

# Agri-food systems contributing to biodiversity objectives



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# Executive Summary

Humanity faces an unprecedented sustainability challenge, with food systems playing a central role. The global food system is among the largest drivers of environmental impacts, including climate change, resource depletion, and biodiversity loss, accounting for over a third of global greenhouse gas emissions and nearly half of the planet's biocapacity. As human populations and consumption demands continue to grow, the stability of key planetary systems is increasingly threatened. This working paper synthesizes knowledge on the European agri-food system, addressing two main objectives: 1) examining its current environmental and biodiversity impacts, alongside the potential of circularity to mitigate these effects, and 2) exploring how biodiversity-friendly agricultural practices are interconnected with and can be supported by shifts in food consumption.

This working paper offers valuable insights into existing monitoring frameworks and methodologies, their ability to effectively capture environmental and biodiversity impacts, and key findings from their evaluations. The literature review and analyses presented here are not intended to be a comprehensive assessment of food systems sustainability; however, they shed light on the scale of Europe's food system, its environmental impacts, and its dependencies. Despite uncertainties in assessing direct biodiversity impacts, the overarching conclusions drawn from existing indicators emphasize both the urgency and the opportunity to transform Europe's food systems to be more environmentally and biodiversity-friendly and operate within the means of our planet.

Building on these findings, the document provides empirical evidence supporting the biodiversity benefits of specific agricultural practices, such as organic farming, agroforestry, and mixed crop-livestock systems. These practices demonstrate potential for reducing environmental harm, yet challenges remain in assessing the synergistic effects of more holistic approaches. This gap highlights the importance of looking beyond individual practices to consider broader approaches such as agroecology. Evidence from policy interventions further underscores the need for a transdisciplinary and integrated strategy that addresses not only environmental concerns but also the socioeconomic, cultural, and political dimensions of food systems.

## **Key findings on the environmental and biodiversity impacts of European food systems:**

- Food systems are a primary driver of ecological overshoot, with Europe's ecological footprint exceeding its biocapacity by 1.5 times. Food consumption alone accounts for nearly a third of Europe's total ecological footprint, highlighting the urgent need for systemic change.
- These impacts are concentrated in the agricultural production stage, with Europe outsourcing over 21% of its food-related environmental impact to other regions through international trade.
- Animal-based products, particularly meat, are the most impactful food category, driving significant land use, greenhouse gas emissions, and biodiversity loss.
- Current footprint assessment methods such as ecological footprint and carbon footprint can quantify human pressure on biodiversity across supply chains using Environmentally Extended Multi-Regional Input-Output (EE-MRIO) analysis. These approaches provide valuable insights into high-impact production and consumption sectors but do not quantify biodiversity outcomes.

- Direct measures of biodiversity outcomes can be calculated through Biodiversity footprints and Life Cycle Assessment (LCA)-based methods. However, further improvements are needed to reduce uncertainties in impact linkages and expand the scope of measurable impacts.
- Circularity principles, including nutrient cycling, waste reduction, and resource efficiency, align closely with biodiversity-friendly practices and offer significant potential to reduce environmental impacts. Optimized circular food systems, combined with dietary shifts, could reduce land use by up to 71% and greenhouse gas emissions per capita by 29% in Europe.

**Key findings on biodiversity-friendly agricultural practices and links with consumption changes:**

- Practices such as organic farming, agroforestry, extensive grazing, urban agriculture, and mixed crop-livestock systems demonstrate significant environmental benefits, including improved soil health, enhanced biodiversity, and reduced pollution and greenhouse gas emissions.
- Organic farming and agroforestry show promise in increasing species richness and ecosystem resilience, though their scalability and yield trade-offs require further investigation.
- Urban agriculture and mixed crop-livestock systems offer additional benefits, such as reduced land use, lower reliance on artificial fertilisers, and increased resource efficiency, but their economic viability and local applicability vary.
- A holistic approach to food systems transformation is essential, integrating environmental, socioeconomic, cultural, and governance dimensions. Agroecology exemplifies such an approach to support sustainable and resilient food systems.
- Effective monitoring frameworks are critical for tracking progress, requiring adaptability to different scales and contexts, and alignment with the principles of transformation.

The knowledge presented here supports transformation by providing a foundation for understanding the current state of European food systems and their environmental and biodiversity impacts. By using appropriate tools, frameworks, and indicators to monitor progress, evaluate impacts, and adapt strategies as needed, stakeholders can identify key leverage points and design targeted interventions. Adopting sustainable production practices, promoting shifts in consumption patterns, and embedding circularity principles will enable Europe to reduce its environmental impacts, enhance biodiversity, and make meaningful progress toward its sustainability goals. While we know that food consumption and especially meat consumption is a major driver of biodiversity loss, more research and data are needed to understand how different consumption changes can directly or indirectly support different biodiversity-friendly production systems. Given the complexity and multi-scale nature of food systems, evidence suggests that effective transformation cannot rely on isolated practices or interventions, and that a systematic approach is essential to ensure that the transformation of European food systems is comprehensive and aligned with long-term sustainability objectives.



# 1 Introduction

## 1.1 Background and context

Humanity is up against a multi-faceted and global sustainability challenge, represented most prominently by climate change, resource depletion, and biodiversity loss. With the continued expansion of human populations, consumption demands now exceed the capacity of the biosphere, leading to the destabilization of key planetary systems (Rockström et al., 2009; Steffen et al., 2015) and ecological overshoot (*Earth Overshoot Day*; Wackernagel et al., 2002).

In our societal transition towards a more sustainable future, food systems are gaining centrality as they are increasingly acknowledged as the single largest reason for our societal transgression of key planetary limits (Willett et al., 2019). Accordingly, food systems have emerged as key transformational component of sustainable development due to both their environmental impacts and societal role. Drawing from the EEA report “Food in a green light: A systems approach to sustainable food”, food systems are here defined as “*All the elements (environment, people, inputs, processes, infrastructures, institutions etc.) and activities that relate to the production, processing, distribution, preparation and consumption of food including waste management, and the outputs of these activities, including socio-economic and environmental outcomes.*”

Food systems represent a basic human need and are associated with multiple SDGs (Sporchia et al., 2024); furthermore, they are deeply embedded in local cultures, while also having contributing to multiple environmental impacts. Accounting for more than a third of the global anthropogenic greenhouse gas emissions (Crippa et al., 2021) and demanding nearly half of the planet’s biocapacity (i.e., the biological capacity to regenerate life supporting resources and services), food systems are a key driver of climate change and ecological overshoot, resulting in deforestation, mineral depletion, desertification, eutrophication, acidification, biodiversity loss, genetic erosion. Meanwhile, food systems (and food security) are particularly vulnerable to the adverse impacts of climate change (IPCC, 2023). Food systems are thus central to the sustainability debate, as well as to the discussions about climate (UNFCCC, 2015) and biodiversity (IPBES, 2019).

Food systems generate a wide range of potential impacts at every stage, from crop production and food processing to packaging, transportation, wholesale, and distribution, and waste disposal. Each stage involves land and resource use, carbon emissions, and waste generation, contributing to a variety of environmental impacts.

This working paper aims to 1) investigate such impacts and externalities and summarize them, with a particular focus on biodiversity impacts, and 2) identify food production and consumption alternatives and the extent to which such alternatives could contribute to a reduced impact on the planet’s biodiversity. In doing so, we also describe the complexity of monitoring food systems from the perspective of environmental impact and put special emphasis on identifying the methodologies that can best capture and monitor the biodiversity impacts deriving from different stages of the food system.

## 1.2 Goal and objectives

The goal of this working paper is to contribute to the development of knowledge of food systems, their impacts, and their contribution to EU biodiversity objectives. The overarching goal is subdivided into two objectives.

**The first objective is to synthesize knowledge on the impacts of current agri-food systems.** We approach this synthesis with a specific focus on the following sub-objectives:

- Conducting a literature review to provide an overview of methodologies used to assess the environmental impacts of food systems and identifying if and what existing methods or approaches can be applied to improve the assessment of biodiversity impacts.
- Describing the current food consumption patterns in Europe and their impacts on biodiversity, ecosystems, and carbon emissions from the perspective of the key methodologies identified in the overview.
- Improving our understanding of circular measures in the food system and how they can reduce the negative impacts of food systems on nature and biodiversity.

**The second objective is to synthesize the knowledge on the links between biodiversity-friendly farming and food consumption changes.** Within this second objective, alternative practices along all the stages of food systems are investigated, with a view to understand how their implementation would contribute lowering the impact of EU food systems on nature and biodiversity. More precisely, a focus is placed on the following sub-objectives:

- How can shifting to sustainable agricultural practices affect biodiversity, simultaneously considering economic aspects and other environmental dimensions? This responds to the supply-side opportunities and focuses on organic farming, agroecology, agroforestry, extensive grazing, mixed livestock-crop production.
- How can dietary shifts to alternative diets affect biodiversity, simultaneously considering economic aspects and other environmental dimensions? This point focuses on the demand-side opportunities and includes diversification, changes in demand for organic, agroecological, environmentally sustainable food from agriculture.

## 1.3 Limitations

This working paper aims to provide the best possible answer to the above questions within the scope of the project. Here we acknowledge some limitations.

The short timeline of this project limits the depth of literature reviews to a rapid assessment, and therefore the selected studies represent the most relevant studies available to answer the research questions within the timeframe (January-May 2024) of this project, rather than an exhaustive compilation of existing studies.

The scope of this study is limited additionally by the current state scientific research in monitoring biodiversity and biodiversity loss (Christie et al., 2021; Hochkirch et al., 2021; Oliver et al., 2021; Hortal et al., 2015), and by the current state of and rapidly evolving understanding of the impacts on biodiversity. While the definition of biodiversity is well defined, the state of biodiversity monitoring

systems has been heavily biased by current ability to make observations, leading to significant focus on specific areas and a lack of knowledge in others. These biases can exist across taxonomic groups, ecosystem types, or geographic regions. Vertebrates, for example, are generally more easily observable than invertebrates or microbes; similarly terrestrial ecosystems are generally more easily observable compared to marine, freshwater, or below ground ecosystems. These monitoring biases and gaps have the potential for the misidentification and underestimation of the relative importance of impacts (Pearman et al., 2024). Such gaps and biases did not affect our evaluation of indicators/methodologies and their respective ability to assess biodiversity impacts that are well understood within the literature. However, it must be acknowledged that to fully understand “biodiversity impact”, a more comprehensive level of biodiversity observation and monitoring is a potentially critical element to the understanding of environmental impacts related to food consumption. To state this directly, we may be able to trace the impact of consumption to a specific location and ecosystem, but if our understanding of the biodiversity that exists in the ecosystem is limited, for example, to terrestrial vertebrates, then this limitation is potentially passed on to assessment of impact on that ecosystem. Further, the information about the state of functional biodiversity, and the complex networks and interrelations between species and populations is limited. Together these limitations also limit our ability to understand the impacts of biodiversity loss on loss of ecosystem services that support the food system itself.

In the effort to drive food system change towards more resource efficient, environmentally and biodiversity friendly production practices, understanding how shifts in consumption can support these changes in production practices is a key question, however the ability of the literature review performed here to answer this question is limited by the available scientific literature. The current state, gaps, and limitations found here as well as the types of studies needed to answer this question are expanded upon in section 4.2.2 “Impact of potential consumption changes on the demand for different agricultural practices”.

Acknowledging the current gaps and challenges this working paper aims to provide a framework to understand the available scientific evidence towards the goals and objectives stated here.

## 1.4 Structure of this document

The main body of this document is structured in three sections.

The first section addresses objective 1: understanding the environmental and biodiversity impacts of current agri-food production and consumption in Europe. This section provides an overview of various indicator frameworks and methodologies and their ability to assess food system impacts on biodiversity and ecosystems based on a rapid literature review. The overview is followed by a summary of the environmental impacts due to current consumption patterns in Europe, and finally an overview of circular measures in Europe and how they can reduce the impact of food systems on biodiversity.

The second section addresses objective 2, to synthesize the knowledge on the links between biodiversity-friendly farming and food consumption changes. This section provides a review of literature to understand potential changes in consumption related to agricultural practices, the



effect of these potential changes on the environment, and how these changes can be implemented in Europe.

The conclusion section brings together key results and discussion from both sections to provide conclusions and synthesis based on the larger picture of the tools we have available to assess impacts on biodiversity, the current state and nature of such impacts, and the potential alternative consumption and production practices that can reduce the environmental impacts of food systems.

## 2 Objective 1: Impacts of current agri-food production-consumption

### 2.1 Methodology

This section describes the methodologies of the major components of objective 1:

- a literature review focused on the environmental and biodiversity impacts of food consumption,
- an Ecological Footprint assessment of EEA32 countries, and
- a literature review focused on circularity within the agri-food system of the European Union.

#### 2.1.1 Literature Review

##### *2.1.1.1 Scope and approach for the review of methodologies to measure the environmental impacts of EU food consumption*

To provide an overview of existing methodologies that can assess the environmental impacts of food consumption in the EU, with a focus on biodiversity impacts, we performed a rapid review of literature, identifying key primary literature, grey literature, secondary reports, and other reference documents with several specific selection criteria. Articles were included or excluded based on geographical region, impact type, consumption type, and publication date as relevant to focus on the research question.

The initial set of literature based on the following criteria: Europe or the EU as the target region, environmental or biodiversity related impacts, food systems as the subject, and publication date no earlier than 2013.

In coordination with the EEA project manager, we then developed an evaluation matrix to identify and classify characteristics of the methodologies, indicators, and results that are relevant to the ability of the various approaches to assess the environmental impacts associated with food systems within the EU. The characteristics were chosen to represent the coverage of the studies across stages of the supply chain (including food waste), stage of impact (e.g., pressure, state, impact), type of biodiversity loss driver, type of biodiversity impact, and result resolution (geographic, temporal, product). For the full list of characteristics and scoring criteria, see the evaluation matrix and

description table provided in the supplementary excel file. A subset of studies that described indicators or methodologies and presented relevant results were then evaluated according to the finalized criteria. Beyond the studies evaluated in the matrix, a larger body of studies were identified and reviewed for this task, including review articles, articles comparing indicators or methodologies, and grey literature summarizing the state of knowledge.

The following section provides a list of the terms and concepts used to define the evaluation matrix developed for this review, along with a description of their usage in the evaluation of articles and within this document.

### *2.1.1.2 Definitions and Conceptual Framework*

This document, and specifically the literature review presented here, covers a cross section of many different research subtopics across academia including food systems, sustainability, impact assessment, and biodiversity. As a result of the merging of various domains we aimed to use simpler and more aggregated common definitions and well-established frameworks wherever possible for the purpose of clarity. This section provides descriptions and definitions of terms and frameworks used in our review and assessment of articles.

**Food Supply chain stages** used in this document follow a simplified version of those identified by the High Level Panel of Experts on Food Security and Nutrition of the Committee on World Food Security (HLPE, 2014), with the addition of the pre-production stage. Here we identify those stages as: pre-production, production, processing, distribution, and consumption.

Pre-production is used in reference to the maintenance of genetic resources, development of agricultural and technological practices, and preparation of agricultural inputs such as machinery, seeds, and agrochemicals. Production refers to growing crops, rearing livestock and harvesting agricultural products. Processing refers to post-harvest modification and preparation of agricultural products. Distribution refers to the transportation of different forms of food and can occur before and after processing. Consumption refers to the final stage when food is obtained by the final consumer through retail purchases and may include both cooking and eating.

**Food loss and food waste** are used here to describe food system wastages. In this document, we refer to “food loss” as wastage that occurs during various pre-consumption stages and we refer to “food waste” as food that has gone through all the stages of the food supply chain up to the stage of consumption, but is not physically consumed, and therefore “wasted” at the point of consumption. From an analysis and assessment point of view, food wastages are assessed separately from the other stages of the food supply chain because wastage can occur throughout all the stages of the food supply chain from production to consumption.

**Pressure, state and impact** are used as defined in the Drivers-Pressures-State -Impact-Response (DPSIR) framework (EEA, 1999; OECD, 2003). This framework is used to define the causal linkages stemming from human activities to the resulting changes in environmental quality and associated policy responses. In this document, food consumption is the overarching activity or driver being investigated, while the review presented here examines various assessments of pressure, states and impacts adopting the terminology as generally defined by the European Environment Agency (Moll et

al., 2005) and specifically applied to biodiversity (Maxim et al., 2009). Pressure is defined as the “[...] consequences of human activities (i.e. release of chemicals, physical and biological agents, extraction and use of resources, patterns of land use, creation of invasion corridors) that have the potential to cause or contribute to adverse effects (impacts)”. Within this framework, a pressure indicator quantifies drivers of biodiversity loss, such as land use, resource use, pollution etc. State is defined as “the quantity of biological features (measured within species, between species and between ecosystems), of physical and chemical features of ecosystems, and/or of environmental functions, vulnerable to pressure(s), in a certain area.” Impact is defined as “changes in the environmental functions, affecting (negatively) the social, economic and environmental dimensions, and which are caused by changes in the state of the biodiversity. Impacts are changes in the environmental functions, affecting the social, economic and environmental dimensions, caused by changes in the state of the biodiversity”.

**Anthropogenic impact drivers of biodiversity loss** refer to the direct drivers identified by IPBES (2019): land use change, direct exploitation of biodiversity, climate change, pollution, and invasive species.

Impact and Impact stage are identified here by ecosystem type and biodiversity impact type.

**Ecosystem Types** follow a simplified six-category version of the EU typology, which include: forests, agroecosystems, urban, wetland and freshwater, marine, and other terrestrial ecosystems.

*Table 1. Aggregation of EU ecosystem typology level 1 (Eurostat, 2023a) to five categories used in this document.*

No.	EU Ecosystem Typology	Simplified Typology (this document)
1	Settlements and other artificial areas	Urban
2	Cropland	Agroecosystems
3	Grassland (pastures, semi-natural and natural grasslands)	Agroecosystems
4	Forest and woodland	Forests
5	Heathland and shrub	Other terrestrial ecosystems
6	Sparsely vegetated ecosystems	Other terrestrial ecosystems
7	Inland wetlands	Wetland and freshwater
8	Rivers and canals	Wetland and freshwater
9	Lakes and reservoirs	Wetland and freshwater
10	Marine inlets and transitional waters	Marine
11	Coastal beaches, dunes and wetlands	Marine
12	Marine ecosystems (coastal waters, shelf and open ocean)	Marine

**Biodiversity Impact** is used in this document to describe the most direct measures of biodiversity, identified as “endpoint impacts on biodiversity” (Damiani et al., 2023) and which include seven categories: genetic composition, species traits, species population, community composition, ecosystem function, ecosystem structure, and species loss.

**Methodologies** are a general term used in science which broadly refers to a system or process that is followed. This review brings together information which covers multiple defined conceptual methodological systems, which we refer to hereafter as “Indicators or Indicator Frameworks” while also identifying established technical methodological processes, which we will refer to hereafter as “methodologies”.

We use “Indicator or Indicator Frameworks” here to describe broader systems which aim to measure particular concepts for example, “Ecological Footprint” or “Carbon Footprint”. This includes defined systems or approaches which use a suite of specific measures or indicators which together are used to answer a broad question, such as in the case of Consumer Footprint (Castellani et al., 2018).

“Methodology” is used in this review to refer to defined technical processes that are often used to calculate the results within the broader “Indicator frameworks”. Examples of methodologies include life cycle assessment (LCA) or environmentally extended multi-regional input-output modelling (EE-MRIO), which are technical processes which can be applied in calculating the value of a given indicator (e.g., carbon footprint, ecological footprint) belonging to the indicator framework.

The set of indicator frameworks and methodologies reviewed for this task are described in detail in section 2.2.1 below.

**Policy links** are presented as part of the literature review framework for section 1, and these links refer only to policies or targets that are explicitly identified and linked within the set of literature reviewed for each indicator/framework. They represent the respective authors’ identification of potential policy links. Because the body of literature reviewed here is not intended to identify policy links, the results presented here should not be considered comprehensive; furthermore, the review framework highlights a potential knowledge gap. The mapping of the broader policy landscape to specific indicators or indicator frameworks would be a useful tool which could complement the results presented here, however development of such a tool is beyond the scope of this working paper.

### 2.1.2 Ecological Footprint analysis of current food consumption in EEA-32 countries

Ecological Footprint and biocapacity accounting are built upon the core concept that Earth is a closed biophysical system and its ecosystems support life through photosynthesis. As such, **biocapacity** is a property of land surfaces and water bodies that represents the finite capacity of these surfaces to generate biomass through photosynthesis, while **Ecological Footprint** represents the appropriation of biocapacity. The results presented here represent the biocapacity within the borders of the EEA32 countries, while the Ecological Footprint focuses on the biocapacity required to meet the demand of food consumed within EEA32 countries.

The Ecological Footprint and biocapacity results presented in this document are produced following the methodology that was used to assess food consumption in Portugal (Galli et al., 2020) and the EU27 (Galli et al., 2023). This method applies national results produced using a biophysical accounting approach (Lin et al., 2018) as an environmental extension to the Global Trade Analysis Project (GTAP10, 2020) multi-regional input-output model (MRIO). The results provide information about biocapacity appropriated across the full-upstream supply chain for 65 economic sectors, 3

final demand categories (household, government, and gross fixed capital formation), with global coverage across 141 regions. The resulting sectoral data is then allocated to household consumption categories following the COICOP classification system (United Nations, 2000). The allocation is based on a GTAP-COICOP concordance table and household consumption expenditures.

