, and the limitations or biases of the valuation methodologies used. In addition, more practical constraints – such as limited budgets for designing and financing the incentives – can also affect the scheme's effectiveness.

Second, the literature acknowledges that PES alone may not be sufficient to incentivize the conservation or enhancement of ecosystem services. Other enabling factors can be equally important. While this report does not provide an exhaustive overview of all possible mechanisms and incentives, Section 4.2.2 highlights several additional considerations for creating an enabling environment that goes beyond PES alone.

4.2.2 Additional elements to promote the enabling environment

As previously noted, PES alone may not be sufficient to conserve or enhance the provision of ecosystem services. The design of more sustainable food systems, where the provision of ecosystem services and public goods is considered, is inherently multidimensional and involves multiple actors (CFS, 2014). Thus, fostering farming systems that deliver a broad range of ecosystem services requires enabling conditions beyond PES alone. Although not a comprehensive list, several key considerations to support the creation of an enabling environment are outlined below.

Legal framework

PES schemes should be embedded in a strong and ambitious policy and legal framework. To ensure human well-being, ecosystem services must be continuously supplied – both now and in the future. This requires long-term, coherent policies underpinned by legislation that provides legitimacy and clear rules for all parties involved (i.e. ecosystem service producers and beneficiaries).

According to the Committee on World Food Security (CFS), states play a unique role in promoting sustainable agriculture and food systems, given their responsibilities in legislation, policy, public administration and the provision of public goods (CFS, 2014). States are therefore encouraged to promote an enabling policy, legal, regulatory and institutional environment, including appropriate safeguards where necessary. This, however, demands coherence, consistency and predictability across the broad set of policies, laws and regulations governing food and agriculture. Achieving such consistency is widely acknowledged as difficult (Muscat *et al.*, 2021), and implementation remains complex (European Court of Auditors, 2020).

Access to credit

Many business models based on alternative farming systems (such as those presented in Section 4.1) depend on financial incentives or subsidies outside of regular markets or are perceived as having limited repayment capacity. As a result, farmers may struggle to access credit from commercial institutions – whether for business development or for transitioning from conventional to alternative systems (Ripoll-Bosch and Schoenmaker, 2021). Sustainable financing and responsible investment are indispensable to support food system transformation and to promote diverse farming approaches that can deliver a wider range of ecosystem services and public goods (CFS, 2014).

Risk mitigation

Alternative farming systems typically rely more on ecological processes than on agrochemical inputs. The literature often contrasts this ecological approach with the conventional approach, which seeks to control environmental and production factors – ensuring optimal availability of nutrients, water and pest control (Erisman *et al.*, 2016). In contrast, alternative approaches promote resilience through biodiversity and ecosystem functioning but may be more vulnerable to climatic variability or pest outbreaks. Therefore, additional mechanisms to mitigate risk may be required.

This need has implications at several levels. For instance, the capacity to mitigate risk can influence access to credit. It also underscores the importance of a supportive policy *framework* that acknowledges the increased uncertainty associated with ecological approaches. Finally, if society demands inherently riskier farming approaches, it should also assume part of the responsibility for mitigating those risks, rather than placing the full burden on farmers.

Knowledge transfer

The field of ecosystem services and its application to livestock agroecosystems is rapidly developing. However, advances are primarily concentrated within the scientific domain, creating a gap between research and practice. There is, therefore, a pressing need to transfer this knowledge to relevant stakeholders to create an enabling environment for land users to act accordingly. Farmers, for example, receive substantial information on how to optimize agricultural performance, but most of this focuses on systems geared toward increasing marketable products (i.e. provisioning ecosystem services). In contrast, alternative systems that aim to deliver multiple ecosystem services are more diverse, less mainstream and often marginalized in terms of knowledge dissemination. As a result, there is a need to (1) generate more knowledge on these systems and their capacity to supply ecosystem services; (2) demonstrate their viability in terms of producing both marketable products and ecosystem services; and (3) ensure effective knowledge transfer to farmers willing to transition to or initiate such alternative farming approaches. The success of this transfer depends on contextual factors but may involve communicating through appropriate channels and in language that stakeholders understand (rather than scientific terminology), establishing pilot demonstration farms or living labs, and relying on trained extension agents who can serve as knowledge intermediaries for farmers.

Visualization and recognition

Many ecosystem services provided by agroecosystems have been sustained by committed farmers who apply traditional or environmentally sound practices without financial compensation – driven by intrinsic motivation (Westerink *et al.*, 2024). These farmers and their diverse systems are not always recognized, and their efforts may go unnoticed, leading to demotivation and reduced adoption of sustainable practices.

Media, policy programmes and knowledge creation and transfer should better acknowledge the heterogeneity of farmers and farming systems and actively recognize the value they bring to ecological and social well-being. According to FAO (2011), PES schemes must include the genuine participation of farmers. If their interests are not considered and they are viewed only as contributors to environmental problems – rather than as part of the solution – then PES programmes are unlikely to lead to lasting improvements.

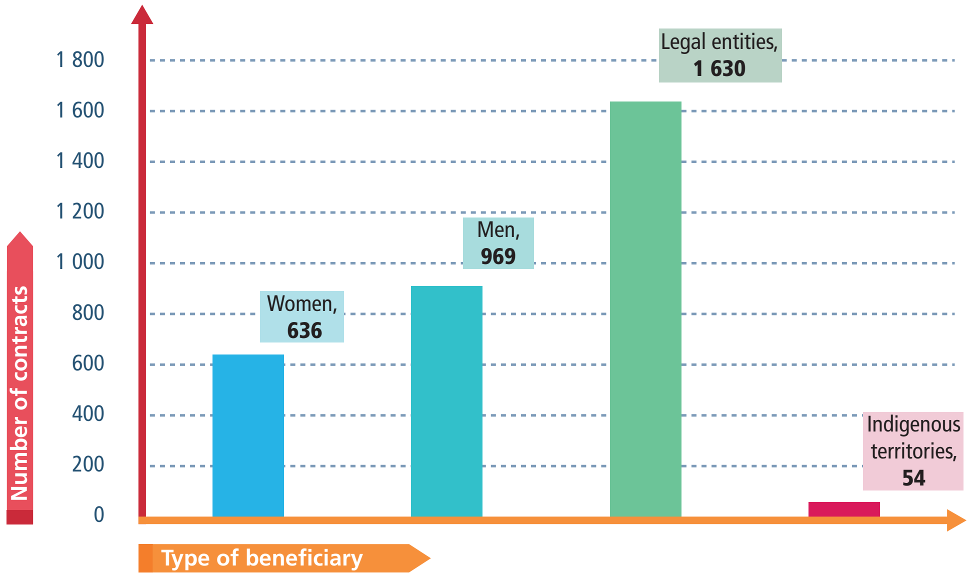
A combination of measures and incentives

Each incentive has value on its own, but the literature has shown that relying on a single measure – such as PES – is rarely sufficient. In contrast, a combination of incentives in a coherent package yields better outcomes for ecosystem services and sustainable farming.

4.2.3 Payments for ecosystem services: insights from the longstanding programme in Costa Rica

A well-known and internationally recognized example of payments for ecosystem services is Costa Rica's Environmental Services Payment Programme, which offers a participatory, monitorable and inclusive model for recognizing and distributing benefits. The text in this subsection was requested by the TAG to provide an example of an existing programme and was contributed by María Elena Herrera Ugalde, Director of Development and Marketing of Environmental Services at the National Forestry Finance Fund (FONAFIFO). References from this section are provided in Box 6.

FIGURE 13
Number of contracts issued per beneficiary type (women, men, legal entities or Indigenous territories)



Source: FONAFIFO. 2023. *Accountability report of the general management.* https://www.fonafifo.go.cr/en/servicios/pago-de-servicios-ambientales/

Since 1997, Costa Rica has implemented the Payment Programme for Environmental Services (PPES; in Spanish, Programa de Pago de Servicios Ambientales), legally established under Forest Law No. 7575 of 1996. Article 3, paragraph k, of this law defines environmental services as including biodiversity protection, water resource conservation for human, rural and hydroelectric consumption, greenhouse gas mitigation and scenic beauty. The law designates the National Forestry Finance Fund (*Fondo Nacional de Financiamiento Forestal,* or Fonafifo), part of the Ministry of Environment and Energy (*Ministerio de Ambiente y Energía,* MINAE), as the implementing agency for the programme. The primary objective of PPES is to finance small- and medium-scale producers and forest owners for the environmental goods and services their lands provide. Fonafifo has since become a recognized leader in designing and implementing financial mechanisms that sustain and enhance ecosystem services to improve Costa Ricans' quality of life.

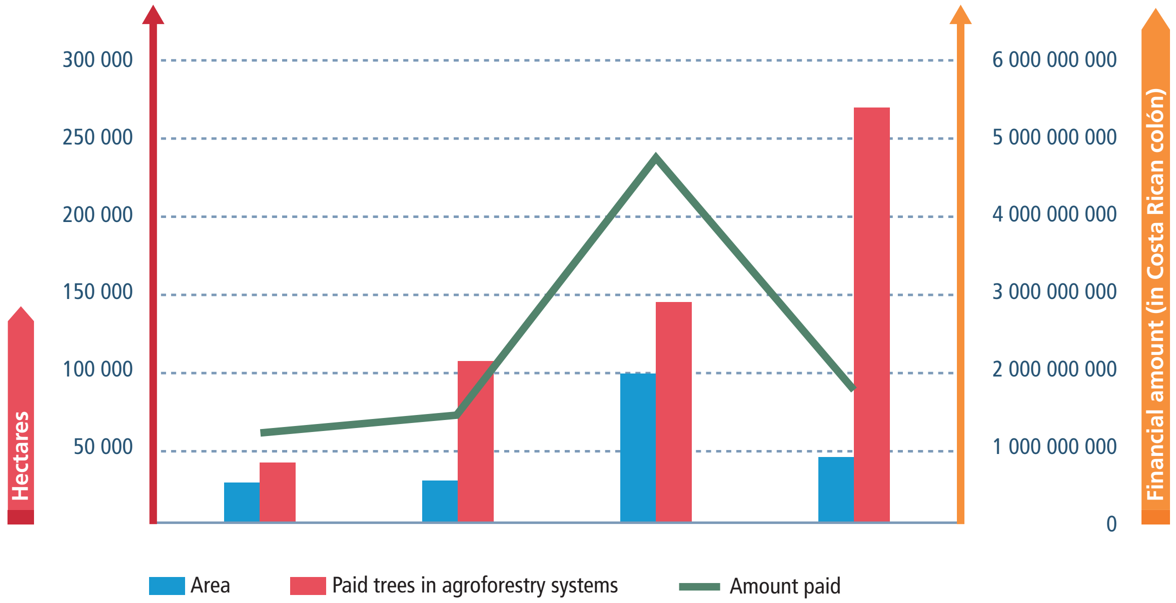
The PPES has evolved over more than 20 years, enabling Costa Rican institutions to progressively refine its implementation across five pillars: regulatory, financial, governance, policy and monitoring. Alongside the founding legislation, the programme operates through an evolving framework of legal instruments, including decrees and annual resolutions. These instruments define the target areas to be covered each year (based on specific modalities and commitments), the payment amounts for each activity, and detailed operational manuals that set out eligibility requirements and activity-specific guidelines for participating landowners.

A key success factor for the PPES has been the allocation of stable financial resources. The programme receives 3.5 percent of Costa Rica's unique fuel tax revenue, which accounts for 92 percent of its financing. As this is not sufficient to meet annual demand from applicants, Fonafifo also mobilizes complementary resources through national and international agreements, donor-funded projects and public–private or public–public partnerships. Since 1997, it has signed 120 agreements with private companies and public institutions, including initiatives linked to water use fees and other environmental services, contributing the remaining 8 percent of programme funds.

Within the country's public policy framework, the PPES plays a crucial role in maintaining more than 50 percent forest cover and supporting biodiversity conservation. It contributes to Costa Rica's commitments under the Paris Agreement, including nationally determined contributions (NDCs), carbon neutrality and decarbonization goals. As such, the programme is a central pillar in Costa Rica's environmental policy and features in national development plans and presidential agendas.

Governance of the PPES is inclusive and participatory, involving small- and medium-scale producers and landowners, private companies and Indigenous territories. Participation also extends to licensed professional foresters, who conduct technical assessments for applicants, as well as multiple state institutions, including the public property registry, the Ministry of Finance and the Office of the Comptroller General, which performs audits.

FIGURE 14
Hectares protected (left), trees paid for, and amount paid far (right)



Source: FONAFIFO. 2023. *Accountability report of the general management.* https://www.fonafifo.go.cr/en/servicios/pago-de-servicios-ambientales/

Since its inception, the PPES has established 3 289 contracts, expanding participation among diverse actors, including 636 contracts with women, as well as with men, corporations and Indigenous Peoples (Figure 13). By 2023, Fonafifo had maintained 188 500 hectares of forest and facilitated the recovery of an additional 17 120 hectares, representing a total public investment of approximately CRC 9.3 billion (USD 18.2 million).

The programme operates through two modalities (Table 12). The first modality focuses on the recovery of forest cover, implemented through three activities: commercial reforestation, natural regeneration and agroforestry systems. Table 12 presents the amounts paid per hectare, the duration of implementation for each activity and the distribution of payments over the contract period, as outlined in the 2023 programme.

The second modality concerns the maintenance of forest cover and includes two activities: general forest protection and forest protection for water resource conservation. As indicated in Table 12, contracts under this modality typically last ten years, while the duration for other activities depends on the modality and species planted, such as in the case of reforestation. Implementation is guided by a manual of procedures, which outlines the requirements each beneficiary must meet and submit to participate in the recognition scheme.

For farms to enter the programme, Fonafifo annually establishes the area to be targeted, as well as the designated economic amount per hectare or tree to be recognized. This largely depends on the budget allocated by the Ministry of Finance. The budget is distributed among the eight regional offices of the institution, proportionally to the existing average forest cover.

Admission into the programme follows prioritization criteria and an assessment matrix defined by the National System of Conservation Areas. For instance, priority is given to: (a) forests on private farms located within protected wild areas; (b) forests within Indigenous territories; (c) forests on farms situated in defined areas of conservation importance; and (d) forests on farms located within officially designated biological corridors.

The development of agroforestry systems, one of the activities promoted by the programme, has resulted in the establishment of more than half a million trees. These trees provide benefits such as shade for crops and livestock, habitats for species that assist in pest control and enhanced ecosystem function. Additionally, these efforts have contributed to the accounting of Nationally Appropriate Mitigation Actions (NAMAs), which represent internationally agreed commitments to reduce greenhouse gas emissions. This alignment has fostered collaboration between the ministries of agriculture and environment, promoting synergies across sectors and broadening stakeholder inclusion in achieving climate goals.

Fonafifo also maintains a Control and Monitoring Department to oversee farms participating in the programme. A series of technological tools is employed to enhance the robustness and transparency of formalized

TABLE 12
The two modalities and activities rewarded under the programme

Modality	Activity	Amount (USD/ha)	Distribution of the amount	Length of the contract
Recovery of forestry cover	Reforestation	1.147	50 percent, 20 percent, 15 percent, 10%, 5% (paid in the first 5 years)	10 to 16 years
		1.293		
		1.940		
	Natural regeneration	187	20% each year	5 years
	Agroforestry systems	1.6[a]	50%, 25%, 25% (paid in years 1-3-5)	5 years
		2.3[a]		
Maintenance of forestry cover	Forest protection	582	10% each year	10 years
	Protection of hydric resources	727	10% each year	10 years

Note: [a] In agroforestry systems, payment is granted for each tree planted.
Source: Authors' own elaboration.

contracts. Monitoring spans from the admission of new farms and payment disbursement to the legal processes regulating participants' rights and duties. It includes field inspections, annual site visits, beneficiary reports, use of high-resolution satellite imagery and geographic positioning systems and external audits.

Through more than 20 years of implementation, Fonafifo has engaged diverse national actors in forest conservation, while channelling direct investment into rural areas. Beyond promoting ecosystem services, the programme's impact in Indigenous territories has included the construction of roads, schools and health centres, as well as the provision of student scholarships, financial aid to older adults and daily meals for families facing extreme poverty.

Fonafifo's strategic vision aims to increase financing from diverse sources to sustain environmental services in priority areas. It seeks to protect ecosystems and biodiversity through environmental financing mechanisms, support rural development and promote a more inclusive and resilient economy for Costa Ricans, while contributing to climate change mitigation and adaptation.

4.3 CONSIDERATIONS AND LIMITS OF THIS REPORT

Terminology

The concept of ecosystem services is complex and has been widely debated (see Schröter *et al.*, 2014, or Section 1.1.2 of this report). Today, the concept is applied across a wide range of disciplines (Braat and de Groot, 2012; Jax *et al.*, 2018). As such, practitioners may use terminology different from that adopted in this report, depending on their disciplinary background and the valuation methods employed. This report was developed by a multidisciplinary team, and terminology-related issues arose throughout the drafting process, particularly when seeking to harmonize and condense information across disciplines. To promote clarity for practitioners, the introductory chapters and glossary should be consulted when interpreting the terms used.

Incomplete knowledge

The study of ecosystem services, and particularly the literature on ecosystem services in livestock agroecosystems, is relatively recent and constantly evolving. Practitioners should therefore be aware of knowledge gaps, uncertainties and the rapid pace of advancement in this field. The guidelines presented here were developed with the best available knowledge of the contributors, but we acknowledge the possibility that some relevant information or literature may have been inadvertently overlooked.

Selection of methods described

This report does not aim to provide an exhaustive list of all possible assessment methods. Rather, it focuses on those methods that the TAG considered most relevant and applicable to the livestock sector, based on their collective expertise and experience.

Classification of methods

The report seeks to guide practitioners through key concepts related to ecosystem services and the methods available for their assessment. To do so, the methods have been grouped into four broad categories: biophysical (Section 3.1), sociocultural (Section 3.2), economic (Section 3.3) and modelling (Section 3.4). These categories are not always clear-cut, and some methodologies span more than one. A selection of such cross-cutting approaches is presented in Section 3.5. Terminology may also vary across scientific disciplines, which is why each methodological chapter begins with a conceptualization section defining its scope and rationale. Where methods are closely linked across chapters or could reasonably be placed in more than one category, cross-references have been provided. We recognize that alternative classifications or chapter sequences may be preferred by some practitioners. Other inventories and typologies have been proposed in the literature (e.g. Dunford *et al.*, 2018; Wolff *et al.*, 2015; Jacobs *et al.*, 2018).

Preference for a method

The guidelines do not recommend or promote one method over another. The selection of a particular method should be based on the intended purpose of the ecosystem services assessment. Nevertheless, we believe that combining multiple methods can offer a more robust and comprehensive understanding of ecosystem services. This is supported by findings in the literature (e.g. Dunford *et al.*, 2018; Harrison *et al.*, 2018; Jacobs *et al.*, 2018), which highlight how the integration of methods allows for the inclusion of a broader range of ecosystem services and value types, facilitates engagement with diverse stakeholder groups and enhances understanding of the complex socioecological systems involved.

4.4 OTHER SUSTAINABILITY CONSIDERATIONS

4.4.1 Introduction

While the primary focus of ecosystem services in livestock agroecosystems is on the benefits that humans derive from these systems, other sustainability concerns – such as biodiversity conservation or climate change – may not be explicitly addressed within traditional ecosystem services frameworks. Nonetheless, these considerations remain vital for shaping effective policies and strategies related to the global role of livestock. When evaluating livestock agroecosystems, it is essential to ensure that system resilience is maintained through outcomes such as the preservation of diversified plant and animal genetic resources and the avoidance of actions that disrupt natural systems – for example, those causing detrimental changes to soil and water quality and flow across the landscape.

Policies and actions that contribute to the resilience of both animals and the environment, even if they fall outside the classical definition of ecosystem services, are crucial for sustainability. This section highlights several important areas that should be taken into account to support a more holistic and forward-looking approach to livestock production.

4.4.2 Animal resilience and genetics

The link between animal resilience and welfare is critical for promoting ecosystem stability through sustainable practices that conserve biodiversity and address environmental threats. Healthy ecosystems with diverse animal populations are better equipped to mitigate the effects of extreme weather events and disease outbreaks. Factors contributing to animal resilience include genetic and phenotypic diversity, as well as physiological and behavioural flexibility. The baseline for assessing resilience is the provision of adequate access to food, water and shelter for all animals.

Resilient animals are more capable of withstanding environmental fluctuations compared to more susceptible ones, increasing their chances of survival and productivity under harsh conditions. Animals that enjoy good welfare – for example, those free from disease, adequately nourished and able to adjust to thermal variation – tend to exhibit higher resilience, as they are better prepared to recover from adverse situations and respond effectively to environmental or health challenges. In contrast, animals in poor welfare conditions may have compromised resilience.

Organizations are increasingly developing detailed ethical standards and guidelines to promote humane treatment of animals and better integration of human, animal and environmental health (e.g. Provenza *et al.*, 2015). Resilient animals play a key role in maintaining ecosystem stability and function, influencing predator–prey dynamics, herbivore population control and nutrient cycling. Conversely, environmental changes – such as habitat loss or climate change – can disrupt animal populations and threaten biodiversity and ecosystem service stability.

Birds and mammals, for example, contribute to seed dispersal by consuming fruit and distributing seeds across landscapes, supporting plant regeneration and diversity. Several factors may influence ecosystem services by promoting stable animal resilience. A decline in animal and insect populations – due to biodiversity loss from habitat destruction, water pollution or rising CO_2 emissions – can reduce pollination and thereby impact food production. Animals also support nutrient cycling, particularly in diverse biomes. Decomposers such as worms, insects and microbes break down organic matter and recycle nutrients into the soil, which is essential for maintaining soil fertility and agricultural productivity.

Maintaining biodiversity and reducing environmental contamination is crucial, as healthy ecosystems rich in biodiversity provide a wider range of ecosystem services than degraded systems that prioritize only one or a few outputs (e.g. food) (Schneiders *et al.*, 2012). More balanced agroecosystems – relying on natural pest control, use of robust and locally adapted breeds, and conservation of natural resources – can reduce dependence on external inputs and veterinary treatments, which may have long-term negative effects on ecosystems and human health (Miller *et al.*, 2022; Boxall, 2004). There is an urgent need to promote the sustainable use of agrochemicals and animal health treatments in order to preserve their effectiveness and avoid both immediate and long-term harm to the environment and human well-being.

4.4.3 Environmental resilience and biodiversity

Integration of livestock in agroecosystems globally

Total global emissions from ruminant livestock have increased from 40 to 120 Tg CH_4 per year over the past century (Zhang, 2022), with further increases expected (FAO, 2018). As such, the pressure exerted by livestock on terrestrial ecosystems remains under scrutiny. Mitigation options – such as the sustainable intensification of livestock systems and adjusting the balance between intensive and extensive ruminant production based on local context – are being explored (FAO, 2018). These considerations raise the broader question of global carrying capacity and the ability of extensive livestock agroecosystems to sustainably support production. Any such potential must be examined alongside global efforts to stabilize or reduce greenhouse gas emissions from ruminants.

Preserving biodiversity contributes to ecosystem services

Ecosystem services are closely tied to the conservation of local and biome-specific biodiversity, which supports the resilience of ecosystems overall. Conserving biodiversity is essential for both human and animal health, as it underpins the provision of many ecosystem services. Healthy ecosystems act as natural regulators by enabling biological control of disease vectors and maintaining ecological balance. For instance, wetlands filter water by removing pollutants and pathogens, reducing the risk of waterborne diseases. Forests improve air quality by absorbing pollutants and serve as important carbon sinks, contributing to climate regulation. Intact habitats can also serve as barriers to zoonotic disease transmission. The resilience of healthy ecosystems enhances their capacity to withstand and recover from disturbances, including disease outbreaks.

Connecting ecosystem services, livestock and the environment

Historically, the focus on production efficiency in farming has contributed to issues such as degraded soils and loss of biodiversity. These challenges have encouraged a shift towards more integrated, systems-based approaches that balance short- and long-term ecosystem services. Globally, farmers operate under tight economic constraints and often face persistent challenges – such as pest and disease buildup – that are exacerbated by reliance on unsustainable systems. Systems optimized solely for high yields tend to generate trade-offs that compromise long-term ecosystem functions. To mitigate these trade-offs, some farmers have shifted to more complex livestock systems with diversified inputs. Integration of such systems offers an opportunity to overcome the physical and biological limitations associated with high-input farming by improving nutrient cycling, diversifying land use, rehabilitating degraded soils and exploring alternative feed sources. Successful implementation of these systems requires informed decision-making, technical support and tools tailored to diversified farming enterprises. Diversification is both a challenge and an opportunity for addressing soil degradation and biodiversity loss while ensuring the long-term sustainability and economic viability of agricultural production.

Preserving pollination as a fundamental ecosystem service

Pollination underpins both managed and natural terrestrial ecosystems, with most agricultural and wild plants depending on a variety of pollinators, including ants, bats, flies, birds, mammals and insects. Among these, bees – especially honeybees and bumblebees – are considered the most important. Managed bee populations are used to pollinate specific crops, often being transported to fields or greenhouses, whereas wild pollinators depend entirely on healthy habitats and ecosystems.

To maintain and enhance pollination services, it is essential to conserve bee populations and promote biodiverse local pollinator communities. This requires diversified cropping systems combined with suitable natural habitats that provide nectar and pollen throughout the bees' active season, as well as nesting areas. Diverse forage and the absence of resource scarcity contribute to higher resilience among pollinators, improving their ability to buffer adverse conditions such as climate change or disease outbreaks.

Food and feed production often involves pesticide use, which can pose risks to pollinators foraging in treated areas. There is a trade-off between the benefits of insecticide use in pest control and potential side effects on bees and other pollinators. Integrated Pest Management (IPM) approaches, including the use of pest thresholds to guide insecticide application only when necessary, can help reduce unnecessary exposure. Understanding pollinator behaviour, foraging activity and timing – in combination with knowledge of pest lifecycles and pesticide properties (e.g. persistence, toxicity, systemic action) – can support targeted treatment strategies that minimize risks to non-target organisms. For example, pesticide applications may be scheduled outside peak foraging times or modified to reduce exposure.

Raising awareness of the needs and benefits of pollinators, investing in capacity building on pollinators and pests, and implementing pollinator-safe pesticide use conditions can help reduce adverse effects on pollinator communities. In doing so, pollination services can be safeguarded, contributing to more sustainable crop production and resilient terrestrial ecosystems – both managed and natural.

4.4.4 Micronutrients and human health

Food from terrestrial animal-source foods (TASF) contributes approximately 21 percent of the global total caloric supply, although regional differences exist. Globally, TASF accounts for around 30 percent of protein consumption, with country-level disparities in both the supply and the specific categories of TASF available (FAO, 2023).