### 2.1.3 Methodology for the overview of circular measures in EU food systems

A rapid literature review was undertaken in March 2024 to enhance comprehension of circularity within the agri-food system of the European Union. This included the identification of barriers and levers, such as EU regulations, to facilitate circularity in this context.

To discover relevant literature on the subject, Google Scholar was utilized as a search engine, limiting findings to those published after 1992. This year was pivotal due to the adoption of "Agenda 21" during the Earth Summit in Rio de Janeiro proposing a comprehensive plan for sustainable development across global, national, and local levels, encompassing various sectors.

The following keywords were applied in combination with Boolean operators during the search process:

- “EU AND Food circularity AND Biodiversity”
- “EU AND Circular agriculture AND Biodiversity”
- “EU AND Circular food AND Biodiversity”
- “EU AND Circular economy AND Food system AND Biodiversity”.

The initial search encompassed the first 20 results from each query. After duplicates were removed, a total of 46 results were left. Subsequently, articles not adequately addressing circularity or the European Union as a cohesive entity were filtered out, resulting in the exclusion of 27 articles. Prioritization was given to 19 articles deemed most relevant, along with those referenced within.

To reach a more comprehensive overview and to ensure up-to-date references to the current status of the relevant EU regulations, additional scientific and grey literature was reviewed in agreement with the EEA project manager. Moreover, recent EU statistics concerning circular measures were used to complete the picture (e. g. regarding food waste in the EU).

## 2.2 Results

### 2.2.1 Overview of Methodologies and Indicator Frameworks

This section presents technical methodologies and indicator frameworks described and applied by studies in the literature review. For a full table describing elements by study, see the accompanying excel file.

#### 2.2.1.1 Technical Methodologies

**Life Cycle Assessment (LCA)** and **Environmentally Extended Input-Output (EE-IO)** are two categories of commonly used and recognized methodologies used for quantifying environmental and/or biodiversity impact. Almost all quantitative results within the reviewed literature adopted one



or both approaches. Marques et al. (2017) provide an in-depth review of **Multi-Regional Input Output (MRIO)** analysis and LCA approaches, including available databases, models, and the current state of their ability to assess biodiversity impacts. Here we list and summarize key characteristics of these two methodologies and related technical methodologies.

**LCA** is a bottom-up approach commonly used to quantify the environmental impacts along a defined boundary of the life cycle of a product (ISO, 2006). It is a broad and widely recognized approach with several international standards, databases, and modelling procedures and is the primary approach among multiple studies examined here. LCA can and has been applied to quantify a broad range of environmental impacts including GHG emissions, pollution, species loss, and other footprint-based metrics. LCA allows the detailed calculation of products and processes and thus can achieve high product-level resolution, providing greater relevance to businesses and other stakeholders. LCA-based studies often refer to the specific terms Life Cycle Inventory (LCI) and Life Cycle Impact Assessment (LCIA) to refer to the specific sub-components or phases of the assessment where LCI or the “inventory” phase refers to the collection of input/output data associated with the life cycle of the target, and LCIA refers to the “impact assessment” or the significance of the result on the environment (ISO, 2006).

Many LCIA methods exist, used to convert either resource use or emissions into impact. Impacts are generally categorized into either mid-point or end-point categories, where mid-point refers to earlier stage impacts that could be considered “pressure” in the DPSIR framework, and end-points refer to more concrete or direct impact associated with from the mid-point impact. Examples of specific LCIA models/methods include ReCiPe and LC-IMPACT.

Life Cycle Sustainability Assessment (LCSA) is an expansion upon the environmental impact focus of LCA to include broader sustainability aspects, examples include Life Cycle Costing (LCC), and Social LCA (S-LCA).

The Basket of Products (BoP; EC-JRC, 2012) approach is a specific LCA-based approach developed to assess the environmental impacts of consumption by representing the per-capita average consumption in the EU. The BoP approach identifies a specific set of products, or the “basket of products”, by their relative importance, either economic or mass, and follows the International Reference Life Cycle Data System (ILCD) methodology. This approach is referenced in a number of studies either in the development of methodology (Notarnicola et al., 2017), the evolution and incorporation as a key element of the consumer footprint (Castellani et al., 2018) or the application of the methodology toward specific impacts (Crenna et al., 2019; García-Herrero et al., 2023). While a standardized basket of goods benefits in providing a reference point for understanding and comparing differences across multiple dimensions of interest such as time, geography, solutions, etc., it has been noted that there may be gaps where products of low importance (by economic value or mass) have relatively high biodiversity impact (Crenna et al., 2019).

**EEIO** analysis or modelling is a widely used top-down methodology in the assessment of environmental impacts based on underlying economic input-output models. While single-region models exist, all studies referenced here use multi-regional models and are therefore referred to as Environmentally Extended Multi-Region Input-Output (EE-MRIO) analyses. Built upon existing economic IO tables, EEIO operates on the general concept that environmental impacts can be traced across economic flows by understanding the impact intensity per unit of economic value. By tracing

economic flows to and from various economic sectors and across multiple regions (here regions refer to countries or country groupings), EE-MRIO allows the tracking of impacts from the country and sector of production to the final consumer. Resulting values thus are globally consistent, in other words, the sum of all impacts at the production stage is fully traced and allocated to final consumers, and thus impact across the full supply chain is included within results. EEIO-based methodologies provide the unique ability to trace impacts through international trade flows, and therefore provide greater relevance for supply chain/sectoral actors, governments, and at national/regional/global scales (Marques et al., 2017). Nonetheless, EE-MRIO does not easily allow the disaggregation of impacts related to certain components such as food waste, which occur across multiple stages and sectors of the supply chain.

**Other technical methodologies/approaches:** Most studies in this review adopt LCA and EEIO-based approaches, however, other approaches exist that apply similar principles to track embodied impacts. Vanham et al. (2013), in an approach more similar to LCA, apply values from existing data/databases to calculate the water footprint of specific products, food consumption statistics, and food intake by source to calculate the footprint of different diets. Similarly, Antonelli et al. (2017) use published data on the virtual water content of commodities, summing the total traded commodities to calculate the total water footprint trade flows between countries. No specific technical terminology is described in these studies, however, for the purpose of this review, we will refer to this as physical accounting of embodied impacts, as it tracks impacts embedded in physical goods, either traded or consumed, as opposed to the tracking of impacts on the basis of financial flows used in EE-IO.

### 2.2.1.2 Indicators and Indicator Frameworks

Table 2. Overview of Indicator Frameworks (see annex A for a detailed evaluation by publication)

Indicator Framework	What does it aim to measure	Unit and/or description of measure(s)	Studies including this indicator
Biodiversity Footprint	Biodiversity pressure and biodiversity impact	Potentially Disappeared Fraction (PDF) of species, species loss per year, others.*	(4) Marques et al. 2017, Crenna et al. 2019, Vanham et al. 2019, Koslowski et al. 2020, Sanyé-Mengual et al. 2023
Carbon footprint	CO <sub>2</sub> emissions	Mass (e.g. Kg CO <sub>2</sub> )	included in multiple studies as a sub-indicator
Consumer Footprint	Average environmental impact of EU citizens	Suite of indicators - see environmental footprint**	(2) Castellani et al. 2018, Notarnicola et al. 2017,
Consumption Footprint	Environmental Impact of EU economies	Suite of indicators - see environmental footprint**	(1) Sala et al. 2023

Indicator / Framework	What does it aim to measure	Unit and/or description of measure(s)	Studies including this indicator
Ecological Footprint	Bioproductive capacity	Bioproductivity-weighted surface area (terrestrial, aquatic); global hectares (gha)	(3) Vanham et al. 2019, Galli et al. 2022, Galli et al. 2023
Environmental Footprint	Environmental Impact (Product or Organization)	Climate Change(kgCO <sub>2</sub> eq), Ozone Depletion(kgCFC-11eq), Human Toxicity, non-cancer(CTUh), Human Toxicity, cancer(CTUh), Particulate Matter(kg PM <sub>2.5</sub> ), Ionising radiation(kBqU235eq), Photochemical ozone formation(kg NMVOCwq), Acidification(mol H <sup>+</sup> eq), Terrestrial eutrophication(mol Neq), Marine eutrophication(kg N eq), Freshwater ecotoxicity(CTUe), Land use(kg C deficit), Water resource depletion(m <sup>3</sup> water eq), Resource use, fossils(MJ), Resource use, minerals and metals (kg Sb eq)	(3) Castellani et al. 2018, Notarnicola et al. 2017, Sala et al. 2023
Multi-Indicator Sustainability Assessment	Ecosystem Stability	Ecosystem Status (composite score) Per capita GHG emissions (composite score), Per capita blue water consumption (composite score), Per capita land use (composite score), Per capita non-renewable energy use (composite score), Per capita biodiversity footprint (composite score)	(1) Chaudhary et al. 2018

Indicator / Framework	What does it aim to measure	Unit and/or description of measure(s)	Studies including this indicator
Environmental Performance (indicator suite)	Ecosystem Status	GHG emissions, consumptive freshwater use (water use), terrestrial land occupation (land use), and nitrogen (N) and phosphorus (P) emissions	(1) Gephart et al. 2021
GHG Footprint	GHG Emissions (CO <sub>2</sub> , CH <sub>4</sub> , N <sub>2</sub> O, HFCs)	Mass or mass equivalent; KgGHG or kgCO <sub>2</sub> eq	(2) Sandstrom et al. 2018, Bajan et al. 2022
Nitrogen Footprint	Reactive Nitrogen loss to the environment	Mass (Kg N)	(4) Leip et al. 2014, Vanham et al. 2019, Galli et al. 2022, Leip et al. 2022
Seafood consumption Footprint	Seafood consumption	Mass (kg of seafood)	(1) Guillen et al. 2019
Social Footprint	Social Impact of products and services	Medium risk hours per worker hour	(1) Mancini et al. 2023
Water Footprint	Water consumption	Volume; cubic meters of water (Blue, Green, Grey)	(4) Vanham et al. 2013, Antonelli et al. 2017, Vanham et al. 2019, Galli et al. 2022, García-Herrero et al. 2023

\* The biodiversity footprint is not a single standardized framework, thus all studies looking at measures of biodiversity loss are included here.

\*\* Consumer and Consumption Footprint aim to adopt the approach and methodology of Environmental Footprint approach and set of impact categories; however, they are not fully compliant with the standards.

### Biodiversity Footprints

Biodiversity Footprint is a term currently used in the literature which describes various metrics for measuring the impact of human activities on biodiversity; however, the term “biodiversity footprint” does not refer to a specific methodology or framework. Multiple methodologies and frameworks exist that calculate different types of “biodiversity footprints”. Common to these methodologies is that they typically rely on determining the relationship between upstream pressures (also referred to as midpoint measures; IPBES classification of drivers also falls into this category) such as land use, emissions, resource consumption with more direct measures of either ecosystem quality or biodiversity/biodiversity loss (also referred to as endpoint measures). Examples of direct measures include Mean Species Abundance (MSA), species richness, and Potentially Disappeared Fraction of

species (PDF). For simplicity, studies that aim to measure biodiversity impact through endpoint measures such as these are included in this section.

In addition to the multiple metrics used in quantifying biodiversity footprints, Sanyé-Mengual et al. (2023) identified and compared multiple LCA models, identifying differences between models by their respective coverage of impact categories, representative products, process, and elementary flow covered by each model.

This review looks at biodiversity footprints as described by (Crenna et al., 2019; Koslowski et al., 2020; Marques et al., 2017; Sanyé-Mengual et al., 2023)

**Impact stage:** Overall, biodiversity footprints are developed as an attempt to achieve more granular and direct measures of biodiversity, and therefore components such as PDF fall firmly in the stage of “impact” as they capture aspects of both ecosystem quality and biodiversity/biodiversity loss measures. A key limitation is that biodiversity impacts are considered from the point of production processes and do not include direct urban expansion and its potential for habitat fragmentation. Additionally, biodiversity of untracked activities such as illegal logging or poaching are not included due to the absence of data (Koslowski et al., 2020).

**Supply Chain Coverage:** Crenna et al. (2019) and Marques et al. (2023) apply an LCA-based approach, which covers the full life cycle of food products from production to consumption including distinct waste flows at each stage and at the end of the life cycle. LCA does not include pre-production activities such as maintenance of genetic material, or development of agricultural or technological practices. Koslowski (2020) uses an EE-MRIO-based approach, providing full supply chain coverage including pre-production impacts and waste.

**Anthropogenic Impact Drivers:** Biodiversity footprints linking consumption to biodiversity impact cover the drivers of land use change, climate change, and pollution.

**Impacted Ecosystems:** Crenna et al. (2019) directly address wetland, freshwater, and marine ecosystems, however the remaining biodiversity footprint studies reviewed here do not explicitly address the type of ecosystems impacted while the methodologies and types of impacts would imply that all ecosystem types are affected.

**Impacts on biodiversity:** All studies here measure potential species loss either directly as species loss per year or PDF, while additionally Sanyé-Mengual et al. (2023) also include approaches assessing MSA, which cover species population and community composition aspects of biodiversity impact.

**Coverage and Resolution:** As various studies and types of biodiversity footprints exist, the spatial coverage, temporal coverage, and target resolution vary greatly among studies. See the accompanying table for additional details for each study.

**Key findings:** Examining differences between urban and rural areas and income levels, Koslowski et al. (2020) found that urban residents had slightly higher footprints while higher income was related to higher biodiversity impacts. Further, they found that the total European biodiversity footprint decreased by 4% from 2005-2010. Unsurprisingly, the economies with the largest GDP in Europe also had the greatest total biodiversity impact. On a per capita basis, differences in results between



models were primarily due to differences in impact categories used, with LC-IMPACT showing the greatest biodiversity impacts associated with mediterranean countries and high per-capita GDP countries, while no clear pattern emerged using ReCiPe (Koslowski et al., 2020). The top contributors of biodiversity impact from EU consumption were food (animal-based products and chocolate), mobility, and household goods. From the perspective of impact types, land use and climate change impacts were responsible for 75% of the overall impact, while ecotoxicity was identified as another major contributor by the LC-IMPACT model.

Similarly, Crenna et al (2019) looked at the impact of European food consumption, finding that the greatest impact “..in the majority of [midpoint] impact categories- are due to the consumption of meat and dairy products.” This result was found to be true both from a per-kg and from a total consumption perspective. In terms of endpoint impacts, they found that “..ecosystem quality and biodiversity are mainly affected by the consumption of pork and beef meat,” and this is mainly due to land transformation and occupation for agricultural production of feed.

From a methodological perspective, they point out that endpoint modelling is highly uncertain, suggesting that the latest, most up-to-date models should be used in order to minimize uncertainty. Still, while the improvement of LCA-based endpoint modelling is needed in terms of impact category coverage (ecotoxicity, resource overexploitation, invasive species), characterization factors refinement, taxonomic coverage, and land use intensity, they emphasize the importance of current modelling in identifying impact hotspots as all methodologies strongly converge on animal-based products as the greatest drivers of biodiversity impact.

Sanyé-Mengual (2023) looked at the biodiversity footprints of EU consumption and compared the results of eight widely used LCIA models/methods finding that general convergence of models identifying land use and climate change as the most impactful driver of biodiversity footprints. Ecotoxicity was also identified as the third largest contributor. Food was found to be the largest consumption category when compared to appliances, household goods, housing and mobility. Within food, meat, specifically beef and pork was identified as having the largest biodiversity footprint, with chocolate also identified as a top contributor in some models.

From a methodological perspective they highlighted the fact that impact category coverage was an important consideration due to the potential underestimation of biodiversity loss drivers such as climate change and ecotoxicity (Sanyé-Mengual et al., 2023). Across the set of LCIA models and methods, endpoint impact categories included land use (occupation, transformation), climate change, ecotoxicity (terrestrial, freshwater, marine), acidification (terrestrial, freshwater), photochemical ozone formation, Eutrophication (terrestrial, freshwater, marine), thermal pollution, ionizing radiation, and water use. The LC-Impact model was one of 2 method/models that covered all 3 types of ecotoxicity and it was noted that “disagreements among methods and models were based on the inclusion of impact categories and the characteristics of the underlying impact assessment model (e.g., consideration of species vulnerability and scope in terms of environmental compartments)” Overall, biodiversity endpoint measures remain a major challenge for LCA (Sanyé-Mengual et al., 2023).

**Policy links identified by the authors:** The studies here present linkages to the CBD Strategic Plan for Biodiversity 2011-2020, Aichi targets, SDG14 (Life below water) and SDG15 (Life on land).

## Consumer Footprint

The consumer footprint is one of two footprint indicators developed by the European Commission's Joint Research Centre (JRC) to understand the environmental impacts of consumption in the EU with a focus on the per-capita consumption associated with the lifestyle of an average EU citizen. It is a bottom-up LCA-based approach which disaggregates consumption into five components of consumption (food, housing, mobility, household goods, and appliances), each of which has an associated Basket of Products (BoP) representative of average per-capita consumption. This review focuses on the food component of the Consumer Footprint, described by Castellani et al. (2018). For a detailed comparison between consumer and consumption footprints, see Sala et al. (2019).

**Indicator components:** The consumer footprint refers to a suite of 15 pressure indicators or “impact categories” including climate change, ozone depletion, human toxicity (non-cancer), human toxicity (cancer), particulate matter, ionizing radiation, photochemical ozone formation, acidification, terrestrial eutrophication, freshwater eutrophication, marine eutrophication, freshwater ecotoxicity, land use, water resource depletion, resource use (fossils), resource use (minerals and metals)

**Supply Chain Coverage:** The LCA-based BoP approach covers the full life cycle of food products from production to consumption including distinct waste flows at each stage and at the end of the life cycle. LCA does not include pre-production activities such as the maintenance of genetic material or the development of agricultural or technological practices.

**Anthropogenic Impact Drivers:** The above impact categories cover land use change, climate change, and pollution.

**Impacted Ecosystems:** The impact categories and the tracking of more detailed elementary flows explicitly address forests, agroecosystems, wetlands, freshwater and marine ecosystems, as well as other terrestrial ecosystems. Urban areas are not directly assessed, however would be implicit to some impact categories that are land-use agnostic.

**Impacts on biodiversity:** The consumption footprint of food does not explicitly cover direct biodiversity impacts.

**Coverage and Resolution:** The BoP indicator approach aims to represent the annual average consumption of an EU citizen, and therefore provides a regional average value for the EU for the reference year 2010.

**Key findings:** Meat products, dairy products and beverages appeared as hotspots in most impact categories. For meat products, ecotoxicity, human toxicity, eutrophication, acidification, water depletion and climate change are among the leading impacts.

**Policy links identified by the authors:** The consumer footprint is linked directly to a number of Sustainable Development Goals (SDGs), including SDG12 Responsible consumption and production, while impact categories directly touch on SDG3 Good health and wellbeing, SDG6 Clean water and sanitation, SDG13 Climate action, SDG14 Life below water, and SDG15 Life on land. Castellani et al. (2018) provide five illustrative usage scenarios that have policy relevance to the Circular Economy Package, Urban Waste Water Directive, Bioeconomy Strategy, Roadmap to a resource efficient Europe, and SDG12.3 Food Waste.

## Consumption Footprint

The consumption footprint (EC-JRC, 2023) is the complementary approach to the consumer footprint. The consumption footprint consists of a set of 16 LCA-based component indicators that can be calculated using either a bottom-up approach such as process-based LCA, or a top-down approach such as EE-MRIO. The consumption footprint aims to quantify the consumption of the entire economy and thus uses trade statistics to capture the full set of import and export trade flows from the target region. In doing so, the consumption footprint has a broader scope than the consumer footprint. The consumer footprint applies LCA based environmental profiles to a basket of products in combination with statistics on individual consumption. The consumption footprint framework includes five components of consumption (food, housing, mobility, household goods, and appliances)<sup>1</sup>, here we look at results from the top-down approach (“Consumption Footprint Europe: top-down approach,” 2024), and more specifically on the food component as described by Sala et al. (2023). Sala et al. (2023) evaluate biodiversity loss using the term “biodiversity footprints” and food waste associated with food using the consumption footprint framework. For a detailed comparison between consumer and consumption footprints, see Sala et al. (2019).

**Indicator Components:** The consumption footprint for food follows the Environmental Footprint (“Environmental Footprint,” 2021) methodology which includes 16 midpoint impact categories: acidification, climate change, ecotoxicity-freshwater, eutrophication-freshwater, eutrophication-marine, eutrophication-terrestrial, human toxicity-cancer, human toxicity-non-cancer, ionising radiation, land use, ozone depletion, particulate matter, photochemical ozone formation, resource use-fossil, resource use-mineral and metals, and water use. Biodiversity impacts can also be calculated using the consumption footprint method as applied by Sala et al. (2023) using the ReCiPe LCA model.

**Supply Chain Coverage:** Derived from an LCA-based approach, the results encompass production, processing, distribution, consumption, and waste treatment.

**Anthropogenic Impact Drivers:** The midpoint impact categories defined in the Environmental Footprint framework cover land use change, climate change, and pollution.

**Impacted Ecosystems:** The impact categories explicitly address freshwater and marine ecosystems, and while land use and terrestrial eutrophication are specified impact categories, specific terrestrial ecosystems are not identified. Nor are urban areas explicitly identified.

**Impacts on biodiversity:** The 16 specific midpoint impact categories are not direct measures of biodiversity impact. However, biodiversity impact (footprints) can be linked to these midpoint categories through ReCiPe LCA model. Sala et al. (2023) present high level results identifying the impact categories and products with the largest biodiversity footprint.

**Coverage and Resolution:** Biodiversity footprint and food waste results are provided for 45 products at the regional (EU27) level for the year 2020.

**Key findings:** The product groups responsible for the majority of environmental impact are animal-based, with meat and dairy representing 65% of the environmental impact of food consumption in

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<sup>1</sup> The (“Consumption Footprint Europe: top-down approach,” 2024) website presents results in 7 domains: food, housing, mobility, household goods, services, clothing and footwear, and changes in stock.

the EU. In 2020, land use, climate change, and terrestrial acidification are the primary drivers of the biodiversity footprint in the EU-27, and roughly 10% of all food supplied to retail, food services and households was wasted. Household and food services are a key focus area because 62% of food is wasted at the consumption stage, representing 70% of the environmental impact from food waste. In terms of biodiversity loss, meat and cheese contributed to over 60% of the biodiversity footprint of EU food consumption while land use, global warming, and terrestrial acidification were the drivers of 95% of the biodiversity footprint (Sala et al., 2023).

**Policy links identified by the authors:** The consumption footprint framework aims at supporting EU policy ambitions, such as by monitoring progress towards the achievement of the European Green Deal (EC, 2021a) ambitions, including those in the New Circular Economy Action Plan (EC, 2020a), the Farm to Fork Strategy (EC, 2020b), the Biodiversity Strategy (EC, 2020c), and the Zero Pollution Action Plan (EC, 2021b).

### Ecological Footprint

See also section 3.1.2 for EEA-32 specific analysis. The ecological footprint framework (Wackernagel and Rees, 1996) was developed to measure the biological regeneration of the biosphere (“biocapacity” or “BC”) and the amount that is appropriated by human activity (“ecological footprint” or “EF”). The national accounting methodology (Lin et al., 2018) is maintained by the Footprint Data Foundation (FODAFO, 2024) and produces results for 190 countries and the world from 1961-2022 following a physical accounting approach. Here we focus on an EE-MRIO based methodology for calculating the ecological footprint of consumption by households; this methodology tracks multiple consumption types (i.e., food, transportation, housing, goods, and services), although we here focus on the sole ecological footprint of food (Galli et al., 2023).

**Indicator Components:** The ecological footprint framework identifies six ecological footprint categories (cropland, grazing land, forest products, fishing grounds, built-up land, and carbon footprint) which represent types of human demand on bioproduction. It also classifies bioproduction into five types of bioproduktive areas (cropland, grazing land, forest, fishing grounds, and built-up land) each of which corresponds to a matching footprint category, with the exception of the forest biocapacity which meets the demand for both forest products such as wood or fiber and carbon sequestration. The various footprint and biocapacity types are quantified in global hectares (gha), which are bioproduktivity weighted hectares.

**Supply Chain Coverage:** The EE-MRIO based approach provides full supply chain coverage, including all inputs related to pre-production and fixed capital investments as well as waste products at all stages. The nature of EE-MRIO approach means that all footprint associated with the production of food is allocated to final consumers and thus all food that is lost or wasted is also included, but the waste component cannot be disaggregated from the approach without additional data and analysis. Results also provide the ability to trace final consumption or demand through the supply chain demand to the country and sector of production.

**Anthropogenic Impact Drivers:** Ecological footprint indicators directly encompass aspects of land use change as both ecological footprint and biocapacity are accounted in global hectares (gha), which represent bioproduktivity-weighted land (or aquatic) area. A global hectare represents a

hectare of bioproductive surface area, with world average bioproductive capacity. For example, a hectare of highly productive forest area would have a biocapacity of 2 gha if that hectare was twice as bioproductive as an average hectare in the world; conversely, a hectare with lower than world average productivity, such as shrub or grassland would have a biocapacity of less than 1 gha. In this way, global hectares capture the primary functional aspect of land (and aquatic) area.

Ecological Footprint represents the total size of the human economy's metabolism, measured against the planet's biocapacity, and it includes humanity's production of waste CO<sub>2</sub> emissions. Thus, the accounting represents also represents a minimum threshold for sustainability through two underlying principles, that 1) resources are not consumed faster than they can be regenerated and 2) waste cannot be produced faster than it is assimilated. While the consumption of a resource, for example, 5 tonnes of wheat, is represented as the global hectares necessary to produce that resource, 5 tonnes of waste CO<sub>2</sub> emissions is also represented as the global hectares necessary to assimilate that waste after the deduction of the percentage sequestered by oceans. See Mancini et al (2016) for a detailed description of the calculation.

Anthropogenic CO<sub>2</sub> emissions can be expressed directly as tonnes or further translated into global hectares, however, neither is a direct representation of the magnitude of radiative forcing nor can they be directly expressed as climate change/climate change effects. Nevertheless, both are informative about the potential contribution to climate change. By deducting the percentage of CO<sub>2</sub> sequestered by oceans, the carbon subcomponent of the Ecological Footprint is directly proportional to the anthropogenic contribution to atmospheric CO<sub>2</sub> accumulation.

**Impacted Ecosystems:** All ecosystem types (forests, agroecosystems, urban, wetland and freshwater, marine, and other terrestrial) are affected and identified in the ecological footprint framework.

**Impacts on biodiversity:** Ecological footprint subcomponents do not include any direct measures of biodiversity impact.

**Coverage and Resolution:** Results are provided for 12 food categories by country for the EU27 region in the years 2004, 2007, 2011, and 2014. Latest year data are based both on base year MRIO model and input production results from the National Footprint and Biocapacity accounts. The next update would bring data to the latest base MRIO year of 2017 with national production data up to 2020. Results are also indicated by origin country. A subset analysis provides the footprint intensity of various food products.

**Key findings:** Per capita ecological footprint decreased by 20% from 2004 to 2014 while per capita biocapacity decreased by 4% during the same period. On a total basis, results over this time period were slightly lower due to the increasing population in the EU with an 18% reduction in ecological footprint and a 2% decrease in biocapacity. The ecological footprint of food represented 28-31% of the total ecological footprint of the EU27. Animal-based food was found to have a much higher EF intensity than plant-based foods. Cropland footprints were the largest component of the EU27's food ecological footprint, making up 57% in 2004 and 2014. A quarter of biocapacity consumed in food originates from non-EU countries.



**Policy links identified by the authors:** This study identifies links to the EU’s Farm to Fork Strategy, Sustainable Development Goals, and the Paris Agreement.

### Environmental Footprint

See also Consumer and Consumption Footprints. The term “Environmental Footprint” has more than one usage in scientific literature. It is a general term used to describe the “Footprint Family” of indicators (Galli et al., 2022; Vanham et al., 2019; Čuček et al., 2015; Hoekstra and Wiedmann, 2014; Hammond, 2006), most prominently known by but not limited to carbon footprint, ecological footprint, water footprint and nitrogen footprint. The Environmental Footprint also refers to a specific methodology (EC, 2013; Manfredi et al., 2012) that was developed for the assessment of products, Product Environmental Footprint (PEF) and organisations, Organisation Environmental Footprint (OEF). While products and organisations are not within the scope of this review, it is relevant to mention the Environmental Footprint because this methodology and its defined impact assessment categories are adopted by the Consumer and Consumption Footprint Indicator frameworks.

### Nitrogen Footprint

While reactive nitrogen ( $N_r$ ) is a crucial input to global food production through its usage in fertilizers,  $N_r$  is also considered a pollutant with wide-ranging environmental impacts including climate change, smog, acid rain, eutrophication, and biodiversity loss (Galloway et al., 2003). The nitrogen footprint (Leach et al., 2012) aims to quantify the amount of  $N_r$  that is emitted into the environment as the result of an activity such as production or consumption of food.

**Indicator Components:** The concept does not define any specific sub-components, however, it can be calculated as part of a broader nitrogen budget to represent nitrogen flows between agricultural sub-pools such as livestock production, manure management, and soil cultivation, (Leip et al., 2014).

**Supply Chain Coverage:** Leip et al. (2014) calculated the nitrogen footprint based on a farm-gate LCA, which did not include NO<sub>x</sub> emissions from energy use, land use change, or that which occurred from food waste after the production stage.

**Anthropogenic Impact Drivers:** Nitrogen footprint is associated with climate change and pollution drivers of biodiversity loss.

**Impacted Ecosystems:** The nitrogen footprint has the potential to impact all ecosystem types, however, as a pressure indicator, it is not explicitly linked to the ecosystems where impact occurs.

**Impacts on biodiversity:** The nitrogen footprint has the potential to result in multiple biodiversity impacts, however, because results are not explicitly linked to the ecosystems where impact occurs there is no explicit measure or quantification of biodiversity impact.

**Coverage and Resolution:** Leip et al. (2022) provide results for the EU27 region for the base year 2015. Leip et al. (2014) provides results for the EU27 (excluding Malta and Cyprus) region with country level results for the reference year of 2004.

**Key findings:** Leip et al. (2022) looked at reducing nitrogen waste from an intervention perspective, finding that the largest improvements in nitrogen use efficiency could be made in the livestock sector and that the majority of potential interventions that could effectively reduce the nitrogen footprint require dietary shifts, while Leip et al. (2014) found that the nitrogen footprint was significantly higher for ruminant meat products (500 gN/kg) compared to pork or poultry meat products (100gN/kg). Vegetable products had lower nitrogen footprints than any animal-based category, with the largest nitrogen footprint associated with oilseeds (20gN/kg).

**Policy links identified by the authors:** A number of linkages to policy were identified, including the National Emission Ceilings Directive, Habitats Directive, Nitrates Directive, Water Framework Directive, Paris Agreement, European Green Deal: Farm to Fork Strategy, Biodiversity Strategy.

### Seafood Consumption Footprint

The seafood consumption footprint (Guillen et al., 2019) is a consumption-based approach that accounts for the total biomass of seafood consumed by a population. The description below refers to the study by Guillen et al. (2019), who apply an EE-MRIO-based approach to track the flows of seafood biomass through the global supply chain.

**Indicator Components:** No subcomponents are described for the seafood consumption footprint

**Supply Chain Coverage:** The authors developed an EE-MRIO model to track the flow of seafood biomass thus covering the entire global supply chain from production to consumption. Products can be disaggregated into four categories (aquaculture species, fisheries species, products for human consumption, and fishmeal and fish oil) across four sectors/industries (aquaculture, capture, fish meal, and fish processing), including intermediate consumption and final use.

**Anthropogenic Impact Drivers:** The seafood consumption footprint tracks seafood biomass and therefore is a measure of direct exploitation of species.

**Impacted Ecosystems:** The seafood consumption footprint covers direct impacts on marine and freshwater ecosystems.

**Impacts on biodiversity:** The seafood consumption footprint does not provide species-level information related to biodiversity, however, the extraction of seafood biomass from aquatic ecosystems can result in impacts on species populations, community composition, ecosystem function, ecosystem structure and species loss.

**Coverage and Resolution:** The model developed by Guillen et al. (2019) provides global results by country for the base year of 2011, with resolution to 4 sectors: aquaculture, capture, fish meal, and fish processing. The base input data for the EE-MRIO model are country level data from FAOstat and COMTRADE which may suggest full resolution for EU countries is available, however the EU is presented as an aggregate in the results.

**Key findings:** Globally, seafood consumption is highly dependent on imports with capture fisheries providing significantly more supply to international consumption than aquaculture products. Seafood consumption in the EU depends significantly on imports.

**Policy links identified by the authors:** No specific policy links are identified by the authors.

### Water Footprint

The water footprint (Hoekstra and Mekonnen, 2012) is a consumption-based approach that accounts for the total consumption of water by an activity or economy. The description below refers to usage results of the water footprint approach by (García-Herrero et al., 2023; Antonelli et al., 2017; Vanham et al., 2013).

**Indicator Components:** The water footprint tracks rainwater (green water footprint), surface water (blue water footprint) and water polluted (grey water footprint).

**Supply Chain Coverage:** Vanham et al. (2013) looked at diets from consumption perspective, inclusive of all stages with the exception of pre-production; Antonelli et al. (2017) looked only at the distribution stage of the supply chain by tracking trade flows; Garcia-Herrero et al. (2023) used LCA based methodologies, tracking the full supply chain, with the exception of pre-production, to look at the water footprint from a production and consumption perspective.

**Anthropogenic Impact Drivers:** Water Footprint assessment does not directly address anthropogenic impact drivers; however, the usage of water would implicitly result in land use change.

**Impacted Ecosystems:** While the methodology does not specify types of ecosystem impacts, Water consumption impacts all ecosystems implicitly.

**Impacts on biodiversity:** In affected ecosystems, water consumption would directly affect water availability, and thus all aspects of biodiversity.

**Coverage and Resolution:** All studies provide coverage of the EU, with category resolution coverage for 13 agricultural products and 4 diet scenarios from 1996-2005 (Vanham et al., 2013), 9 agricultural product categories from 1993-2011 (Antonelli et al., 2017), and 45 BoP Food products in 2014 (García-Herrero et al., 2023).

**Key findings:** Vanham et al. (2013) found that the largest fraction of water footprint was associated with edible agricultural foods, and that reduction of meat intake would be the most effective at reducing WF. Antonelli et al. (2017) found that intra-regional EU trade represented the majority of virtual water trade in the EU, while water-scarce EU countries have become increasingly large exporters of blue water in the region over the past 33 years. Garcia-Herrero et al. (2023) applied two methodologies, water footprint and AWARE, and found that cashew and almonds were among the highest impact foods for EU water consumption when considering aligned scopes of both methodologies, while wine and chocolate were also found to have the highest water footprint.

**Policy links identified by the authors:** Water Footprint Assessment was linked to the Water Framework Directive 2000, Common Agricultural Policy (CAP) (Antonelli et al., 2017); and more recently to SDGs, European Green Deal, and Farm to Fork Strategy (García-Herrero et al., 2023).

## 2.2.2 Environmental Impact of Food Consumption in Europe

### *Environmental Impact of European Countries*

From 2004 to 2014, the average per-person ecological footprint of the EEA-32 region decreased by 18 percent from 4.13 to 3.38 gha per capita (Figure 1). The ecological footprint of the EEA-32 was 1.8 times larger than the region's biocapacity in 2004, decreasing over time to 1.5 times the size in 2014.

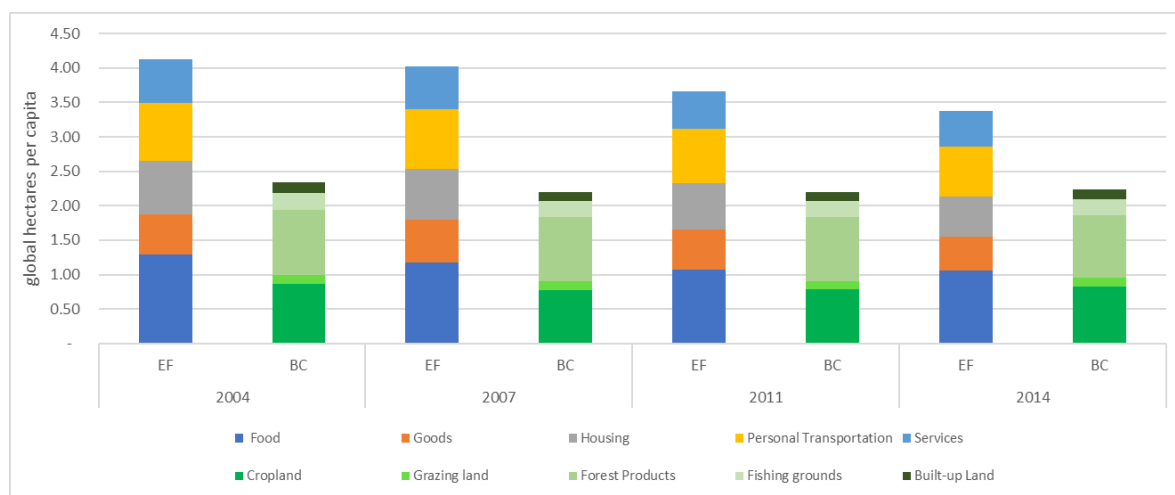


Figure 1. EEA-32 Ecological Footprint (EF) by household consumption category and biocapacity (BC) by land type (2004-2014)

Food was the largest household ecological footprint category, on average, making up between 29-31% of the total ecological footprint across all years (Fig 1).

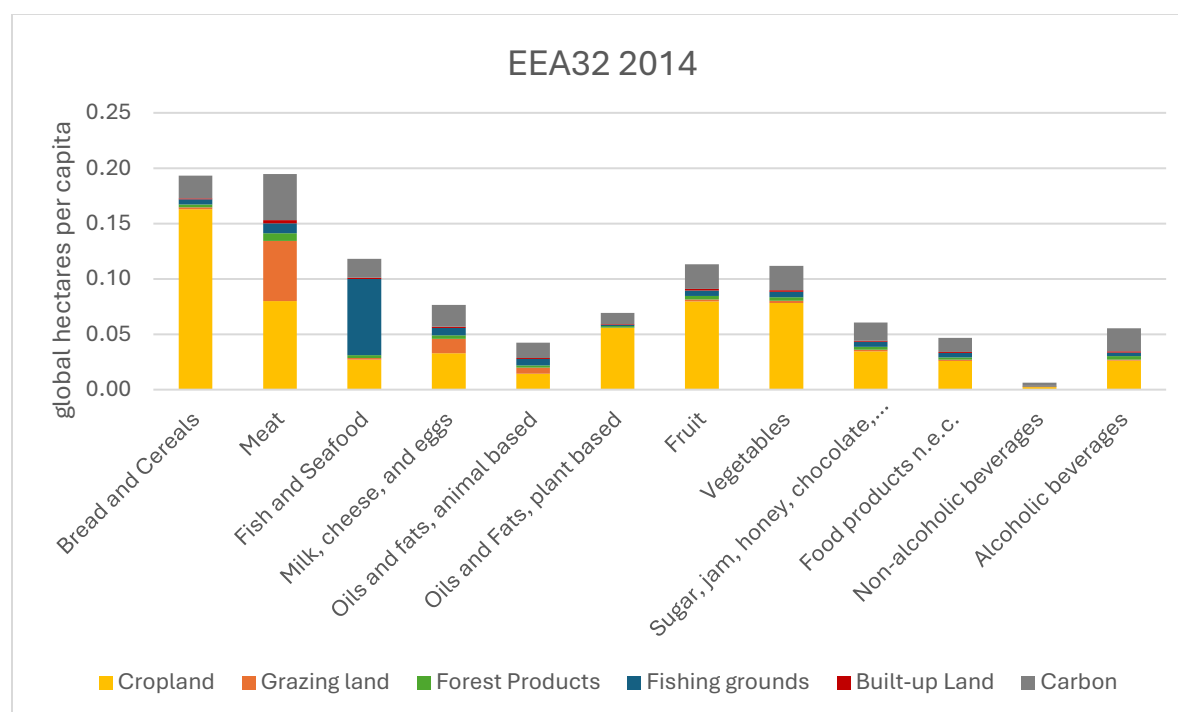


Figure 2. EEA32 Ecological Footprint of Food by COICOP food category, year 2014

***Animal products were consistently found to have high environmental and biodiversity impacts.***

Meat consumption in the EEA32 had the largest ecological footprint among food categories (Figure 2). The bread and cereals category follows closely behind in ecological footprint per capita, however, a supplemental analysis by Galli et al. (2023) found that the footprint intensity (gha/1000kcal) was up to 3 orders of magnitude higher for meat as well as fish and seafood products. This is consistent with the above results drawn from the literature reviewed, as meat and/or animal-based products, including dairy, represented the most consistent product categories identified among multiple studies to have the highest impacts and/or the food category of which reduction could have the greatest positive impact (Castellani et al., 2018; Vanham et al., 2013; Crenna et al., 2019; Galli et al., 2023; Giosuè et al., 2022; Leip et al., 2014, 2022; Notarnicola et al., 2017; Sala et al., 2023; Sandström et al., 2018; Sanyé-Mengual et al., 2023; Vanham et al., 2023). These impacts included pressure indicators such as greenhouse gas footprints, ecological footprint, water footprint, and nitrogen footprint, impact indicators such as ecotoxicity, human toxicity, eutrophication, and acidification associated with the consumer footprint, as well as more direct measures of biodiversity loss as captured in the biodiversity footprint.

***Agriculture production, in particular for the production of meat and animal products, was consistently identified as the food production stage with the greatest impact*** by studies looking at GHG emissions, biodiversity footprints, and consumer footprints (Bajan et al., 2022; Crenna et al., 2019; Notarnicola et al., 2017; Sanyé-Mengual et al., 2023). The primary drivers during this stage were land use change and greenhouse gas emissions. In terms of food waste, Sala et al. (2023) found that the largest share of waste (62%) is generated in the consumption stage, followed by processing and manufacturing (20%), primary production (10%), and retail and distribution (7%). Additionally, while meat and dairy generate less than 20% of food waste by mass, they are responsible for over 50% of the environmental impact associated with waste.