Animal-source foods – including meat, fish (encompassing all aquatic ASFs), eggs and dairy – provide high-quality proteins and a balanced range of essential nutrients (Beal *et al.*, 2023). Their contribution to human nutrition is significant not only because of the superior quality of these proteins, but also due to their rich and diverse provision of micronutrients.

Foods of animal origin, such as meat, eggs, milk and dairy products, deliver concentrated amounts of essential, highly bioavailable nutrients to human diets – including protein, iron, vitamin A, vitamin B_{12} and zinc – all of which are particularly important for vulnerable populations. Milk, in particular, provides macronutrients (proteins, carbohydrates and fats) that contribute to its nutritional and biological value (Claeys *et al.*, 2014), along with micronutrients such as vitamins and minerals that play critical roles in vital bodily functions (Cashman, 2006; Paparo *et al.*, 2021; Trinchese, 2021).

ASFs offer an important supplement and diversification to predominantly plant-based diets and are particularly effective in combating malnutrition and various forms of micronutrient deficiency. According to Pethick *et al.* (2023), meat supply remains critical for ensuring adequate global micronutrient intake. While access to animal-source foods remains limited in many regions, overconsumption is reported in others. Both underconsumption and overconsumption are matters of concern and pose risks to human health.

Appendices

Appendix 1
Case studies on the application of the steps to design and perform the valuation of ecosystem services

This appendix presents examples of how the valuation of ecosystem services is conducted in livestock agroecosystems. It illustrates the application of the approach outlined in Section 2.1. Roadmap for the evaluation of ecosystem services from livestock agroecosystems, including the steps that guide the assessment process, and provides practical examples of how individual methodologies described in Part 3 can be implemented.

The two case studies presented were developed by different authors and in distinct contexts. **Case study 1**, focusing on pasture-based systems in Uruguay, covers the full assessment of ecosystem services through the five steps defined in Section 2.1. **Case study 2**, based on pasture-based systems in Brazil, highlights the process of designing and selecting methods and indicators for assessing ecosystem services:

CASE STUDY 1: ECOSYSTEM SERVICES PROVIDED BY PASTURE-BASED SYSTEMS IN URUGUAY (EXAMPLE OF FIVE STEPS TAKEN)

José Paruelo,[1,2,3] Pablo Baldassini,[1,2,] Gonzalo Camba Sans,[2] Federico Gallego,[3] Luciana Staiano,[2] Hernán Dieguez,[2] Oscar Bkumetto,[1] Santiago Baeza[4]

1. Purpose of the evaluation

This study aims to evaluate the environmental performance of extensive agricultural systems in Uruguay, with particular emphasis on livestock systems. The objective is to inform the development of public policies that support biodiversity conservation and the provision of ecosystem services, highlight the contributions of pastoral systems and stimulate further research in this area. The indicators used in the evaluation were developed under the 2022 initiative Huella Ambiental Ganadera, led by the Ministries of Agriculture and Environment of Uruguay (Ministerio de Ambiente de Uruguay, 2022).

2. Framing the valuation

The livestock and agricultural-livestock areas in 2022 in Uruguay were around 11 647 million hectares, which represent 71 percent of the total area (Ministerio de Ganadería, Agricultura y Pesca, 2023). Total stock was 11.6 and 6.2 million heads of cattle and sheep, respectively (Ministerio de Ganadería, Agricultura y Pesca, 2023). Native grasslands are the main forage base for meat and wool production in Uruguay, while for dairy production, the forage base is sown pastures and forage annual crops. According to the land use cartography of the National Direction of Territorial Planning (Ministerio de Ganadería, Agricultura y Pesca, 2020) and MapBiomas (MapBiomas Uruguay, n.d.), native grasslands occupy 51 percent of the country, which is equivalent to 9.13 million hectares. Sown pastures and forage crops occupy 6.72 percent and 16.8 percent, respectively.

3. Selection of valuation frameworks and methods

For this document, the results of five indicators are presented for all *police sections* (PS) – administrative territorial divisions. These indicators can be updated annually, allowing for the analysis of temporal trends at the police section scale. In this context, livestock areas refer to natural grasslands or sown pastures.

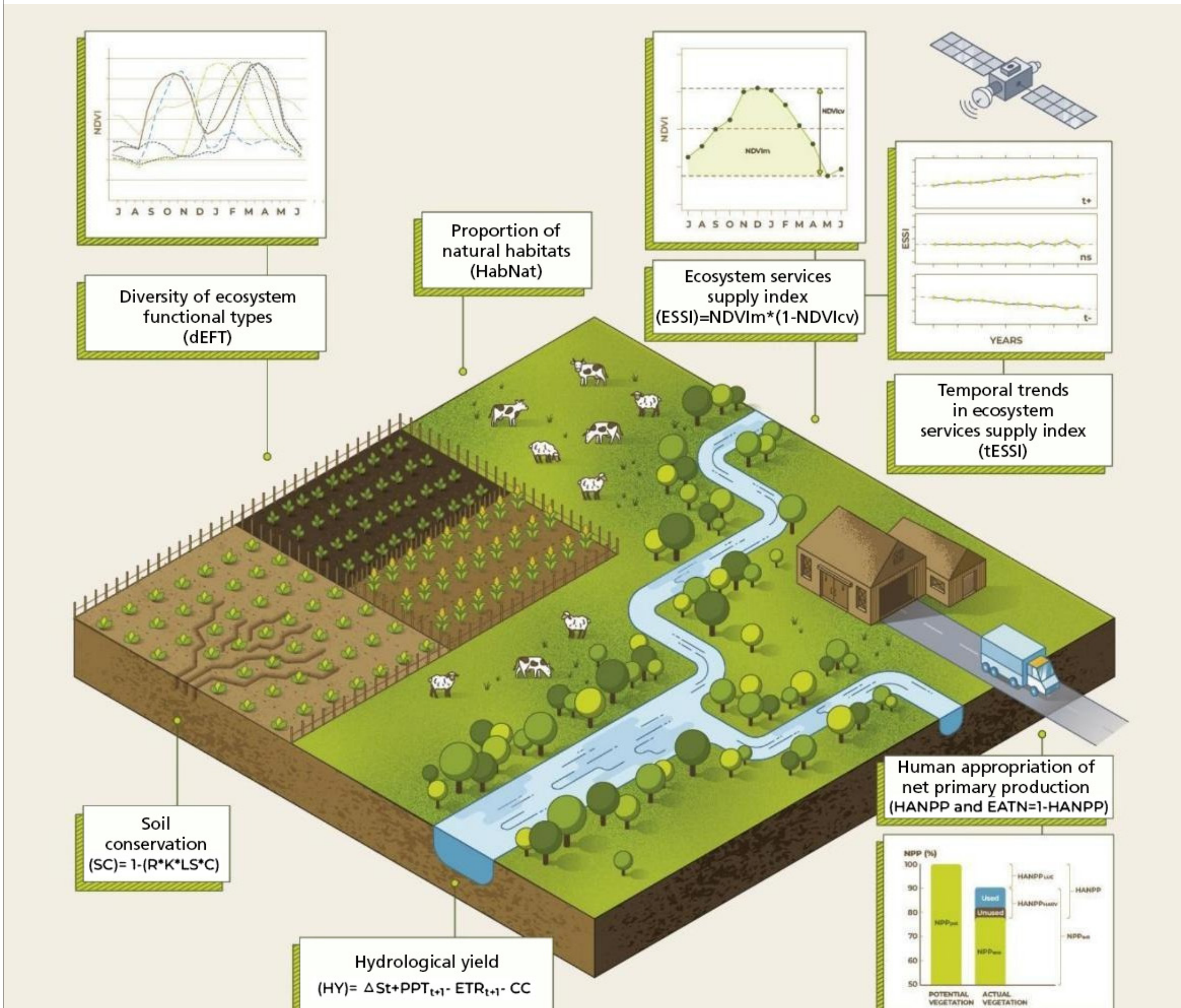
The selected indicators consist of synoptic metrics (Figure 1) derived from remote sensing data and biophysical modelling. Data were processed at the police section scale using Geographic Information Systems (QGIS and ArcGIS). The models were developed on the Google Earth Engine platform (Gorelick *et al.*, 2017). The information presented does not differentiate among livestock subsystems, such as beef cattle, sheep or dairy production.

The indicators were selected based on the following criteria: i) scientific evidence and local validation; ii) existence of a protocol for estimation or calculation, including data requirements; iii) relevance to key environmental processes (e.g. biodiversity, ecosystem service provision, carbon dynamics); iv) known limitations regarding the scalability or generalization of the indicators across the study region; and v) degree of legitimacy among stakeholders.

The latter was assessed based on the inclusion of the indicators in the national Environmental Footprint of the

[1] Área Recursos Naturales, Producción y Ambiente. Instituto Nacional de Investigación Agropecuaria (INIA). Uruguay
[2] IFEVA- Facultad de Agronomía, Universidad de Buenos Aires y CONICET, Argentina.
[3] IECA. Facultad de Ciencias, Universidad de la República, Uruguay.
[4] Departamento de Sistemas Ambientales. Facultad de Agronomía, Universidad de la República, Uruguay.

FIGURE A1.1
Indicators measured for Uruguay at the country/police section scale



Source: Authors' own elaboration.

Livestock Sector initiative (Huella Ambiental de la Ganadería) in Uruguay (Ministerio de Ambiente de Uruguay, 2022).

3.1. Proportion of natural habitats

This indicator identifies the proportion of natural environments – including native grasslands, wetlands and native forests – within each police section.

To quantify the proportion of natural habitat, land use cartography produced by MapBiomas (Baeza *et al.*, 2022) was used. The indicator is calculated as the sum of the area occupied by native grasslands, forests and wetlands divided by the total area of each police section.

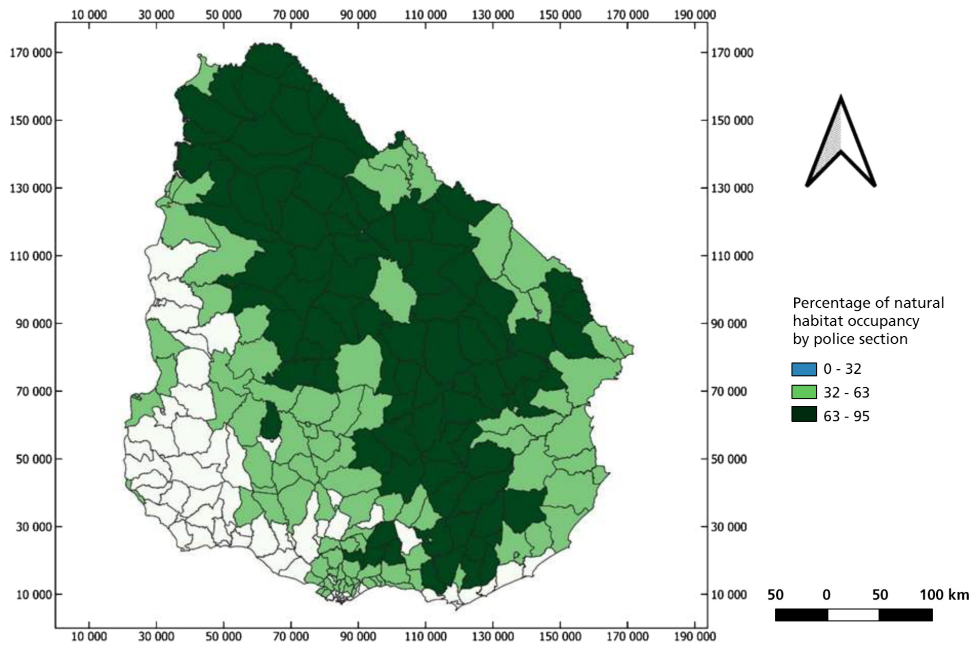
3.2. Diversity of ecosystems functional types

This indicator (dEFTs) evaluates the functional biodiversity of the landscape based on the seasonal dynamics of carbon gains (Gallego *et al.*, 2023). Ecosystem functional types (EFTs), as defined by Paruelo *et al.* (2001), represent land units with similar annual dynamics in carbon gains, characterized by three attributes: the annual average, the intra-annual coefficient of variation and the timing of peak productivity. These attributes were assessed using spectral vegetation indices, specifically the Enhanced Vegetation Index (EVI), derived from MODIS imagery (product MOD13Q1), with a spatial resolution of 230 metres and a temporal resolution of 16 days.

EFT diversity, described using the Shannon index, reflects the functional heterogeneity of anthropized land uses (i.e. areas other than natural habitats) within each police section (Alcaraz-Segura *et al.*, 2013). This approach enables the classification of land cover based on productivity, seasonality and phenology. For example, wheat crops typically peak in spring, soybean crops in summer, while double crops show a summer peak with a lower coefficient of variation.

EFTs were generated for the year 2015. Each of the

FIGURE A1.2
Percentage of the surface of each police section classified as natural habitat (natural grassland, forest, wetland)



Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.
Source: MapBiomas Uruguay. N.D. *Uruguay land use and coverage mapping platform.* [Cited 4 June 2025]. https://uruguay.mapbiomas.org/

three functional attributes was categorized into four levels, following the protocols of Alcaraz-Segura *et al.* (2013) and Gallego *et al.* (2023), resulting in 64 EFT categories. Pixels corresponding to grasslands (250 metre resolution) were excluded, and the Shannon index was calculated to describe the functional diversity of the anthropized portion of each police section (i.e. agricultural areas).

3.3. Ecosystem services supply index

The Ecosystem Services Supply Index (ESSI) is calculated as the product of the mean annual Enhanced Vegetation Index (EVI) and the complement of the EVI annual coefficient of variation (i.e. 1 – CV EVI) (Paruelo *et al.*, 2016; Staiano *et al.*, 2021). ESSI values were averaged at the police section level for the period 2020–2021. The index is based on data with a spatial resolution of 250 metres and a temporal resolution of 16 days.

ESSI provides information on the supply of regulating and supporting ecosystem services, particularly those associated with carbon dynamics.

3.4. Temporal trends of the ESSI

This indicator reflects the temporal trend of the Ecosystem Services Supply Index (ESSI) over the period 2000–2020 (Paruelo *et al.,* 2016; Staiano *et al.,* 2021). The index, referred to as tESSI, is expressed as the percentage of the police section area showing statistically significant negative trends. It serves as a measure of changes in the supply of regulating and supporting ecosystem services over time.

3.5. Human appropriation of net primary production

This indicator captures the human impact on ecosystems and characterizes land use intensity (Baeza and Paruelo, 2018; Paruelo and Sierra, 2020). Human appropriation of net primary production (HANPP) is calculated as the difference between the net primary production (NPP) in the absence of human influence – i.e. the potential vegetation (NPP_o) – and the NPP remaining after human harvest (NPP_{act}). The NPP_{rem} is estimated as the NPP of the current vegetation (NPP_{act}) minus the harvested NPP (NPP_h), which includes biomass appropriated by humans as agricultural products (e.g. grain, wood, meat) or destroyed during harvest. The formula is:

$$\mathbf{HANPP = NPP_o - NPP_{rem} = NPP_o - (NPP_{act} - NPP_h)}$$

The complement of HANPP (1 – HANPP) indicates the proportion of energy remaining available to other trophic levels, which in turn supports the supply of ecosystem services.

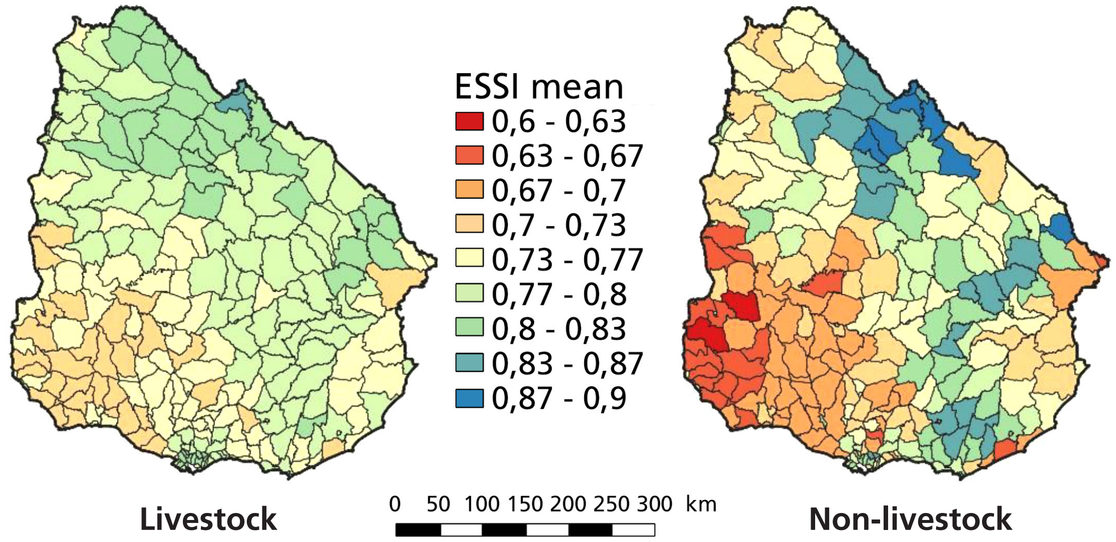
4 Implementation of valuation methods

4.1. Percentage of natural habitats

The administrative units with the largest livestock areas had between 63 and 95 percent of their surface covered by natural habitats (Figure A1.2). Most of the natural habitats within the police sections corresponded to native grasslands (Figure A1.3).

FIGURE A1.3

Average values of the Ecosystem Services Supply Index for livestock and non-livestock areas by police section for the 2020–2021 campaign. Higher ESSI values represent a greater supply of ecosystem services



Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.

Source: NASA. N.D. *Moderate Resolution Imaging Spectroradiometer (MODIS) – About MODIS.* [Cited 4 June 2025]. https://modis.gsfc.nasa.gov/about/

4.2. and 4.3. Ecosystem services supply index and its temporal trends

Figure 3 shows that the supply of ecosystem services (ES) in the portions of police sections (PS) dedicated to livestock is higher than when the entire section is considered. This pattern shifts in PSs with significant forest cover, such as Rivera. The proportion of the PS displaying negative trends is also lower in areas occupied by livestock production systems (Figure 4).

4.4. Diversity of ecosystems functional types

EFT diversity (dEFTs) showed a spatial pattern that contrasts with that of the ESSI (Figure 5), thus providing a complementary perspective on the environmental performance of agricultural systems. Greater diversity and heterogeneity are associated with a higher supply of habitats and resources. A more diverse and heterogeneous non-native system is likely to demonstrate greater resilience and more efficient resource use. EFT diversity serves as a proxy for the multifunctionality of non-natural agricultural landscapes.

4.5. Human appropriation of the net primary production

In Uruguay's agricultural systems, HANPP exceeds 40 percent when all land cover and land uses are considered (Baeza and Paruelo, 2018). Appropriation can reach 70–80 percent in areas dominated by cropland and tree plantations. In contrast, HANPP in livestock systems based on native grasslands is less than 11 percent, indicating a greater availability of energy to support biodiversity and maintain ecosystem integrity (Figure A1.6).

5. Application of the outcomes of the valuation

The analysis of the results allows for a spatially explicit description of the heterogeneity in environmental performance and ecosystem services supply across Uruguay. Environmental performance is higher in areas dominated by livestock production systems. These areas exhibit greater preservation of native habitats, higher functional diversity, stronger ecosystem service supply, and more energy available to sustain biodiversity and ecosystem functions.

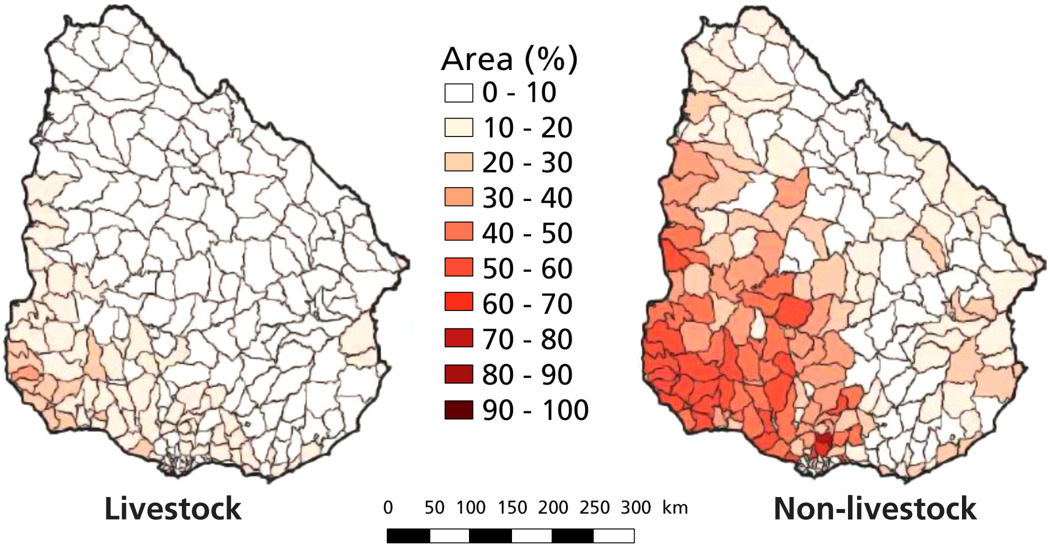
Although there is a general trend of declining ecosystem service supply, the police sections where livestock systems predominate show either neutral or only slightly negative trends. In contrast, areas dominated by non-livestock activities display a more generalized decline.

The lowest values of the various indicators were observed in different police sections, highlighting spatial variability. Differences in the spatial patterns of the indicators indicate low redundancy among them. This is further confirmed by the correlation coefficients calculated between indicators (Table 1) (Paruelo *et al.*, 2023). The strongest correlations were found between ESSI and HabNat (0.68) and between ESSI and dEFT (0.66). Despite these associations, less than 50 percent of the variability in one indicator can be explained by another, underscoring the need for a multi-indicator approach to environmental valuation.

The indicators presented at the police section level can

FIGURE A1.4

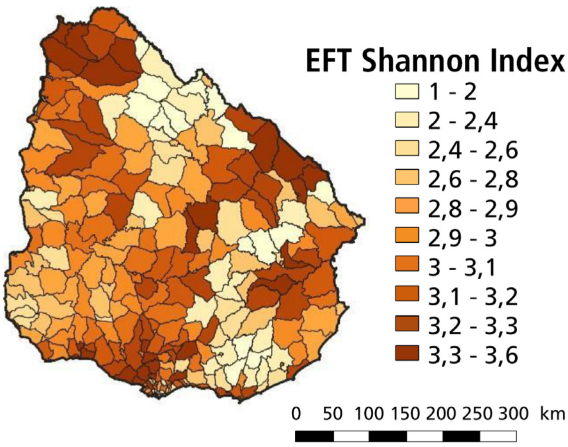
Percentage of the surface with a negative trend in the values of the Ecosystem Service Supply Index for livestock and non-livestock areas at the police section. The trend was evaluated for a time series from 2000 to 2021. The negative trend means that the supply of ecosystem services has decreased



Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.
Source: NASA. N.D. *Moderate Resolution Imaging Spectroradiometer (MODIS) – About MODIS.* [Cited 4 June 2025]. https://modis.gsfc.nasa.gov/about/

FIGURE A1.5

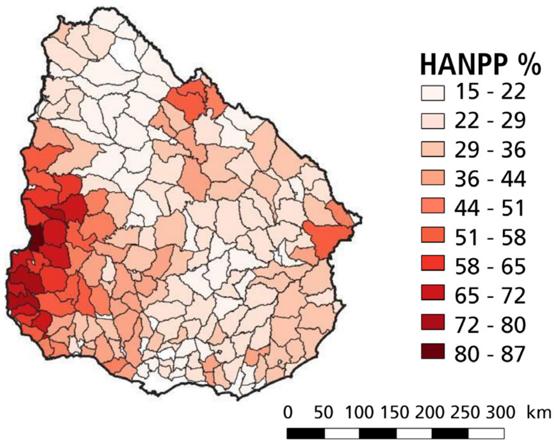
Diversity of Functional Types of Ecosystems that do not correspond to natural habitats described using the Shannon index for each police section. A higher index corresponds to greater diversity



Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.
Source: NASA. N.D. *Moderate Resolution Imaging Spectroradiometer (MODIS) – About MODIS.* [Cited 4 June 2025]. https://modis.gsfc.nasa.gov/about/

FIGURE A1.6

Percentage of appropriation of net primary production by police section. Higher levels of appropriation are associated with more intensive use of natural resources and greater human intervention



Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.
Source: NASA. N.D. *Moderate Resolution Imaging Spectroradiometer (MODIS) – About MODIS.* [Cited 4 June 2025]. https://modis.gsfc.nasa.gov/about/

also be generated at finer spatial resolutions (e.g. higher-order watersheds or individual farms) and linked to management practices and productive or economic performance. In addition to the advantages of achieving full territorial coverage and tracking temporal changes through a common protocol, synoptic indicators serve as a complement to field-based evaluations (e.g. Blumetto *et al.*, 2019), which remain an essential step in developing a comprehensive assessment of environmental performance.

The environmental performance of a territorial unit (e.g. a farm or police section), assessed across multiple dimensions, can be directly linked to the production outputs from

TABLE A1.1
Correlation among indicators at the "Police Section" level. In bold are those correlations statistically significant at p<0.05

	dEFT	HabNat	ESSI	tESSI	1-HANPP
dEFT	1				
HabNat	0.26	1			
ESSI	0.66	0.68	1		
tESSI	0.33	0.62	0.62	1	
1-HANPP	-0.26	0.27	-0.08	0.00025	1

Source: Authors' own elaboration.

that unit. In Uruguay, the combination of environmental performance indicators with existing livestock traceability systems enables the classification of products based on their environmental sustainability. This typification of products provides an effective and explicit connection to consumer behaviour, thereby offering the potential to influence consumption patterns by differentiating products according to their environmental and social attributes.

CASE STUDY 2: ECOSYSTEM SERVICES PROVIDED BY PASTURE-BASED SYSTEMS IN BRAZIL (EXAMPLE OF THREE STEPS TAKEN)

Title: Integrated production systems and metrics for ecosystem services evaluation in Brazilian livestock

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Keywords: ecosystem services framework, Brazilian livestock sustainability, policies, integrated crop-livestock-forestry.