***The temporal trend of the impact of EEA32 countries' consumption varied with the type of impact.*** Several studies that provided time series results indicate a reduction in impact over time including a decrease in the per capita (20%) and total Ecological Footprint of food consumption from 2004 to 2014 (Galli et al., 2023), and a 2% decrease in total GHG emissions and 6% decrease in emissions intensity (CO<sub>2</sub>-eq per 1 GDP (PPP)) of food production for the EU-28 between the periods of 2010-2013 and 2014-2017 (Bajan et al., 2022). However, a study combining 16 impact indicators into a single impact score found that the Consumption Footprint of EU food consumption increased by 20% from 2010-2020 (Sala et al., 2023). The discrepancy in trends still exists when looking at overlapping time periods (2010-2014 and 2010-2017) for the Consumption Footprint. It is not surprising that the Ecological Footprint and the GHG Footprint results show similar trends since both frameworks include CO<sub>2</sub> emissions, which generally represent the largest component of each. Similarly, while some of the primary components of the ecological footprint and GHG footprint are conceptually covered in the Consumption footprint as climate change, fossil resource use, and land use, the Consumption Footprint is a combined indicator that includes over 16 indicators of various types. Moreover, the diverging results between Ecological Footprint/GHG Footprint compared to Consumption footprint suggests that while total demand on ecosystem biocapacity has decreased along with GHG emissions, the impacts or effects on ecosystem state and quality as measured through consumption footprint scores may be increasing. Assuming that GHG emissions sources are the same and decreasing over this period, this would mean that the other indicators of state and



quality may be increasing to a greater degree. A deeper investigation into the time trends of the Consumption footprint subcomponents would reveal which of the indicators are causing the net increase.

Another source of differences may be due to the technical methodology used, which can have a significant impact on results, especially when it comes to the accounting of livestock grazing (Vanham et al., 2023), one of the largest environmental impacts identified here in all categories of both pressure type indicators such as ecological, water, and GHG footprints and also impact indicators such as ecotoxicity and eutrophication.

**The EU displacement of impacts through international trade is significant.** Consumption and EE-MRIO based approaches allow us to track the origin of resources ultimately consumed in the EU and identify the location and type of impact. Here we find that over 21% of the biocapacity demanded by EEA food consumption activities is imported from countries outside of the EEA32, with China, USA, Brazil, Russian Federation, and Ukraine as the top sources. Sandstrom et al. (2018) found that most animal products consumed in the EU are produced in the EU or nearby, and the majority of food and feed crops are imported from more distant countries, in particular soybeans from Latin America, which in addition to vegetable oils and oil seeds, make up 76% of the EU's land use change emissions. Guillen et al. (2019) when looking at the seafood consumption footprint in 2011, found that the EU's consumption of seafood (by mass[kg]) depended significantly on imports; here we find that 22% of the fishing grounds footprint (by embedded primary-production [gha]) consumed in the EEA32 is imported from outside the EEA 32. In terms of water, the EU has seen a historical trend of increasing internal reliance, also from water scarce countries in terms of blue water, seeing increased internal trade and decreasing trade in virtual water outside the EU (Antonelli et al., 2017).

## 2.2.3 Circularity and EU food systems

### 2.2.3.1 *Circularity concepts and definitions*

A noteworthy strategy for driving the transition towards sustainability in the food system is converting to a circular design, which has seen a surge in interest in recent times (Jurgilevich et al., 2016; van Zanten et al., 2023). Incorporating circular principles into EU food systems would be a shift from the current agricultural system in Europe which largely relies on linear processes (Vlajic et al., 2021), leading to excessive utilisation and depletion of resources, significant waste generation across various stages of the value chain (Hamam et al., 2021), and ultimately negative impacts on biodiversity (Vlajic et al., 2021). In fact, the agri-food sector is one of the major drivers of biodiversity loss, endangering approximately two-thirds of species through agriculture and aquaculture (SITRA, 2022).

In general, transitioning to a circular economy is recognised to contribute positively to biodiversity (IPBES, 2019) by reducing the primary demand for resources (e. g. through recycling of resources), preventing pollution (e. g. through closing nutrient loops), and biodiversity-friendly sourcing (e. g. agricultural practices) (EEA, 2023a). Among the sectors deemed most impactful on terrestrial biodiversity - such as buildings and construction, forests, and fibres and textiles - the agri-food sector holds the greatest potential to halt and reverse biodiversity loss by applying circular economy principles. With a possible contribution of 73% to the overall positive biodiversity impact in a circular

economy scenario, its potential is significantly higher than, for instance, the building and construction sector with 10% (SITRA, 2022). This section aims to provide a better understanding of circular practices in the food system, their potential barriers and levers (including EU regulations), and their possible impacts on biodiversity.

In the scientific literature, circular economy is often viewed as a combination of activities related to reduction of use, recycling, and reuse (Kirchherr et al., 2017). These are so called R-principles. In regard to the food system, Vlajic et al. (2021) exemplified several circular practices on the basis of eight R-principles reoccurring in the literature, namely “Rethink”, “Redesign”, “Reduce”, “Replace”, “Reuse”, “Repurpose”, “Recycle”, “Recovery of other resources”.

Various approaches exist to define circularity within the food systems. From a narrow perspective, “avoiding and reusing waste and by-product streams to close biomass and nutrient cycles are the key principles of circular food systems” (van Zanten et al., 2023). The groundwork for circular agriculture was laid by Dutch researcher Jaap van Bruchem in the late 1990s when he acknowledged the importance of closed nutrient cycles in dairy farms (Hoeven, 2019).

From a broader perspective, circular economy in food systems encompasses all activities, practices and technology that help use and value resources efficiently. One such definition with a broader perspective emphasise the link to biodiversity, stating that circular economy in agriculture encompasses “the set of activities designed to not only ensure economic, environmental and social sustainability in agriculture through practices that pursue the efficient and effective use of resources in all phases of the value chain, but also guarantee the regeneration of biodiversity in agro-ecosystems and the surrounding ecosystems” (Velasco-Muñoz et al., 2021). Another broader definition is the one by van Zanten et al. (2019) who state that: “moving towards circularity in the food system implies searching for practices and technology that minimise the input of finite resources [...], encourage the use of regenerative ones [...], prevent leakage of natural resources from the food system [...], and stimulate reuse/recycling of inevitable resource losses [...] in a way that adds the highest value to the food system.” Both definitions involve the importance of valuing resources and using them efficiently. However, the first one specifically helps to understand several goals of the circular design in terms of impact, whereas the second one clarifies how to handle different types of resources. Another important aspect mentioned in the definition by Muscio and Sisto (2020) is the reduction of “negative discharges, such as waste and emissions in the environment”.

The definition of the circular economy concept for a specific part of the agri-food sector, such as fisheries and aquaculture, can help to better illustrate and understand the concept. In this context, the circular economy can be understood as “using all parts of the creatures harvested in aquaculture (including tails, shells, guts and frames) for human or animal consumption, or as fertiliser to grow more food. It means minimising the waste materials that end up in landfills, or as harmful gas or water emissions. It means using vessels, equipment and gear for longer, and making sure materials can be recycled at the end of their useful lives. It means decarbonising the energy and materials used along every link in the supply chain” (Cunningham et al., 2022). In aquaculture, it can be distinguished between direct circularity which replaces conventional monoculture with integrated multitrophic aquaculture, and indirect circularity which includes practices such as the valorisation of emerging waste (Eroldoğan et al., 2023).

The principles of circularity can be generally measured as the degree of maximisation of resource use efficiency or minimisation of waste, thus circular systems are generally quantified in a relative sense; for example, as the improvement of efficiency from the current average for the broad sector/activity or as the improvement from a previous non-circular state of the same system. However, there have been few quantifiable definitions of a circular system aside from a theoretical system that is fully efficient in using resources and producing no waste. Global Footprint Network has also proposed an absolute and quantifiable conceptual definition of a circular system (Global Footprint Network, 2020) based on a systems framework that is biological and regenerative in nature. Such a framework identifies a circular system as one whose activities ultimately result in the reduction in the ratio of human demand on biological resources compared to the biological resource regeneration of the planet. This can be described more simply as a system that produces a net reduction in ecological overshoot. Such a system implies that the net benefit from resource provisioning services (such as provision of recycled goods) and biological regeneration provided by that system, when compared to conventional practices, exceeds the resources needed to operate that system. This designation is most applicable to actors whose purpose is to provide services based on circular principles, for example, such as a plastics/wood/paper recycling company, whose provision of recycled resources results in a net reduction in total ecological footprint compared to the production of new plastics or primary harvest of wood. The net reduction would be calculated after deducting the resources consumed during the operation of the actor, which could, for example, include scope 3 CO<sub>2</sub> emissions (carbon footprint), paper usage (forest product footprint), and infrastructure area (built-up land footprint). This could apply also to food system actors that engage in regenerative practices or recycling of waste where the practices measurably reduce ecological footprint or increase biocapacity.

Sustainable agricultural systems and practices such as agroecology, organic farming or conservation agriculture are strongly aligned with circularity principles in the food systems since they aim to use resources efficiently and promote nutrient cycling. See section 3.2.1 for further information on environment and biodiversity-friendly production practices.

### *2.2.3.2 Literature Review*

The results of the literature review are structured by first outlining some highlights of the research as well as key principles from the narrow perspective mentioned in the introduction, then expanding to broader measures. It starts by explaining and visualising the principle of cascading use of biomass through a food use hierarchy. Next, the importance of nutrient cycling, closely linked to biomass cascading is discussed. The section then takes a broader perspective, examining waste streams beyond food and by-products, changes in protein production, strategies for reducing and decarbonising energy use, and the role of localised food systems.

The literature advocates for a radical redesign of the food system along the supply chain (van Zanten et al., 2023) while suggesting to rethink the definition of economic growth (van Zanten et al., 2019). Even though it has been acknowledged that achieving a fully circular food system might be too idealistic due to the unavoidable resource losses stemming from the system's complexities (van Zanten et al., 2019), a circular design holds significant potential for halting biodiversity loss and promoting biodiversity recovery (SITRA, 2022). Various concepts associated with circular food

systems include agroecology (Zarbà et al., 2021), closed loops (Barros et al., 2020), restorative and regenerative agriculture (Calisto Friant et al., 2021; Hamam et al., 2021), circular bioeconomy (Barros et al., 2020), clean production models (Hamam et al., 2021), and localised supply chains (Giudice et al., 2020).

Through a biophysical optimisation model, van Zanten et al. (2023) explored different circular scenarios applied to the EU27 and the UK food system, revealing the potential for a 71% reduction in land use and a 29% decrease in greenhouse gas (GHG) emissions per capita. Importantly, this analysis only considered direct land use changes, as data on indirect land use change is highly uncertain. It is suggested that real-world circular design could yield even greater environmental benefits. Reducing land use (Davison et al., 2021) and GHG emissions driving climate change (Habibullah et al., 2022) is vital, given their significant threats to nature and biodiversity. Moreover, reducing land use may create opportunities for biodiversity conservation (van Zanten et al., 2023).

However, while the findings presented by van Zanten et al. (2023) highlight promising outcomes, it is essential to interpret the numbers with caution. Firstly, the scenarios are based on a hypothetical self-sufficient European food system, while currently, the EU is highly involved in international trade (ESTAT, 2024). Secondly, a notable weakness of the modelling approach lies in data inconsistencies, prompting the authors to stress the importance of enhanced data collection and management at both the EU and global levels. Thus, results should be interpreted simply as circularity optimised scenarios with assumed self-sufficiency in all scenarios and dietary shifts in some scenarios.

Many circular measures highlighted in the literature revolve around the prevention, reduction, reuse, recycling, or recovery of waste and by-products as well as nutrient cycling, which can be associated with the narrow concepts of circularity in the food system. Additionally, a diverse range of measures, extending beyond waste, by-product, and nutrient management are mentioned. These include, for instance, favouring the consumption of certain livestock or animal-based products over others, transitioning to renewable energy instead of fossil fuels and localised food systems. The R-principles from Vlajic et al. (2021) reappear in the literature multiple times. However, terms like recycling, recovery, and reuse are often used interchangeably without clear distinction. Furthermore, discrepancies exist regarding the categorisation of by-products and food waste. While sometimes they are mentioned/considered as separate categories (e. g. in Hamam et al. 2021), elsewhere food losses and waste are incorporated within the broader category of by-products (van Zanten et al., 2019).

### **The principle of the cascading use of biomass in the food system**

The cascade use of biomass has been identified as key for circular food systems (Donner et al., 2020). The concept aims to maximise biomass use efficiency through sequential valorisation processes utilising various material forms (Keegan et al., 2013). Figure 3 illustrates a food use hierarchy for surplus food, food waste and by-products from processing which reflects several measures mentioned in the literature. The highest priority is given to the prevention of food waste. Following prevention, the hierarchy includes the reuse of surplus food for human consumption, reuse of surplus food for animal consumption, recycling of food waste and by-products from processing, recovery of nutrients and energy, and finally, disposal of food waste, which is the least preferred option.

Through the cascade use, several products can be gained. In the agri-food system, one can utilise food waste and by-products, such as crop residues like straw from cereal production, co-products from industrial processing, and livestock excreta, to create value by using them as a new source of food or by transforming them into, e. g., energy, fertiliser, or biochar (Barros et al., 2020; van Zanten et al., 2019). Regarding the aquatic food system, food waste and by-products such as fins, bones, shells, and blood water, can be valorised in the form of medicine, materials, fuels or animal feed to reduce the significant amount of seafood mass currently discarded (over 50%), thereby benefitting the environment and creating jobs (Hayes and Gallagher, 2019; Vieira Veríssimo et al., 2021).

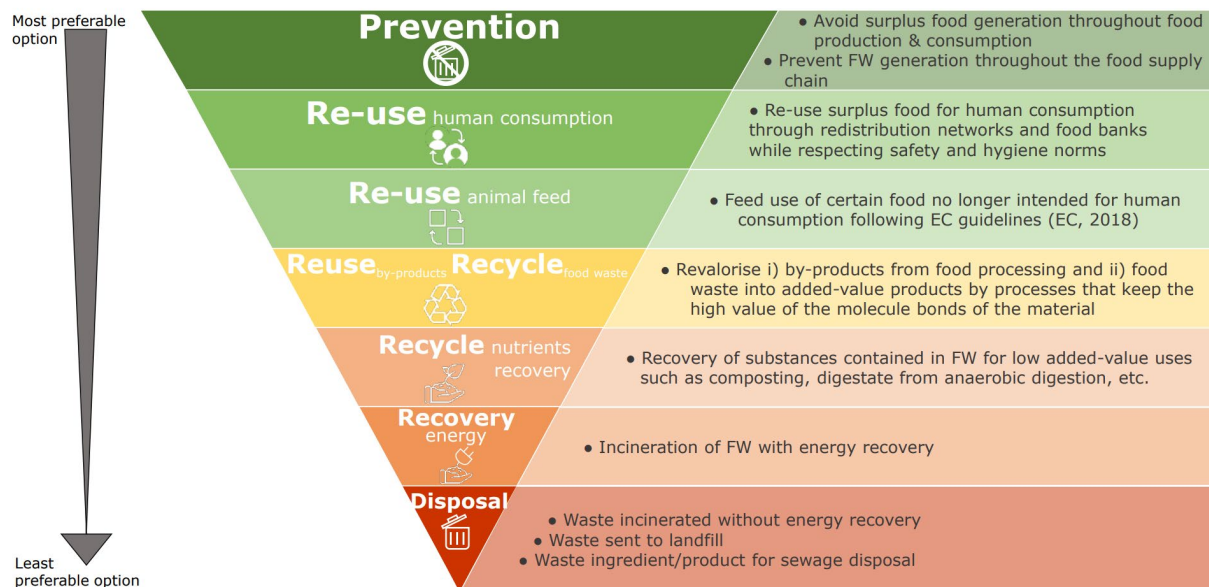


Figure 3: Food waste hierarchy, reproduced from Sanchez Lopez et al. (2020)

**Prevention** of food waste is the first priority of the hierarchy. Food waste in the EU needs to be massively reduced, as it is estimated that 20% of food is lost or wasted (EC, 2020a). In total, food waste amounted to 131kg per person per year in 2021, primarily from households (54%), followed by the manufacturing of food products and beverages (21%) and restaurants and food services (9%) (EC, 2022a). It contributes to approximately 16% of the GHG emissions from the EU food system (EC, 2024a). The major causes of food loss and waste can be attributed to four primary factors: overproduction, poor forecasting, technical inefficiencies, and consumer behaviour (Dora et al., 2021). At the household level, reasons include a lack of planning for grocery shopping and meals as well as impulsive purchases influenced through marketing strategies (European Parliament, 2024a). Further, consumers expect certain aesthetics and sizes which leads to retailers setting up rules and food not matching those requirements being thrown out even though they are perfectly fine for consumption (Mason et al., forthcoming). The variety of reasons for food loss and waste requires different approaches to tackle the issue. In general, optimising logistics across the supply chain is paramount for waste prevention and the efficient utilisation of resources (Teigiserova et al., 2020). One strategy to prevent surplus food is the reduction of overproduction (Garske et al., 2020a) as well as effective planning of product utilisation (Barros et al., 2020). This can be done through the improvement of contract conditions enabling better planning for producers (Thapa Karki et al., 2021).



Moreover, food labelling indicating the durability of food products often leads to avoidable food waste since consumers do not trust to eat the products after these dates, even though they might still be safe for consumption (Giudice et al., 2020). It is estimated that around 10% of the food waste generated in the EU can be attributed to date marking (EC, 2018). Changing policies in regard to labelling could be a lever for circularity (Giudice et al., 2020).

In general, to ensure a positive impact on the environment it is vital to take possible rebound effects into consideration since the reduction of food loss and waste could lead to lower prices and increased consumption, potentially offsetting 53-71% of the benefits (Hegwood et al., 2023).

**Use of surplus food and by-products for human consumption** is the second priority. When surplus food cannot be prevented, the food should be reused for human consumption, for instance through donations to social services, food banks (Hamam et al., 2021) or food sharing (Giudice et al., 2020). A potential barrier for the reuse of food could be the established hygiene rules (Teigiserova et al., 2020). Further, there is a risk that these measures may detract from the primary goal of waste prevention (Giudice et al., 2020). To avoid this from happening, shifting behaviours and habits by consumer education can act as a lever (Teigiserova et al., 2020).

Another way of reusing food is to utilise human edible by-products that emerge throughout the food value chain, such as during the production stage. Wheat middlings from flour production, for instance, can be utilised in mueslis. A further example is an EU-funded project that turns vegetable waste into functional powders, e. g. for food colouring or flavouring (Bas-Bellver et al., 2020). In regard to innovative foods fortified with by-products, it might be necessary to establish consumer trust (Hamam et al., 2021).

**Use of food waste and by-products as animal feed** is the third priority. By-products and food waste not suitable for human consumption can be used as animal feed for livestock, aquaculture or pets (van Zanten et al., 2019). These by-products may result from crops, livestock, and fish (van Zanten et al., 2023). Repurposing food waste as livestock feed contributes to more sustainable and circular production of animal-based food for human consumption and additionally results in manure that can be used as fertiliser to further increase the circularity of the system (van Zanten et al., 2023, 2019). As an example, the study by Dao et al. (2023) demonstrated that repurposing food waste as feed for laying hens increased feed efficiency while sustaining egg production. Moreover, the application of food surplus and by-products extends beyond traditional farm animals. For example, feed for aquaculture can be derived from food waste generated onboard cruise ships (Strazza et al., 2015). Technologies and methods to convert food waste into feed include, for instance, anaerobic digestion, insect-based conversion, and microbial fermentation (Hasan and Lateef, 2024). Further exploration of potentials will be addressed in subsequent paragraphs focusing on nutrient cycling. However, it should be noted that by-products and surplus food must be carefully managed when using it as animal feed, for instance through heat treatment. Health risks could emerge due to potential contaminations such as pathogenic bacteria, viruses (Shurson, 2020) or plastic pieces (Grant, 2018).

**Recycling of food waste and by-products including nutrient and energy recovery** encompasses the final possibilities prior to disposal. When all of the above is not an option, the by-products including food waste can be recycled in the form of new products. Recycling food waste and by-products as well as using biorefinery processes can yield a variety of products, including



biomaterials (Hamam et al., 2021), biochar (Barros et al., 2020), bio-pesticides (Duquennoi and Martinez, 2022), biowaste fertiliser (Barros et al., 2020), biopolymers (Hamam et al., 2021), various forms of bioenergy (Barros et al., 2020), and even medicine (Ellen MacArthur Foundation, 2024). Beyond reducing food waste, these products can bring various environmental benefits. Biogas for renewable energy production, for instance, can contribute to mitigating climate change and eutrophication (Winqvist et al., 2019). A further example is the application of biochar to soils as a promising method to reduce CO<sub>2</sub> emissions (Tan, 2019) and enhance soil quality (Amoah-Antwi et al., 2020). Research has also underscored the multifaceted advantages of substituting fossil fuels with bioalcohol, spanning environmental, social, and economic realms (Ometto et al., 2007). Moreover, aquatic biorefineries can reduce environmental pressures emerging from nutrient leakage in coastal areas that are related to aquaculture expansion (Günther et al., 2023). Encouraging the development of new biomaterials and biorefineries by policies can boost circularity (Muscat et al., 2021).

Due to the fast degradation rates of food, especially in warmer climates, management cycles are inevitably short when reusing or recycling food loss and waste (Teigiserova et al., 2020). Proximity in this regard can reduce costs in different ways, as it can reduce transportation and energy costs (Teigiserova et al., 2020). The eco-efficiency of biogas plants is influenced by factors such as the type of raw materials and transportation distances, with closer proximity to food processing plants allowing for higher eco-efficiency (Muradin et al., 2018). Farm-scale biogas plants can enhance the use of manure and offer several environmental benefits (Winqvist et al., 2019).

Industrial symbiosis, which involves collaboration between various sectors to valorise waste and increase resource efficiency (Trokanas et al., 2014), is also a potential lever for the circular use of waste in the food system (Zarbà et al., 2021). Additionally, partnerships between retailers could facilitate the collective collection of certain types of waste or enhance consumer participation in waste collection (Barros et al., 2020).

### **Nutrient cycling**

In addition to reducing and valorising food waste and by-products, nutrient cycling is a recurrent theme in the literature and a fundamental principle of circular food systems. The two elements are intertwined because food waste and by-products include nutrients and therefore can be valorised to increase circularity of nutrients in the food system. Composting food waste, for example, can substitute artificial fertilisers (van Zanten et al., 2023). Further, livestock can be fed with food waste and act as ‘nutrient recyclers’ since their manure can be used as fertiliser (van Zanten et al., 2019). Depending on the nutritional value of waste and by-products, it can be more beneficial to either feed them to livestock or directly apply them to the soil (van Zanten et al., 2019). In order to find out which of those two options is more beneficial in different circumstances, more research is needed (van Zanten et al., 2019).

Not only animal excreta but also human excreta contain valuable nutrients such as phosphorus or nitrogen. In a circular agri-food system, these waste streams could be used for organic fertilisation to increase nutrient cycling (Mhatre et al., 2021; van Zanten et al., 2019). Recovering nutrients from human excreta needs to be done with caution due to potential health hazards (van Zanten et al., 2023). To avoid risks it is important to ensure appropriate sanitation systems and redesign where necessary (van Zanten et al., 2023). Especially cities represent a large pool of human excreta and

opportunities for harnessing these resources in innovative urban farming models such as combining plant, insect and fish production (van Zanten et al., 2019). Also, sewage sludge can be used for fertilisation (Mosquera-Losada et al., 2017).

When dealing with livestock manure as fertiliser, several aspects need to be considered. To positively impact soil biodiversity and prevent negative effects such as nitrogen leakage, it is important to prioritise manure quality over quantity (Köninger et al., 2021). As a lower priority, manure could also be recycled through anaerobic digestion, converting it into biogas and nutrient-rich digestate for fertilisation (Köninger et al., 2021). Further, the potential of livestock as nutrient recyclers in a circular scenario depends on human diets since it can influence the amount of by-products available as feed for livestock. For example, if human consumption switches to whole grains instead of refined grains, fewer by-products would be available for feed (van Selm et al., 2022; van Zanten et al., 2023). Another uncertainty concerning the role of farm animals in circular food systems is that some animal species are less efficient in upcycling low-opportunity-cost feed than others. For instance, some species are bred for efficiency in consuming high-opportunity-cost feed which can be consumed also by humans such as grains (van Zanten et al., 2019). In contrast, low-opportunity-cost feed includes by-products, food waste or grassland assets (van Selm et al., 2022). However, in circular scenarios that involve feeding animals exclusively with low-opportunity-cost feed, the modelled quantity of protein provision (23g/person/day) was less than half of the current European average consumption (51g/person/day), suggesting that a reduction in animal-based protein is critical to enhancing the circularity of European food systems (van Zanten et al., 2023). To maximise the efficient use of animal manure, mixed crop-livestock farming (UN DESA, 2021) and enhanced collaboration between livestock farmers, cooperatives and crop farmers is recommended (Barros et al., 2020). Overall, reconnecting livestock and crop farming is essential to closing nutrient cycles and promoting sustainability (Muscio and Sisto, 2020).

Using the abovementioned methods for organic fertilisation and nutrient cycling can maintain or improve soil fertility, decrease pressure on land and reduce the need to apply artificial fertilisers (van Zanten et al., 2023). The latter is of particular relevance since excessive synthetic fertilisation in the realm of agricultural intensification in Europe poses a threat to aquatic and terrestrial biodiversity (Billen et al., 2021). At the same time, there are challenges and risks associated to the use of manure or other organic fertilisers as well. The environmental impacts depend on the nutrient source, the application method and quantities as well as the area and crops where it is applied (Köninger et al., 2021). Furthermore, in a circular scenario in the EU, the available quantities of organic fertilisers may not meet the demand (van Zanten et al., 2023). Lastly, nutrient release in organic fertilisers is delayed compared to mineral fertilisers, potentially leading to nutrient losses and a mismatch with crop demand if not planned and implemented correctly (van Zanten et al., 2019).

### **Additional circularity measures from a broader perspective**

Next to food waste and by-products, it is also essential to explore other, non-biotic waste streams within the food system and their potential for valorisation to reduce environmental impact through e. g. marine littering (EEA, 2023b). Other waste streams include, for instance, food packaging waste (Geueke et al., 2018), fishing gear (Gilman, 2015) and waste from agricultural machinery (Zarbà et al., 2021). Recognising these waste streams underscores the holistic approach needed to achieve circularity in the food system.

**Shifting protein consumption patterns** can further enhance circularity and the environmental impact of food systems. This involves reducing the consumption of animal-based food which is essential when considering low-opportunity cost feed such as food waste instead of high-opportunity cost feed (van Zanten et al., 2019) and prioritising arable land for biomass production intended for human consumption over production for animal feed (van Selm et al., 2022). Moreover, the consumption of alternative protein sources (SITRA, 2022) and moving towards low-trophic species (van Zanten et al., 2019) can positively contribute to the environment.

**Energy** use in the food sector accounting for 17% of the EU's gross energy consumption in 2013 (EC, 2015) causes high costs (Zarbà et al., 2021) and can substantially harm biodiversity (EEA, 2023a). Impacts can be reduced when renewable energy options such as solar, wind or bio-energy are utilised instead of non-renewable fossil fuels (Zarbà et al., 2021) (Barros et al., 2020).

In 2019, the agriculture sector accounted for 28% of the total water abstraction in the EU-27 (202,000 million m<sup>3</sup>) (EEA, 2022a), which is why the reduction of its use is needed as a first priority (Del Borghi et al., 2020). Additionally, wastewater should be reused or recovered to increase water cycling (Duquennoi and Martinez, 2022) or recover nutrients (Garske et al., 2020b). Hydroponic systems, for instance, promote water cycling by combining aquaculture and crop production (Aleksić and Šušteršič, 2020). To prevent risks to human and environmental health from diverse pollutants, it is important to treat the wastewater adequately. Even though in the EU urban waste waters are collected and treated according to certain standards to ensure safety (EC, 2021c), currently, urban wastewater treatments for pollutants like PFAs need to be improved which is why the European Parliament recently approved new rules for enhanced urban wastewater treatment (EP, 2024).

**Transitioning to localised food systems and short supply chains** can further benefit circularity in the food system through e. g. short management cycles (Jurgilevich et al., 2016; Teigiserova et al., 2020). Nevertheless, it is crucial to acknowledge that local food systems might not consistently decrease GHG emissions, especially if local production doesn't align with local environments and involves non-environmentally friendly practices or technologies (Coelho et al., 2018).

**Sustainable agricultural systems and farming approaches** such as organic farming (Selvan et al., 2023), agroecology (Wezel et al., 2020), mixed crop-livestock farming, or agroforestry (UN DESA, 2021) strongly overlap with circular principles. Using resources sustainably and closing biomass and nutrient cycles are among their basic principles, which are crucial to ensure environmental sustainability (Selvan et al., 2023; Wezel et al., 2020).

## **EU efforts to enhance circularity in Europe**

The abovementioned measures are important to establish a circular food system in the EU and thereby enhance biodiversity. While some data related to measures have been cited to give an idea about advancements in the EU, it is important to have a look at current policies and strategies being implemented. Recently, the concept of a circular economy has gained importance in EU policies (Zarbà et al., 2021) and several regulations and strategies are shaping the transition towards circular food systems. At the heart of these efforts lies the Circular Economy Action Plan (CEAP), which aims to establish circularity in the EU through strategies such as promoting sustainable products and reducing waste (EC, 2022b). Notably, the [CEAP from 2015](#) has already taken measures to reduce food loss and waste. This includes the establishment of a [methodology for measuring food waste](#), the

creation of an [EU Platform on Food Loss and Waste](#), clarification of legislation on waste, food, and feed to facilitate [food donation](#) and [repurposing of food for animal feed](#), alongside efforts to enhance [date marking](#) practices (EC, 2024b).

Further, as an action stemming from the CEAP from 2015, the [EU Waste Framework Directive](#) was revised in 2018, urging EU countries to reduce food waste along the supply chain, monitor levels and provide updates regarding advancements made (EC, 2024b). The Directive obliges Member States to take measures to achieve adherence to the hierarchy for waste management, with a certain extent of flexibility. Member States can deviate from the hierarchy if they can achieve a higher environmental benefit in that way (2008). Member States shall use economic tools and other incentives for the application of the hierarchy. The Directive also requires separation or recycling of biowaste at source or separate collection, which could enhance recycling, e. g. as fertiliser (Garske et al., 2020a). In 2023, the Directive was revised again proposing legally binding food waste reduction targets including the reduction by 10% in processing and manufacturing and by 30% per person at the retail and consumption level by 2030 (EC, 2024c). In March 2024, the Parliament stated that it aims to increase these targets to achieve a reduction of 20% and 40% compared to 2020-2022, with potential increases to 30% and 50% by 2035 (European Parliament, 2024b).

In the new [CEAP](#) from 2020, food is one of the focus sectors as it is considered one of the most resource-intensive sectors (EC, 2022b). The CEAP serves as a foundational pillar of the [European Green Deal](#) (EGD) (EC, 2022b), which aims to achieve a modern, resource-efficient, and competitive economy while reducing emissions, decoupling economic growth from resources, and ensuring inclusion for all (EC, 2021a). As part of the EGD, [Directive 2024/825](#) (EC, 2021a) came into force. It is directed towards creating transparency about circularity aspects of goods such as durability, or recyclability so that consumers can make better-informed decisions preventing misleading information about the sustainability of goods (EU, 2024).

The [Farm to Fork Strategy](#) (F2F), adopted in 2020, is the most relevant element of the EGD for the food system, aiming at creating fair, healthy, and environmentally friendly food systems (EC, 2020b). The F2F and the CEAP are interrelated since they both aim to promote the sustainability and circularity of food systems. The F2F strategy promoted a revision of the EU date marking rules to prevent misunderstandings and misinterpretations by consumers (EC, 2024d). It further aims to improve sustainable nutrient management via an integrated nutrient management action plan to encourage sustainable agricultural practices, including the recycling of organic waste for fertilisation (EC, 2020b). However, that initiative has not been implemented.

The concept of bioeconomy, closely related to the circular economy (Kardung et al., 2021), is also integral to the EGD (EUBA, 2023). Bioeconomy entails “the sustainable production of renewable resources and their conversion into food, feed, fibres, materials, chemicals and energy” (EUBA, 2023). The [Bioeconomy Strategy](#) of the EU, aiming at sustainability, includes, sustainable management of natural resources, reduction of non-renewable resources, climate change adaptation and mitigation as well as food and nutrition security (EC, 2024e). Among others, the [bioeconomy action plan](#) from 2018 aims to support sustainable biorefineries, sustainable food and farming systems and education across the bioeconomy (EC, 2024e).

The [Common Agricultural Policy](#) (CAP) is the sectoral EU policy that aims to contribute to the goals set by the F2F strategy and the [biodiversity](#) strategy related to agriculture (EC, 2022c). It aims to ensure environmental, social, and economic sustainability in the agricultural sector (EC, 2021d). The CAP [2023-2027](#) offers policy interventions to be implemented by EU member states to support sustainable farming practices such as organic farming, agroecological and other environmentally-friendly practices (EC, 2023a) which are strongly aligned with circular measures.

The [Common Fisheries Policy](#) aims at enhancing the sustainability and resilience of the EU's fisheries and aquaculture sectors (EC, 2022d). Circularity in fisheries promotes reducing fish overexploitation and implementing the landing obligation by 2019 (EC, 2022d). This obligation requires that all catches of regulated fish species be landed, with some exemptions, allowing undersized fish to be repurposed for non-human consumption products (EC, 2024f).

A lever for circularity in the food systems is research and innovation. [Horizon Europe](#), as the major funding program for research and innovation in the EU supports research on innovative solutions that enhance sustainability in the food system. It supports food system-related research on circularity, nutrition, quality of food, climate and communities and alternative proteins (EC, 2020b).

The [Circular Economy Stakeholder Platform](#), initiated by the European Commission and the European Economic and Social Committee, is a hub for exchanging insights on various circular economy topics. It connects stakeholders, disseminates information on best practices, national, regional, and local strategies, European and international networks, and announces relevant events. Additionally, it provides a toolbox comprising financing opportunities, educational resources, training materials, measurement possibilities, and guidelines.

There are additional policies and strategies in the EU that impact circularity in the food system. Due to continuous changes and the variety, this working paper will not cover all of them. However, to provide an idea, some examples include the [Renewable Energy Directive](#), the [Water Reuse Regulation](#), the [EU Single Use Plastic Directive](#), or the [Sewage Sludge Directive](#). Additionally, Figure 4 provides a comprehensive overview of key EU policies, strategies and visions that influence the European food system.

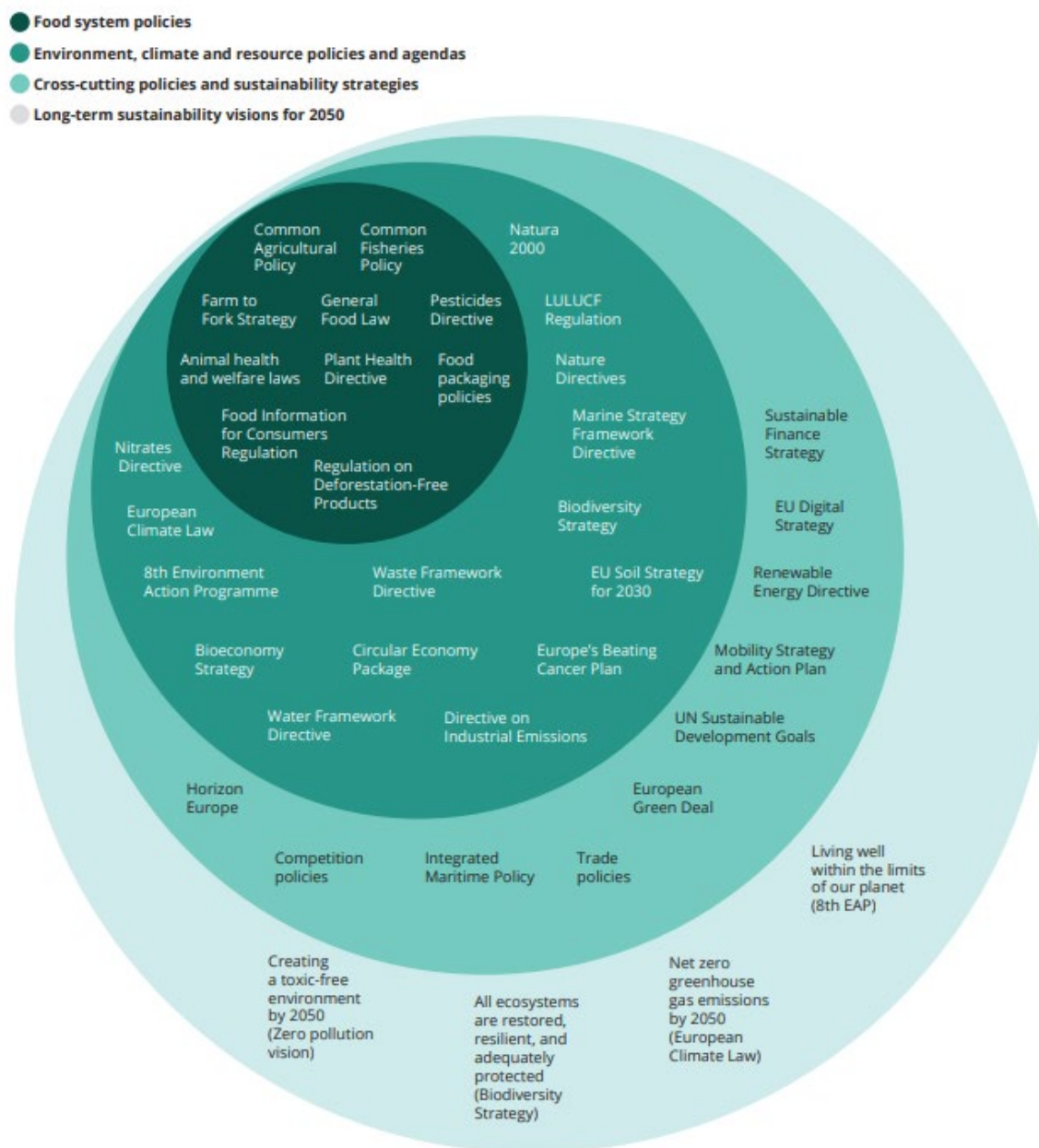


Figure 4 Key EU Policies, strategies and visions influencing Europe's Food System. Source: (EEA, 2022b)



## **General levers to further enhance the implementation of circularity in the food system of the EU**

Since the application of a circular design to the food system has a high potential to positively contribute to biodiversity (SITRA, 2022), it is recommended to further promote its implementation. Although some barriers and levers specific to certain measures have been mentioned earlier, the literature identifies several general levers. These can further enhance the EU's performance in this area.

The mix of policy tools to promote circular principles while fostering awareness and accountability among stakeholders is crucial (Hamam et al., 2021). In general, there is a necessity for collaboration between policy makers and diverse stakeholders and for comprehending the obstacles and motivations (Hamam et al., 2021). Regulatory measures play a crucial role, such as bans and specific production standards, as well as public procurement directives, taxes, and fees (Giudice et al., 2020). Economic instruments, such as taxes and cap-and-trade systems offer viable strategies for mitigating food waste (Garske et al., 2020a). Policymakers should prioritise making sustainable foods financially appealing to consumers and shift costs from sustainable to non-sustainable options (Giudice et al., 2020). In that sense, also retailers can play a role in favouring circular food through pricing strategies (Barros et al., 2020). To exemplify this matter, "premium prices" often associated with organic food products are frequently cited as the main reason for consumers to hesitate to purchase them, despite acknowledging their benefits and having a positive attitude towards these products (Melovic et al., 2020). Conversely, the consumption and environmental footprint of highly unsustainable food products, such as beef, could be reduced through significant taxation. Even taxing only the most impactful animal products could make a substantial difference (Bonnet et al., 2018). According to a pan-European survey, around 62% of respondents expressed support for tax-free healthy and sustainable food products (UCPH, 2023).

Moreover, education initiatives (Giudice et al., 2020), subsidies for sustainable initiatives, transparent information dissemination (van Zanten et al., 2019) and improved food labelling (Mhatre et al., 2021) are instrumental in steering behaviours towards circularity. The development of new marketing strategies to enhance consumer interest in upcycled products is essential, along with innovative business models such as those involving farmers with biogas plants (Donner et al., 2020). Technology and innovation, if aligned with environmental sustainability goals, hold promise in advancing circularity within the food system (Muscio and Sisto, 2020). By improving efficiencies, technologies can save resources (Muscio and Sisto, 2020), with examples including advancements in indoor aquaculture for seaweed production or meat substitutes (Kardung et al., 2021). However, van Zanten et al. (2023) argue that achieving circularity in the food system doesn't necessarily require new technologies. Instead, its success would depend largely on social acceptance and economic sector transformation. Support for research endeavours is vital in deepening our understanding of the complexities associated with transitioning to a circular food system (Muscio and Sisto, 2020). The policy-science interface can be crucial and can be addressed through e. g. policy briefs or policy labs (Duquennoi and Martinez, 2022).

## 2.3 Discussion

The set of environmental impact assessments reviewed here covers a broad range of topics, but also evolving methodologies and indicators. In synthesising the results, there was a clear consistency among studies highlighting the high negative environmental impact of the consumption of animal-based products due to land use and climate impacts occurring during the agricultural stage.

### 2.3.1 How do current methodologies capture environmental and biodiversity impacts

Starting from the earliest stages of impact, pressure indicators such as the Ecological Footprint, which captures bioproductivity-weighted land demand including that from CO<sub>2</sub> emissions, and the Carbon Footprint are able to best capture the pressure exerted in relation to land use and climate change. Water remains a clear requirement for the existence of biodiversity in ecosystems, but the linkages between water consumption and its biodiversity impacts have not been clearly established by the studied footprint frameworks.

State and impact stage indicators, which are comprised mostly of LCA-based approaches can achieve more direct measures of impact, although the current ability to link endpoint measures of biodiversity remains a challenge.