1. Introduction

Brazil has led global beef exports since 2003 (Oliveira *et al.*, 2016). In 2023, cattle production increased by 3 percent compared to the same quarter of the previous year. Over the past decades, Brazil has achieved major scientific, technological and policy advances in sustainable livestock management. These developments include:

i. **Genetic improvement of forage cultivars** – Since the 1970s, beef production has increased more than fivefold while the total pasture area has decreased by 3 percent. Pasture area, which peaked in the 1990s, has since declined by almost 30 million hectares (Landau *et al.*, 2020). Research by Embrapa and partners has supported the development and adaptation of forage cultivars suited to Brazilian conditions, enhancing not only biomass production but also resistance to pests and diseases, as well as improving nutritional quality.

ii. **Improved livestock nutrition** – Brazilian forages tend to be low in mineral content, making supplementation essential. Supplementation of pasture-based diets is a key practice for reducing GHG emissions (Feltran-Barbieri and Féres, 2021).

iii. **Advances in animal breeding** – Selective breeding has significantly contributed to improvements in beef production. The use of expected progeny differences (EPDs), consanguinity control and embryo selection has been further enhanced by genomic tools, improving traits such as feed efficiency, carcass quality, and resistance to pests and diseases Embrapa, 2023).

iv. **Animal health and welfare** – Brazilian beef production typically involves low external input use. Awareness of animal welfare is growing among Brazilian consumers, similar to trends in developed countries (Hötlez and Vandresen, 2022). Encouragingly, more farmers are adopting "rational management" practices that simultaneously improve animal welfare and production efficiency.

v. **Low-carbon livestock policy** – The Sectoral Plan for Mitigation and Adaptation to Climate Change for the Consolidation of a Low-Carbon Economy in Agriculture (ABC Plan) was launched to help meet Brazil's NDCs. A dedicated credit line with preferential financing was created to support implementation. Between 2010 and 2020, the ABC Plan mitigated approximately 170 million tonnes of CO_2 equivalent over 52 million hectares, exceeding its target by 46.5 percent. The plan was updated in 2020 and is now known as ABC+ (2020–2030) (Brasil, 2021).

In response to increasing global demands for socioenvironmental safeguards in meat production, Brazil continues to adapt its practices to meet stricter sustainability requirements. This includes efforts to curb deforestation, which remains central to maintaining trade relations and strengthening the agricultural sector (Imaflora, 2023). Forest restoration and recovery of degraded lands are critical strategies for achieving food security and sustainability goals. The agricultural sector plays a pivotal role in this effort (Feltran-Barbieri and Féres, 2021), along with improved resource management that preserves soils, water and biodiversity. These actions contribute to reducing GHG emissions by lowering methane intensity per unit of output, avoiding deforestation and increasing soil organic carbon (Gouvello *et al.*, 2011).

A major area of progress has been the adoption of integrated production systems. These are defined as the simultaneous, rotational, or successive cultivation of different plant and animal species in the same area. Integrated systems aim to enhance resource use efficiency, reduce environmental impacts and improve productivity (Guimarães Júnior *et al.*, 2020).

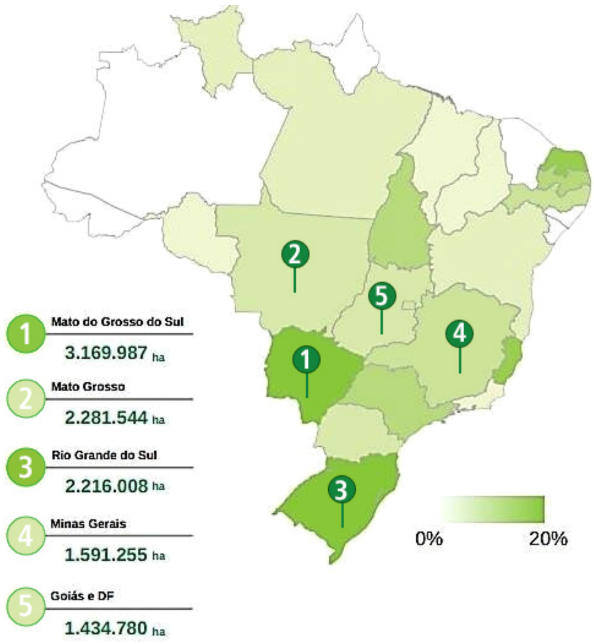
Among these, integrated crop–livestock–forestry (ICLF) systems stand out. ICLF combines agricultural, livestock and/or forestry activities on the same land – intercropped, rotated, or in succession – to generate synergistic benefits across the agroecosystem. Variants include integrated crop–livestock (ICL or agropastoral), crop–livestock–forestry (ICLF or agrosilvipastoral), livestock–forestry (ILF or silvopastoral) and crop–forestry (ICF or agrosilvicultural) systems (Brasil, 2023). While all are practised in Brazil, ICLF systems are particularly relevant for ecosystem services (ES) provision.

By 2020/2021, the area under ICLF systems was estimated at 17 million hectares, with the potential to expand to 48 million hectares, especially through the rehabilitation of degraded pastures (Embrapa, 2023). These systems can help mitigate – or even neutralize – GHG emissions, particularly where trees are present, thereby making livestock production more sustainable. Key benefits include increased carbon sequestration, improved animal comfort and welfare, enhanced forage quality and diversified income streams (Bungenstab and Almeida, 2014). Leite *et al.* (2023) offer a detailed comparison of ICLF and conventional systems, emphasizing their relative advantages and disadvantages in relation to GHG emissions and carbon sinks.

However, it is essential to disseminate these technologies and monitor their impacts. In this context, the ecosystem services approach is highly relevant, as it addresses both human needs and the imperative to protect ecosystems (MEA, 2005). The concept of ecosystem services facilitates the integration of various sectors of society and plays a key role in advancing livestock sustainability.

Despite ongoing efforts, there remain significant gaps in the availability of efficient metrics and standardized protocols to assess the impacts of sustainable livestock practices on ecosystem services (Dumont *et al.*, 2018; Figueroa *et al.*, 2022). The objective of this case study is to highlight progress made in livestock sustainability in Brazil – particularly through integrated production systems – and to propose a preliminary methodological framework for evaluating their impacts on ecosystem services.

FIGURE A1.7
Integrated crop-livestock-forestry area in the Brazilian states (2020/2021)



Note: Refer to the disclaimer on page ii for the names and boundaries used in this map.
Source: Rede ILPF. N.D. *Integrated crop–livestock–forestry network: ILPF in numbers.* [Cited 1 June 2025]. https://redeilpf.org.br/ilpf-em-numeros/#

2. Metrics to measure the impacts of livestock on ecosystem services in Brazil

A study carried out by Dumont *et al.* (2019) mentions that the various ecosystem services provided by livestock are rarely quantified. In Brazil, the situation is no different. Many advances have been made in the sustainability area, but there is still no standardization of metrics to assess and value ecosystem services provided by livestock. This is challenging since Brazil is a continental country, with wide-ranging production systems, climates, soils and management.

A literature review on cattle ranching sustainability in Latin America (Figueroa *et al.*, 2022) found that ecological analyses tend to focus on characterizing production systems and evaluating the impacts of livestock on ecosystems – particularly in relation to climate change (e.g. greenhouse gas emissions), land-use change, soil degradation (e.g. nutrient depletion and erosion), and pollution from the use of nitrogen- and phosphate-based fertilizers.

In response to these challenges, the Brazilian Carbon Neutral Beef project (Carne Carbono Neutro, or CCN) proposed a national protocol for certifying carbon-neutral meat (Zanasi *et al.*, 2020). Initiated by the Brazilian Agricultural Research Corporation (Embrapa), in partnership with MARFRIG – one of the world's largest beef producers – the CCN project supports the sustainable intensification of beef production through the adoption of ICLF or LF systems. The approach incorporates good agricultural practices, thermal comfort for animals, soil conservation, carbon sequestration through tree biomass and monitoring of pasture management (Zanasi *et al.*, 2020).

3. Preliminary methodological framework to evaluate integrated production systems impacts on ecosystem services

3.1 Methodology

The present preliminary framework to evaluate the impacts of integrated production systems on ecosystem services considers the provisioning ecosystem services (biotic and abiotic) and regulation and maintenance ecosystem services (biotic and abiotic), according to the Common International Classification of Ecosystem Services. It considers some of the biophysical methods presented in this report. The CICES classification level used was "Class".

This proposal also considers two scales: farm and regional landscape. Dumont *et al.* (2019) and Figueroa *et al.* (2022) highlight the importance of evaluating ecosystem services related to livestock at different spatial and temporal scales. At the farm scale, various ecosystem services are provided by grasslands to farmers, such as soil fertility, biological regulations and erosion control, which benefit to some extent from the functional diversity of grassland species and the duration of the pasture phase in the crop rotation. At the landscape scale, review papers (e.g. Lüscher *et al.*, 2014; Herrero-Jáuregui and Oesterheld, 2018) have quantified the main effects of grassland management and landscape heterogeneity on biodiversity. They show how farming practices interact with landscape heterogeneity in a multiscale process to shape grassland biodiversity and ecosystem services provision. "Provisioning ecosystem services" are linked to the ability of natural ecosystems to provide food, fibre and energy for human consumption through processes such as photosynthesis, nutrient sequestration and others, and are also related to semi-natural ecosystems, which involve human interference, as is the case with agriculture and livestock (Groot *et al.*, 2002). Then, water, food, wood, milk, meat and other goods are some of the examples of provisioning services. Many provisioning services are traded in markets. However, in many regions, rural households also directly depend on provisioning services for their livelihoods.

"Regulation ecosystem services" relate to the characteristic regulatory processes of ecosystems, such as maintaining air quality, climate regulation, erosion control, water purification and flow regulation, water self-purification (the process of degradation of nutrients contained in bodies of water due to sources of pollution, generally sewage), regulation of human diseases and pests in agriculture, pollination and mitigation of natural damage (MEA, 2005). In the CICES classification, the term "maintenance" was added, which relates to the maintenance of biodiversity, leaving the class called "regulation and maintenance". All these services work together to make ecosystems clean, sustainable, functional and resilient to change.

Finally, we surveyed the literature for studies that evaluate some of the ecosystem services proposed in our methodological framework, specifically regarding integrated crop–livestock–forestry impacts on Brazil, identifying the indicators and methods of evaluation (all direct biophysical methods) they applied and the scale of work. In this step, only papers published in English were considered. It is believed that the result of this step could complement and help in putting the proposed framework into practice.

3.2 Results

Figure A1.8 presents the Preliminary methodological framework to evaluate the impacts of integrated production systems on ecosystem services. The application of this framework is intended to quantify and monitor the impacts of integrated production systems on provisioning and regulation, and maintenance of ecosystem services. It can be improved and adapted to different Brazilian regions. The framework considers indicators related to soil and water conservation practices and pasture management at the farm scale, as well as actions and policies at the municipal level, which influence the increase or improvement of selected ecosystem services.

Valani *et al.* (2020) found that, from a total of 92 papers, Brazil's prominent focus of research is on soil quality and integrated crop–livestock–forest systems, with significant contributions from the central and southern regions. Embrapa was the main publishing institution, presenting one-third of the studies. Crop–livestock was the most common integrated system; ferralsols were the most common soil group, and most of the studied soils were clayey. No-tillage was the main tillage system. Most studies focused on the topsoil, assessing physical and/or chemical soil quality indicators. More emphasis is needed on biological indicators of soil quality, as well as on assessments that integrate biological, physical and chemical indicators of soil quality. Table A2.1 compiles the indicators, types of methods and scale used in some studies found in the literature that evaluate the impacts of integrated crop–livestock–forestry systems on ecosystem services in Brazil, even if not explicitly using the ecosystem services approach.

4. Reflection/take-home message

Many advances have been made in relation to Brazilian livestock sustainability. Research, technology and innovation are the factors that most contribute to Brazilian livestock's

TABLE A1.2
Ecosystem services (CICES classification), indicators, analytics, methods or reference and scale to evaluate the impacts of integrated crop-livestock-forestry on ecosystem services in Brazil

Ecosystem service	Indicators	Analytics	Scale	Reference study
Decomposition and fixing processes and their effect on soil quality	Microbial carbon and biochemical activity	Total organic carbon and soil organic matter	Farm	Zago, L.M.S., Ramalho, W.P., Caramori, S.S. et al. 2020. Biochemical indicators drive soil quality in integrated crop–livestock–forestry systems. *Agroforestry Systems*, 94: 2249–2260. https://doi-org.fao.idm.oclc.org/10.1007/s10457-020-00547-w
		Total nitrogen		
		B-glucosidase and acid phosphatase activities		
		Glycine aminopeptidase activity		
		Arylsulfatase activity		
		Phenoloxidase		
		C content of microbial biomass		
	Soil physicochemical attributes	Physicochemical properties		
	Soil biological attributes	Soil organic carbon	Farm	Assis, P.C.R., Stone, L.F., Silveira, A.L.R.D., Oliveira, J.D.M., Wruck, F.J. & Madari, B.E. 2017. Biological soil properties in integrated crop–livestock–forest systems. *Revista Brasileira de Ciência do Solo*, 41(0). https://doi.org/10.1590/18069657rbcs20160209
		Microbial biomass carbon and nitrogen		
		Soil basal respiration		
		Metabolic quotient and microbial quotient		
	Systemic soil fertility	Not mentioned		Anghinoni, I. & Vezzani, F.M. 2021. Systemic soil fertility as product of system sel-forganization resulting from management. *Revista Brasileira de Ciência do Solo*, 45. https://doi.org/10.36783/18069657rbcs20210090
Control of erosion rates/Hydrological cycle and water flow regulation	Soil physical attributes	Water clay dispersed, soil bulk density, macroporosity, microporosity, porosity	Farm	Moreira, G.M., Neves, J.C.L., Rocha, G.C., Magalhães, C.A.D.S., Farias Neto, A.L., Meneguci, J.L.P. & Fernandes, R.B.A. 2018. Physical quality of soils under a crop–livestock– forest system in the Cerrado/Amazon transition region. *Revista Árvore*, 42(2). https://doi.org/10.1590/1806- 90882018000200013
		Soil water retention curve		
		Least limiting water range		
		S index		
		Soil organic matter		
Regulation of chemical composition of atmosphere and oceans (carbon storage by plants and soil)/ filtration-sequestration-storage-accumulation by micro-organisms, algae, plants and animals	Carbon stock in eucalyptus	Diameter at breast height of tree average of the tree diameters	Farm	Morales, M.M., Tonini, H., Behling, M. & Hoshide, A.K. 2023. Eucalyptus carbon stock research in an integrated livestock–forestry system in Brazil. *Sustainability*, 15(10): 7750. https://doi.org/10.3390/su15107750
		Canopy biomass in separate compartments: leaves, dead branches, fresh branches and the tree trunk		
		Carbon dioxide equivalent stock		

Source: Authors' own elaboration.

continuous increase in efficiency to meet growing global demand, sustainably, while mitigating climate change.

The application of metrics and indicators to evaluate and monitor Brazilian livestock performance regarding the provision of ecosystem services, mainly in integrated production systems, is essential to guide more appropriate actions. This practice is essential so that Brazilian products from these systems can conquer more stringent international markets, not to mention the impacts on the environment, food security and increased quality of life for everyone, at the national level.

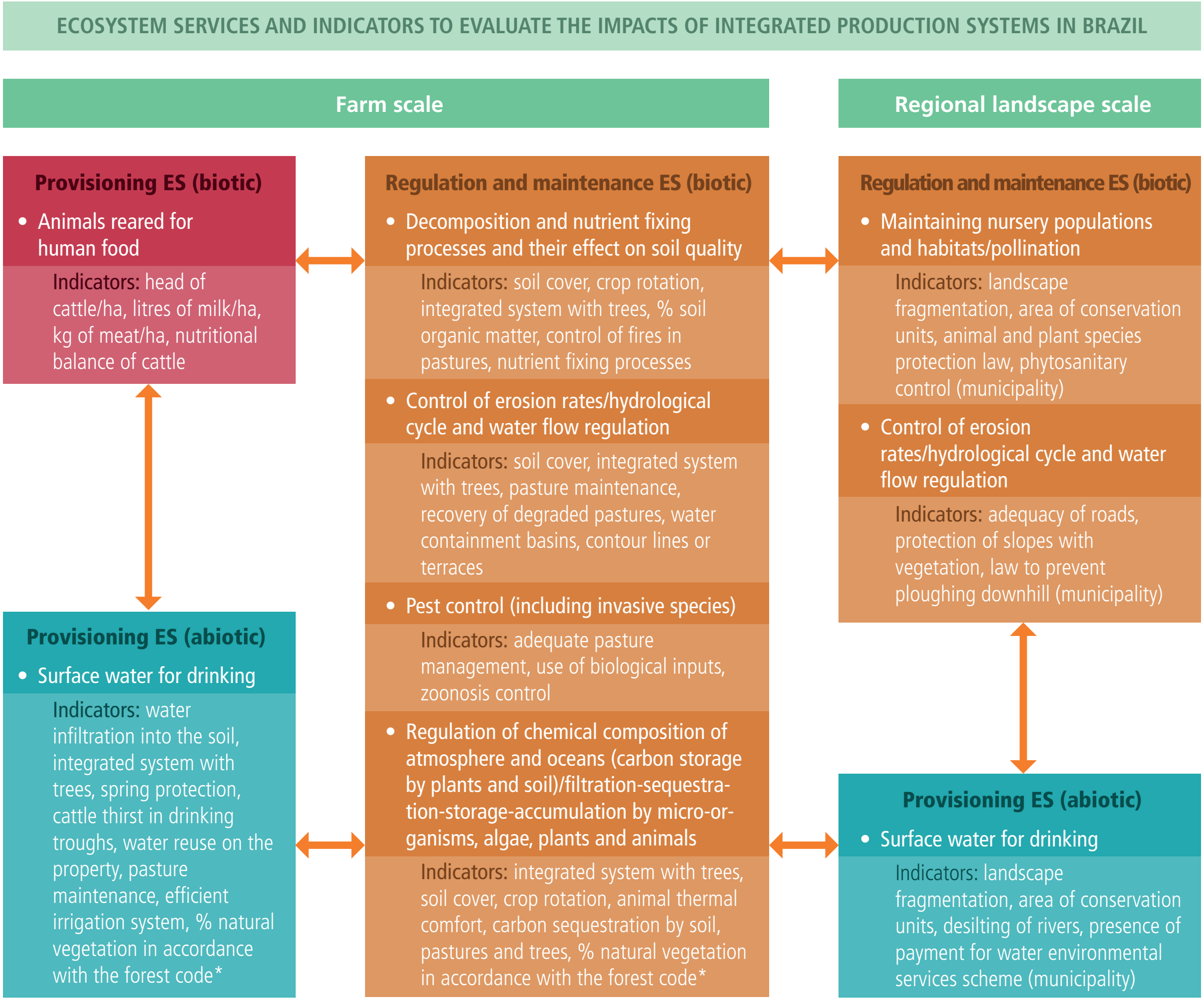
There are national policies aimed at the sustainability of Brazilian livestock, mainly the ABC+ Plan, which is part of Brazil's climate policy to reduce GHG emissions. However, livestock sustainability indicators are scarce in the literature and policies.

Of the studies surveyed in the literature that evaluated the impacts of ICLF on ecosystem services in Brazil, the majority used soil quality indicators, applying direct biophysical methods of analysis and on a farm scale. This shows the need to evaluate other ecosystem services, indicators and methods and expand the scale of study to the landscape.

The advantage of this framework is to present grouped indicators into the language of ecosystem services, using the international CICES classification and farm and landscape scale, which can obviously be improved and adapted continuously to different Brazilian regions.

FIGURE A1.8

Preliminary methodological framework to evaluate the impacts of integrated production systemson ecosystem services



Note: * Forest Code: Law 12,651/2012, which aims to protect natural vegetation on Brazilian private farms.

Source: Embrapa. N.D. *International Cooperation – Brazilian Agricultural Research Corporation.* [Cited 1 June 2025]. https://www.embrapa.br/en/international

Appendix 2
Ecosystem services classification following the CICES classification (VERSION CICES 5.1)

TABLE A2.1
Ecosystem services classification (CICES classification)

Section	Division	Group	Class	Code	Class type
Provisioning (biotic)	Biomass	Cultivated terrestrial plants for nutrition, materials or energy	Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes	1.1.1.1	*Crops by amount, type (e.g. cereals, root crops, soft fruit, etc.)*
Provisioning (biotic)	Biomass	Cultivated terrestrial plants for nutrition, materials or energy	Fibres and other materials from cultivated plants, fungi, algae and bacteria for direct use or processing (excluding genetic materials)	1.1.1.2	*Material by amount, type, use, media (land, soil, freshwater, marine)*
Provisioning (biotic)	Biomass	Cultivated terrestrial plants for nutrition, materials or energy	Cultivated plants (including fungi, algae) grown as a source of energy	1.1.1.3	*By amount, type, source*
Provisioning (biotic)	Biomass	Cultivated aquatic plants for nutrition, materials or energy	Plants cultivated by in- situ aquaculture grown for nutritional purposes	1.1.2.1	*Plants, algae by amount, type*
Provisioning (biotic)	Biomass	Cultivated aquatic plants for nutrition, materials or energy	Fibres and other materials from in-situ aquaculture for direct use or processing (excluding genetic materials)	1.1.2.2	*Plants, algae by amount, type*
Provisioning (biotic)	Biomass	Cultivated aquatic plants for nutrition, materials or energy	Plants cultivated by in- situ aquaculture grown as an energy source	1.1.2.3	*Plants, algae by amount, type*
Provisioning (biotic)	Biomass	Reared animals for nutrition, materials or energy	Animals reared for nutritional purposes	1.1.3.1	*Animals, products by amount, type (e.g. beef, dairy)*
Provisioning (biotic)	Biomass	Reared animals for nutrition, materials or energy	Fibres and other materials from reared animals for direct use or processing (excluding genetic materials)	1.1.3.2	*Material by amount, type, use, media (land, soil, freshwater, marine)*
Provisioning (biotic)	Biomass	Reared animals for nutrition, materials or energy	Animals reared to provide energy (including mechanical)	1.1.3.3	*By amount, type, source*
Provisioning (biotic)	Biomass	Reared aquatic animals for nutrition, materials or energy	Animals reared by in-situ aquaculture for nutritional purposes	1.1.4.1	*Animals by amount, type*
Provisioning (biotic)	Biomass	Reared aquatic animals for nutrition, materials or energy	Fibres and other materials from animals grown by in-situ aquaculture for direct use or processing (excluding genetic materials)	1.1.4.2	*Animals by amount, type*
Provisioning (biotic)	Biomass	Reared aquatic animals for nutrition, materials or energy	Animals reared by in-situ aquaculture as an energy source	1.1.4.3	*Animals by amount, type*
Provisioning (biotic)	Biomass	Wild plants (terrestrial and aquatic) for nutrition, materials or energy	Wild plants (terrestrial and aquatic, including fungi, algae) used for nutrition	1.1.5.1	*Plants, algae by amount, type*
Provisioning (biotic)	Biomass	Wild plants (terrestrial and aquatic) for nutrition, materials or energy	Fibres and other materials from wild plants for direct use or processing (excluding genetic materials)	1.1.5.2	*Plants, algae by amount, type*
Provisioning (biotic)	Biomass	Wild plants (terrestrial and aquatic) for nutrition, materials or energy	Wild plants (terrestrial and aquatic, including fungi, algae) used as a source of energy	1.1.5.3	*Material by type/source*
Provisioning (biotic)	Biomass	Wild animals (terrestrial and aquatic) for nutrition, materials or energy	Wild animals (terrestrial and aquatic) used for nutritional purposes	1.1.6.1	*Animals by amount, type*
Provisioning (biotic)	Biomass	Wild animals (terrestrial and aquatic) for nutrition, materials or energy	Fibres and other materials from wild animals for direct use or processing (excluding genetic materials)	1.1.6.2	*Material by type/source*

(Cont.)

TABLE A2.1 *(Cont.)*
Ecosystem services classification (CICES classification)

Section	Division	Group	Class	Code	Class type
Provisioning (biotic)	Biomass	Wild animals (terrestrial and aquatic) for nutrition, materials or energy	Wild animals (terrestrial and aquatic) used as a source of energy	1.1.6.3	*By amount, type, source*
Provisioning (biotic)	Genetic material	Genetic material from plants, algae or fungi	Seeds, spores and other plant materials collected for maintaining or establishing a population	1.2.1.1	*By species or varieties*
Provisioning (biotic)	Genetic material	Genetic material from plants, algae or fungi	Higher and lower plants (whole organisms) used to breed new strains or varieties	1.2.1.2	*By species or varieties*
Provisioning (biotic)	Genetic material	Genetic material from plants, algae or fungi	Individual genes extracted from higher and lower plants for the design and construction of new biological entities	1.2.1.3	*Material by type*
Provisioning (biotic)	Genetic material	Genetic material from animals	Animal material collected for the purposes of maintaining or establishing a population	1.2.2.1	*By species or varieties*
Provisioning (biotic)	Genetic material	Genetic material from animals	Wild animals (whole organisms) used to breed new strains or varieties	1.2.2.2	*By species or varieties*
Provisioning (biotic)	Genetic material	Genetic material from organisms	Individual genes extracted from organisms for the design and construction of new biological entities	1.2.2.3	*Material by type*
Provisioning (biotic)	Other types of provisioning service from biotic sources	Other	Other	1.3.X.X	*Use nested codes to allocate other provisioning services from living systems to appropriate Groups and Classes*
Provisioning (abiotic)	Water	Surface water used for nutrition, materials or energy	Surface water for drinking	4.2.1.1	*By amount, type, source*
Provisioning (abiotic)	Water	Surface water used for nutrition, materials or energy	Surface water used as a material (non-drinking purposes)	4.2.1.2	*By amount and source*
Provisioning (abiotic)	Water	Surface water used for nutrition, materials or energy	Freshwater surface water used as an energy source	4.2.1.3	*By amount, type, source*
Provisioning (abiotic)	Water	Surface water used for nutrition, materials or energy	Coastal and marine water used as energy source	4.2.1.4	*By amount, type, source*
Provisioning (abiotic)	Water	Ground water for used for nutrition, materials or energy	Ground (and subsurface) water for drinking	4.2.2.1	*By amount, type, source*
Provisioning (abiotic)	Water	Ground water for used for nutrition, materials or energy	Ground water (and subsurface) used as a material (non-drinking purposes)	4.2.2.2	*By amount and source*
Provisioning (abiotic)	Water	Ground water for used for nutrition, materials or energy	Ground water (and subsurface) used as an energy source	4.2.2.3	*By amount and source*
Provisioning (abiotic)	Water	Other aqueous ecosystem outputs	Other	4.2.X.X	*Use nested codes to allocate other provisioning services from non-living systems to appropriate Groups and Classes*
Regulation and maintenance (biotic)	Transformation of biochemical or physical inputs to ecosystems	Mediation of wastes or toxic substances of anthropogenic origin by living processes	Bio-remediation by micro-organisms, algae, plants and animals	2.1.1.1	*By type of living system or by waste or subsistence type*
Regulation and maintenance (biotic)	Transformation of biochemical or physical inputs to ecosystems	Mediation of wastes or toxic substances of anthropogenic origin by living processes	Filtration/sequestration/ storage/accumulation by micro-organisms, algae, plants and animals	2.1.1.2	*By type of living system, or by water or substance type*
Regulation and maintenance (biotic)	Transformation of biochemical or physical inputs to ecosystems	Mediation of nuisances of anthropogenic origin	Smell reduction	2.1.2.1	*By type of living system*
Regulation and maintenance (biotic)	Transformation of biochemical or physical inputs to ecosystems	Mediation of nuisances of anthropogenic origin	Noise attenuation	2.1.2.2	*By type of living system*
Regulation and maintenance (biotic)	Transformation of biochemical or physical inputs to ecosystems	Mediation of nuisances of anthropogenic origin	Visual screening	2.1.2.3	*By type of living system*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Control of erosion rates	2.2.1.1	*By reduction in risk, area protected*

(Cont.)