Biodiversity footprints are the only frameworks that have linked direct measures of biodiversity loss with food consumption and production, however, while results are mostly consistent, the highlighting of mixed results (one study found ecotoxicity to have the highest impact in contrast to others) suggests that the assessment methods for biodiversity impact are not clearly defined and further clarifications are needed (possibly because there are diverse valid ways to address it). In terms of understanding the impacts of food systems on the environment and especially biodiversity, it is helpful to use the impact stages (DPSIR) as a hierarchy, even though the specific usage of the DPSIR framework in reference to the delineation between stages is not standardised nor is it consistently used in the literature. Nevertheless, considering food production and consumption as the driver, pressure indicators generally provide more concrete and accurate quantitative assessments of the subject. For example, water use, CO<sub>2</sub> emission, mass of extracted material, and land use are concepts that can be concretely and easily understood and measured for 1 kg of consumed wheat. Additionally, the ability to compare environmental resource use or impact against a relevant and defined environmental threshold at different spatial scales is a critical aspect of sustainability assessment (Baue, 2019). For example, the ecological footprint of a population can be compared to the biocapacity of a country, region or world in total or per capita basis to provide various contextual information about overuse, dependency, and other contextual aspects. Similarly, GHG emissions can be evaluated against a 1.5-degree climate change threshold on a global scale. On one hand, these thresholds represent critical boundaries for the stability of planetary systems and therefore they are implicit prerequisites for the maintenance of biodiversity. On the other hand, it is difficult to translate and link these quantified pressures to specific, spatially explicit, and tangible effects both on the environment and even more so on biodiversity. This leads to important questions such as: how does having a large water footprint manifest in environmental or biodiversity damage? At what point does the size of an ecological footprint or carbon footprint result in significantly harmful impacts on biodiversity that should be avoided?

Moving to state and impact type indicators, they generally provide information on ecosystem quality or changes to ecosystem quality by measuring the impacts of production and consumption activities such as ecotoxicity, via the flow of resources/emissions/pollutants. In general, LCA-based methodologies or models are needed to translate pressures into impacts in a standardized manner. While there is not a clear concordance between LCA impact categorisation and the impact stages of the DPSIR framework, the categorisation of “midpoint” indicators as used across LCA-based studies generally falls within the state and impact categories. with some overlaps with “pressure” type indicators.

As primary flows or emissions followed down the cause-effect stream are ultimately translated to measures of ecosystem quality and impacts we inherently introduce interpretation and uncertainty in both concept (what is being measured) and in measurement (units of the indicators). Ultimately, however, these types of indicators can provide more information with specific relevance to an endpoint target. For stakeholders and policies that are most concerned with the target, endpoint measures can directly quantify these outcomes or targets. Due to the degree of interpretation, weighting, and current state of scientific knowledge of the cause-effect chain on biodiversity, however, this also leads to important questions, for example: what type of biodiversity loss does a biodiversity footprint measure? Are they different with each model and how does the ultimate stakeholder interpret the result? If we value biodiversity for the purely functional aspect of ecosystem function, does a greater biodiversity footprint mean a greater loss in ecosystem resilience and function if not all species are captured?

As our ability to gather more biodiversity-relevant data improves, the question of why biodiversity is important and what aspects of biodiversity are important to the stakeholder becomes more and more relevant. This is because multi-dimensional composite indicators become harder to interpret as increasing impact factors are combined. Increasing impact categories has resulted in additional weighting and scoring for the sake of providing a high level of simplicity and interpretability.

The combined result of the consumption footprint for example, brings together 16 indicators across a wide range of environmental impact types into a single value. As discussed earlier in section 3.2.2, the consumption footprint suggests that the environmental impact related to EU food consumption has increased by 20% from 2010-2020. While this tells us that some calculated value of impact has been increasing, it is not possible to understand which of the 16 impact categories may be causing this increase and which categories may even be decreasing. Interpreting this result is particularly challenging when the impacts range from climate change to ecotoxicity to human toxicity and the time evolution of individual impacts is not presented. These results may only be accessible and easily interpreted by subject matter experts and only with a deeper review of the primary study. Ultimately stakeholders may have a different value system in their interpretation of relative importance, for example viewing human health impact as being much more important than climate change compared to the weighting scheme built into the indicator.

In regard to the biodiversity footprint, some additional uncertainty exists in both translating midpoints to endpoints (e.g. how does land use change translate quantitatively to biodiversity loss) and in the comprehensiveness of capturing all midpoints that contribute to biodiversity loss (does the biodiversity footprint include all measurable midpoints, such as ecotoxicity, that lead to

biodiversity loss). As scientific knowledge is continuously improving in these areas, uncertainty will decrease and there may be a greater convergence in the common interpretation of these indicators.

Still, there remain serious challenges in understanding environmental and biodiversity impacts related to food systems. Some challenges are simply due to the complexity of ecosystems. For example, it is unclear still whether the existing biodiversity impact measures account for non-linear effects on biodiversity or accurately assess the relative importance of impact types on different elements of biodiversity (as monitoring gaps exist). It is unclear if or how much the frameworks can address or account for temporal and threshold effects, related to extreme climate events or habitat fragmentation.

### 2.3.2 What do the results of these methodologies tell us?

We know with relative certainty about the overall level of pressure exerted in many areas in terms of resource flows and waste/emission/pollutant flows related to food systems. The available indicators also consistently highlight specific sectors, food types, and food system stages that are the most impactful.

Detailed LCA-based studies provide an indication of specific products that have a high impact relative to consumption, and these are identified for multiple impact categories. End-point indicators with the greatest uncertainty such as those related to biodiversity consistently converge on the same impact hotspots. Current results suggest that improvements in the coverage of impact and biodiversity monitoring have the potential to extend and refine our knowledge of biodiversity impact hotspots. Indeed, there remain many important questions for scientific advancement, however the existing knowledge can provide substantive support for stakeholders and policy development around environmental and biodiversity impacts.

From an overarching sustainability perspective ecological footprint results for Europe show a decreasing trend of per capita and total consumption, however, the gap between Europe's ecological footprint and biocapacity is still large, with consumption exceeding biocapacity by nearly 1.5 times. The ecological footprint associated with food consumption is the largest demand category across all categories (food, housing, mobility, goods, and services), making up nearly a third of the total ecological footprint. This deficit or quantitative imbalance implies potential risks if there are disruptions to supply chains or production systems and highlights a need for increased capacity and resilience for domestic agri-food systems. From a quantitative perspective, the maintenance of biodiversity can be viewed from a utilitarian perspective with biodiversity representing a key attribute of both natural and human-managed ecosystems which supports the resilient structure and function of ecosystems.

Within the ecological footprint of food, the largest subcategory was found to be “meat”, followed closely by “bread and cereals”. While nearly equivalent in terms of total footprint, the footprint efficiency, or ecological footprint per calorie of food consumed, of “meat” is up to several orders of magnitudes larger than that of plant-based food found in “breads and cereals”. Other indicators and indicator frameworks support these findings, with water footprint studies indicating both a dependence on imported water footprint, or water used within the supply chain of imported products, from outside Europe (Dolganova et al., 2019) and an increasing water demand on water-

scarce countries within Europe. Marine resources in the form of seafood appear also to be dependent on international trade when looking at the total mass of consumed seafood. Together, these dependencies underscore the importance of understanding Europe's food-system related supply chain sources and the resilience of those sources in the context of potential climate-related or geopolitical disruptions.

With the exception of the seafood consumption footprint, which does not look broadly at all food categories, all studies providing results from pressure indicators (the ecological footprint, GHG footprint, carbon footprint, nitrogen footprint, and water footprint) indicate that animal-based food products are the category of food with the greatest footprint and/or where reduction in consumption has the greatest potential to reduce footprints.

The results from these pressure indicators are linked to the drivers of biodiversity loss but they do not allow direct attribution to impact on biodiversity in a quantifiable or standardised manner. It can be argued that while humanity's consumption is in overshoot of biocapacity, all pressure can potentially translate to additional biodiversity loss. This would imply that any pressure hotspots identified, such as the consumption of animal-based products, or production of livestock, can be considered a biodiversity impact.

LCA-based midpoint indicators provide a broader range of environmental impacts, including and overlapping with the pressure indicators and can be directly linked to biodiversity loss through cause-effect relationships and translated into endpoint indicators. Capturing a broad range of environmental impacts, recent results from the consumption footprint suggest that total environmental impact from European food consumption is increasing even though pressure seems to be decreasing. Compared to pressure indicators described above, the consumption footprint incorporates many more categories of environmental damage or pollution and thus the results generally suggest that the overall quantity of consumption in Europe is decreasing over time while damage to ecosystems appears to be increasing.

Similar to the results of pressure indicators, the hotspots identified by LCA-based midpoint and endpoint indicators also point to animal-based products as having the largest environmental and biodiversity impact. The fact that almost all indicators point to this hotspot goes beyond confirmation of animal-based food products as a hotspot. They suggest that the production and consumption of animal-based products result in the greatest pressure on biodiversity and additionally result in a broad range of environmental impacts.

Among LCA-based studies, climate change, land-use change, and ecotoxicity were consistently identified as the highest impact categories associated with European food consumption.

A single statement can summarize the most consistent finding across almost all indicators and frameworks. Consumption of animal-based products appears to drive the greatest negative environmental and biodiversity impacts, which occur with the greatest impact at the agricultural production stage.

From a solutions perspective, this means that agricultural production, especially the livestock sector, must change. Knowledge and practical experience are available to implement sustainable agricultural approaches, although implementation can face challenges depending on local

environmental and socio-economic conditions. Enablers to overcome challenges and scale up sustainable production systems include supportive policies, financial and economic tools, knowledge transfer and systemic changes along the food value chain including retail and consumption.

One of the horizontal sustainability principles is circularity, which has gained significant attention in the EU. It offers a holistic approach to address biodiversity impacts of food systems, directly by improving resource use efficiency and reducing waste, and indirectly by supporting environmentally and biodiversity-friendly agricultural production practices. Adopting a circular design in the food system can greatly benefit terrestrial, freshwater and marine biodiversity and the environment. It can reduce land- and sea use, pollution and greenhouse gas emissions. Circular measures can enable freeing up and repurposing land for biodiversity conservation or converting to less harmful agricultural and aquaculture practices using fewer inputs and resources while maintaining or improving production. At the centre of this circular design are nutrient cycling and the avoidance and valorisation of food waste and by-products. Further resource efficiency and reduced negative impact on biodiversity can potentially be achieved through enhancements of other waste streams along the supply chain such as packaging or fishing gear, shifts in protein consumption patterns, decarbonisation and minimising energy use, and the promotion of localised food systems. Circularity is intrinsic to sustainable agricultural practices such as agroecology, organic farming, and regenerative agriculture as their principles include strategies to increase resource efficiency, material-, nutrient- and water cycling.

Examining potential scenarios for applying circularity in the EU food system is a new research area, currently highlighting several gaps and uncertainties. Increasing research in this field is crucial to better understand potential trade-offs. Future models should combine biophysical models with socio-economic models for a more holistic understanding of potential implications. More research is needed to fill knowledge gaps in circular measures and to understand better what tangible impacts the complex mix of EU strategies and policies can have on circularity. At the same time, actions in the policy-science interface are critical to support an ongoing transition to a resource-efficient and environmentally friendly agri-food system.



## 3 Objective 2 - Links between biodiversity-friendly farming and food consumption changes

### 3.1 Methodology

To identify the links between biodiversity-friendly farming and food consumption changes we performed a literature review and synthesized the findings in a structured evaluation matrix. Below we describe the review criteria and conceptual elements adopted in the evaluation of the literature.

#### 3.1.1 Scope (Review targets/study type/territorial scope/temporal coverage)

The literature review was carried out in February 2024 through Google Scholar. While in principle the focus was on EU countries, studies from other countries were included as well. The research focused primarily on peer-reviewed papers providing quantitative results on the assessment of the effect of alternative farming practices compared to traditional ones; however, papers with qualitative results were also scrutinized and included those with conclusive and clearly pointed out results. The review looked at studies published since 1990 up to February 2024.

#### 3.1.2 Approach (Criteria for selection, keywords used)

The keywords used were the following: ("Organic farming" AND "Biodiversity"), ("Agro-ecology" AND "Biodiversity"), ("Agroecology" AND "Biodiversity"), ("Extensive grazing" AND "Biodiversity"), ("Mixed livestock-crop" AND "Biodiversity"), ("Permaculture" AND "Biodiversity"), ("Agricultural synergies" AND "Biodiversity"), ("Multilayer agriculture " AND "Biodiversity"), ("Multilayer farming" AND "Biodiversity"), ("Multilayer cultivation" AND "Biodiversity"), ("Urban agriculture" AND "Biodiversity"), ("Urban farming " AND "Biodiversity"), ("Eating habits" AND "Biodiversity"), ("Eating choices" AND "Biodiversity"), ("Dietary choices" AND "Biodiversity"), ("Dietary habits" AND "Biodiversity"), ("Plant-based" AND "Biodiversity"), ("Vegetarian diet" AND "Biodiversity"), ("Vegan diet"), ("Alternative diet" AND "Biodiversity"), ("Vegetarianism" AND "Biodiversity"), ("Veganism" AND "Biodiversity"), ("Sustainable diet" AND "Biodiversity"), ("Alternative protein" AND "Biodiversity"), ("Organic food" AND "Biodiversity"). As Google Scholar searches all sections of the papers (including the references), a manual filtering operation was performed to ensure that the objective of the identified papers was to assess the impact of alternative agricultural practices or human dietary habits on biodiversity. The keywords and indicators to focus on were chosen after carrying out an expert consultation. The selected indicators are meant to capture key aspects of biodiversity by focusing on the most widespread indicators, avoiding the use of too specific ones, on one hand, and on the other hand, they were selected to reflect the potential connection with EU legal frameworks on environmental matters. Details on the selected agricultural practices and indicators are provided in the following sections.

### 3.1.3 Conceptual framework

To investigate the relationship between the demand-side and the supply-side of food systems, the biodiversity-friendly agricultural practices accounted for are first introduced and described. Second, the biodiversity aspects considered are described by providing details about the biodiversity indicators selected and investigated, followed by a description of other environmental aspects (and related indicators) that are commonly implied in research projects targeting the assessment of the effects of agricultural practices on biodiversity. Furthermore, an overview of the general current EU food consumption patterns is provided. These first sections work as an introduction to the review's results that are subsequently provided.

The findings guide the following sections where, first, an overview of the current EU organic food dynamics and trends are shown in terms of both production and consumption. The results of studies addressing simultaneous changes on the demand- and supply-side of the EU food systems are presented highlighting linkages with environmental benefits. Moreover, the potential impacts of different diets and changes in consumption patterns in EU on biodiversity are presented and described.

Finally, the potential means to implement shifts in terms of food consumption are presented. The specific and overall findings are then discussed to draw conclusions about the linkages between EU food systems and the related impacts on biodiversity.

Sustainable agriculture can be defined as a system of farming that strives to provide the resources necessary for present human populations while conserving the planet's ability to sustain future generations (Dubey, 2024). Agricultural practices focused exclusively on crop yields can be transformed into sustainable practices focused on multiple outcomes, including environmental health.

To investigate the impacts of potential consumption changes on the demand for different agricultural practices, this section focuses on six selected agricultural systems/practices (agroecology, organic farming, agroforestry, extensive grazing, urban farming and practices increasing landscape diversity) defined below.

**Agroecology** is a broadly used term that describes the concept of applying ecological principles to agricultural practice. More recently, it has also evolved to describe a general principled approach that is inclusive of aspects of its own implementation. The approach has evolved to cover socioeconomic, cultural and political aspects. Wezel et.al., (2020) provide a detailed review of the evolution of agroecology as a science, a set of practices, and a social movement that has led to the dynamic and transdisciplinary nature of the concept. HPLE (2019) identifies 13 consolidated principles (recycling, input reduction, soil health, animal health, biodiversity, synergy, economic diversification, co-creation of knowledge, social values and diets, fairness, connectivity) that underpin the agroecology approach.

For disambiguation, this section refers to the principled approach of agroecology, as described by HPLE (2019) and Wezel et.al., (2020), as a single holistic biodiversity-friendly “practice” or approach. We acknowledge that agroecology is a very broad concept and that most of the following specific practices discussed here are also considered agroecological practices.

**Organic farming** is a production system that avoids or largely excludes the use of synthetically compounded fertilisers, pesticides, growth regulators and livestock feed additives. To the maximum extent feasible, crop protection in organic farming systems relies on crop rotations, crop residues, animal manures, green manures, off-farm organic wastes and aspects of biological pest control (EC, 2024g).

**Agroforestry** is a land-use system and technology where woody perennials (trees, shrubs, etc.) are deliberately used on the same land management unit as agricultural crops and/or animals (EC, 2024g). Agroforestry can be considered as a farming approach, a “farm practice”, a group of farm practices, or a “farming system”. Agroforestry can occur at a variety of spatial scales (e.g., field or woodlot, farm, watershed) in different agricultural systems. When properly applied, agroforestry can potentially improve livelihoods through enhanced health and nutrition, increased economic growth, and strengthened environmental resilience and ecosystem sustainability (Castle et al., 2022). Agroforestry systems also yield proven strategies for long-term carbon sequestration, soil enrichment, biodiversity conservation, and air- and water-quality improvements, benefiting both the landowners and society (Petruzzello and Gold, 2016).

**Extensive grazing** is the predominant form of land use on at least a quarter of the world’s land surface (FAO, 1991), in which livestock are raised on food that comes mainly from rangelands. The term livestock includes domesticated animals such as cattle, sheep, goats, camels, horses, llamas, alpacas, etc. Extensive grazing differs from intensive grazing, where animal feed comes mainly from artificial, seeded pastures and not from unmanaged rangeland (FAO, 1991).

**Urban farming** is the practice of cultivating crops, livestock, or types of food in an urban environment. In recent years, urban farming has become a hot topic in the fields of agricultural sustainability and social justice (Yuan et al., 2022). Urban and peri-urban agriculture can be defined as the production of food and other outputs and related processes, taking place on land and other spaces within cities and surrounding regions (FAO, 2022).

Urban landscapes are typically highly simplified, intensively developed ecosystems with low levels of native biodiversity. Urban farming may contribute to biodiversity if it is incorporated e.g. by creating/restoring green spaces in cities.

**Increasing landscape diversity in agricultural landscapes:** A landscape is "a mosaic of heterogeneous landforms, vegetation types, and land uses" (Urban et al., 1987). Therefore, assemblages of different ecosystems create landscapes on Earth. Although there is no standard definition of the size of a landscape, they are usually in the hundreds or thousands of square kilometres. Species composition and population viability are often affected by the structure of the landscape; for example, the size, shape, and connectivity of individual patches of ecosystems within the landscape (Noss, 1990). Conservation management should be directed at whole landscapes to ensure the survival of species that range widely across different ecosystems (species of plants that have widely dispersed pollen and seeds) (Hunter Jr and Gibbs, 2006).

Landscape is one of the most precious assets contributing to Europe's cultural identity. As landscape is determined to a large extent by land use, the identification of land use changes, especially through changes in the land cover in the European Union, provides clues to the drivers of the transitions that the landscape is currently going through. Land use shapes our environment in positive and negative ways. Most scenarios for global economic and societal development show a strong territorial

polarisation of land functions in Europe soon. Although multifunctional land use is widely seen as a promising solution for the liveability of the European landscape and for balancing the provision of ecosystem services, there are not many proactive policy alternatives to set the boundaries for such use and at the same time address environmental management that invariably requires system- and place-based adaptation (Buckwell et al., 2017).

### *3.1.3.1 Consumption patterns*

Food consumption patterns are changing in response to shifting lifestyles and preferences across the world. Current patterns in food consumption in Europe are considered unsustainable (Holden et al., 2018). The EU shows higher consumption levels in terms of calories, protein, and fat intake compared to the world average, exceeding the latter by 19%, 25%, and 75%, respectively (FAOSTAT, 2024). In particular, EU consumption levels for animal fats and milk are 4 and 2.5 times higher compared to the global average consumption with the per capita supply of both increasing by 13% from 2010 to 2021 (FAOSTAT, 2024).

In this context, dietary choices play a key role in affecting environmental burdens (Danso-Abbeam et al., 2021). Consumption patterns have enormous implications for biodiversity; as described in section 2.2.2, a more meat-based diet requires more resources (land, water, energy) than a plant-based diet, thus affecting more habitat area, therefore there is an urgent need for more sustainable eating patterns. Sustainable diets are meant as those diets with low environmental impacts that contribute to food and nutrition security and to healthy life for present and future generations (Danso-Abbeam et al., 2021). Among animal-based diets, red meat has a higher environmental impact. Some diets characterized by a low intake of meat and saturated fat are considered healthier and more sustainable. For instance, the Mediterranean diet has been appreciated to have a lower environmental impact, mainly as it includes the consumption of more plant-derived products and fewer animal products, with respect to other current dietary patterns (Danso-Abbeam et al., 2021). Some alternative and popular self-imposed dietary patterns such as vegetarian and vegan diets are also considered more sustainable as all meat and animal products are excluded (Kraak and Aschemann-Witzel, 2024; Clark et al., 2019; Poore and Nemecek, 2018).

### *3.1.3.2 Biodiversity indicators*

The Driver-Pressure-State-Impact-Response (DPSIR) Framework (EEA, 1999; OECD, 2003) provides a structure within which to present the indicators needed to inform policy makers on environmental issues and the effect of the political choices in the short/long term. The DPSIR framework assumes a chain of causal links starting with ‘driving forces’ (economic sectors, human activities) through ‘pressures’ (emissions, waste) to ‘states’ (physical, chemical and biological) and ‘impacts’ on ecosystems, human health and functions, eventually leading to political ‘responses’ (prioritization, target setting, indicators). Establishing a DPSIR framework for a particular setting is a complex task as all the various cause-effect relationships must be carefully described and environmental changes can rarely be attributed to a single cause. Here, five biodiversity indicators such as species abundance, biodiversity footprint, land use, number of species and richness are selected.

Biodiversity indicators help us measure and monitor the pressures or threats, such as trends in land and water use, habitat loss or invasive species and the state of species and ecosystems, such as the health of species or integrity of ecosystems. Moreover, biodiversity indicators provide information about relevant policy responses, such as the protection of important biodiversity areas and the benefits they provide to people, such as the ecosystem services that freshwater provides. Fine-scale indicators may be developed to inform local decisions on the ground, such as determining the degree to which restoration or management practices are working. In practice, biodiversity indicators enable the quantifiable assessment and comparison of biodiversity in space and time, and they are essential for the design and implementation of effective policy.

**Species abundance** represents the sum total of individuals from a given species within a given area (Dubey, 2024). Species are considered abundant when they have a high population relative to the size of the area they inhabit. It can also include other measures of performance for plants, animals, or other forms of life in a given area, including number of breeding pairs, population density, and even biomass. In ecology, species abundance can serve as a tool to measure biological diversity and inform conservation efforts (Dubey, 2024).

**Biodiversity footprint** measures the integral impact of a commodity, company, person or community on global biodiversity, expressed in terms of biodiversity change, as a result of production and consumption of particular goods and services (Marques et al., 2017). This involves measuring a number of impact factors, including habitat destruction, species loss, ecosystem disruption, and other relevant factors. Each human activity's contribution to the overall biodiversity footprint is assessed by this important indicator. See section 2.2.1 for a detailed description and summary of findings on biodiversity footprint studies.

**Land use** is the total of arrangements, activities and inputs that people undertake in a certain land type. Land use is not only one of the most important drivers of climate change, but it also affects biodiversity, the functioning of ecosystems, and the services they provide (Koellner et al., 2013). Land use as a biodiversity indicator quantifies regional species loss due to land occupation and transformation by humans. The **number of taxonomic units** defined by Sokal (1963) (usually the number of species, but also, for example, the number of genera or families) is the most common biodiversity indicator used by scientists. Counting the species, genera or families is therefore a typical and informative activity used by numerous scientists.

The species **richness indicator** assesses the total number of unique species within a given biological community, ecosystem, biome, or other defined area (Feistner, 2004). Species richness is often used to compare the biodiversity of different biological communities, compare the number of species within a particular taxonomic grouping (such as birds or mammals) at different locations, and monitor changes in a particular biological community over time.

### 3.1.3.3 *Links with additional aspects*

Here we describe the key aspects we identified, their significance and linkages to the contents of the selected and analysed studies. Among the studies linked with our keywords and within the context of our scopes and objectives, studies including five additional aspects of such as land use change, carbon stock, pollinators, economic aspects and EU policies were identified. These aspects were selected because of their synergic relevance in the context of biodiversity.

**Land use change** refers to two major processes. The first process is a change in land cover associated with the expansion or contraction of the area of land used for different purposes (e.g., pasture, cropland, urban). The second process is a change in the type of management on existing land cover (e.g., changes in irrigation, fertilizer use, crop type, harvesting practices, or impermeable surfaces). Land use change related to management can occur without changing the extent of different land covers. The degradation and conversion of forests to alternative land uses, such as agriculture, is one of the leading causes of biodiversity loss, especially in the tropics. Furthermore, the impacts of land-use change are likely to be exacerbated by the looming climate crisis.

**Carbon stock** is defined as a natural reservoir that accumulates and stores some carbon-containing chemical compound for an indefinite period. For instance, forest carbon stock is the amount of carbon that has been sequestered from the atmosphere and is now stored within the forest ecosystem, mainly within living biomass and soil, and to a lesser extent also in dead wood and litter. This is directly linked with the carbon footprint which is an indicator belonging to the footprint family and measures the amount of carbon dioxide emissions associated with all the activities of an entity including direct and indirect emissions. Protecting aboveground carbon stocks in tropical forests is essential for mitigating global climate change and is assumed to simultaneously conserve biodiversity.

**Pollinators** help wild plants reproduce, and help crops to produce seeds, fruit and nuts, and other tasty foods for us to eat. In Europe, pollinators are mostly insects, such as bees, hoverflies, butterflies, moths, wasps, beetles and other fly species. Some pollinators, such as the honeybee, are domesticated. Having a species-rich community of pollinators is essential to keep ecosystems healthy and resilient so it is important that we understand and protect pollinators. A good example of pollinators are butterflies and birds that play an important role in ecosystems and provide a range of ecosystem services, including pollination. Butterflies are sensitive to environmental change and are a good indicator of the health of the environment. Fluctuations in numbers between years are typical features of butterfly populations. The assessment of change is therefore made on an analysis of the underlying trend. Birds are sensitive to environmental pressures and their populations can reflect changes in the health of the environment. Indeed, the Farmland Bird Index is an environmental context indicator that represents the state of health of agricultural environments, aggregating information deriving from individual indices, such as population trends of bird species typical of agricultural environments and open mountain environments.

Biodiversity underpins all economic activities and human well-being. It provides critical life-supporting ecosystem services, including the provision of food and clean water, but also largely invisible services such as flood protection, nutrient cycling, water filtration and pollination. Yet humanity is destroying natural capital at an unprecedented rate, posing significant but often



overlooked risks to the economy, the financial sector and the well-being of current and future generations (OECD, 2021).

The European Commission has been putting in place a new European biodiversity governance framework (EC, 2020c) This will help map obligations and commitments and set out a roadmap to guide their implementation. In particular, the EU's biodiversity strategy for 2030 is a comprehensive, ambitious and long-term plan to protect nature and reverse the degradation of ecosystems. The strategy aims to put Europe's biodiversity on a path to recovery by 2030 and contains specific actions and commitments.

#### *3.1.3.4 Consumption patterns and diets in the EU*

Food consumption patterns across countries and cultures in the EU are largely variable in terms of per capita meat consumption and energy intake (Holden et al., 2018). For instance, the Mediterranean diet, compared to the average EU diet, is generally considered to be beneficial to both health and the environment by contributing to health maintenance and helping mitigate biodiversity degradation (Mattas et al., 2023). Mediterranean countries record the highest consumption in terms of fruit, vegetable and pulses, followed by northern central European countries, whereas eastern European countries record the lowest levels (SAPEA, 2023). Broadly speaking, a north/south divide between European countries can be observed, reflecting territorial and economic differences, as well as cultural and historical reasons. National food-based dietary guidelines in EU countries all suggest that a shift towards a healthy diet requires predominantly plant-based foods, rich in vegetables, fruits, whole grains, pulses, and fish, with only moderate low-fat dairy products and limited red and processed meat, salt, added sugar and high-fat animal products (SAPEA, 2023).

Food consumption patterns therefore represent important drivers of environmental burdens including biodiversity loss. The variety of diets we find in Europe makes this aspect interesting from a food sustainability point of view.

Dietary guidelines have to combine the health-nutrition aspect with the environmental aspect to provide food for all in a healthy and sustainable way. In some countries, environmental sustainability is being incorporated into dietary recommendations, for example, Nordic nutrition recommendations have been updated for sustainability in 2023 (Blomhoff et al., 2023). The Finnish national nutrition recommendations included some notions of sustainability already before. The recommendations steer also the school meals provision in Finland, where a lot of sustainability actions are being taken forward at the moment. The law for basic education secures a right to wholesome meal for all pupils during the school day and is a key lever for sustainable diets as well.

In terms of environmental sustainability, meat and egg consumption covers on average 56% (49%–64% across different EU countries), while dairy consumption covers about 27% (16%–36%) of the total emissions linked with the EU food supply, whereas the consumption of grains covers just 4% of EU food supply-related greenhouse gas emissions (Sandström et al., 2018). Due to enteric fermentation, manure management, and feed consumption, protein from beef, pork, fish, chicken and egg generate about twenty, four, three, three, and two times higher greenhouse gas emissions per kilogram than plant-based protein sources like cereals and legumes, respectively (Poore and

Nemecek, 2018), indicating that the consumption of plant-based proteins would be preferable in terms of GHG emission.

Local food consumption is becoming prevalent among consumers, who perceive it as more environmentally sustainable and healthier, despite its minor direct positive environmental benefit due to the marginal role played by transport to overall emissions from the food system (6% for EU diets) (SAPEA, 2023).

Current EU consumption patterns (Eurostat, 2023b) indicate that around two-thirds of the EU population eat at least one portion of fruit and vegetables on a daily basis, Romania being the country with the lowest daily consumption and Belgium the one with the highest one. It is remarkable that EU citizens with higher educational levels (tertiary education) were most likely to eat at least five portions of fruit and vegetables daily. Income seems to affect this behaviour too since the share of people eating at least five portions of fruit and vegetables per day was greater among EU citizens with higher income. There is also a well-defined difference in fruit consumption between men and women, with women having higher levels of consumption in all EU countries. The consumption of vegetables shows similar trends, but with an overall larger consumption compared to fruits, with Romanian citizens consuming the least amounts, and Belgium the most. Just over half (50.6 %) of EU population ate vegetables at least once a day. The sex difference persists, being women more likely to eat vegetables at least once a day compared to men. Higher income seems to play the same role in boosting vegetable consumption as in boosting fruit consumption. Fruit or vegetable juice was less likely to be consumed among EU citizens since less than 10% of them drank juice at least once a day, however without significant differences between sexes. Instead, income seems to be affecting juice consumption as for fruit and vegetables.

At the EU level, according to the FAO food balance sheet, dairy products, vegetables, and cereals are the most consumed food items, on a mass basis in 2021 (Figure 5). However, by considering the different nutritional values of the various food items it is possible to gain further insights. Comparing them in terms of energy content it is evident that vegetal products cover most of the calories supplied to the average EU citizen in 2021, the energy intake from animal products is lower, making up 58% of the total. Within plant-based food, most energy intake is from cereals (Figure 6). Considering fat supply, animal products and vegetal products made up 53 and 47 percent of total calories respectively in 2021 (Figure 7). Ultimately, most of the protein supply was sourced from animal products, primarily from meat, with 37% coming from vegetal products (Figure 8).

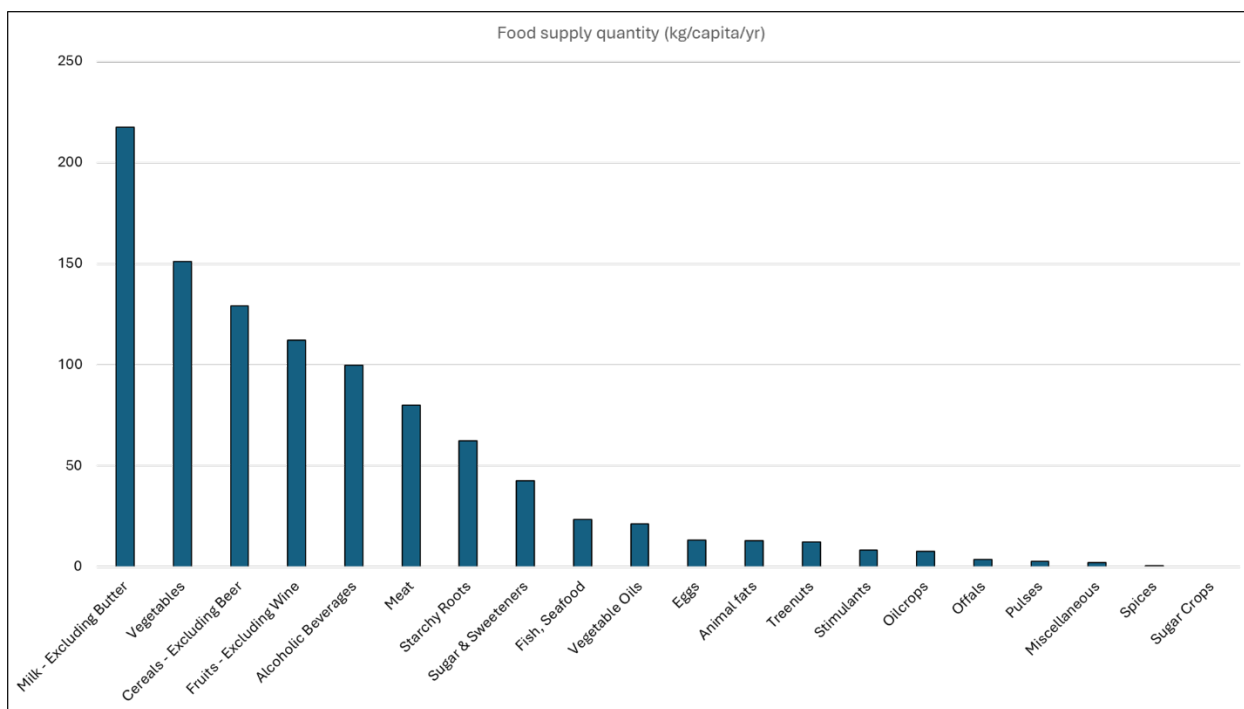


Figure 5. EU food supply by aggregate food item category in 2021. Values are expressed in terms of kg/capita per year. Data sourced from FAOSTAT.

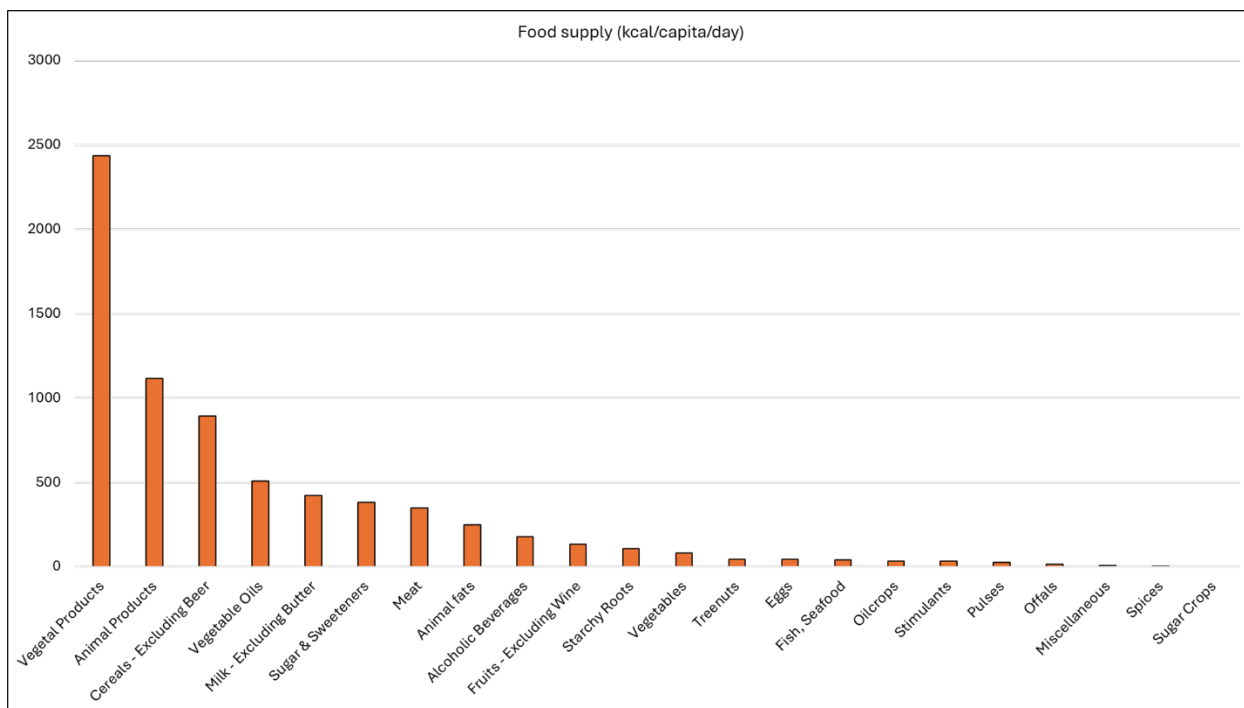


Figure 6. EU food supply by aggregate food item category in terms of energy in 2021. Values are expressed in terms of kcal/capita per day. Data sourced from FAOSTAT.

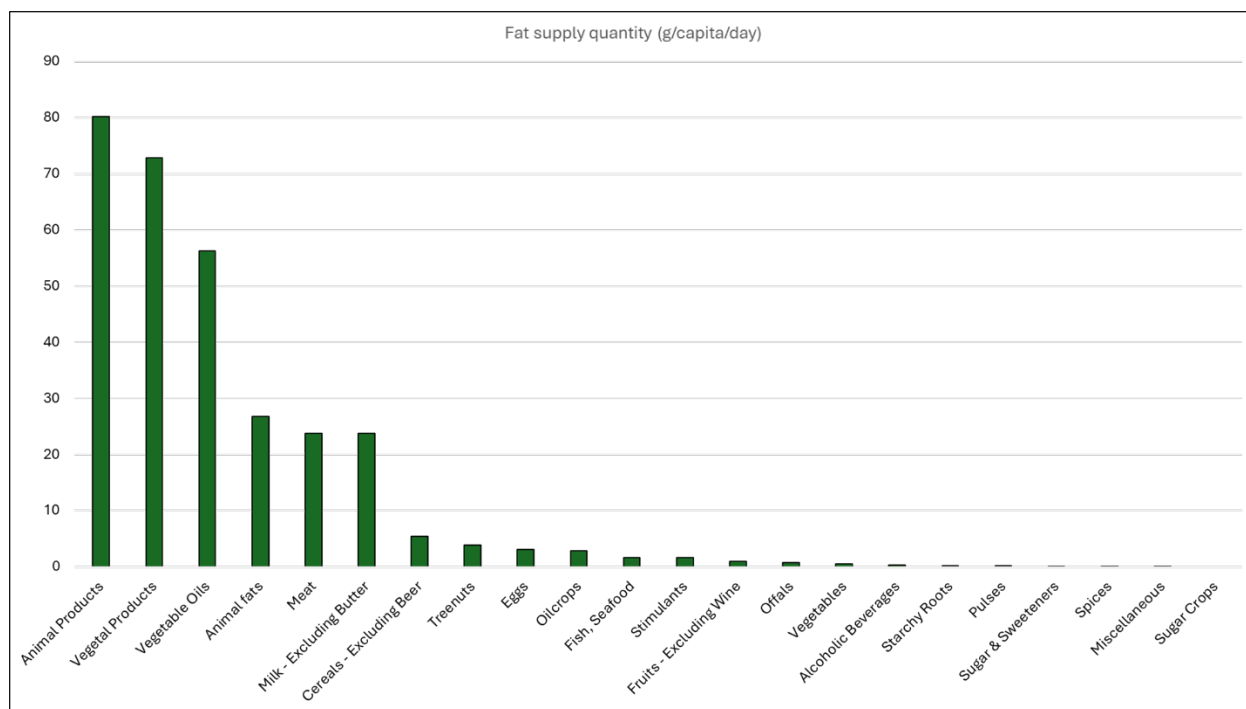


Figure 7. EU food supply by aggregate food item category in terms of fat in 2021. Values are expressed in terms of grams of fat/capita per day. Data sourced from FAOSTAT.