TABLE A2.1 *(Cont.)*

Ecosystem services classification (CICES classification)

Section	Division	Group	Class	Code	Class type
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Buffering and attenuation of mass movement	2.2.1.2	*By reduction in risk, area protected*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Hydrological cycle and water flow regulation (Including flood control and coastal protection)	2.2.1.3	*By depth/volumes*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Wind protection	2.2.1.4	*By reduction in risk, area protected*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Fire protection	2.2.1.5	*By reduction in risk, area protected*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection	Pollination (or "gamete" dispersal in a marine context)	2.2.2.1	*By amount and pollinator*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection	Seed dispersal	2.2.2.2	*By amount and dispersal agent*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection	Maintaining nursery populations and habitats (including gene pool protection)	2.2.2.3	*By amount and source*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Pest and disease control	Pest control (including invasive species)	2.2.3.1	*By reduction in incidence, risk, area protected by type of living system*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Pest and disease control	Disease control	2.2.3.2	*By reduction in incidence, risk, area protected by type of living system*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Regulation of soil quality	Weathering processes and their effect on soil quality	2.2.4.1	*By amount/concentration and source*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Regulation of soil quality	Decomposition and fixing processes and their effect on soil quality	2.2.4.2	*By amount/concentration and source*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Water conditions	Regulation of the chemical condition of freshwaters by living processes	2.2.5.1	*By type of living system*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Water conditions	Regulation of the chemical condition of salt waters by living processes	2.2.5.2	*By type of living system*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Atmospheric composition and conditions	Regulation of chemical composition of atmosphere and oceans	2.2.6.1	*By contribution of type of living system to amount, concentration or climatic parameter*
Regulation and maintenance (biotic)	Regulation of physical, chemical, biological conditions	Atmospheric composition and conditions	Regulation of temperature and humidity, including ventilation and transpiration	2.2.6.2	*By contribution of type of living system to amount, concentration or climatic parameter*
Regulation and maintenance (biotic)	Other types of regulation and maintenance service by living processes	Other	Other	2.3.X.X	*Use nested codes to allocate other regulating and maintenance services from living systems to appropriate Groups and Classes*
Cultural (biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Physical and experiential interactions with natural environment	Characteristics of living systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions	3.1.1.1	*By type of living system or environmental setting*
Cultural (biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Physical and experiential interactions with natural environment	Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions	3.1.1.2	*By type of living system or environmental setting*
Cultural (biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Intellectual and representative interactions with natural environment	Characteristics of living systems that enable scientific investigation or the creation of traditional ecological knowledge	3.1.2.1	*By type of living system or environmental setting*

(Cont.)

TABLE A2.1 *(Cont.)*

Ecosystem services classification (CICES classification)

Section	Division	Group	Class	Code	Class type
Cultural (biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Intellectual and representative interactions with natural environment	Characteristics of living systems that enable education and training	3.1.2.2	*By type of living system or environmental setting*
Cultural (biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Intellectual and representative interactions with natural environment	Characteristics of living systems that are resonant in terms of culture or heritage	3.1.2.3	*By type of living system or environmental setting*
Cultural (biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Intellectual and representative interactions with natural environment	Characteristics of living systems that enable aesthetic experiences	3.1.2.4	*By type of living system or environmental setting*
Cultural (biotic)	Indirect, remote, often indoor interactions with living systems that do not require presence in the environmental setting	Spiritual, symbolic and other interactions with natural environment	Elements of living systems that have symbolic meaning	3.2.1.1	*By type of living system or environmental setting*
Cultural (biotic)	Indirect, remote, often indoor interactions with living systems that do not require presence in the environmental setting	Spiritual, symbolic and other interactions with natural environment	Elements of living systems that have sacred or religious meaning	3.2.1.2	*By type of living system or environmental setting*
Cultural (biotic)	Indirect, remote, often indoor interactions with living systems that do not require presence in the environmental setting	Spiritual, symbolic and other interactions with natural environment	Elements of living systems used for entertainment or representation	3.2.1.3	*By type of living system or environmental setting*
Cultural (biotic)	Indirect, remote, often indoor interactions with living systems that do not require presence in the environmental setting	Other biotic characteristics that have a non-use value	Characteristics or features of living systems that have an existence value	3.2.2.1	*By type of living system or environmental setting*
Cultural (biotic)	Indirect, remote, often indoor interactions with living systems that do not require presence in the environmental setting	Other biotic characteristics that have a non-use value	Characteristics or features of living systems that have an option or bequest value	3.2.2.2	*By type of living system or environmental setting*
Cultural (biotic)	Other characteristics of living systems that have cultural significance	Other	Other	3.3.X.X	*Use nested codes to allocate other cultural services from living systems to appropriate Groups and Classes*

Note: The classification here provided, in Appendix 2, refers to CICES Version 5.1. At the time of writing this report, a draft version of V5.2 was released. Since V5.2 was only a draft, it was decided to keep V5.1. When using CICES, therefore, it is advised to consult the CICES website for potential updates.

Source: Haines-Young, R. & Potschin-Young, M. 2018. Common International Classification of Ecosystem Services (CICES) V5.1 and guidance on the application of the revised structure. *One Ecosystem*, 3: e27108. https://doi.org/10.3897/oneeco.3.e27108

References

References

Accatino, F., Tonda, A., Dross, C., Léger, F. & Tichit, M. 2019. Trade-offs and synergies between livestock production and other ecosystem services. *Agricultural Systems*, *168*: 58–72. https://doi.org/10.1016/j.agsy.2018.08.002

Alcamo, J., Bennett, E. M. & Millennium Ecosystem Assessment (Program), eds. 2003. *Ecosystems and human well-being: a framework for assessment*. Island Press.

Alcaraz-Segura, D., Paruelo, J., Epstein, H. & Cabello, J. 2013. Environmental and human controls of ecosystem functional diversity in temperate South America. *Remote Sensing*, 5(1): 127–154. https://doi.org/10.3390/rs5010127

Alders, R. G., Campbell, A., Costa, R., Guèye, E. F., Ahasanul Hoque, M., Perezgrovas-Garza, R., Rota, A. & Wingett, K. 2021. Livestock across the world: diverse animal species with complex roles in human societies and ecosystem services. *Animal Frontiers*, *11*(5): 20–29. https://doi.org/10.1093/af/vfab047

Alejandre, E. M., Scherer, L., Guinée, J. B., Aizen, M. A., Albrecht, M., Balzan, M. V., Bartomeus, I., Bevk, D., Burkle, L. A., Clough, Y., Cole, L. J., Delphia, C. M., Dicks, L. V., Garratt, M. P. D., Kleijn, D., Kovács-Hostyánszki, A., Mandelik, Y., Paxton, R. J., Petanidou, T. Van Bodegom, P. M. 2023. Characterization factors to assess land use impacts on pollinator abundance in life cycle assessment. *Environmental Science & Technology*, *57*(8): 3445–3454. https://doi.org/10.1021/acs.est.2c05311

Alejandre, E. M., Van Bodegom, P. M. & Guinée, J. B. 2019. Towards an optimal coverage of ecosystem services in LCA. *Journal of Cleaner Production*, *231*: 714–722.

Antle, J. M. & Stoorvogel, J. J. 2006. Predicting the supply of ecosystem services from agriculture. *American Journal of Agricultural Economics*, *88*(5): 1174–1180. https://doi.org/10.1111/j.1467-8276.2006.00929.x

Aspinall, R. & Staiano, M. 2019. Ecosystem services as the products of land system dynamics: lessons from a longitudinal study of coupled human–environment systems. *Landscape Ecology*, *34*(7): 1503–1524. https://doi.org/10.1007/s10980-018-0752-7

Atkinson, G., Bateman, I. & Mourato, S. 2014. Valuing ecosystem services and biodiversity. In *Nature in the balance: the economics of biodiversity*. OUP Oxford.

Azimi, M. S., Haghdadi, M., Riyazinia, V. & Molnár, Z. 2020. Expert understandings on rangeland ecosystem services and their sustainable management (Atrak River Basin, NE Iran). *Environmental Resources Research*, *8*(2): 109–120. https://doi.org/10.22069/ijerr.2020.5275

Baeza, S., Vélez-Martin, E., De Abelleyra, D., Banchero, S., Gallego, F., Schirmbeck, J., Veron, S., Vallejos, M., Weber, E., Oyarzabal, M., Barbieri, A., Petek, M., Guerra Lara, M., Sarrailhé, S. S., Baldi, G., Bagnato, C., Bruzzone, L., Ramos, S. & Hasenack, H. 2022. Two decades of land cover mapping in the Río de la Plata grassland region: the MapBiomas Pampa initiative. *Remote Sensing Applications: Society and Environment*, *28*: 100834. https://doi.org/10.1016/j.rsase.2022.100834

Bailey, D. W., Trotter, M. G., Tobin, C. & Thomas, M. G. 2021. Opportunities to apply precision livestock management on rangelands. *Frontiers in Sustainable Food Systems*, *5*: 611915. https://doi.org/10.3389/fsufs.2021.611915

Baldwin, T., Ritten, J. P., Derner, J. D., Augustine, D. J., Wilmer, H., Wahlert, J., Anderson, S., Irisarri, G. & Peck, D. E. 2022. Stocking rate and marketing dates for yearling steers grazing rangelands: can producers do things differently to increase economic net benefits? *Rangelands*, *44*(4): 251–257. https://doi.org/10.1016/j.rala.2022.04.002

Barbier, E. B. 2007. Valuing ecosystem services as productive inputs. *Economic Policy*, *22*(49): 178–229. https://doi.org/10.1111/j.1468-0327.2007.00174.x

Barnaud, C. & Van Paassen, A. 2013. Equity, power games, and legitimacy: dilemmas of participatory natural resource management. *Ecology and Society*, *18*(2): art21. https://doi.org/10.5751/ES-05459-180221

Barton, D. N., Harrison, P. A., Dunford, R., Gomez-Baggethun, E., Jacobs, S., Kelemen, E., Martin-Lopez, B., Antunes, P., Aszalós, R., Badea, O., Baro, F., Berry, P., Carvalho, L., Czúcz, B., Demeyer, R., Dick, J., Blanco, G. G., Garcia-Llorente, M., Giuca, R. Yli-Pelkonen, V. 2017. Integrated assessment and valuation of ecosystem services: guidelines and experiences. https://doi.org/10.13140/RG.2.2.15429.35043

Batáry, P., Dicks, L. V., Kleijn, D. & Sutherland, W. J. 2015. The role of agri-environment schemes in conservation and environmental management. *Conservation Biology*, *29*(4): 1006–1016. https://doi.org/10.1111/cobi.12536

Bateki, C. A., Cadisch, G. & Dickhoefer, U. 2019. Modelling sustainable intensification of grassland-based ruminant production systems: a review. *Global Food Security*, *23*: 85–92. https://doi.org/10.1016/j.gfs.2019.04.004

Baylis, K., Peplow, S., Rausser, G. & Simon, L. 2008. Agri-environmental policies in the EU and United States: a comparison. *Ecological Economics*, *65*(4): 753–764. https://doi.org/10.1016/j.ecolecon.2007.07.034

Beal, T., Gardner, C. D., Herrero, M., Iannotti, L. L., Merbold, L., Nordhagen, S. & Mottet, A. 2023. Friend or foe? The role of animal-source foods in healthy and environmentally sustainable diets. *The Journal of Nutrition*, *153*(2): 409–425. https://doi.org/10.1016/j.tjnut.2022.10.016

Bekele, E. G., Lant, C. L., Soman, S. & Misgna, G. 2013. The evolution and empirical estimation of ecological-economic production possibilities frontiers. *Ecological Economics*, *90*: 1–9. https://doi.org/10.1016/j.ecolecon.2013.02.012

Belward, A. S. & Skøien, J. O. 2015. Who launched what, when and why: trends in global land-cover observation capacity from civilian Earth observation satellites. *ISPRS Journal of Photogrammetry and Remote Sensing*, *103*: 115–128. https://doi.org/10.1016/j.isprsjprs.2014.03.009

Benavides, R. A. M., Gaona, R. C., Atzori, A. S., Sánchez, L. F. & Guerrero, H. S. 2023. Application of a system dynamics model to evaluate the implementation of payment for environmental services as a reconversion mechanism in high mountain farming. *Ecological Modelling*, *484*: 110469. https://doi.org/10.1016/j.ecolmodel.2023.110469

Bennett, E. M., Peterson, G. D. & Gordon, L. J. 2009. Understanding relationships among multiple ecosystem services. *Ecology Letters*, *12*(12): 1394–1404. https://doi.org/10.1111/j.1461-0248.2009.01387.x

Berg, H., Dang, S. & Tam, N. T. 2023. Assessing stakeholders' preferences for future rice farming practices in the Mekong Delta, Vietnam. *Sustainability*, *15*(14): 10873. https://doi.org/10.3390/su151410873

Bernués, A., Alfnes, F., Clemetsen, M., Eik, L. O., Faccioni, G., Ramanzin, M., Ripoll-Bosch, R., Rodríguez-Ortega, T. & Sturaro, E. 2019. Exploring social preferences for ecosystem services of multifunctional agriculture across policy scenarios. *Ecosystem Services*, *39*: 101002. https://doi.org/10.1016/j.ecoser.2019.101002

Bernués, A., Rodríguez-Ortega, T., Ripoll-Bosch, R. & Alfnes, F. 2014. Socio-cultural and economic valuation of ecosystem services provided by Mediterranean mountain agroecosystems. *PLoS ONE*, *9*(7): e102479. https://doi.org/10.1371/journal.pone.0102479

Bernués, A., Tello-García, E., Rodríguez-Ortega, T., Ripoll-Bosch, R. & Casasús, I. 2016. Agricultural practices, ecosystem services and sustainability in High Nature Value farmland: unraveling the perceptions of farmers and nonfarmers. *Land Use Policy*, *59*: 130–142. https://doi.org/10.1016/j.landusepol.2016.08.033

Berrio-Giraldo, L., Villegas-Palacio, C. & Arango-Aramburo, S. 2021. Understating complex interactions in socio-ecological systems using system dynamics: a case in the tropical Andes. *Journal of Environmental Management*, *291*: 112675. https://doi.org/10.1016/j.jenvman.2021.112675

Bettencourt, E. M. V., Tilman, M., Narciso, V., Carvalho, M. L. D. S. & Henriques, P. D. D. S. 2015. *The livestock roles in the wellbeing of rural communities of Timor-Leste*. https://doi.org/10.22004/AG.ECON.212441

Blumetto, O. 2022. Los agroecosistemas ganaderos, importante hábitat para las aves: análisis cualitativo del efecto del manejo productivo en especies prioritarias para la conservación en Uruguay. *Recursos Rurais*, *18*: 5–15. https://doi.org/10.15304/rr.id8567

Blumetto, O., Castagna, A., Cardozo, G., García, F., Tiscornia, G., Ruggia, A., Scarlato, S., Albicette, M. M., Aguerre, V. & Albin, A. 2019. Ecosystem Integrity Index, an innovative environmental evaluation tool for agricultural production systems. *Ecological Indicators*, *101*: 725–733. https://doi.org/10.1016/j.ecolind.2019.01.077

Boer, I. J. M. de & Ittersum, M. K. van. 2018. *Circularity in agricultural production.* Mansholt lecture 2018. Wageningen, Wageningen University & Research. https://library.wur.nl/WebQuery/wurpubs/547719

Bond, W. J. 2019. Vertebrate herbivory and open ecosystems. In W. J. Bond, *Open ecosystems*, pp. 121–140. Oxford, Oxford University Press. https://doi.org/10.1093/oso/9780198812456.003.0008

Boone, R. B. & Galvin, K. A. 2014. Simulation as an approach to social-ecological integration, with an emphasis on agent-based modelling. In M. J. Manfredo, J. J. Vaske, A. Rechkemmer & E. A. Duke, eds. *Understanding society and natural resources*, pp. 179–202. Dordrecht, Springer Netherlands. https://doi.org/10.1007/978-94-017-8959-2_9

Boone, R. B., Galvin, K. A., BurnSilver, S. B., Thornton, P. K., Ojima, D. S. & Jawson, J. R. 2011. Using coupled simulation models to link pastoral decision making and ecosystem services. *Ecology and Society*, *16*(2): art6. https://doi.org/10.5751/ES-04035-160206

Booth, P., Turnley, J., Law, S., Ma, J. & boyd, J. 2014. Implementation of EcoAIMTM – A multi-objective decision support tool for ecosystem services at Department of Defense installations.

Boxall, A.B.A. 2004. The environmental side effects of medication: how are human and veterinary medicines in soils and water bodies affecting human and environmental health? *EMBO Reports*, *5*(12): 1110–1116. https://doi.org/10.1038/sj.embor.7400307

Boyd, J. & Banzhaf, S. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, *63*(2–3): 616–626. https://doi.org/10.1016/j.ecolecon.2007.01.002

Braat, L.C. & De Groot, R. 2012. The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. *Ecosystem Services*, *1*(1): 4–15. https://doi.org/10.1016/j.ecoser.2012.07.011

Brander, L.M., Bräuer, I., Gerdes, H., Ghermandi, A., Kuik, O., Markandya, A., Navrud, S., Nunes, P.A.L.D., Schaafsma, M., Vos, H. & Wagtendonk, A. 2012. Using meta-analysis and GIS for value transfer and scaling up: valuing climate change induced losses of European wetlands. *Environmental and Resource Economics*, *52*(3): 395–413. https://doi.org/10.1007/s10640-011-9535-1

Brander, L.M., De Groot, R., Schägner, J.P., Guisado-Goñi, V., Van 't Hoff, V., Solomonides, S., McVittie, A., Eppink, F., Sposato, M., Do, L., Ghermandi, A., Sinclair, M. & Thomas, R. 2024. Economic values for ecosystem services: a global synthesis and way forward. *Ecosystem Services*, *66*: 101606. https://doi.org/10.1016/j.ecoser.2024.101606

Brandle, J.R., Hodges, L. & Zhou, X.H. 2004. Windbreaks in North American agricultural systems. *Agroforestry Systems*, *61–62*(1–3): 65–78. https://doi.org/10.1023/B:AGFO.0000028990.31801.62

Bredin, Y.K., Lindhjem, H., Van Dijk, J. & Linnell, J.D.C. 2015. Mapping value plurality towards ecosystem services in the case of Norwegian wildlife management: a Q analysis. *Ecological Economics*, *118*(C): 198–206. https://doi.org/10.1016/j.ecolecon.2015.07.005

Bringezu, S. & Moriguchi, Y. 2018. Material flow analysis. In: Bartelmus, P. & Seifert, E.K., eds. *Green accounting*, 1st ed., pp. 149–166. Routledge. https://doi.org/10.4324/9781315197715-6

Briones, M.J.I. & Schmidt, O. 2017. Conventional tillage decreases the abundance and biomass of earthworms and alters their community structure in a global meta-analysis. *Global Change Biology*, *23*(10): 4396–4419. https://doi.org/10.1111/gcb.13744

Brown, M.T. & Ulgiati, S. 2016. Emergy assessment of global renewable sources. *Ecological Modelling*, *339*: 148–156. https://doi.org/10.1016/j.ecolmodel.2016.03.010

Brück, M., Fischer, J., Law, E., Schultner, J. & Abson, D. 2023. Drivers of ecosystem service specialization in a smallholder agricultural landscape of the Global South: a case study in Ethiopia. *Ecology and Society*, *28*(3): art1. https://doi.org/10.5751/ES-14185-280301

Burns, J.G., Eory, V., Butler, A., Simm, G. & Wall, E. 2022. Review: preference elicitation methods for appropriate breeding objectives. *Animal*, *16*(6): 100535. https://doi.org/10.1016/j.animal.2022.100535

Campbell, E.T. & Brown, M.T. 2012. Environmental accounting of natural capital and ecosystem services for the US National Forest System. *Environment, Development and Sustainability*, *14*(5): 691–724. https://doi.org/10.1007/s10668-012-9348-6

Cárdenas-Mamani, Ú. & Perrotti, D. 2022. Understanding the contribution of ecosystem services to urban metabolism assessments: an integrated framework. *Ecological Indicators*, *136*: 108593. https://doi.org/10.1016/j.ecolind.2022.108593

Cashman, K.D. 2006. Milk minerals (including trace elements) and bone health. *International Dairy Journal*, *16*(11): 1389–1398. https://doi.org/10.1016/j.idairyj.2006.06.017

Castelletti, A., Lotov, A.V. & Soncini-Sessa, R. 2010. Visualization-based multi-objective improvement of environmental decision-making using linearization of response surfaces. *Environmental Modelling & Software*, *25*(12): 1552–1564. https://doi.org/10.1016/j.envsoft.2010.05.011

Caussade, S., Ortúzar, J.D.D., Rizzi, L.I. & Hensher, D.A. 2005. Assessing the influence of design dimensions on stated choice experiment estimates. *Transportation Research Part B: Methodological*, *39*(7): 621–640. https://doi.org/10.1016/j.trb.2004.07.006

CFS. 2014. *Principles for responsible investment in agriculture and food systems*. https://www.fao.org/3/a-au866e.pdf

Chapin, F.S., Matson, P.A. & Vitousek, P.M. 2011. *Principles of terrestrial ecosystem ecology*. Springer New York. https://doi.org/10.1007/978-1-4419-9504-9

Charteris, A.F., Harris, P., Marsden, K.A., Harris, I.M., Guo, Z., Beaumont, D.A., Taylor, H., Sanfratello, G., Jones, D.L., Johnson, S.C.M., Whelan, M.J., Howden, N., Sint, H., Chadwick, D.R. & Cárdenas, L.M. 2021. Within-field spatial variability of greenhouse gas fluxes from an extensive and intensive sheep-grazed pasture. *Agriculture, Ecosystems & Environment*, *312*: 107355. https://doi.org/10.1016/j.agee.2021.107355

Chen, X., Lupi, F., An, L., Sheely, R., Viña, A. & Liu, J. 2012. Agent-based modelling of the effects of social norms on enrollment in payments for ecosystem services. *Ecological Modelling*, *229*: 16–24. https://doi.org/10.1016/j.ecolmodel.2011.06.007

Christie, M., Fazey, I., Cooper, R., Hyde, T. & Kenter, J.O. 2012. An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. *Ecological Economics*, *83*: 67–78. https://doi.org/10.1016/j.ecolecon.2012.08.012

CICES. 2025. *Towards a common classification of ecosystem services*. https://cices.eu/

Ciftcioglu, G.C. 2020. Using a combination of Q-methodology and survey-based approach for assessing forest ecosystem services of Five Finger Mountains in Northern Cyprus. *Sustainability Science*, *15*(6): 1789–1805. https://doi.org/10.1007/s11625-020-00824-8

Claassen, R., Cattaneo, A. & Johansson, R. 2008. Cost-effective design of agri-environmental payment programs: U.S. experience in theory and practice. *Ecological Economics*, *65*(4): 737–752. https://doi.org/10.1016/j.ecolecon.2007.07.032

Claeys, W.L., Verraes, C., Cardoen, S., De Block, J., Huyghebaert, A., Raes, K., Dewettinck, K. & Herman, L. 2014. Consumption of raw or heated milk from different species: an evaluation of the nutritional and potential health benefits. *Food Control*, *42*: 188–201. https://doi.org/10.1016/j.foodcont.2014.01.045

Coase, R.H. 1960. The problem of social cost. *The Journal of Law & Economics*, *3*: 1–44. https://doi.org/10.1086/674872

Commission on Genetic Resources for Food and Agriculture, ed. 2010. *The second report on the state of the world's plant genetic resources for food and agriculture*. Commission on Genetic Resources for Food and Agriculture, Food and Agriculture Organization of the United Nations.

Conway, G.R. 1987. The properties of agroecosystems. *Agricultural Systems*, *24*(2): 95–117. https://doi.org/10.1016/0308-521X(87)90056-4

Cooper, T., Hart, K. & Baldock, D. 2009. *The provision of public goods through agriculture in the European Union*. Institute for European Environmental Policy, London.

Coscieme, L., Pulselli, F.M., Marchettini, N., Sutton, P.C., Anderson, S. & Sweeney, S. 2014. Emergy and ecosystem services: a national biogeographical assessment. *Ecosystem Services*, *7*: 152–159. https://doi.org/10.1016/j.ecoser.2013.11.003

Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. & Van Den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature*, *387*(6630): 253–260. https://doi.org/10.1038/387253a0

Cozim-Melges, F., Ripoll-Bosch, R., Veen, G.F., Oggiano, P., Bianchi, F.J.J.A., Van Der Putten, W.H. & Van Zanten, H.H.E. 2024. Farming practices to enhance biodiversity across biomes: a systematic review. *npj Biodiversity*, *3*(1): 1. https://doi.org/10.1038/s44185-023-00034-2

Crossman, N.D. & Bryan, B.A. 2009. Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality. *Ecological Economics*, *68*(3): 654–668. https://doi.org/10.1016/j.ecolecon.2008.05.003

Crossman, N.D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., Drakou, E.G., Martín-Lopez, B., McPhearson, T., Boyanova, K., Alkemade, R., Egoh, B., Dunbar, M.B. & Maes, J. 2013. A blueprint for mapping and modelling ecosystem services. *Ecosystem Services*, *4*: 4–14. https://doi.org/10.1016/j.ecoser.2013.02.001

Crouzat, E., Martín-López, B., Turkelboom, F. & Lavorel, S. 2016. Disentangling trade-offs and synergies around ecosystem services with the influence network framework: illustration from a consultative process over the French Alps. *Ecology and Society*, *21*(2): art32. https://doi.org/10.5751/ES-08494-210232

Crouzat, E., Mouchet, M., Turkelboom, F., Byczek, C., Meersmans, J., Berger, F., Verkerk, P.J. & Lavorel, S. 2015. Assessing bundles of ecosystem services from regional to landscape scale: insights from the French Alps. *Journal of Applied Ecology*, *52*(5): 1145–1155. https://doi.org/10.1111/1365-2664.12502

Daily, G.C., ed. 1997. *Nature's services: societal dependence on natural ecosystems*. Island Press.

De Araujo Barbosa, C.C., Atkinson, P.M. & Dearing, J.A. 2015. Remote sensing of ecosystem services: a systematic review. *Ecological Indicators*, *52*: 430–443. https://doi.org/10.1016/j.ecolind.2015.01.007

De Groot, R.S., Wilson, M.A. & Boumans, R.M.J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, *41*(3): 393–408. https://doi.org/10.1016/S0921-8009(02)00089-7

De Oliveira, L.E.C. & Berkes, F. 2014. What value São Pedro's procession? Ecosystem services from local people's perceptions. *Ecological Economics*, *107*: 114–121. https://doi.org/10.1016/j.ecolecon.2014.08.008

De Oliveira Silveira, E.M.D.O., Terra, M.D.C.N.S., Ter Steege, H., Maeda, E.E., Acerbi Júnior, F.W. & Scolforo, J.R.S. 2019. Carbon-diversity hotspots and their owners in Brazilian southeastern Savanna, Atlantic Forest and Semi-Arid Woodland domains. *Forest Ecology and Management*, *452*: 117575. https://doi.org/10.1016/j.foreco.2019.117575

De Santiago, M.F., Barrios, M., D'Anatro, A., García, L.F., Mailhos, A., Pompozzi, G., Rehermann, S., Simó, M., Tesitore, G., Teixeira De Mello, F., Valtierra, V. & Blumetto, O. 2022. From theory to practice: can LEAP/FAO biodiversity assessment guidelines be a useful tool for knowing the environmental status of livestock systems? *Sustainability*, *14*(23): 16259. https://doi.org/10.3390/su142316259

Dernat, S., Dumont, B. & Vollet, D. 2023. La Grange®: a generic game to reveal trade-offs and synergies among stakeholders in livestock farming areas. *Agricultural Systems*, *209*: 103685. https://doi.org/10.1016/j.agsy.2023.103685

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J.R., Arico, S., Báldi, A., Bartuska, A., Baste, I.A., Bilgin, A., Brondizio, E., Chan, K.M., Figueroa, V.E., Duraiappah, A., Fischer, M., Hill, R. Zlatanova, D. 2015. The IPBES conceptual framework – connecting nature and people. *Current Opinion in Environmental Sustainability*, *14*: 1–16. https://doi.org/10.1016/j.cosust.2014.11.002

Díaz, S., Settele, J., Brondízio, E.S., Ngo, H.T., Agard, J., Arneth, A., Balvanera, P., Brauman, K.A., Butchart, S.H.M., Chan, K.M.A., Garibaldi, L.A., Ichii, K., Liu, J., Subramanian, S.M., Midgley, G.F., Miloslavich, P., Molnár, Z., Obura, D., Pfaff, A. Zayas, C.N. 2019. Pervasive human-driven decline of life on Earth points to the need for transformative change. *Science*, *366*(6471): eaax3100. https://doi.org/10.1126/science.aax3100

Dietz, T., Stern, P.C. & Dan, A. 2009. How deliberation affects stated willingness to pay for mitigation of carbon dioxide emissions: an experiment. *Land Economics*, *85*(2): 329–347. https://doi.org/10.3368/le.85.2.329

Druille, M., Oyarzabal, M. & Oesterheld, M. 2019. Radiation use efficiency of forage resources: a meta-analysis. *Agronomy Journal*, *111*(4): 1770–1778. https://doi.org/10.2134/agronj2018.10.0645

Duffy, K.P., Cipparone, H.C., Johnson, E.S., Rickard, L.N., Beard, K. & Nascimento, F. 2020. Leveraging spatial dimensions of news media content analysis to explore place-based differences in natural resource issues. *Journal of Environmental Studies and Sciences*, *10*(3): 303–309. https://doi.org/10.1007/s13412-020-00595-9

Dumont, B., Ryschawy, J., Duru, M., Benoit, M., Chatellier, V., Delaby, L., Donnars, C., Dupraz, P., Lemauviel-Lavenant, S., Méda, B., Vollet, D. & Sabatier, R. 2019. Review: associations among goods, impacts and ecosystem services provided by livestock farming. *Animal*, *13*(8): 1773–1784. https://doi.org/10.1017/S1751731118002586

Dunford, R., Harrison, P., Smith, A., Dick, J., Barton, D.N., Martin-Lopez, B., Kelemen, E., Jacobs, S., Saarikoski, H., Turkelboom, F., Verheyden, W., Hauck, J., Antunes, P., Aszalós, R., Badea, O., Baró, F., Berry, P., Carvalho, L., Conte, G. Yli-Pelkonen, V. 2018. Integrating methods for ecosystem service assessment: experiences from real world situations. *Ecosystem Services*, *29*: 499–514. https://doi.org/10.1016/j.ecoser.2017.10.014

Dunlap, R.E. & Van Liere, K.D. 2008. The "new environmental paradigm". *The Journal of Environmental Education*, *40*(1): 19–28. https://doi.org/10.3200/JOEE.40.1.19-28

Duru, M., Donnars, C., Rychawy, J., Therond, O. & Dumont, B. 2018. La « grange »: un cadre conceptuel pour appréhender les bouquets de services rendus par l'élevage dans les territoires. *INRA Productions Animales*, *30*(4): 273–284. https://doi.org/10.20870/productions-animales.2017.30.4.2259

Edens, B., Maes, J., Hein, L., Obst, C., Siikamaki, J., Schenau, S., Javorsek, M., Chow, J., Chan, J.Y., Steurer, A. & Alfieri, A. 2022. Establishing the SEEA ecosystem accounting as a global standard. *Ecosystem Services*, *54*: 101413. https://doi.org/10.1016/j.ecoser.2022.101413

Embrapa. 2023. *Brasil em 50 alimentos*. Embrapa.

Engel, S. 2016. The Devil in the Detail: A practical guide on designing payments for environmental services. *International Review of Environmental and Resource Economics*, *9*(1–2): 131–177. https://doi.org/10.1561/101.00000076

Engel, S., Pagiola, S. & Wunder, S. 2008. Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics*, *65*(4): 663–674. https://doi.org/10.1016/j.ecolecon.2008.03.011

European Court of Auditors. 2020. *Biodiversity on farmland: CAP contribution has not halted the decline*. Publications Office. https://data.europa.eu/doi/10.2865/336742

EUROSTAT. 2018. *Economy-wide material flow accounts handbook*. https://ec.europa.eu/eurostat/web/products-manuals-and-guidelines/-/ks-gq-18-006

Faccioni, G., Sturaro, E., Ramanzin, M. & Bernués, A. 2019. Socio-economic valuation of abandonment and intensification of Alpine agroecosystems and associated ecosystem services. *Land Use Policy*, *81*: 453–462. https://doi.org/10.1016/j.landusepol.2018.10.044

FAO. N.D. *Incentives for ecosystem services*. https://www.fao.org/in-action/incentives-for-ecosystem-services/en/ (cited 27 May 2025).

FAO. 2008. An international technical workshop Investing in sustainable crop intensification: The case for improving soil health. *Integrated Crop Management,* Vol.6-2008. FAO, Rome: 22-24 July 2008 https://www.fao.org/4/i0951e/i0951e.pdf

FAO. 2011. *Payments for ecosystem services and food security*. https://www.fao.org/4/i2100e/i2100e00.htm

FAO. 2020. *Biodiversity and the livestock sector – Guidelines for quantitative assessment*. FAO. https://doi.org/10.4060/ca9295en

FAO & ILRI (Eds). 2011. *Global livestock production systems*. FAO.

FAO/WHO Codex Alimentarius. 2004. Code of practice on good animal feeding – CAC/RCP 54-2004. https://www.fao.org/feed-safety/resources/resources-details/en/c/1054052/

Felthoven, R.G. & Morrison Paul, C.J. 2004. Multi-output, nonfrontier primal measures of capacity and capacity utilization. *American Journal of Agricultural Economics*, *86*(3): 619–633. https://doi.org/10.1111/j.0002-9092.2004.00605.x

Ferraro, P.J. & Simpson, R.D. 2002. The cost-effectiveness of conservation payments. *Land Economics*, *78*(3): 339–353. https://doi.org/10.2307/3146894

Figueroa, D., Galicia, L. & Suárez Lastra, M. 2022. Latin American cattle ranching sustainability debate: An approach to social–ecological systems and spatial–temporal scales. *Sustainability*, 14(14): 8924. https://doi.org/10.3390/su14148924

Fisher, B., Turner, K., Zylstra, M., Brouwer, R., De Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirkby, C., Morling, P., Mowatt, S., Naidoo, R., Paavola, J., Strassburg, B., Yu, D. & Balmford, A. 2008. Ecosystem services and economic theory: Integration for policy-relevant research. *Ecological Applications*, *18*(8): 2050–2067. https://doi.org/10.1890/07-1537.1

Fisher, B., Turner, R.K. & Morling, P. 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics*, *68*(3): 643–653. https://doi.org/10.1016/j.ecolecon.2008.09.014

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N. & Snyder, P.K. 2005. Global consequences of land use. *Science*, *309*(5734): 570–574. https://doi.org/10.1126/science.1111772

Forio, M.A.E., Villa-Cox, G., Van Echelpoel, W., Ryckebusch, H., Lock, K., Spanoghe, P., Deknock, A., De Troyer, N., Nolivos-Alvarez, I., Dominguez-Granda, L., Speelman, S. & Goethals, P.L.M. 2020. Bayesian belief network models as trade-off tools of ecosystem services in the Guayas River Basin in Ecuador. *Ecosystem Services*, *44*: 101124. https://doi.org/10.1016/j.ecoser.2020.101124

Fortnam, M., Brown, K., Chaigneau, T., Crona, B., Daw, T.M., Gonçalves, D., Hicks, C., Revmatas, M., Sandbrook, C. & Schulte-Herbruggen, B. 2019. The gendered nature of ecosystem services. *Ecological economics*, *159*: 312-325. https://doi.org/10.1016/j.ecolecon.2018.12.018

Fox, N., Graham, L.J., Eigenbrod, F., Bullock, J.M. & Parks, K.E. 2021. Enriching social media data allows a more robust representation of cultural ecosystem services. *Ecosystem Services*, *50*: 101328. https://doi.org/10.1016/j.ecoser.2021.101328

Francesconi, W., Srinivasan, R., Pérez-Miñana, E., Willcock, S.P. & Quintero, M. 2016. Using the Soil and Water Assessment Tool (SWAT) to model ecosystem services: A systematic review. *Journal of Hydrology*, *535*: 625–636. https://doi.org/10.1016/j.jhydrol.2016.01.034

Franzese, P.P., Buonocore, E., Donnarumma, L. & Russo, G.F. 2017. Natural capital accounting in marine protected areas: The case of the Islands of Ventotene and S. Stefano (Central Italy). *Ecological Modelling*, *360*: 290–299. https://doi.org/10.1016/j.ecolmodel.2017.07.015

Freeman III, A.M., Herriges, J.A. & Kling, C.L. 2014. *The measurement of environmental and resource values*. 0 ed. Routledge. https://doi.org/10.4324/9781315780917

Fritz, S., See, L., Bayas, J.C.L., Waldner, F., Jacques, D., Becker-Reshef, I., Whitcraft, A., Baruth, B., Bonifacio, R., Crutchfield, J., Rembold, F., Rojas, O., Schucknecht, A., Van Der Velde, M., Verdin, J., Wu, B., Yan, N., You, L., Gilliams, S. McCallum, I. 2019. A comparison of global agricultural monitoring systems and current gaps. *Agricultural Systems*, *168*: 258–272. https://doi.org/10.1016/j.agsy.2018.05.010

Gallego, F., Bagnato, C., Baeza, S., Camba-Sans, G. & Paruelo, J. 2023. Río de la Plata grasslands: How did land-cover and ecosystem functioning change in the twenty-first century? In G.E. Overbeck, V.D.P. Pillar, S.C. Müller & G.A. Bencke, eds. *South Brazilian Grasslands*, pp. 475–493. Springer International Publishing. https://doi.org/10.1007/978-3-031-42580-6_18

García-Llorente, M., Iniesta-Arandia, I., Willaarts, B.A., Harrison, P.A., Berry, P., Bayo, M.D.M., Castro, A.J., Montes, C. & Martín-López, B. 2015. Biophysical and sociocultural factors underlying spatial trade-offs of ecosystem services in semiarid watersheds. *Ecology and Society*, *20*(3): art39. https://doi.org/10.5751/ES-07785-200339

García-Llorente, M., Martín-López, B., Iniesta-Arandia, I., López-Santiago, C.A., Aguilera, P.A. & Montes, C. 2012. The role of multi-functionality in social preferences toward semi-arid rural landscapes: An ecosystem service approach. *Environmental Science & Policy*, *19–20*: 136–146. https://doi.org/10.1016/j.envsci.2012.01.006

Gebbels, J.N., Kragt, M.E., Thomas, D.T. & Vercoe, P.E. 2022. Improving productivity reduces methane intensity but increases the net emissions of sheepmeat and wool enterprises. *Animal*, 16(4): 100490. https://doi.org/10.1016/j.animal.2022.100490

Geng, W., Li, Y., Zhang, P., Yang, D., Jing, W. & Rong, T. 2022. Analyzing spatio-temporal changes and trade-offs/synergies among ecosystem services in the Yellow River Basin, China. *Ecological Indicators*, 138: 108825. https://doi.org/10.1016/j.ecolind.2022.108825

Gentry, R.R., Alleway, H.K., Bishop, M.J., Gillies, C.L., Waters, T. & Jones, R. 2020. Exploring the potential for marine aquaculture to contribute to ecosystem services. *Reviews in Aquaculture*, 12(2): 499–512. https://doi.org/10.1111/raq.12328

Ghermandi, A. 2018. Integrating social media analysis and revealed preference methods to value the recreation services of ecologically engineered wetlands. *Ecosystem Services*, 31: 351–357. https://doi.org/10.1016/j.ecoser.2017.12.012

Gómez-Baggethun, E., Barton, D., Berry, P., Dunford, R. & Harrison, P. 2016. Concepts and methods in ecosystem services valuation. In *Routledge Handbook of Ecosystem Services*, pp. 99–111. https://doi.org/10.4324/9781315775302-9

Gómez-Baggethun, E., De Groot, R., Lomas, P.L. & Montes, C. 2010. The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics*, 69(6): 1209–1218. https://doi.org/10.1016/j.ecolecon.2009.11.007

González, L.A., Bishop-Hurley, G., Henry, D. & Charmley, E. 2014. Wireless sensor networks to study, monitor and manage cattle in grazing systems. *Animal Production Science*, 54(10): 1687. https://doi.org/10.1071/AN14368

González, L.A., Kyriazakis, I. & Tedeschi, L.O. 2018. Review: Precision nutrition of ruminants: Approaches, challenges and potential gains. *Animal*, 12: s246–s261. https://doi.org/10.1017/S1751731118002288

González, L.A., Shirvan, M.B. & Molfino, J. 2023. 312 Emissions intensity of growing cattle with contrasting residual methane production, average daily gain, feed conversion ratio, residual feed intake, and feed intake. *Journal of Animal Science*, 101(Suppl. 3): 240–241. https://doi.org/10.1093/jas/skad281.290

González-García, E., Alhamada, M., Pradel, J., Douls, S., Parisot, S., Bocquier, F., Menassol, J.B., Llach, I. & González, L.A. 2018. A mobile and automated walk-over-weighing system for a close and remote monitoring of liveweight in sheep. *Computers and Electronics in Agriculture*, 153: 226–238. https://doi.org/10.1016/j.compag.2018.08.022

Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D. & Moore, R. 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. *Remote Sensing of Environment*, 202: 18–27. https://doi.org/10.1016/j.rse.2017.06.031

Gorosábel, A., Estigarribia, L., Lopes, L.F., Martinez, A.M., Martínez-Lanfranco, J.A., Adenle, A.A., Rivera-Rebella, C. & Oyinlola, M.A. 2020. Insights for policy-based conservation strategies for the Rio de la Plata Grasslands through the IPBES framework. *Biota Neotropica*, 20(Suppl. 1): e20190902. https://doi.org/10.1590/1676-0611-bn-2019-0902

Götzl, M., Tiefenbach, Tramberend & Cond'e. 2013. Review of recent literature on mapping ecosystem services and analysis of methods used. *ETC/BD report for the EEA*. http://bd.eionet.europa.eu/

Gould, R.K., Klain, S.C., Ardoin, N.M., Satterfield, T., Woodside, U., Hannahs, N., Daily, G.C. & Chan, K.M. 2015. A protocol for eliciting nonmaterial values through a cultural ecosystem services frame. *Conservation Biology*, 29(2): 575–586. https://doi.org/10.1111/cobi.12407

Groot, J.C.J. & Rossing, W.A.H. 2011. Model-aided learning for adaptive management of natural resources: An evolutionary design perspective. *Methods in Ecology and Evolution*, 2(6): 643–650. https://doi.org/10.1111/j.2041-210X.2011.00114.x

Groot, R.S., Wilson, M.A. & Boumans, R.M.J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41(3): 393–408. https://doi.org/10.1016/S0921-8009(02)00089-7

Gutiérrez, F., Gallego, F., Paruelo, J.M. & Rodríguez, C. 2020. Damping and lag effects of precipitation variability across trophic levels in Uruguayan rangelands. *Agricultural Systems*, 185: 102956. https://doi.org/10.1016/j.agsy.2020.102956

Hamilton, S.H., Fu, B., Guillaume, J.H.A., Badham, J., Elsawah, S., Gober, P., Hunt, R.J., Iwanaga, T., Jakeman, A.J., Ames, D.P., Curtis, A., Hill, M.C., Pierce, S.A. & Zare, F. 2019. A framework for characterising and evaluating the effectiveness of environmental modelling. *Environmental Modelling & Software*, 118: 83–98. https://doi.org/10.1016/j.envsoft.2019.04.008

Hamilton, S.H., Pollino, C.A., Stratford, D.S., Fu, B. & Jakeman, A.J. 2022. Fit-for-purpose environmental modelling: Targeting the intersection of usability, reliability and feasibility. *Environmental Modelling & Software*, 148: 105278. https://doi.org/10.1016/j.envsoft.2021.105278

Hansson, H. & Lagerkvist, C.J. 2016. Dairy farmers' use and non-use values in animal welfare: Determining the empirical content and structure with anchored best-worst scaling. *Journal of Dairy Science*, 99(1): 579–592. https://doi.org/10.3168/jds.2015-9755

Harrison, P.A., Dunford, R., Barton, D.N., Kelemen, E., Martín-López, B., Norton, L., Termansen, M., Saarikoski, H., Hendriks, K., Gómez-Baggethun, E., Czúcz, B., García-Llorente, M., Howard, D., Jacobs, S., Karlsen, M., Kopperoinen, L., Madsen, A., Rusch, G., Van Eupen, M. Zulian, G. 2018. Selecting methods for ecosystem service assessment: A decision tree approach. *Ecosystem Services*, 29: 481–498. https://doi.org/10.1016/j.ecoser.2017.09.016

Hazell, P. & Wood, S. 2008. Drivers of change in global agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363(1491): 495–515. https://doi.org/10.1098/rstb.2007.2166

Helbing, D. 2012. Agent-based modelling. In D. Helbing, ed. *Social self-organization*, pp. 25–70. Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-24004-1_2

Hendriks, C., Obernosterer, R., Müller, D., Kytzia, S., Baccini, P. & Brunner, P.H. 2000. Material flow analysis: A tool to support environmental policy decision making. Case-studies on the city of Vienna and the Swiss lowlands. *Local Environment*, 5(3): 311–328. https://doi.org/10.1080/13549830050134257

Hensher, D.A. 2006. How do respondents process stated choice experiments? Attribute consideration under varying information load. *Journal of Applied Econometrics*, 21(6): 861–878. https://doi.org/10.1002/jae.877

Hermelingmeier, V. & Nicholas, K.A. 2017. Identifying five different perspectives on the ecosystem services concept using Q methodology. *Ecological Economics*, 136: 255–265. https://doi.org/10.1016/j.ecolecon.2017.01.006

Hermes, J., Van Berkel, D., Burkhard, B., Plieninger, T., Fagerholm, N., Von Haaren, C. & Albert, C. 2018. Assessment and valuation of recreational ecosystem services of landscapes. *Ecosystem Services*, 31: 289–295. https://doi.org/10.1016/j.ecoser.2018.04.011

Herrero-Jáuregui, C. & Oesterheld, M. 2018. Effects of grazing intensity on plant richness and diversity: A meta-analysis. *Oikos*, 127(6): 757–766. https://doi.org/10.1111/oik.04893

Hocquette, J.-F., Ellies-Oury, M.-P., Lherm, M., Pineau, C., Deblitz, C. & Farmer, L. 2018. Current situation and future prospects for beef production in Europe – A review. *Asian-Australasian Journal of Animal Sciences*, 31(7): 1017–1035. https://doi.org/10.5713/ajas.18.0196

Horne, P., Boxall, P.C. & Adamowicz, W.L. 2005. Multiple-use management of forest recreation sites: A spatially explicit choice experiment. *Forest Ecology and Management*, 207(1–2): 189–199. https://doi.org/10.1016/j.foreco.2004.10.026

Hubatova, M., McGinlay, J., Parsons, D.J., Morris, J. & Graves, A.R. 2023. Assessing preferences for cultural ecosystem services in the English countryside using Q methodology. *Land*, 12(2): 331. https://doi.org/10.3390/land12020331

Huitema, D., Cornelisse, C. & Ottow, B. 2010. Is the jury still out? Toward greater insight in policy learning in participatory decision processes – the case of Dutch citizens' juries on water management in the Rhine Basin. *Ecology and Society*, 15(1): art16. https://doi.org/10.5751/ES-03260-150116

Huntsinger, L. & Oviedo, J.L. 2014. Ecosystem services are social–ecological services in a traditional pastoral system: The case of California's Mediterranean rangelands. *Ecology and Society*, 19(1): art8. https://doi.org/10.5751/ES-06143-190108

Iniesta-Arandia, I., García-Llorente, M., Aguilera, P.A., Montes, C. & Martín-López, B. 2014. Socio-cultural valuation of ecosystem services: Uncovering the links between values, drivers of change, and human well-being. *Ecological Economics*, 108: 36–48. https://doi.org/10.1016/j.ecolecon.2014.09.028

IPBES. 2013. Summary of the second session of the plenary of the intergovernmental science-policy platform on biodiversity and ecosystem services: 9-14 December 2013. *Earth Negotiations Bulletin*. https://enb.iisd.org/events/2nd-session-ipbes-plenary/summary-report-9-14-december-2013 (cited 25 July 2025)

IPBES. 2019. Summary of the seventh session of the plenary of the intergovernmental science-policy platform on biodiversity and ecosystem services: 29 April – 4 May 2019. *Earth Negotiations Bulletin*. https://enb.iisd.org/events/stakeholder-day-and-7th-session-plenary-intergovernmental-platform-biodiversity-and/summary (cited 25 July 2025)

IPBES. 2022. *Methodological assessment of the diverse values and valuation of nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Zenodo. https://doi.org/10.5281/ZENODO.6522522

Irisarri, J.G., Derner, J.D., Ritten, J.P. & Peck, D.E. 2019. Beef production and net revenue variability from grazing systems on semiarid grasslands of North America. *Livestock Science*, 220: 93–99. https://doi.org/10.1016/j.livsci.2018.12.009

Irisarri, J.G.N. & Oesterheld, M. 2020. Temporal variation of stocking rate and primary production in the face of drought and land use change. *Agricultural Systems*, 178: 102750. https://doi.org/10.1016/j.agsy.2019.102750

Irisarri, J.G.N., Oesterheld, M., Golluscio, R.A. & Paruelo, J.M. 2014. Effects of animal husbandry on secondary production and trophic efficiency at a regional scale. *Ecosystems*, 17(4): 738–749. https://doi.org/10.1007/s10021-014-9756-6

Jacobs, S., Kelemen, E., O'Farrell, P., Martin, A., Schaafsma, M., Dendoncker, N., Pandit, R., Mwampamba, T.H., Palomo, I., Castro, A.J., Huambachano, M.A., Filyushkina, A. & Gunimeda, H. 2023. The pitfalls of plural valuation. *Current Opinion in Environmental Sustainability*, 64: 101345. https://doi.org/10.1016/j.cosust.2023.101345

Jax, K., Furman, E., Saarikoski, H., Barton, D.N., Delbaere, B., Dick, J., Duke, G., Görg, C., Gómez-Baggethun, E., Harrison, P.A., Maes, J., Pérez-Soba, M., Saarela, S.-R., Turkelboom, F., Van Dijk, J. & Watt, A.D. 2018. Handling a messy world: Lessons learned when trying to make the ecosystem services concept operational. *Ecosystem Services*, 29: 415–427. https://doi.org/10.1016/j.ecoser.2017.08.001

Jopke, C., Kreyling, J., Maes, J. & Koellner, T. 2015. Interactions among ecosystem services across Europe: Bagplots and cumulative correlation coefficients reveal synergies, trade-offs, and regional patterns. *Ecological Indicators*, 49: 46–52. https://doi.org/10.1016/j.ecolind.2014.09.037

Kanninen, B.J., ed. 2007. *Valuing environmental amenities using stated choice studies: A common sense approach to theory and practice*, Vol. 8. Springer Netherlands. https://doi.org/10.1007/1-4020-5313-4

Kareem, A., Farooqi, Z.U.R., Kalsom, A., Mohy-Ud-Din, W., Hussain, M.M., Raza, M. & Khursheed, M.M. 2022. Organic farming for sustainable soil use, management, food production and climate change mitigation. In S.A. Bandh, ed. *Sustainable agriculture*, pp. 39–59. Springer International Publishing. https://doi.org/10.1007/978-3-030-83066-3_3

Kaye-Zwiebel, E. & King, E. 2014. Kenyan pastoralist societies in transition: Varying perceptions of the value of ecosystem services. *Ecology and Society*, 19(3): art17. https://doi.org/10.5751/ES-06753-190317

Keane, A. 2016. A review of conceptual frameworks arising from the ESPA programme.

Kendall, M.G. 1938. A new measure of rank correlation. *Biometrika*, 30(1–2): 81–93. https://doi.org/10.1093/biomet/30.1-2.81

Kenter, J.O., Hyde, T., Christie, M. & Fazey, I. 2011. The importance of deliberation in valuing ecosystem services in developing countries – Evidence from the Solomon Islands. *Global Environmental Change*, 21(2): 505–521. https://doi.org/10.1016/j.gloenvcha.2011.01.001

Kim, J., Ale, S., Kreuter, U.P. & Teague, W.R. 2023. Grazing management impacts on ecosystem services under contrasting climatic conditions in Texas and North Dakota. *Journal of Environmental Management*, 347: 119213. https://doi.org/10.1016/j.jenvman.2023.119213

Kimoto, C., DeBano, S.J., Thorp, R.W., Taylor, R.V., Schmalz, H., DelCurto, T., Johnson, T., Kennedy, P.L. & Rao, S. 2012. Short-term responses of native bees to livestock and implications for managing ecosystem services in grasslands. *Ecosphere*, 3(10): 1–19. https://doi.org/10.1890/ES12-00118.1

Klain, S.C., Satterfield, T.A. & Chan, K.M.A. 2014. What matters and why? Ecosystem services and their bundled qualities. *Ecological Economics*, 107: 310–320. https://doi.org/10.1016/j.ecolecon.2014.09.003

Koemle, D. & Yu, X. 2020. Choice experiments in non-market value analysis: Some methodological issues. *Forestry Economics Review*, 2(1): 3–31. https://doi.org/10.1108/FER-04-2020-0005

Koetse, M.J., Brouwer, R. & Van Beukering, P.J.H. 2015. Economic valuation methods for ecosystem services. In J.A. Bouma & P.J.H. Van Beukering, eds. *Ecosystem services*, 1st ed., pp. 108–131. Cambridge University Press. https://doi.org/10.1017/CBO9781107477612.009

Kok, A., Oostvogels, V.J., De Olde, E.M. & Ripoll-Bosch, R. 2020. Balancing biodiversity and agriculture: Conservation scenarios for the Dutch dairy sector. *Agriculture, Ecosystems & Environment*, 302: 107103. https://doi.org/10.1016/j.agee.2020.107103

Kovanda, J. 2021. Economy-wide material system analysis: Mapping material flows through the economy. *Journal of Industrial Ecology*, 25(5): 1121–1135. https://doi.org/10.1111/jiec.13142

Lamarque, P., Quétier, F. & Lavorel, S. 2011. The diversity of the ecosystem services concept and its implications for their assessment and management. *Comptes Rendus. Biologies*, 334(5–6): 441–449. https://doi.org/10.1016/j.crvi.2010.11.007

Lamothe, K.A. & Sutherland, I.J. 2018. Intermediate ecosystem services: The origin and meanings behind an unsettled concept. *International Journal of Biodiversity Science, Ecosystem Services & Management*, 14(1): 179–187. https://doi.org/10.1080/21513732.2018.1524399

Lancaster, K.J. 1966. A new approach to consumer theory. *Journal of Political Economy*, 74(2): 132–157. https://doi.org/10.1086/259131

Lankoski, J. 2003. Agri-environmental externalities: A framework for designing targeted policies. *European Review of Agriculture Economics*, 30(1): 51–75. https://doi.org/10.1093/erae/30.1.51

Larsen, D., Tyndall, J.C., Schulte, L.A. & Grudens-Schuck, N. 2019. Exploring stakeholder consensus for multiple outcomes in agriculture: An Iowa case study. *Frontiers in Sustainable Food Systems*, 3. https://doi.org/10.3389/fsufs.2019.00110

Le Clec'h, S., Finger, R., Buchmann, N., Gosal, A.S., Hörtnagl, L., Huguenin-Elie, O., Jeanneret, P., Lüscher, A., Schneider, M.K. & Huber, R. 2019. Assessment of spatial variability of multiple ecosystem services in grasslands of different intensities. *Journal of Environmental Management*, 251: 109372. https://doi.org/10.1016/j.jenvman.2019.109372

Lebacq, T., Baret, P.V. & Stilmant, D. 2013. Sustainability indicators for livestock farming: A review. *Agronomy for Sustainable Development*, 33(2): 311–327. https://doi.org/10.1007/s13593-012-0121-x

Lecegui, A., Olaizola, A.M. & Varela, E. 2022. Disentangling the role of management practices on ecosystem services delivery in Mediterranean silvopastoral systems: Synergies and trade-offs through expert-based assessment. *Forest Ecology and Management*, 517: 120273. https://doi.org/10.1016/j.foreco.2022.120273

Leduc, G., Manevska-Tasevska, G., Hansson, H., Arndt, M., Bakucs, Z., Böhm, M., Chitea, M., Florian, V., Luca, L., Martikainen, A., Pham, H.V. & Rusu, M. 2021. How are ecological approaches justified in European rural development policy? Evidence from a content analysis of CAP and rural development discourses. *Journal of Rural Studies*, 86: 611–622. https://doi.org/10.1016/j.jrurstud.2021.06.009

Leite, F.F.G.D., Nóbrega, G.N., Baumgärtner, L.C., Alecrim, F.B., Da Silveira, J.G., Cordeiro, R.C. & Rodrigues, R.D.A.R. 2023. Greenhouse gas emissions and carbon sequestration associated with integrated crop–livestock–forestry (ICLF) systems. *Environmental Reviews*, 31(4): 589–604. https://doi.org/10.1139/er-2022-0095

Leroy, G., Hoffmann, I., From, T., Hiemstra, S.J. & Gandini, G. 2018. Perception of livestock ecosystem services in grazing areas. *Animal*, 12(12): 2627–2638. https://doi.org/10.1017/S1751731118001027

Lettenmeier, M., Rohn, H., Liedtke, C., Schmidt-Bleek, F., Bienge, K., Urbaneja, D.M. & Buddenberg, J. 2009. *Resource productivity in 7 steps: How to develop eco-innovative products and services and improve their material footprint*. Wuppertal Spezial, 41. https://ideas.repec.org//b/zbw/wupspe/41.html

Lindborg, R., Hartel, T., Helm, A., Prangel, E., Reitalu, T. & Ripoll-Bosch, R. 2023. Ecosystem services provided by semi-natural and intensified grasslands: Synergies, trade-offs and linkages to plant traits and functional richness. *Applied Vegetation Science*, 26(2): e12729. https://doi.org/10.1111/avsc.12729

Liu, J., **Dietz, T.**, **Carpenter, S.R.**, **Alberti, M.**, **Folke, C.**, **Moran, E.**, **Pell, A.N.**, **Deadman, P.**, **Kratz, T.**, **Lubchenco, J.**, **Ostrom, E.**, **Ouyang, Z.**, **Provencher, W.**, **Redman, C.L.**, **Schneider, S.H. & Taylor, W.W.** 2007. *Complexity of coupled human and natural systems*. Science, 317(5844): 1513–1516. https://doi.org/10.1126/science.1144004

Liu, J., **Herzberger, A.**, **Kapsar, K.**, **Carlson, A.K. & Connor, T.** 2019. *What is telecoupling?* In: Friis, C. & Nielsen, J.Ø., eds. *Telecoupling*, pp. 19–48. Springer International Publishing. https://doi.org/10.1007/978-3-030-11105-2_2

Liu, J., **Hull, V.**, **Batistella, M.**, **DeFries, R.**, **Dietz, T.**, **Fu, F.**, **Hertel, T.W.**, **Izaurralde, R.C.**, **Lambin, E.F.**, **Li, S.**, **Martinelli, L.A.**, **McConnell, W.J.**, **Moran, E.F.**, **Naylor, R.**, **Ouyang, Z.**, **Polenske, K.R.**, **Reenberg, A.**, **De Miranda Rocha, G.**, **Simmons, C.S. & Zhu, C.** 2013. *Framing sustainability in a telecoupled world*. Ecology and Society, 18(2): art26. https://doi.org/10.5751/ES-05873-180226

Liu, J., **Yang, W. & Li, S.** 2016. *Framing ecosystem services in the telecoupled Anthropocene*. Frontiers in Ecology and the Environment, 14(1): 27–36. https://doi.org/10.1002/16-0188.1

Lliso, B., **Lenzi, D.**, **Muraca, B.**, **Chan, K.M. & Pascual, U.** 2022. *Nature's disvalues: What are they and why do they matter?* Current Opinion in Environmental Sustainability, 56: 101173. https://doi.org/10.1016/j.cosust.2022.101173

Lüscher, A., Mueller-Harvey, I., Soussana, J.F., Rees, R.M. & Peyraud, J.L. 2014. Potential of legume-based grassland–livestock systems in Europe: A review. *Grass and Forage Science*, 69(2): 206–228. https://doi.org/10.1111/gfs.12124

Lyytimäki, J. 2014. *Bad nature: Newspaper representations of ecosystem disservices*. Urban Forestry & Urban Greening, 13(3): 418–424. https://doi.org/10.1016/j.ufug.2014.04.005

Ma, L., **Derner, J.D.**, **Harmel, R.D.**, **Tatarko, J.**, **Moore, A.D.**, **Rotz, C.A.**, **Augustine, D.J.**, **Boone, R.B.**, **Coughenour, M.B.**, **Beukes, P.C.**, **Van Wijk, M.T.**, **Bellocchi, G.**, **Cullen, B.R. & Wilmer, H.** 2019. *Application of grazing land models in ecosystem management: Current status and next frontiers*. In: *Advances in Agronomy*, Vol. 158, pp. 173–215. Elsevier. https://doi.org/10.1016/bs.agron.2019.07.003

Maccagnani, B., **Veromann, E.**, **Ferrari, R.**, **Boriani, L. & Boecking, O.** 2020. *Agroecosystem design supports the activity of pollinator networks*. In: Smagghe, G., Boecking, O., Maccagnani, B., Mänd, M. & Kevan, P.G., eds. *Entomovectoring for precision biocontrol and enhanced pollination of crops*, pp. 1–17. Springer International Publishing. https://doi.org/10.1007/978-3-030-18917-4_1

Mack, G., **El Benni, N.**, **Spörri, M.**, **Huguenin-Elie, O.**, **Tindale, S.**, **Hunter, E.**, **Newell Price, P. & Frewer, L.J.** 2023. *Perceived feasibility of sward management options in permanent grassland of Alpine regions and expected effects on delivery of ecosystem services*. Environment, Development and Sustainability. https://doi.org/10.1007/s10668-022-02899-y

Maczka, K., **Matczak, P.**, **Pietrzyk-Kaszyńska, A.**, **Rechciński, M.**, **Olszańska, A.**, **Cent, J. & Grodzińska-Jurczak, M.** 2016. *Application of the ecosystem services concept in environmental policy – A systematic empirical analysis of national level policy documents in Poland*. Ecological Economics, 128: 169–176. https://doi.org/10.1016/j.ecolecon.2016.04.023

Maes, J., **Egoh, B.**, **Willemen, L.**, **Liquete, C.**, **Vihervaara, P.**, **Schägner, J.P.**, **Grizzetti, B.**, **Drakou, E.G.**, **Notte, A.L.**, **Zulian, G.**, **Bouraoui, F.**, **Paracchini, M.L.**, **Braat, L. & Bidoglio, G.** 2012. *Mapping ecosystem services for policy support and decision making in the European Union*. Ecosystem Services, 1(1): 31–39. https://doi.org/10.1016/j.ecoser.2012.06.004

Malinga, R., **Gordon, L.J.**, **Lindborg, R. & Jewitt, G.** 2013. *Using participatory scenario planning to identify ecosystem services in changing landscapes*. Ecology and Society, 18(4): art10. https://doi.org/10.5751/ES-05494-180410

Mancini, M.S., **Galli, A.**, **Coscieme, L.**, **Niccolucci, V.**, **Lin, D.**, **Pulselli, F.M.**, **Bastianoni, S. & Marchettini, N.** 2018. *Exploring ecosystem services assessment through Ecological Footprint accounting*. Ecosystem Services, 30: 228–235. https://doi.org/10.1016/j.ecoser.2018.01.010

Maniatakou, S., **Berg, H.**, **Maneas, G. & Daw, T.M.** 2020. *Unravelling diverse values of ecosystem services: A socio-cultural valuation using Q methodology in Messenia, Greece*. Sustainability, 12(24): 10320. https://doi.org/10.3390/su122410320

Manning, J.K., **Cronin, G.M.**, **González, L.A.**, **Hall, E.J.S.**, **Merchant, A. & Ingram, L.J.** 2017. *The effects of global navigation satellite system (GNSS) collars on cattle (Bos taurus) behaviour*. Applied Animal Behaviour Science, 187: 54–59. https://doi.org/10.1016/j.applanim.2016.11.013

Manzano, P. & Malo, J.E. 2006. *Extreme long-distance seed dispersal via sheep*. Frontiers in Ecology and the Environment, 4(5): 244–248. https://doi.org/10.1890/1540-9295(2006)004[0244:ELSDVS]2.0.CO;2

MapBiomas Uruguay. N.d. *Uruguay land use and coverage mapping platform*. [Cited 4 June 2025]. https://uruguay.mapbiomas.org/

Martin, A.J. & Doucet, T. 2022. *Communication of ecosystem services and disservices in local newspapers in Winnipeg, Canada*. Urban Forestry & Urban Greening, 74: 127653.

Martín-López, B., **Gómez-Baggethun, E.**, **García-Llorente, M. & Montes, C.** 2014. *Trade-offs across value-domains in ecosystem services assessment*. Ecological Indicators, 37: 220–228. https://doi.org/10.1016/j.ecolind.2013.03.003

Martín-López, B., **Gómez-Baggethun, E.**, **Lomas, P.L. & Montes, C.** 2009. *Effects of spatial and temporal scales on cultural services valuation*. Journal of Environmental Management, 90(2): 1050–1059. https://doi.org/10.1016/j.jenvman.2008.03.013

Mavrommati, G., **Borsuk, M.E. & Howarth, R.B.** 2017. *A novel deliberative multicriteria evaluation approach to ecosystem service valuation*. Ecology and Society, 22(2): art39. https://doi.org/10.5751/ES-09105-220239

Mavrommati, G., **Borsuk, M.E.**, **Kreiley, A.I.**, **Larosee, C.**, **Rogers, S.**, **Burford, K. & Howarth, R.B.** 2021. *A methodological framework for understanding shared social values in deliberative valuation*. Ecological Economics, 190: 107185. https://doi.org/10.1016/j.ecolecon.2021.107185

McAuliffe, G.A., **Takahashi, T.**, **Orr, R.J.**, **Harris, P. & Lee, M.R.F.** 2018. *Distributions of emissions intensity for individual beef cattle reared on pasture-based production systems*. Journal of Cleaner Production, 171: 1672–1680. https://doi.org/10.1016/j.jclepro.2017.10.113

McClelland, S.C., **Haddix, J.D.**, **Azad, S.**, **Boughton, E.H.**, **Boughton, R.K.**, **Miller, R.S.**, **Swain, H.M. & Dillon, J.A.** 2023. *Quantifying biodiversity impacts of livestock using life-cycle perspectives*. Frontiers in Ecology and the Environment, 21(6): 275–281. https://doi.org/10.1002/fee.2636

McLellan, V. & Shackleton, C.M. 2019. *The relative representation of ecosystem services and disservices in South African newspaper media*. Ecosystems and People, 15(1): 247–256. https://doi.org/10.1080/26395916.2019.1667442

McNaughton, S.J., **Oesterheld, M.**, **Frank, D.A. & Williams, K.J.** 1989. *Ecosystem-level patterns of primary productivity and herbivory in terrestrial habitats*. Nature, 341(6238): 142–144. https://doi.org/10.1038/341142a0

MEA (Ed.) 2005. *Ecosystems and human well-being: Synthesis*. Island Press.

Mennig, P. & Sauer, J. 2008. Integration von Ökologie und Bioökonomie am Beispiel von Agrarumweltmaßnahmen.

Mensah, S., **Veldtman, R.**, **Assogbadjo, A.E.**, **Ham, C.**, **Glèlè Kakaï, R. & Seifert, T.** 2017. *Ecosystem service importance and use vary with socio-environmental factors: A study from household-surveys in local communities of South Africa*. Ecosystem Services, 23: 1–8. https://doi.org/10.1016/j.ecoser.2016.10.018

Merida, V.E., **Cook, D.**, Ögmundarson**, Ó. & Davíðsdóttir, B.** 2022. *Ecosystem services and disservices of meat and dairy production: A systematic literature review*. Ecosystem Services, 58: 101494. https://doi.org/10.1016/j.ecoser.2022.101494

Millennium Ecosystem Assessment (Program) (Ed.) 2005. *Ecosystems and human well-being: Synthesis*. Island Press.

Miller, S.A., **Ferreira, J.P. & LeJeune, J.T.** 2022. *Antimicrobial use and resistance in plant agriculture: A one health perspective*. Agriculture, 12(2): 289. https://doi.org/10.3390/agriculture12020289

Ministerio de Ambiente de Uruguay. 2022. *Huella ambiental de la ganadería en Uruguay*. https://www.gub.uy/ministerio-ambiente/comunicacion/noticias/huella-ambiental-ganaderia-uruguay (cited 27 May 2025).

Modernel, P., **Rossing, W.A.H.**, **Corbeels, M.**, **Dogliotti, S.**, **Picasso, V. & Tittonell, P.** 2016. *Land use change and ecosystem service provision in Pampas and Campos grasslands of southern South America*. Environmental Research Letters, 11(11): 113002. https://doi.org/10.1088/1748-9326/11/11/113002

Monteith, J.L. 1972. *Solar radiation and productivity in tropical ecosystems*. The Journal of Applied Ecology, 9(3): 747. https://doi.org/10.2307/2401901

Moonen, A.-C. & Bàrberi, P. 2008. *Functional biodiversity: An agroecosystem approach*. Agriculture, Ecosystems & Environment, 127(1–2): 7–21. https://doi.org/10.1016/j.agee.2008.02.013

Moritz, M., **Cross, B. & Hunter, C.E.** 2023. *Artificial pastoral systems: A review of agent-based modelling studies of pastoral systems*. Pastoralism, 13(1): 31. https://doi.org/10.1186/s13570-023-00293-5

Mottet, A., **De Haan, C.**, **Falcucci, A.**, **Tempio, G.**, **Opio, C. & Gerber, P.** 2017. *Livestock: On our plates or eating at our table? A new analysis of the feed/food debate*. Global Food Security, 14: 1–8. https://doi.org/10.1016/j.gfs.2017.01.001

Mouchet, M.A., **Lamarque, P.**, **Martín-López, B.**, **Crouzat, E.**, **Gos, P.**, **Byczek, C. & Lavorel, S.** 2014. *An interdisciplinary methodological guide for quantifying associations between ecosystem services*. Global Environmental Change, 28: 298–308. https://doi.org/10.1016/j.gloenvcha.2014.07.012

Mouchet, M.A., **Paracchini, M.L.**, **Schulp, C.J.E.**, **Stürck, J.**, **Verkerk, P.J.**, **Verburg, P.H. & Lavorel, S.** 2017. *Bundles of ecosystem (dis)services and multifunctionality across European landscapes*. Ecological Indicators, 73: 23–28. https://doi.org/10.1016/j.ecolind.2016.09.026

Mukherjee, N., **Hugé, J.**, **Sutherland, W.J.**, **McNeill, J.**, **Van Opstal, M.**, **Dahdouh-Guebas, F. & Koedam, N.** 2015. *The Delphi technique in ecology and biological conservation: Applications and guidelines*. Methods in Ecology and Evolution, 6(9): 1097–1109. https://doi.org/10.1111/2041-210X.12387

Muñoz-Ulecia, E., **Bernués, A.**, **Ondé, D.**, **Ramanzin, M.**, **Soliño, M.**, **Sturaro, E.** & **Martín-Collado, D.** 2022. *People's attitudes towards the agrifood system influence the value of ecosystem services of mountain agroecosystems*. PLOS ONE, 17(5): e0267799. https://doi.org/10.1371/journal.pone.0267799

Murphy, M.B., **Mavrommati, G.**, **Mallampalli, V.R.**, **Howarth, R.B. & Borsuk, M.E.** 2017. *Comparing group deliberation to other forms of preference aggregation in valuing ecosystem services*. Ecology and Society, 22(4): art17. https://doi.org/10.5751/ES-09519-220417

Murray-Rust, D., **Dendoncker, N.**, **Dawson, T.P.**, **Acosta-Michlik, L.**, **Karali, E.**, **Guillem, E.**, **& Rounsevell, M.** 2011. *Conceptualising the analysis of socio-ecological systems through ecosystem services and agent-based modelling*. Journal of Land Use Science, 6(2–3): 83–99. https://doi.org/10.1080/1747423X.2011.558600

Muscat, A., **De Olde, E.M.**, **De Boer, I.J.M.**, **& Ripoll-Bosch, R.** 2020. *The battle for biomass: A systematic review of food-feed-fuel competition*. Global Food Security, 25: 100330. https://doi.org/10.1016/j.gfs.2019.100330

Muscat, A., **De Olde, E.M.**, **Kovacic, Z.**, **De Boer, I.J.M.**, **& Ripoll-Bosch, R.** 2021. *Food, energy or biomaterials? Policy coherence across agro-food and bioeconomy policy domains in the EU*. Environmental Science & Policy, 123: 21–30. https://doi.org/10.1016/j.envsci.2021.05.001

Nadalini, A.C.V., **Kalid, R.D.A.**, **& Torres, E.A.** 2021. *Emergy as a Tool to Evaluate Ecosystem Services: A Systematic Review of the Literature*. Sustainability, 13(13): 7102. https://doi.org/10.3390/su13137102

NASA. N.D. *Moderate Resolution Imaging Spectroradiometer (MODIS) – About MODIS*. [Cited 4 June 2025]. https://modis.gsfc.nasa.gov/about/

Nelson, E., **Mendoza, G.**, **Regetz, J.**, **Polasky, S.**, **Tallis, H.**, **Cameron, Dr.**, **Chan, K.M.**, **Daily, G.C.**, **Goldstein, J.**, **Kareiva, P.M.**, **Lonsdorf, E.**, **Naidoo, R.**, **Ricketts, T.H.**, **& Shaw, Mr.** 2009. *Modelling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales*. Frontiers in Ecology and the Environment, 7(1): 4–11. https://doi.org/10.1890/080023

Nemec, K.T. & **Raudsepp-Hearne, C.** 2013. *The use of geographic information systems to map and assess ecosystem services*. Biodiversity and Conservation, 22(1): 1–15. https://doi.org/10.1007/s10531-012-0406-z

Newman, L., **Newell, R.**, **Mendly-Zambo, Z.**, **& Powell, L.** 2022. *Bioengineering, telecoupling, and alternative dairy: Agricultural land use futures in the Anthropocene*. The Geographical Journal, 188(3): 342–357. https://doi.org/10.1111/geoj.12392

Newton, P., **Civita, N.**, **Frankel-Goldwater, L.**, **Bartel, K.**, **& Johns, C.** 2020. *What Is Regenerative Agriculture? A Review of Scholar and Practitioner Definitions Based on Processes and Outcomes*. Frontiers in Sustainable Food Systems, 4: 577723. https://doi.org/10.3389/fsufs.2020.577723

NGFS-INSPIR. 2021. *Central banking and supervision in the biosphere: An agenda for action on biodiversity loss, financial risk and system stability*. https://www.ngfs.net/en/publications-and-statistics/publications/central-banking-and-supervision-biosphere-agenda-action-biodiversity-loss-financial-risk-and-system

Nguyen, T.H., **Nong, D.**, **& Paustian, K.** 2019. *Surrogate-based multi-objective optimization of management options for agricultural landscapes using artificial neural networks*. Ecological Modelling, 400: 1–13. https://doi.org/10.1016/j.ecolmodel.2019.02.018

Notenbaert, A. M. O., Herrero, M., Kruska, R. L., You, L., Wood, S., Thornton, P. K., & Omolo, A. 2009. Classifying livestock production systems for targeting agricultural research and development in a rapidly changing world. ILRI Discussion Paper 19. *ILRI (International Livestock Research Institute)*, Nairobi, Kenya. 41 pp. https://hdl.handle.net/10568/589

Nyberg, Y., **Wetterlind, J.**, **Jonsson, M.**, **& Öborn, I.** 2020. *The role of trees and livestock in ecosystem service provision and farm priorities on smallholder farms in the Rift Valley, Kenya*. Agricultural Systems, 181: 102815. https://doi.org/10.1016/j.agsy.2020.102815

Odum, H.T. 1996. *Environmental accounting: EMERGY and environmental decision making*. Wiley.

Odum, H.T. (Howard T. with Internet Archive) 1983. *Systems ecology: An introduction*. New York: Wiley. http://archive.org/details/systemsecologyin0000odum

OECD (2001). *Environmental indicators for agriculture: Volume 3: Methods and results* (OECD Environmental Indicators). OECD Publishing. https://doi.org/10.1787/9789264188556-en

Oesterheld, M., **Sala, O.E.**, **& McNaughton, S.J.** 1992. *Effect of animal husbandry on herbivore-carrying capacity at a regional scale*. Nature, 356(6366): 234–236. https://doi.org/10.1038/356234a0

Okoli, C. & Pawlowski, S.D. 2004. *The Delphi method as a research tool: An example, design considerations and applications*. Information & Management, 42(1): 15–29. https://doi.org/10.1016/j.im.2003.11.002

Ominski, K., **Gunte, K.**, **Wittenberg, K.**, **Legesse, G.**, **Mengistu, G.**, **& McAllister, T.** 2021. *The role of livestock in sustainable food production systems in Canada*. Canadian Journal of Animal Science, 101(4): 591–601. https://doi.org/10.1139/cjas-2021-0005

Oostvogels, V.J., **Dumont, B.**, **Nijland, H.J.**, **De Boer, I.J.M.**, **& Ripoll-Bosch, R.** 2024. *What about the negatives? An integrated framework for revealing diverse values of nature and its conservation*. People and Nature, 6(6): 2633–2646. https://doi.org/10.1002/pan3.10750

Orchard-Webb, J., **Kenter, J.O.**, **Bryce, R.**, **& Church, A.** 2016. *Deliberative Democratic Monetary Valuation to implement the Ecosystem Approach*. Ecosystem Services, 21: 308–318. https://doi.org/10.1016/j.ecoser.2016.09.005

Orenstein, D.E., **& Groner, E.** 2014. *In the eye of the stakeholder: Changes in perceptions of ecosystem services across an international border*. Ecosystem Services, 8: 185–196. https://doi.org/10.1016/j.ecoser.2014.04.004

Oteros-Rozas, E., **Martín-López, B.**, **Daw, T.M.**, **Bohensky, E.L.**, **Butler, J.R.A.**, **Hill, R.**, **Martin-Ortega, J.**, **Quinlan, A.**, **Ravera, F.**, **Ruiz-Mallén, I.**, **Thyresson, M.**, **Mistry, J.**, **Palomo, I.**, **Peterson, G.D.**, **Plieninger, T.**, **Waylen, K.A.**, **Beach, D.M.**, **Bohnet, I.C.**, **Hamann, M.**, **& Vilardy, S.P.** 2015. *Participatory scenario planning in place-based social-ecological research: Insights and experiences from 23 case studies*. Ecology and Society, 20(4), art32. https://doi.org/10.5751/ES-07985-200432

Pagiola, S., **& Platais** (n.d.). *Payments for environmental services* [Text/HTML]. World Bank. Retrieved February 10, 2025, from https://documents.worldbank.org/en/publication/documents-reports/documentdetail/983701468779667772/Payments-for-environmental-services

Palomo, I., **Martín-López, B.**, **Potschin, M.**, **Haines-Young, R.**, **& Montes, C.** 2013. *National Parks, buffer zones and surrounding lands: Mapping ecosystem service flows*. Ecosystem Services, 4: 104–116. https://doi.org/10.1016/j.ecoser.2012.09.001

Paparo, L., **Nocerino, R.**, **Ciaglia, E.**, **Di Scala, C.**, **De Caro, C.**, **Russo, R.**, **Trinchese, G.**, **Aitoro, R.**, **Amoroso, A.**, **Bruno, C.**, **Di Costanzo, M.**, **Passariello, A.**, **Messina, F.**, **Agangi, A.**, **Napolitano, M.**, **Voto, L.**, **Gatta, G.D.**, **Pisapia, L.**, **Montella, F.**, **& Berni Canani, R.** 2021. *Butyrate as a bioactive human milk protective component against food allergy*. Allergy, 76(5): 1398–1415. https://doi.org/10.1111/all.14625

Parel, A. 1969. *The political symbolism of the cow in India*. Journal of Commonwealth Political Studies, 7(3): 179–203. https://doi.org/10.1080/14662046908447106

Paruelo, J.M., **Texeira, M.**, **Staiano, L.**, **Mastrángelo, M.**, **Amdan, L.**, **& Gallego, F.** 2016. *An integrative index of Ecosystem Services provision based on remotely sensed data*. Ecological Indicators, 71: 145–154. https://doi.org/10.1016/j.ecolind.2016.06.054

Pascual, U., **Balvanera, P.**, **Anderson, C.B.**, **Chaplin-Kramer, R.**, **Christie, M.**, **González-Jiménez, D.**, **Martin, A.**, **Raymond, C.M.**, **Termansen, M.**, **Vatn, A.**, **Athayde, S.**, **Baptiste, B.**, **Barton, D.N.**, **Jacobs, S.**, **Kelemen, E.**, **Kumar, R.**, **Lazos, E.**, **Mwampamba, T.H.**, **Nakangu, B.**, **& Zent, E.** 2023. *Diverse values of nature for sustainability*. Nature, 620(7975): 813–823. https://doi.org/10.1038/s41586-023-06406-9

Pascual, U., **Muradian, R.**, **Rodríguez, L.C.**, **& Duraiappah, A.** 2010. *Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach*. Ecological Economics, 69(6): 1237–1244. https://doi.org/10.1016/j.ecolecon.2009.11.004

Patterson, M., **McDonald, G.**, **& Hardy, D.** 2017. *Is there more in common than we think? Convergence of ecological footprinting, emergy analysis, life cycle assessment and other methods of environmental accounting*. Ecological Modelling, 362: 19–36. https://doi.org/10.1016/j.ecolmodel.2017.07.022

Peh, K.S.H., **Balmford, A.**, **Bradbury, R.B.**, **Brown, C.**, **Butchart, S.H.M.**, **Hughes, F.M.R.**, **Stattersfield, A.**, **Thomas, D.H.L.**, **Walpole, M.**, **Bayliss, J.**, **Gowing, D.**, **Jones, J.P.G.**, **Lewis, S.L.**, **Mulligan, M.**, **Pandeya, B.**, **Stratford, C.**, **Thompson, J.R.**, **Turner, K.**, **Vira, B.**, **& Birch, J.C.** 2013. *TESSA: A toolkit for rapid assessment of ecosystem services at sites of biodiversity conservation importance*. Ecosystem Services, 5: 51–57. https://doi.org/10.1016/j.ecoser.2013.06.003

Pethick, D.W., **Bryden, W.L.**, **Mann, N.J.**, **Masters, D.G.**, **& Lean, I.J.** 2023. *The societal role of meat: The Dublin Declaration with an Australian perspective*. Animal Production Science, 63(18): 1805–1826. https://doi.org/10.1071/AN23061

Pfeiffer, M., **Hoffmann, M. P.**, **Scheiter, S.**, **Nelson, W.**, **Isselstein, J.**, **Ayisi, K.**, **Odhiambo, J. J.**, **& Rötter, R.** 2022. *Modelling the effects of alternative crop–livestock management scenarios on important ecosystem services for smallholder farming from a landscape perspective*. Biogeosciences, 19(16): 3935–3958. https://doi.org/10.5194/bg-19-3935-2022

Pike, K., **Wright, P.**, **Wink, B.**, **& Fletcher, S.** 2015. *The assessment of cultural ecosystem services in the marine environment using Q methodology*. Journal of Coastal Conservation, 19(5): 667–675. https://doi.org/10.1007/s11852-014-0350-z

Plummer, M. L. 2009. *Assessing benefit transfer for the valuation of ecosystem services*. Frontiers in Ecology and the Environment, 7(1): 38–45. https://doi.org/10.1890/080091

Potschin, M. B., **& Haines-Young, R. H.** 2011. *Ecosystem services: Exploring a geographical perspective*. Progress in Physical Geography: Earth and Environment, 35(5): 575–594. https://doi.org/10.1177/0309133311423172

Potschin, M., **& Haines-Young, R.** 2016. *Defining and Measuring Ecosystem Services*. In Routledge Handbook of Ecosystem Services. Routledge.

Poveda, J. 2021. *Insect frass in the development of sustainable agriculture. A review*. Agronomy for Sustainable Development, 41(1): 5. https://doi.org/10.1007/s13593-020-00656-x

Power, A. G. 2010. *Ecosystem services and agriculture: Tradeoffs and synergies*. Philosophical Transactions of the Royal Society B: Biological Sciences, 365(1554): 2959–2971. https://doi.org/10.1098/rstb.2010.0143

Provenza, F. D., **Meuret, M.**, **& Gregorini, P.** 2015. *Our landscapes, our livestock, ourselves: Restoring broken linkages among plants, herbivores, and humans with diets that nourish and satiate*. Appetite, 95: 500–519. https://doi.org/10.1016/j.appet.2015.08.004

Pulselli, F. M., **Coscieme, L.**, **& Bastianoni, S.** 2011. *Ecosystem services as a counterpart of emergy flows to ecosystems*. Ecological Modelling, 222(16): 2924–2928. https://doi.org/10.1016/j.ecolmodel.2011.04.022

Queiroz, C., **Meacham, M.**, **Richter, K.**, **Norström, A. V.**, **Andersson, E.**, **Norberg, J.**, **& Peterson, G.** 2015. *Mapping bundles of ecosystem services reveals distinct types of multifunctionality within a Swedish landscape*. AMBIO, 44(S1): 89–101. https://doi.org/10.1007/s13280-014-0601-0

Quinlan, R. J., **Rumas, I.**, **Naiskye, G.**, **Quinlan, M.**, **& Yoder, J.** 2016. *Searching for Symbolic Value of Cattle: Tropical Livestock Units, Market Price, and Cultural Value of Maasai Livestock*. Ethnobiology Letters, 7(1). https://doi.org/10.14237/ebl.7.1.2016.621

Ramirez-Reyes, C., **Brauman, K. A.**, **Chaplin-Kramer, R.**, **Galford, G. L.**, **Adamo, S. B.**, **Anderson, C. B.**, **Anderson, C.**, **Allington, G. R. H.**, **Bagstad, K. J.**, **Coe, M. T.**, **Cord, A. F.**, **Dee, L. E.**, **Gould, R. K.**, **Jain, M.**, **Kowal, V. A.**, **Muller-Karger, F. E.**, **Norriss, J.**, **Potapov, P.**, **Qiu, J.**, **& Wright, T. M.** 2019. *Reimagining the potential of Earth observations for ecosystem service assessments*. Science of The Total Environment, 665: 1053–1063. https://doi.org/10.1016/j.scitotenv.2019.02.150

Raudsepp-Hearne, C., **Peterson, G. D.**, **Tengö, M.**, **Bennett, E. M.**, **Holland, T.**, **Benessaiah, K.**, **MacDonald, G. K.**, **& Pfeifer, L.** 2010. *Untangling the Environmentalist's Paradox: Why Is Human Well-being Increasing as Ecosystem Services Degrade?* BioScience, 60(8): 576–589. https://doi.org/10.1525/bio.2010.60.8.4

Raynor, E. J., **Derner, J. D.**, **Hartman, M. D.**, **Dorich, C. D.**, **Parton, W. J.**, **Hendrickson, J. R.**, **Harmoney, K. R.**, **Brennan, J. R.**, **Owensby, C. E.**, **Kaplan, N. E.**, **Lutz, S. M.**, **Hoover, D. L.**, **& Augustine, D. J.** 2024. *Secondary production of the central rangeland region of the United States*. Ecological Applications, 34(5), e2978. https://doi.org/10.1002/eap.2978

Rees, W. E., **& Wackernagel, M.** 2023. *Ecological Footprint Accounting: Thirty Years and Still Gathering Steam*. Environment: Science and Policy for Sustainable Development, 65(5): 5–18. https://doi.org/10.1080/00139157.2023.2225405

Retka, J., **Jepson, P.**, **Ladle, R. J.**, **Malhado, A. C. M.**, **Vieira, F. A. S.**, **Normande, I. C.**, **Souza, C. N.**, **Bragagnolo, C.**, **& Correia, R. A.** 2019. *Assessing cultural ecosystem services of a large marine protected area through social media photographs*. Ocean & Coastal Management, 176: 40–48. https://doi.org/10.1016/j.ocecoaman.2019.04.018

Richter, F., **Jan, P.**, **El Benni, N.**, **Lüscher, A.**, **Buchmann, N.**, **& Klaus, V. H.** 2021. *A guide to assess and value ecosystem services of grasslands*. Ecosystem Services, 52: 101376. https://doi.org/10.1016/j.ecoser.2021.101376

Rieb, J. T., **Chaplin-Kramer, R.**, **Daily, G. C.**, **Armsworth, P. R.**, **Böhning-Gaese, K.**, **Bonn, A.**, **Cumming, G. S.**, **Eigenbrod, F.**, **Grimm, V.**, **Jackson, B. M.**, **Marques, A.**, **Pattanayak, S. K.**, **Pereira, H. M.**, **Peterson, G. D.**, **Ricketts, T. H.**, **Robinson, B. E.**, **Schröter, M.**, **Schulte, L. A.**, **Seppelt, R.**, **& Bennett, E. M.** 2017. *When, Where, and How Nature Matters for Ecosystem Services: Challenges for the Next Generation of Ecosystem Service Models*. BioScience, 67(9): 820–833. https://doi.org/10.1093/biosci/bix075

Rincón-Ruiz, A., **Arias-Arévalo, P.**, **Núñez Hernández, J. M.**, **Cotler, H.**, **Aguado Caso, M.**, **Meli, P.**, **Tauro, A.**, Ávila Akerberg, V. D., **Avila-Foucat, V. S.**, **Cardenas, J. P.**, **Castillo Hernández, L. A.**, **Castro, L. G.**, **Cerón Hernández, V. A.**, **Contreras Araque, A.**, **Deschamps-Lomeli, J.**, **Galeana-Pizaña, J. M.**, **Guillén Oñate, K.**, **Hernández Aguilar, J. A.**, **Jimenez, A. D.**, **& Waldron, T.** 2019. *Applying integrated valuation of ecosystem services in Latin America: Insights from 21 case studies*. Ecosystem Services, 36: 100901. https://doi.org/10.1016/j.ecoser.2019.100901

Ripoll-Bosch, R., **De Boer, I. J. M.**, **Bernués, A.**, **& Vellinga, T. V.** 2013. *Accounting for multi-functionality of sheep farming in the carbon footprint of lamb: A comparison of three contrasting Mediterranean systems*. Agricultural Systems, 116: 60–68. https://doi.org/10.1016/j.agsy.2012.11.002

Roberts, G. W., **Meng, X.**, **Psimoulis, P.**, **& Brown, C. J.** 2020. *Time Series Analysis of Rapid GNSS Measurements for Quasi-static and Dynamic Bridge Monitoring*. In J.-P. Montillet & M. S. Bos (Eds.), Geodetic Time Series Analysis in Earth Sciences (pp. 345–417). Springer International Publishing. https://doi.org/10.1007/978-3-030-21718-1_12

Robertson, A. D., **Paustian, K.**, **Ogle, S.**, **Wallenstein, M. D.**, **Lugato, E.**, **& Cotrufo, M. F.** 2019. *Unifying soil organic matter formation and persistence frameworks: The MEMS model*. Biogeosciences, 16(6): 1225–1248. https://doi.org/10.5194/bg-16-1225-2019

Rodríguez-Ortega, T., Olaizola, A. M., & Bernués, A. 2018. *A novel management-based system of payments for ecosystem services for targeted agri-environmental policy*. Ecosystem Services, 34: 74–84. https://doi.org/10.1016/j.ecoser.2018.09.007

Rodríguez-Ortega, T., Oteros-Rozas, E., Ripoll-Bosch, R., Tichit, M., Martín-López, B., & Bernués, A. 2014. *Applying the ecosystem services framework to pasture-based livestock farming systems in Europe*. Animal, 8(8): 1361–1372. https://doi.org/10.1017/S1751731114000421

Rositano, F., & Ferraro, D. O. 2014. *Ecosystem Services Provided by Agroecosystems: A Qualitative and Quantitative Assessment of this Relationship in the Pampa Region, Argentina*. Environmental Management, 53(3): 606–619. https://doi.org/10.1007/s00267-013-0211-9

Rótolo, G. C., Montico, S., Francis, C. A., & Ulgiati, S. 2015. *How land allocation and technology innovation affect the sustainability of agriculture in Argentina Pampas: An expanded life cycle analysis*. Agricultural Systems, 141: 79–93. https://doi.org/10.1016/j.agsy.2015.08.005

Rótolo, G. C., Rydberg, T., Lieblein, G., & Francis, C. 2007. *Emergy evaluation of grazing cattle in Argentina's Pampas*. Agriculture, Ecosystems & Environment, 119(3–4): 383–395. https://doi.org/10.1016/j.agee.2006.08.011

Rugani, B., & Benetto, E. 2012. *Improvements to Emergy Evaluations by Using Life Cycle Assessment*. Environmental Science & Technology, 46(9): 4701–4712. https://doi.org/10.1021/es203440n

Ryschawy, J., Disenhaus, C., Bertrand, S., Allaire, G., Aznar, O., Plantureux, S., Josien, E., Guinot, C., Lasseur, J., Perrot, C., Tchakerian, E., Aubert, C., & Tichit, M. 2017. *Assessing multiple goods and services derived from livestock farming on a nation-wide gradient*. Animal, 11(10): 1861–1872. https://doi.org/10.1017/S1751731117000829

Ryschawy, J., Dumont, B., Therond, O., Donnars, C., Hendrickson, J., Benoit, M., & Duru, M. 2019. *Review: An integrated graphical tool for analysing impacts and services provided by livestock farming*. Animal, 13(8): 1760–1772. https://doi.org/10.1017/S1751731119000351

Sarkar, D., Kar, S. K., Chattopadhyay, A., Shikha, Rakshit, A., Tripathi, V. K., Dubey, P. K., & Abhilash, P. C. 2020. *Low input sustainable agriculture: A viable climate-smart option for boosting food production in a warming world*. Ecological Indicators, 115: 106412. https://doi.org/10.1016/j.ecolind.2020.106412

Satterfield, T. 2001. *In Search of Value Literacy: Suggestions for the Elicitation of Environmental Values*. Environmental Values, 10(3): 331–359. https://doi.org/10.3197/096327101129340868

Sauer, J., & Wossink, A. 2013. *Marketed outputs and non-marketed ecosystem services: The evaluation of marginal costs*. European Review of Agricultural Economics, 40(4): 573–603. https://doi.org/10.1093/erae/jbs040

Sawyer, G., Roberts, I., Imaz, J., & Refshauge, G. n.d. *Automatic monitoring of body weight of Poll Dorset ewes in late gestation and lactation*.

Schaafsma, M., Bartkowski, B., & Lienhoop, N. 2018. *Guidance for Deliberative Monetary Valuation Studies*. International Review of Environmental and Resource Economics, 12(2–3): 267–323. https://doi.org/10.1561/101.00000103

Schils, R. L. M., Bufe, C., Rhymer, C. M., Francksen, R. M., Klaus, V. H., Abdalla, M., Milazzo, F., Lellei-Kovács, E., Berge, H. T., Bertora, C., Chodkiewicz, A., Dǎmǎtîrcǎ , C., Feigenwinter, I., Fernández-Rebollo, P., Ghiasi, S., Hejduk, S., Hiron, M., Janicka, M., Pellaton, R., & Price, J. P. N. 2022. *Permanent grasslands in Europe: Land use change and intensification decrease their multifunctionality*. Agriculture, Ecosystems & Environment, 330: 107891. https://doi.org/10.1016/j.agee.2022.107891

Schneiders, A., Van Daele, T., Van Landuyt, W., & Van Reeth, W. 2012. *Biodiversity and ecosystem services: Complementary approaches for ecosystem management?* Ecological Indicators, 21: 123–133. https://doi.org/10.1016/j.ecolind.2011.06.021

Scholte, S. S. K., Van Teeffelen, A. J. A., & Verburg, P. H. 2015. *Integrating socio-cultural perspectives into ecosystem service valuation: A review of concepts and methods*. Ecological Economics, 114: 67–78. https://doi.org/10.1016/j.ecolecon.2015.03.007

Schröter, M., Zanden, E. H. van der, Oudenhoven, A. P. E. van, Remme, R. P., Serna-Chavez, H. M., Groot, R. S. de, & Opdam, P. 2014. *Ecosystem Services as a Contested Concept: A Synthesis of Critique and Counter-arguments*. Conservation Letters, 7(6): 514–523. https://doi.org/10.1111/conl.12091

Scowen, M., Athanasiadis, I. N., Bullock, J. M., Eigenbrod, F., & Willcock, S. 2021. *The current and future uses of machine learning in ecosystem service research*. Science of The Total Environment, 799: 149263. https://doi.org/10.1016/j.scitotenv.2021.149263

Senf, C. 2022. *Seeing the System from Above: The Use and Potential of Remote Sensing for Studying Ecosystem Dynamics*. Ecosystems, 25(8): 1719–1737. https://doi.org/10.1007/s10021-022-00777-2

Seppelt, R., Lautenbach, S., & Volk, M. 2013. *Identifying trade-offs between ecosystem services, land use, and biodiversity: A plea for combining scenario analysis and optimization on different spatial scales*. Current Opinion in Environmental Sustainability, 5(5): 458–463. https://doi.org/10.1016/j.cosust.2013.05.002

Seré, C., & Steinfeld, H. (with FAO) 1996. *World livestock production systems: Current status, issues and trends*. FAO.

Shi, Y., **Pinsard, C.**, **& Accatino, F.** 2021. *Land sharing strategies for addressing the trade-off between carbon storage and crop production in France*. Regional Environmental Change, 21(4): 92. https://doi.org/10.1007/s10113-021-01818-7

Shipley, N. J., **Johnson, D. N.**, **Van Riper, C. J.**, **Stewart, W. P.**, **Chu, M. L.**, **Suski, C. D.**, **Stein, J. A.**, **& Shew, J. J.** 2020. *A deliberative research approach to valuing agro-ecosystem services in a worked landscape*. Ecosystem Services, 42: 101083. https://doi.org/10.1016/j.ecoser.2020.101083

Sieber, R. 2006. *Public Participation Geographic Information Systems: A Literature Review and Framework*. Annals of the Association of American Geographers, 96(3): 491–507. https://doi.org/10.1111/j.1467-8306.2006.00702.x

Silveira, E. M. D. O., **Terra, M. D. C. N. S.**, **Ter Steege, H.**, **Maeda, E. E.**, **Acerbi Júnior, F. W.**, **& Scolforo, J. R. S.** 2019. *Carbon-diversity hotspots and their owners in Brazilian southeastern Savanna, Atlantic Forest and Semi-Arid Woodland domains*. Forest Ecology and Management, 452: 117575. https://doi.org/10.1016/j.foreco.2019.117575

Silvertown, J., **Poulton, P.**, **Johnston, E.**, **Edwards, G.**, **Heard, M.**, **& Biss, P. M.** 2006. *The Park Grass Experiment 1856–2006: Its contribution to ecology*. Journal of Ecology, 94(4): 801–814. https://doi.org/10.1111/j.1365-2745.2006.01145.x

Silvestri, S., **Osano, P. M.**, **Leeuw, J. de**, **Herrero, M.**, **Ericksen, P. J.**, **Kariuki, J. B.**, **Njuki, J.**, **Bedelian, C.**, **& Notenbaert, A. M. O.** 2012. *Greening livestock: Assessing the potential of payment for environmental services in livestock inclusive agricultural production systems in developing countries*. https://hdl.handle.net/10568/21188

Slovák, Ľ., **Daněk, J.**, **& Daněk, T.** 2023. *The use of focus groups in cultural ecosystem services research: A systematic review*. Humanities and Social Sciences Communications, 10(1): 45. https://doi.org/10.1057/s41599-023-01530-3

Smith, P., **Ashmore, M. R.**, **Black, H. I. J.**, **Burgess, P. J.**, **Evans, C. D.**, **Quine, T. A.**, **Thomson, A. M.**, **Hicks, K.**, **& Orr, H. G.** 2013. *REVIEW: The role of ecosystems and their management in regulating climate, and soil, water and air quality*. Journal of Applied Ecology, 50(4): 812–829. https://doi.org/10.1111/1365-2664.12016

Smith, S., **Rowcroft, P.**, **Everard, M.**, **& Couldrick, L.** 2013. *Payments for ecosystem services: A best practice guide*.

Sousa, B., **Silveira, V.**, **& Rocca, V.** 2020. *Sustentabilidade de sgroecossistemas familiares no sul da Itália*.

Spash, C. L. 2007. *Deliberative monetary valuation (DMV): Issues in combining economic and political processes to value environmental change*. Ecological Economics, 63(4): 690–699. https://doi.org/10.1016/j.ecolecon.2007.02.014

Staiano, L., **Camba Sans, G. H.**, **Baldassini, P.**, **Gallego, F.**, **Texeira, M. A.**, **& Paruelo, J. M.** 2021. *Putting the Ecosystem Services idea at work: Applications on impact assessment and territorial planning*. Environmental Development, 38: 100570. https://doi.org/10.1016/j.envdev.2020.100570

Steinfeld, H., **Wassenaar, T.**, **& Jutzi, S.** 2006. *Situación, fuerzas motrices y tendencias de los sistemas de producción agropecuaria en los países en desarrollo: -EN- Livestock production systems in developing countries: status, drivers, trends -FR- Les systèmes de production animale dans les pays en développement : statuts, moteurs, tendances -ES-*. Revue Scientifique et Technique de l'OIE, 25(2): 505–516. https://doi.org/10.20506/rst.25.2.1677

Sterman, J. 2000. *Business Dynamics, System Thinking and Modelling for a Complex World*. Http://Lst-liep.liep-Unesco.Org/Cgi-Bin/Wwwi32.Exe/[In=epidoc1.in]/?T2000=013598/(100), 19.

Sun, Z., **& Müller, D.** 2013. *A framework for modelling payments for ecosystem services with agent-based models, Bayesian belief networks and opinion dynamics models*. Environmental Modelling & Software, 45: 15–28. https://doi.org/10.1016/j.envsoft.2012.06.007

Sutton, M. A., **Oenema, O.**, **Erisman, J. W.**, **Leip, A.**, **Van Grinsven, H.**, **& Winiwarter, W.** 2011. *Too much of a good thing*. Nature, 472(7342): 159–161. https://doi.org/10.1038/472159a

Swinnen, E., **Sterckx, S.**, **Wirion, C.**, **Verbeiren, B.**, **& Wens, D.** 2022. *Harmonization of Multi-Mission High-Resolution Time Series: Application to BELAIR*. Remote Sensing, 14(5): 1163. https://doi.org/10.3390/rs14051163

Taoumi, H., **& Lahrech, K.** 2023. *Economic, environmental and social efficiency and effectiveness development in the sustainable crop agricultural sector: A systematic in-depth analysis review*. Science of The Total Environment, 901: 165761. https://doi.org/10.1016/j.scitotenv.2023.165761

Taube, F., **Nyameasem, J. K.**, **Fenger, F.**, **Alderkamp, L.**, **Kluß, C.**, **& Loges, R.** 2024. *Eco-efficiency of leys – The trigger for sustainable integrated crop-dairy farming systems*. Grass and Forage Science, 79(2): 108–119. https://doi.org/10.1111/gfs.12639

Tauro, A., **Gómez-Baggethun, E.**, **García-Frapolli, E.**, **Lazos Chavero, E.**, **& Balvanera, P.** 2018. *Unraveling heterogeneity in the importance of ecosystem services: Individual views of smallholders*. Ecology and Society, 23(4): art11. https://doi.org/10.5751/ES-10457-230411

TEEB. 2010. *TEEB | UNEP - UN Environment Programme*. https://www.unep.org/topics/teeb

Tenza-Peral, A., **Ripoll-Bosch, R.**, **Casasús, I.**, **Martín-Collado, D.**, **& Bernués, A.** 2023. *Sustaining biodiversity and ecosystem services with agricultural production*. In *Sustainable Development and Pathways for Food Ecosystems: Integration and Synergies* (pp. 129–146). Elsevier. https://doi.org/10.1016/B978-0-323-90885-6.00013-2

Tian, T., **Sun, L.**, **Peng, S.**, **Sun, F.**, **& Che, Y.** 2021. *Understanding the process from perception to cultural ecosystem services assessment by comparing valuation methods*. Urban Forestry & Urban Greening, 57: 126945. https://doi.org/10.1016/j.ufug.2020.126945

Tomscha, S. A., **& Gergel, S. E.** 2016. *Ecosystem service trade-offs and synergies misunderstood without landscape history*. Ecology and Society, 21(1): art43. https://doi.org/10.5751/ES-08345-210143

Trinchese, G., **Cimmino, F.**, **Cavaliere, G.**, **Rosati, L.**, **Catapano, A.**, **Sorriento, D.**, **Murru, E.**, **Bernardo, L.**, **Pagani, L.**, **Bergamo, P.**, **Scudiero, R.**, **Iaccarino, G.**, **Greco, L.**, **Banni, S.**, **Crispino, M.**, **& Mollica, M. P.** 2021. *Heart Mitochondrial Metabolic Flexibility and Redox Status Are Improved by Donkey and Human Milk Intake*. Antioxidants, 10(11), Article 11. https://doi.org/10.3390/antiox10111807

Tucker, C. J. 1979. *Red and photographic infrared linear combinations for monitoring vegetation*. Remote Sensing of Environment, 8(2), 127–150. https://doi.org/10.1016/0034-4257(79)90013-0

Tukey, J. W. 1977. *Exploratory Data Analysis*. Addison-Wesley Publishing Company.

Turner, B., **Menendez, H.**, **Gates, R.**, **Tedeschi, L.**, **& Atzori, A.** 2016. *System Dynamics Modelling for Agricultural and Natural Resource Management Issues: Review of Some Past Cases and Forecasting Future Roles*. Resources, 5(4), 40. https://doi.org/10.3390/resources5040040

Turnhout, E., **Metze, T.**, **Wyborn, C.**, **Klenk, N.**, **& Louder, E.** 2020. *The politics of co-production: Participation, power, and transformation*. Current Opinion in Environmental Sustainability, 42, 15–21. https://doi.org/10.1016/j.cosust.2019.11.009

Uehara, T., **Nagase, Y.**, **& Wakeland, W.** 2016. *Integrating Economics and System Dynamics Approaches for Modelling an Ecological–Economic System*. Systems Research and Behaviour al Science, 33(4), 515–531. https://doi.org/10.1002/sres.2373

UK-NEA. 2014. *UK National Ecosystem Assessment*. UNEP-WCMC, LWEC, UK.

Ulgiati, S., **Ascione, M.**, **Bargigli, S.**, **Cherubini, F.**, **Federici, M.**, **Franzese, P. P.**, **Raugei, M.**, **Viglia, S.**, **& Zucaro, A.** 2010. *Multi-method and Multi-scale Analysis of Energy and Resource Conversion and Use*. In F. Barbir & S. Ulgiati (Eds.), *Energy Options Impact on Regional Security* (pp. 1–36). Springer Netherlands. https://doi.org/10.1007/978-90-481-9565-7_1

Ulgiati, S., **Raugei, M.**, **& Bargigli, S.** 2006. *Overcoming the inadequacy of single-criterion approaches to Life Cycle Assessment*. Ecological Modelling, 190(3), 432–442. https://doi.org/10.1016/j.ecolmodel.2005.03.022

UNECE. 2012. Decision 2012 / 10 Adoption of guidance document on national nitrogen budgets ECE/EB.AIR.

US-EPA. 2015. *National Ecosystem Services Classification System (NESCS): Framework Design and Policy Application*. UNEP.

Valani, G.P., Martíni, A.F., Da Silva, L.F.S., Bovi, R.C. & Cooper, M. 2020. Soil quality assessments in integrated crop–livestock–forest systems: A review. *Soil Use and Management*, 37(1): 22–36. https://doi.org/10.1111/sum.12667

Vallet, A., **Locatelli, B.**, **Levrel, H.**, **Wunder, S.**, **Seppelt, R.**, **Scholes, R. J.**, **& Oszwald, J.** 2018. *Relationships Between Ecosystem Services: Comparing Methods for Assessing Tradeoffs and Synergies*. Ecological Economics, 150, 96–106. https://doi.org/10.1016/j.ecolecon.2018.04.002

Van Der Schatte Olivier, A., **Jones, L.**, **Vay, L. L.**, **Christie, M.**, **Wilson, J.**, **& Malham, S. K.** 2020. *A global review of the ecosystem services provided by bivalve aquaculture*. Reviews in Aquaculture, 12(1), 3–25. https://doi.org/10.1111/raq.12301

Van Oudenhoven, A. P. E., **Schröter, M.**, **Drakou, E. G.**, **Geijzendorffer, I. R.**, **Jacobs, S.**, **Van Bodegom, P. M.**, **Chazee, L.**, **Czúcz, B.**, **Grunewald, K.**, **Lillebø, A. I.**, **Mononen, L.**, **Nogueira, A. J. A.**, **Pacheco-Romero, M.**, **Perennou, C.**, **Remme, R. P.**, **Rova, S.**, **Syrbe, R.-U.**, **Tratalos, J. A.**, **Vallejos, M.**, **& Albert, C.** 2018. *Key criteria for developing ecosystem service indicators to inform decision making*. Ecological Indicators, 95, 417–426. https://doi.org/10.1016/j.ecolind.2018.06.020

Van Zanten, H. H. E., **Herrero, M.**, **Van Hal, O.**, **Röös, E.**, **Muller, A.**, **Garnett, T.**, **Gerber, P. J.**, **Schader, C.**, **& De Boer, I. J. M.** 2018. *Defining a land boundary for sustainable livestock consumption*. Global Change Biology, 24(9), 4185–4194. https://doi.org/10.1111/gcb.14321

Velasco-Muñoz, J. F., **Aznar-Sánchez, J. A.**, **Schoenemann, M.**, **& López-Felices, B.** 2022. *An Analysis of the Worldwide Research on the Socio-Cultural Valuation of Forest Ecosystem Services*. Sustainability, 14(4), 2089. https://doi.org/10.3390/su14042089

Vihervaara, P., **Mononen, L.**, **Nedkov, S.**, **Viinikka, A.**, **Adamescu, C. M.**, **Arnell, A.**, **Balzan, M.**, **Bicking, S.**, **Broekx, S.**, **Burkhard, B.**, **Cazacu, C.**, **Czúcz, B.**, **Geneletti, D.**, **Gret-Regamey, A.**, **Harmáčková, Z.**, **Karvinen, V.**, **Kruse, M. K.**, **Liekens, I.**, **Ling, M.**, **Zulian, G.** 2018. *Biophysical Mapping and Assessment Methods for Ecosystem Services*. https://doi.org/10.13140/RG.2.2.25697.48483

Vissers, L. S. M., **Jongeneel, R. A.**, **Saatkamp, H. W.**, **& Oude Lansink, A. G. J. M.** 2022. *A multiple-standards framework to address externalities resulting from meat production*. Applied Economic Perspectives and Policy, 44(2), 946–959. https://doi.org/10.1002/aepp.13152

Wackernagel, M., **& Rees, W. E.** 1996. *Our ecological footprint: Reducing human impact on the earth*. New Society Publishers.

Walz, A., **Schmidt, K.**, **Ruiz-Frau, A.**, **Nicholas, K. A.**, **Bierry, A.**, **De Vries Lentsch, A.**, **Dyankov, A.**, **Joyce, D.**, **Liski, A. H.**, **Marbà, N.**, **Rosário, I. T.**, **& Scholte, S. S. K.** 2019. *Sociocultural valuation of ecosystem services for operational ecosystem management: Mapping applications by decision contexts in Europe*. Regional Environmental Change, 19(8), 2245–2259. https://doi.org/10.1007/s10113-019-01506-7

Wang, C., **Li, X.**, **Yu, H.**, **& Wang, Y.** 2019. *Tracing the spatial variation and value change of ecosystem services in Yellow River Delta, China*. Ecological Indicators, 96, 270–277. https://doi.org/10.1016/j.ecolind.2018.09.015

Wang, Q., **Xiao, H.**, **Ma, Q.**, **Yuan, X.**, **Zuo, J.**, **Zhang, J.**, **Wang, S.**, **& Wang, M.** 2020. *Review of Emergy Analysis and Life Cycle Assessment: Coupling Development Perspective*. Sustainability, 12(1), 367. https://doi.org/10.3390/su12010367

Wang, Z., Kumar, J., Weintraub-Leff, S. R., Todd-Brown, K., Mishra, U., & Sihi, D. 2024. Upscaling soil organic carbon measurements at the continental scale using multivariate clustering analysis and machine learning. *Journal of Geophysical Research: Biogeosciences*, 129(2), e2023JG007702. https://doi.org/10.1029/2023JG007702

Watson, R., **Dixon, J.**, **Hamburg, M.**, **Janetos, A.**, **& Moss, R.** 1998. *Protecting our planet, securing our future. Linkages among global environmental issues and human needs*. United Nations Environment Programme, U.S. National Aeronautics and Space Administration, World Bank. https://www.ecolex.org/details/literature/protecting-our-planet-securing-our-future-linkages-among-global-environmental-issues-and-human-needs-mon-064273/

Weber, M. A., **Caplan, S.**, **Ringold, P.**, **& Blocksom, K.** 2017. Rivers and streams in the media: a content analysis of ecosystem services. *Ecology and society: a journal of integrative science for resilience and sustainability*, 22(3), 15. https://doi.org/10.5751/ES-09496-220315

Weiler, V., **Udo, H. M.**, **Viets, T.**, **Crane, T. A.**, **& De Boer, I. J.** 2014. *Handling multi-functionality of livestock in a life cycle assessment: The case of smallholder dairying in Kenya*. Current Opinion in Environmental Sustainability, 8, 29–38. https://doi.org/10.1016/j.cosust.2014.07.009

Weiskopf, S. R., **Myers, B. J. E.**, **Arce-Plata, M. I.**, **Blanchard, J. L.**, **Ferrier, S.**, **Fulton, E. A.**, **Harfoot, M.**, **Isbell, F.**, **Johnson, J. A.**, **Mori, A. S.**, **Weng, E.**, **Harmáčková, Z. V.**, **Londoño-Murcia, M. C.**, **Miller, B. W.**, **Pereira, L. M.**, **& Rosa, I. M. D.** 2022. *A Conceptual Framework to Integrate Biodiversity, Ecosystem Function, and Ecosystem Service Models*. BioScience, 72(11), 1062–1073. https://doi.org/10.1093/biosci/biac074

Weitzman, J. 2019. *Applying the ecosystem services concept to aquaculture: A review of approaches, definitions, and uses*. Ecosystem Services, 35, 194–206. https://doi.org/10.1016/j.ecoser.2018.12.009

Westerink, J., **Hassink, J.**, **Plomp, M.**, **& van Os, J.** 2024. *Towards more biodiverse agricultural landscapes: How to make species-rich grassland a desirable and feasible option for dairy farmers*. Journal of Rural Studies, 105, 103195. https://doi.org/10.1016/j.jrurstud.2023.103195

Westman, W. E. 1977. *How Much Are Nature's Services Worth?: Measuring the social benefits of ecosystem functioning is both controversial and illuminating*. Science, 197(4307), 960–964. https://doi.org/10.1126/science.197.4307.960

White, B., **& Hanley, N.** 2016. *Should We Pay for Ecosystem Service Outputs, Inputs or Both?* Environmental and Resource Economics, 63(4), 765–787. https://doi.org/10.1007/s10640-016-0002-x

Willcock, S., **Hooftman, D. A. P.**, **Balbi, S.**, **Blanchard, R.**, **Dawson, T. P.**, **O'Farrell, P. J.**, **Hickler, T.**, **Hudson, M. D.**, **Lindeskog, M.**, **Martinez-Lopez, J.**, **Mulligan, M.**, **Reyers, B.**, **Shackleton, C.**, **Sitas, N.**, **Villa, F.**, **Watts, S. M.**, **Eigenbrod, F.**, **& Bullock, J. M.** 2019. *A Continental-Scale Validation of Ecosystem Service Models*. Ecosystems, 22(8), 1902–1917. https://doi.org/10.1007/s10021-019-00380-y

Wilson, M. 2013. *The Green Economy: The Dangerous Path of Nature Commoditization*. Consilience, 10, Article 10. https://doi.org/10.7916/consilience.v0i10.3934

Wong, C. P., **Jiang, B.**, **Kinzig, A. P.**, **Lee, K. N.**, **& Ouyang, Z.** 2015. *Linking ecosystem characteristics to final ecosystem services for public policy*. Ecology Letters, 18(1), 108–118. https://doi.org/10.1111/ele.12389

Wossink, A., **& Swinton, S. M.** 2007. *Jointness in production and farmers' willingness to supply non-marketed ecosystem services*. Ecological Economics, 64(2), 297–304. https://doi.org/10.1016/j.ecolecon.2007.07.003

Wu, L., **Zhang, X.**, **Griffith, B. A.**, **& Misselbrook, T. H.** 2016. *Sustainable grassland systems: A modelling perspective based on the North Wyke Farm Platform*. European Journal of Soil Science, 67(4), 397–408. https://doi.org/10.1111/ejss.12304

Wunder, S. 2005. *Payments for environmental services: Some nuts and bolts*. https://doi.org/10.17528/cifor/001765

Wunder, S. 2007. *The Efficiency of Payments for Environmental Services in Tropical Conservation*. Conservation Biology, 21(1), 48–58. https://doi.org/10.1111/j.1523-1739.2006.00559.x

Wunder, S. 2015. *Revisiting the concept of payments for environmental services*. Ecological Economics, 117, 234–243. https://doi.org/10.1016/j.ecolecon.2014.08.016

Wunder, S., Börner, J., Ezzine-de-Blas, D., Feder, S., & Pagiola, S. 2020. Payments for environmental services: Past performance and pending potentials. *Annual Review of Resource Economics*, 12(1), 209-234. https://doi.org/10.1146/annurev-resource-100518-094206

Xu, W., & Mage, J. A. 2001. A review of concepts and criteria for assessing agroecosystem health including a preliminary case study of southern Ontario. *Agriculture, ecosystems & environment*, 83(3), 215-233. https://doi.org/10.1016/S0167-8809(00)00159-6

Young, J. C., **Rose, D. C.**, **Mumby, H. S.**, **Benitez-Capistros, F.**, **Derrick, C. J.**, **Finch, T.**, **Garcia, C.**, **Home, C.**, **Marwaha, E.**, **Morgans, C.**, **Parkinson, S.**, **Shah, J.**, **Wilson, K. A.**, **& Mukherjee, N.** 2018. *A methodological guide to using and reporting on interviews in conservation science research*. Methods in Ecology and Evolution, 9(1), 10–19. https://doi.org/10.1111/2041-210X.12828

Yu, H., **Wang, Y.**, **Li, X.**, **Wang, C.**, **Sun, M.**, **& Du, A.** 2019. *Measuring ecological capital: State of the art, trends, and challenges*. Journal of Cleaner Production, 219, 833–845. https://doi.org/10.1016/j.jclepro.2019.02.014

Zabala, A., **Sandbrook, C.**, **& Mukherjee, N.** 2018. *When and how to use Q methodology to understand perspectives in conservation research*. Conservation Biology, 32(5), 1185–1194. https://doi.org/10.1111/cobi.13123

Zanasi, C., **Rabboni, C.**, **Rota, C.**, **Bungenstab, D. J.**, **& Laura, V. A.** 2020. *The Carne Carbono Neutro Accordance to Brazilian Consumers' Attitude towards Beef*. International Journal on Food System Dynamics, 11, 360–376. https://doi.org/10.18461/IJFSD.V11I4.60

Zeng, Y., **Hao, D.**, **Huete, A.**, **Dechant, B.**, **Berry, J.**, **Chen, J. M.**, **Joiner, J.**, **Frankenberg, C.**, **Bond-Lamberty, B.**, **Ryu, Y.**, **Xiao, J.**, **Asrar, G. R.**, **& Chen, M.** 2022. *Optical vegetation indices for monitoring terrestrial ecosystems globally*. Nature Reviews Earth & Environment, 3(7), 477–493. https://doi.org/10.1038/s43017-022-00298-5

Zhang, C., **Su, B.**, **Beckmann, M.**, **& Volk, M.** 2024. *Emergy-based evaluation of ecosystem services: Progress and perspectives*. Renewable and Sustainable Energy Reviews, 192, 114201. https://doi.org/10.1016/j.rser.2023.114201

Zhang, L., **Tian, H.**, **Shi, H.**, **Pan, S.**, **Chang, J.**, **Dangal, S. R. S.**, **Qin, X.**, **Wang, S.**, **Tubiello, F. N.**, **Canadell, J. G.**, **& Jackson, R. B.** 2022. *A 130-year global inventory of methane emissions from livestock: Trends, patterns, and drivers*. Global Change Biology, 28(17), 5142–5158. https://doi.org/10.1111/gcb.16280

Zhang, W., **Ricketts, T. H.**, **Kremen, C.**, **Carney, K.**, **& Swinton, S. M.** 2007. *Ecosystem services and dis-services to agriculture*. Ecological Economics, 64(2), 253–260. https://doi.org/10.1016/j.ecolecon.2007.02.024

Zhao, S., **Li, Z.**, **& Li, W.** 2005. *A modified method of ecological footprint calculation and its application*. Ecological Modelling, 185(1), 65–75. https://doi.org/10.1016/j.ecolmodel.2004.11.016

Zhao, S., **& Wu, C.** 2015. *Valuation of mangrove ecosystem services based on emergy: A case study in China*. International Journal of Environmental Science and Technology, 12(3), 967–974. https://doi.org/10.1007/s13762-013-0458-y

Zimmerer, K. S., **Lambin, E. F.**, **& Vanek, S. J.** 2018. *Smallholder telecoupling and potential sustainability*. Ecology and Society, 23(1), art30. https://doi.org/10.5751/ES-09935-230130

Zoderer, B. M., **Lupo Stanghellini, P. S.**, **Tasser, E.**, **Walde, J.**, **Wieser, H.**, **& Tappeiner, U.** 2016. *Exploring socio-cultural values of ecosystem service categories in the Central Alps: The influence of sociodemographic factors and landscape type*. Regional Environmental Change, 16(7), 2033–2044. https://doi.org/10.1007/s10113-015-0922-y

Appendix 1 References

Alcaraz-Segura, D., Paruelo, J., Epstein, H. & Cabello, J. 2013. Environmental and human controls of ecosystem functional diversity in temperate South America. *Remote Sensing*, 5(1): 127–154. https://doi.org/10.3390/rs5010127

Baeza, S. & Paruelo, J.M. 2018. Spatial and temporal variation of human appropriation of net primary production in the Rio de la Plata grasslands. *ISPRS Journal of Photogrammetry and Remote Sensing*, 145: 238–249. https://doi.org/10.1016/j.isprsjprs.2018.07.014

Baeza, S., Vélez-Martin, E., De Abelleyra, D., Banchero, S., Gallego, F., Schirmbeck, J., Veron, S., Vallejos, M., Weber, E., Oyarzabal, M., Barbieri, A., Petek, M., Guerra Lara, M., Sarrailhé, S.S., Baldi, G., Bagnato, C., Bruzzone, L., Ramos, S. & Hasenack, H. 2022. Two decades of land cover mapping in the Río de la Plata grassland region: The MapBiomas Pampa initiative. *Remote Sensing Applications: Society and Environment*, 28: 100834. https://doi.org/10.1016/j.rsase.2022.100834

Blumetto, O., Castagna, A., Cardozo, G., García, F., Tiscornia, G., Ruggia, A., Scarlato, S., Albicette, M.M., Aguerre, V. & Albin, A. 2019. Ecosystem Integrity Index, an innovative environmental evaluation tool for agricultural production systems. *Ecological Indicators*, 101: 725–733. https://doi.org/10.1016/j.ecolind.2019.01.077

Brasil (Instituto Brasileiro de Geografia e Estatística). 2021. *Ecosystem accounts: species threatened with extinction in Brazil* – 2014. Brasília. Available at: https://www.ibge.gov.br/en/statistics/economic/national-accounts/28954-ecosystem-accounting.html

Bungenstab, D. & Almeida, R. 2014. *Integrated crop–livestock–forestry systems: A Brazilian experience for sustainable farming.* DIEA, 2023.

Dumont, B., Groot, J.C.J. & Tichit, M. 2018. Review: Make ruminants green again – how can sustainable intensification and agroecology converge for a better future? *Animal*, 12: s210–s219. https://doi.org/10.1017/S1751731118001350

Dumont, B., Ryschawy, J., Duru, M., Benoit, M., Chatellier, V., Delaby, L., Donnars, C., Dupraz, P., Lemauviel-Lavenant, S., Méda, B., Vollet, D. & Sabatier, R. 2019. Review: Associations among goods, impacts and ecosystem services provided by livestock farming. *Animal*, 13(8): 1773–1784. https://doi.org/10.1017/S1751731118002586

Embrapa. N.d. *International cooperation – Brazilian Agricultural Research Corporation.* https://www.embrapa.br/en/international (cited 1 June 2025).

Feltran-Barbieri, R. & Féres, J.G. 2021. Degraded pastures in Brazil: Improving livestock production and forest restoration. *Royal Society Open Science*, 8(7): 201854. https://doi.org/10.1098/rsos.201854

Figueroa, D., Galicia, L. & Suárez Lastra, M. 2022. Latin American cattle ranching sustainability debate: An approach to social–ecological systems and spatial–temporal scales. *Sustainability*, 14(14): 8924. https://doi.org/10.3390/su14148924

Gallego, F., Bagnato, C., Baeza, S., Camba-Sans, G. & Paruelo, J. 2023. Río de la Plata grasslands: How did land-cover and ecosystem functioning change in the twenty-first century? In **Overbeck, G.E., Pillar, V.D.P., Müller, S.C. & Bencke, G.A.,** eds. *South Brazilian grasslands*, pp. 475–493. Springer International Publishing. https://doi.org/10.1007/978-3-031-42580-6_18

Ministerio de Ganadería, Agricultura y Pesca. 2021. *Actualización de cobertura y uso del suelo de Uruguay al año 2020/2021*. Montevideo: MGAP. [Cited 10 September 2025]. Available at: https://www.mgap.gub.uy

Ministerio de Ganadería, Agricultura y Pesca. 2023. *Anuario estadístico agropecuario 2023.* DIEA.

Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D. & Moore, R. 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. *Remote Sensing of Environment,* 202: 18–27. https://doi.org/10.1016/j.rse.2017.06.031

Gouvello, C., Soares Filho, S. & Nassar, A. 2011. *Brazil low-carbon country case study. Land use, land-use change and forestry: technical synthesis report.* Washington, DC: World Bank. Available at: https://documents1.worldbank.org/curated/en/753311468013874292/pdf/698690ESW0P1050020110English0report.pdf

De Groot, R.S., Wilson, M.A. & Boumans, R.M.J. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics,* 41(3): 393–408. https://doi.org/10.1016/S0921-8009(02)00089-7

Guimarães Junior, R., Martha Junior, G.B., Macedo, M.C.M., Marchão, R.L., Guimarães Júnior, R., Pulrolnik, K. & Maciel, G.A. 2020. Integrated crop–livestock systems in the Cerrado region. *Pesquisa Agropecuária Brasileira,* 46: 1127–1138. https://doi.org/10.1590/S0100-204X2011001000003

Herrero-Jáuregui, C. & Oesterheld, M. 2018. Effects of grazing intensity on plant richness and diversity: A meta-analysis. *Oikos,* 127(6): 757–766. https://doi.org/10.1111/oik.04893

Hötzel, M.J. & Vandresen, B. 2022. Brazilians' attitudes to meat consumption and production: Present and future challenges to the sustainability of the meat industry. *Meat Science,* 192: 108893. https://doi.org/10.1016/j.meatsci.2022.108893

Imaflora. 2023. Website of the Instituto de Manejo e Certificação Florestal e Agrícola. In: *Imaflora.* São Paulo, Brazil, Imaflora. [Cited 10 September 2025]. https://www.imaflora.org/

Landau, E.C., with Silva, G.A. da, Moura, L., Hirsch, A. & Guimarães, D.P. 2020. *Dinâmica da produção agropecuária e da paisagem natural no Brasil nas últimas décadas.* Embrapa.

MapBiomas Uruguay. N.D. *Uruguay land use and coverage mapping platform.* [Cited 4 June 2025]. https://uruguay.mapbiomas.org/

MEA (Ed.) 2005. *Ecosystems and human well-being: Synthesis.* Island Press.

Ministerio de Ambiente de Uruguay. 2022. *Huella ambiental de la ganadería en Uruguay.* https://www.gub.uy/ministerio-ambiente/comunicacion/noticias/huella-ambiental-ganaderia-uruguay (cited 27 May 2025).

Ministerio de Ganadería, Agricultura y Pesca. 2021. Actualización de cobertura y uso del suelo de Uruguay al año 2020/2021. Montevideo: MGAP. [Cited 10 September 2025]. Available at: https://www.mgap.gub.uy/

De Oliveira Silva, R., Barioni, L.G., Hall, J.A.J., Folegatti Matsuura, M., Zanett Albertini, T., Fernandes, F.A. & Moran, D. 2016. Increasing beef production could lower greenhouse gas emissions in Brazil if decoupled from deforestation. *Nature Climate Change,* 6(5): 493–497. https://doi.org/10.1038/nclimate2916

Paruelo, J.M., Jobbágy, E.G. & Sala, O.E. 2001. Current distribution of ecosystem functional types in temperate South America. *Ecosystems*, 4(7): 683–698. https://doi.org/10.1007/s10021-001-0037-9

Paruelo, J.M. & Sierra, M. 2023. Sustainable intensification and ecosystem services: How to connect them in agricultural systems of southern South America. *Journal of Environmental Studies and Sciences*, 13(1): 198–206. https://doi.org/10.1007/s13412-022-00791-9

Paruelo, J.M., Texeira, M., Staiano, L., Mastrángelo, M., Amdan, L. & Gallego, F. 2016. An integrative index of ecosystem services provision based on remotely sensed data. *Ecological Indicators*, 71: 145–154. https://doi.org/10.1016/j.ecolind.2016.06.054

Potschin, M.B. & Haines-Young, R.H. 2011. Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography: Earth and Environment*, 35(5): 575–594. https://doi.org/10.1177/0309133311423172

Staiano, L., Camba Sans, G.H., Baldassini, P., Gallego, F., Texeira, M.A. & Paruelo, J.M. 2021. Putting the ecosystem services idea at work: Applications on impact assessment and territorial planning. *Environmental Development*, 38: 100570. https://doi.org/10.1016/j.envdev.2020.100570

Valani, G.P., Martíni, A.F., Da Silva, L.F.S., Bovi, R.C. & Cooper, M. 2020. Soil quality assessments in integrated crop–livestock–forest systems: A review. *Soil Use and Management,* 37(1): 22–36. https://doi.org/10.1111/sum.12667

Zanasi, C., Rabboni, C., Rota, C., Bungenstab, D.J. & Laura, V.A. 2020. The Carne Carbono Neutro accordance to Brazilian consumers' attitude towards beef. *International Journal on Food System Dynamics*, 11(4): 360–376. https://doi.org/10.18461/IJFSD.V11I4.60

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