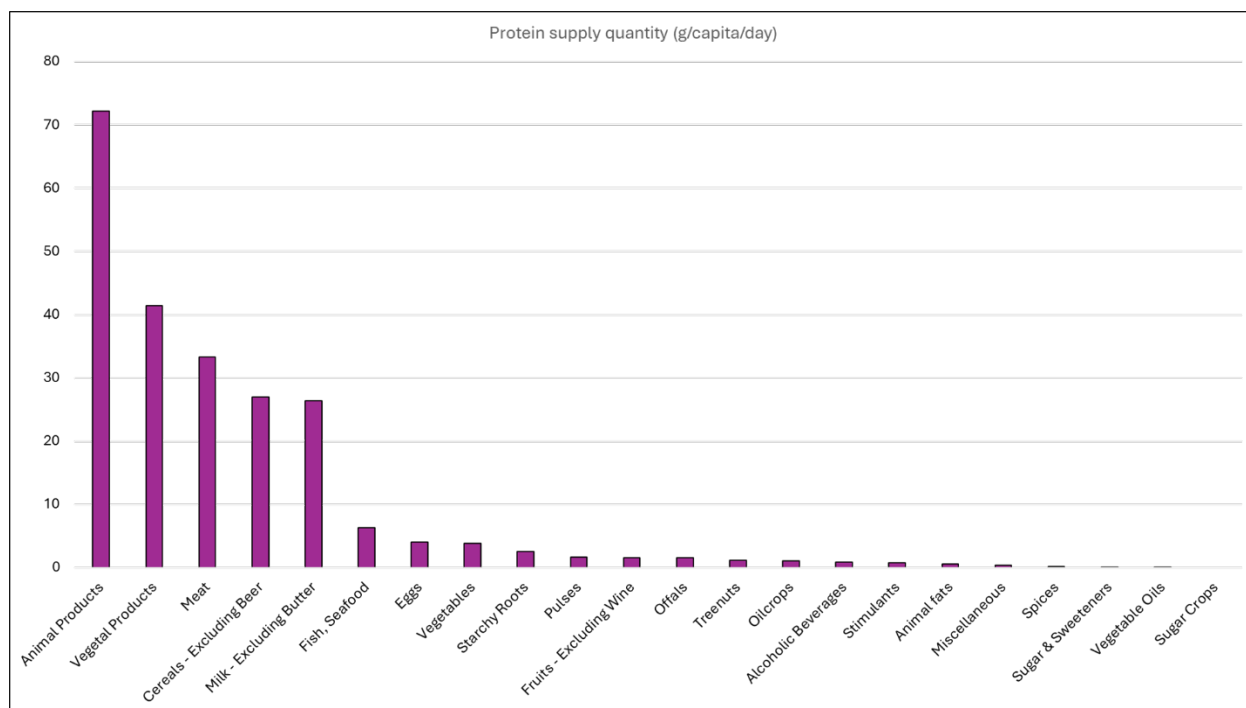


Figure 8. EU food supply by aggregate food item category in terms of protein in 2021. Values are expressed in terms of grams of protein/capita per day. Data sourced from FAOSTAT

The proportions of marketing channels for organic food sales vary remarkably across the different EU countries (Figure 9). Generally, the preferred channel is general retailers. Exceptions are Denmark and Czechia, where direct marketing and other channels represent the top choice, respectively. In many cases the sales via direct marketing are negligible, as well as the sales via other channels, while specialised retailers generally cover a significant part of the sales, except for Denmark and the Czech Republic.

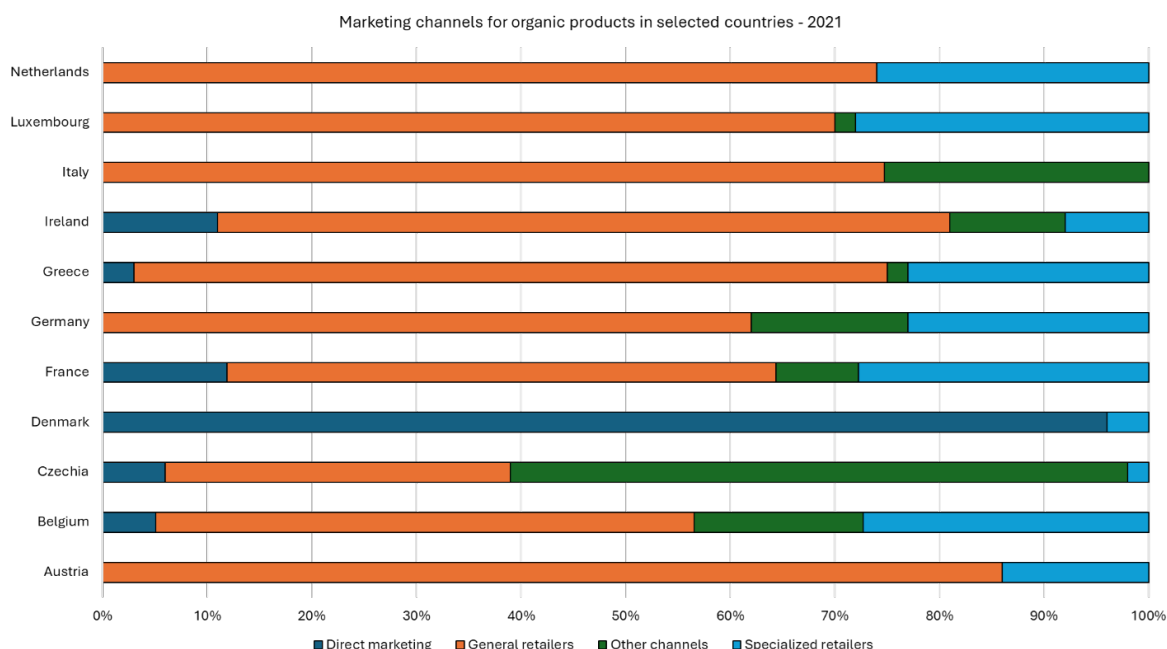


Figure 9. Marketing channels for organic products in selected countries in 2021. Adapted from Willer et al. (2023).

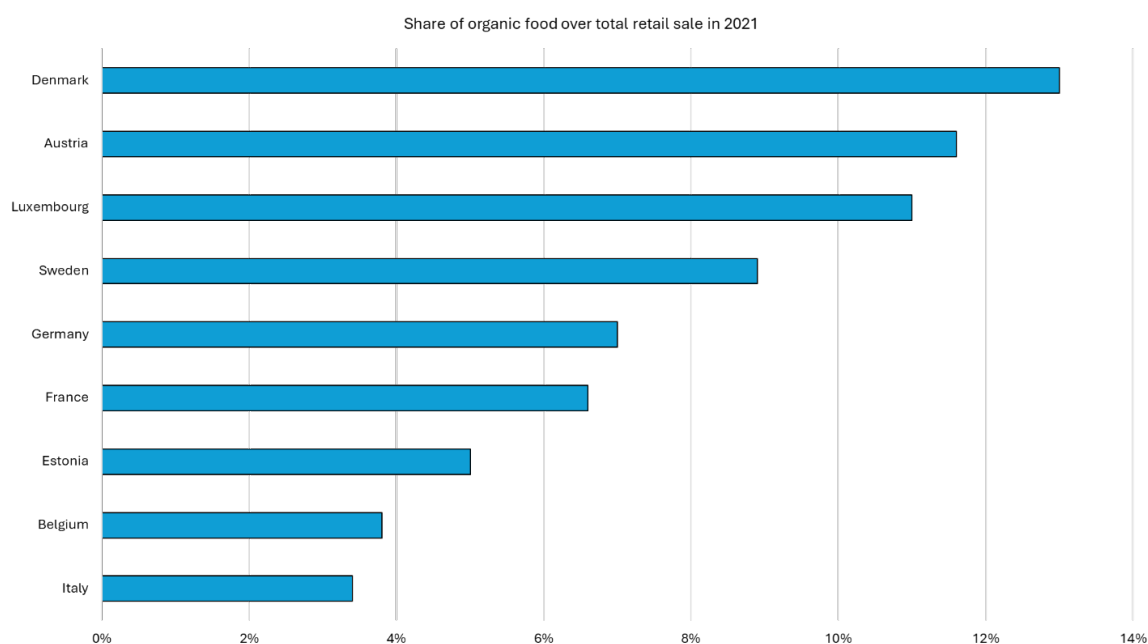


Figure 10. The share of organic food sales over the total food retail sales in selected countries in 2021. Adapted from Willer et al. (2023).

The share of organic food sales over the total food retail sales in 2021 varied across different EU countries. It spanned between 3.4% in Italy and 13% in Denmark (Figure 10).

## 3.2 Results

### 3.2.1 Biodiversity impacts of the selected environmentally sustainable agricultural systems and practices

#### **Agroecology:**

Biodiversity is addressed by the principles of agroecology both directly and indirectly. The fifth of thirteen agroecology principles (HLPE, 2019) directly aims to “maintain and enhance diversity of species, functional diversity and genetic resources and thereby maintain overall agroecosystem biodiversity in time and space at field, farm and landscape scales.”

By fostering beneficial biological interactions and synergies among different components of the agroecosystem (such as crops, animals, trees, soil, and water), agroecology enhances functional biodiversity. This includes practices like intercropping, agroforestry, and the integration of livestock, which create more complex and resilient ecosystems.

Among the 13 principles, five of them are related to environmental sustainability in the areas of resource use and resilience, which indirectly provide positive impacts on biodiversity. These include the key principles of recycling, input reduction, soil health and animal health, and synergy. These principles aim to reduce the consumption pressure on biodiversity and improve ecosystem quality in support of biodiversity.

The remaining principles address economic or social aspects around equity and responsibility. These include the principles of economic diversity, co-creation of knowledge, social values and diets, fairness, connectivity, land and natural resource governance, and participation. These principles provide social and governance support to enable the uptake of biodiversity-friendly agroecological practices.

As a practice, the approach of agroecology is inclusive of and extends beyond the specific practices discussed below, for which we assess the quantitative impacts identified in the literature.

#### **Organic farming:**

Two review papers (Hole et al., 2005; Winqvist et al., 2012) showed positive effects of organic farming on biodiversity. A study conducted at the Lodge Nature Reserve (U.K.) revealed that the biodiversity benefits of organic management are likely to accrue through the provision of a greater quantity/quality of both crop and non-crop habitat than on conventional farms (Hole et al., 2005). The study pinpointed several aspects of the holistic approach that require further research. It also emphasized three key management practices: limiting or eliminating the use of chemical pesticides and inorganic fertilisers, managing non-cropped habitats sympathetically, and maintaining mixed farming systems. These practices, while not exclusive to organic farming, are fundamental to it and offer significant benefits for farmland wildlife. The review was timely given policy developments in



the UK and Europe promoting organic agriculture as one of the potential solutions to the continuing loss of biodiversity in agricultural landscapes.

Winqvist et al. (2012) investigated the effects of organic farming on the arable biodiversity of plants, arthropods, soil biota, birds, and mammals in a review paper. The ecosystem services of pollination, biological control, seed predation, and decomposition were also included. Their review concluded that organic farming positively impacts biodiversity for common species while noting that these effects are often specific to certain species and depend on traits or context. Further, they suggest that while organic farming is not the sole solution to the issues posed by modern agriculture, it does provide biodiversity benefits compared to conventional farming.

Three other studies included in the literature review focused on the impact of organic farming on biodiversity in Europe.

One of the studies revealed contrasting effects of organic and conventional farming on biodiversity across multiple trophic groups (Ostandie et al., 2021). Among the seven groups studied, organic farming at the field scale enhanced the abundance of arthropods, specifically springtails (+ 31.6%) and spiders (+ 84%). Conversely, it had detrimental effects on pollinator abundance and soil microbial biomass. The study was not able to determine clear links between specific organic farming practices and biodiversity conservation performance.

The second study focused on the assessment of organic and minimal agricultural practices on different soil cultivation types, including conventional, minimum till, mulch, no-till and organic farming (Houšková et al., 2021). Among the different cultivation and agricultural practices, they assessed the quantity and health of earthworms and soil microbial activity to evaluate biodiversity conditions. Additionally, they examined basic soil properties and structure to determine the impact of the tested agricultural practices on the soil environment. The tested practices were found to improve soil moisture content, biodiversity, and soil structure stability. In a third case study in Europe, Szilágyi et al. (2021) examined the impact of organic and permaculture farming practices compared to conventional systems on the diversity of pollinator species. They found that the abundance of specific pollinator groups was highest for permaculture farms while highlighting the importance of a broader range of additional farm management considerations, such as plant protection, flower resources, active biodiversity management, and social factors such attitude of the farmers towards protection of pollinators.

Concerning studies outside Europe, Tuck et al. (2014) conducted a hierarchical meta-analysis of 94 studies mostly located in northern America that compared biodiversity under organic and conventional farming methods. Among the functional groups, which included producers or plants, herbivores, pollinators, predators, soil-living decomposers and others, they found that organic farming increased species richness by an average of 30%.

More broadly, a summary of global meta-analysis/synthesis studies from 2005-2010 (EC, 2024g) found a significant positive effect of organic cropping systems compared to conventional cropping systems on biodiversity in 6 of 8 meta-analyses. Among these studies, addition of compost, crop field size, crop type, diversity of cover crops, experiment scale, herbicide application, landscape structure and heterogeneity, organism group, pest management strategy, proportion of arable land

in the surrounding landscape, and taxon were the reported factors with a significant effect on size and direction of biodiversity change.

### **Practices increasing landscape diversity:**

Tscharntke et al. (2021) highlighted the key role of landscape-level species pools and suggested that landscape-level changes provide greater benefits than locally incentivised management practices. They provided evidence supporting the restoration of seminatural habitats at the landscape level and also suggested improving landscape heterogeneity through small and diversified crop fields. Similarly, Estrada-Carmona et al. (2022) performed a comprehensive meta-analysis of complex agricultural landscapes and biodiversity, and found that complex landscapes supported more biodiversity, thus concluding that synergistic management of agriculture, applied at the landscape level, is critical to biodiversity conservation and sustainable production. In a global analysis, Lichtenberg et al. (2017) quantified the effects of organic farming and plant diversification on species abundance, local biodiversity, and regional biodiversity. They found that organic farming and increased plant diversity increased arthropod abundance, particularly for rare taxa. Species richness and abundance were found to increase at both local and regional scales. While farms embedded in complex landscapes were found to have greater increases in species richness. Organic farming and in-field plant diversification were found to have the strongest effects on beneficial groups such as pollinators and pest-predators, suggesting double benefits to these management schemes in the form of improved ecosystem service providers without the proliferation of pest populations. Belfrage et al. (2005) argued that the consideration of organic agriculture's effect on biodiversity should include factors affected by farm size. Indeed, they compared the diversity and abundance of birds, and abundance of butterflies, bumblebees and herbaceous plants between six small farms (<52 ha arable land) and six large farms (>135 ha arable land) in Roslagen in southeastern Sweden (two of the large and four of the small farms were organic), finding different results. More than twice as many bird species, butterflies, and herbaceous plant species, and five times more bumblebees were found on the small compared to the large farms. The largest differences were found between small organic and large conventional farms. Differences were also noted between small and large organic farms: 56% more bird species were found on small organic than on large organic farms, although none of the farms used any pesticides. The authors identified that smaller farms had greater crop diversity per area than larger farms, both organic and conventional. They argued that smaller field sizes for each crop created greater landscape heterogeneity, which in turn resulted in greater biodiversity.

### **Agroforestry:**

Agroforestry has been proposed as a sustainable agricultural approach, conserving biodiversity and enhancing ecosystem service provision while not compromising productivity. Agroforestry systems can be distinguished between traditional silvopastoral agroforestry systems that maintain semi-natural habitats (dehesas, montado, wooded pastures, etc) and other types of agroforestry. Agroforestry can yield a variety of food products such as almonds, walnuts, chestnuts, corn, wheat, rice, tomatoes, peppers, squash, basil, mint and mushrooms grown in a wooded area. Honey produced from bees that pollinate trees and crops in agroforestry systems is another important food product potentially yielded by agroforestry.

A meta-analysis by Torralba et al. (2016) revealed an overall positive effect of agroforestry (effect size = 0.454,  $p < 0.01$ ) over conventional agriculture and forestry, based on 365 comparisons from 53

publications. They concluded that agroforestry can enhance biodiversity and ecosystem service provision relative to conventional agriculture and forestry.

The potential of agroforestry is particularly relevant in the European context, where agri-environment interventions are often addressed at a farm-, rather than at a catchment or landscape-scale (Plieninger et al., 2015).

Santos et al. (2022) emphasized the crucial role of agroforestry in boosting biodiversity within agricultural landscapes, advocating for its prioritisation in European agri-environmental funding schemes. The study tested the habitat amount hypothesis, which posits that the total habitat area in a landscape determines its species richness. This hypothesis inherently considers the isolation and connectivity of habitat patches and the complexity of landscapes, which are key factors in predicting biodiversity.

Applying the habitat amount hypothesis to virtual agricultural landscapes, the study identified general mechanisms through which agroforestry systems enhance biodiversity, significantly increasing both functional diversity and overall biodiversity. However, the authors noted that these normalised values should be adjusted to reflect real-world conditions, as factors such as crop type, tree species, and agroecological management practices can influence outcomes.

This study reinforced the idea that agroforestry is critical to halt biodiversity loss in agricultural landscapes and, despite the limitations inherent to a preliminary demonstration, the methodology developed provides a starting point to anticipate the changes in landscape biodiversity induced by land use change, guiding strategies to integrate crop production with biodiversity conservation.

Allan et al. (2014) tested the effects of land-use intensity (LUI) which they measured as the combined intensity of fertilisation, grazing, and mowing on biodiversity. Highlighting the complexity of practical measurements of biodiversity, they introduced the concept of multidiversity, a unique measure of whole-ecosystem biodiversity. The concept was applied to synthesize a measure of biodiversity across 49 taxonomic groups of plants, animals, fungi, and bacteria from 150 grasslands. Assessing the multidiversity of 150 grassland landscapes, they found that multidiversity declined with increasing LUI among grasslands, particularly for rarer species and aboveground organisms, while the effects on common species and belowground biodiversity were less pronounced. Interannual variation in land-use intensity was found to be an important factor that was positively correlated with overall multidiversity. In addition to decreasing overall land-use intensity, the study recommended varying land-use intensity across years as a complementary strategy to increase biodiversity.

Traditional orchards are not agroforestry systems in a strict sense but can be considered agroforestry according to definitions with a broader scope. Traditional orchards can be beneficial for biodiversity as well. Cohen et al. (2023) verified the extent to which biodiversity in Mediterranean orchards, specifically olive groves, depends on past land use, similar to some wooded ecosystems, as opposed to current conditions only, which is the case for farmlands. Researchers analysed land-use changes over two centuries in 67 olive plots in the South of France using historical records and aerial photographs. They found that plant community parameters and species composition are significantly influenced by past land use. Orchards initially covered with natural vegetation and recently restored had the highest species diversity, while long-cultivated, abandoned, and burnt orchards had lower species counts due to intensive management and plant competition. The study

suggested that the restoration of olive groves should be encouraged, as they provide important ecosystem services and represent a promising prospect for Europe as a whole. More broadly, they recommended that ecosystem services should be given greater consideration in public policy.

### **Functional agrobiodiversity measures:**

Maskell et al. (2023) focused on ten functional agrobiodiversity (FAB) measures in Europe, which balance food production with minimised impacts on nature. These included conservation tillage techniques; mixed crops and crop rotations, including sward diversity (herbal leys); cover and catch crops, including legumes; organic matter input; modified manure quality and diversity; agroforestry; hedgerow management; field margin management; and reduction in the use of plant protection products. The review found that the measures generally resulted in positive outcomes, with a number of mixed effects. Outcomes for biodiversity include maintaining high populations of soil-ameliorate fauna and insect pest predators and small mammals. They found a general decrease in fertilisers and pesticides, improving above- and below-ground biodiversity, soil and water quality as well as water conservation. However, evidence was also conflicting for some of the measures, with both positive and negative effects observed on yield, soil organic carbon and greenhouse gas emissions. The rapid assessment was not able to derive conclusive results in these areas in part due to a lack of suitable evidence at the requisite spatial and temporal scale, for example, lag times and uncertainty between cause and effect of treatment and change in soil organic carbon mean that longer observation times would be required for more reliable evidence.

Bianchi et al. (2013) discussed opportunities and limitations for FAB in Europe, highlighting four key limitations: translation of knowledge into management practice, uncertainty in the effectiveness of FAB measures for crop yields, crop quality, profitability, and reduction of agrochemical inputs, lack of financial systems to allow fair accounting of investments and public benefits, and lack of coordination of local actors to facilitate implementation of FAB at proper spatial scales. The authors identified linkages and alignment with five policy developments in Europe. The Thematic Programme on Agricultural Biodiversity within the United Nations Convention on Biological Diversity underscores the importance of agriculture as an essential provider of ecosystem services while also driving biodiversity loss, thus providing an opportunity to promote FAB practices. The EU common Agricultural policy also presents an opportunity for FAB to be a method to preserve long-term productivity and ecosystems and provide benefits to farmers who practice them. EU Directive 2009/128/EC aims to minimise health and environmental risks from pesticides, which presents an opportunity to apply FAB methods such as nectar-rich flower strips, planting trap crops or conserving non-crop habitats as a natural replacement for pesticides. The EU Water Framework Directive 2000/60/EC requires water quality standards to which FAB can be used to support. Field margins, for example, can act as a buffer to reduce pesticide drift and nutrient flows into surface water. Lastly, the International Treaty on Plant Resource for Food and Agriculture and the European Seed Legislation Directive 2009/145/EC aimed to provide food security through the maintenance of local plant breeds and varieties. Such support of local plants in agriculture is well aligned with FAB.

### **Extensive grazing:**

Adapting grazing management (livestock species, method and intensity) to the carrying capacity of the managed ecosystems and to the local biodiversity conservation or restoration needs is important for the development of sustainable grazing systems.

Based on a meta-analysis of 116 studies in temperate grasslands globally, the response of species richness of plants, arthropods, and microbes to grazing intensity agreed with the intermediate disturbance hypothesis in grasslands; species richness increased with light and moderate grazing intensities, while it decreased at heavy intensity (Wang and Tang, 2019). In addition, plant and microbe species richness increased with light and moderate grazing and declined with heavy grazing intensity. The species richness of arthropods (divided into pollinators and carabids) monotonously declined with increasing grazing intensity. Importantly, structural equation modelling showed that grazing resulted in enhanced plant species richness mainly through its negative effects on plant biomass. Grazing had negative effects on plant coverage and arthropod abundance, so arthropod species richness declined with increased grazing intensity. Moreover, increased grazing intensity caused an increase in soil pH, a decrease in soil moisture, and then a decrease in microbe species richness. These findings confirmed that different taxa exhibit diverse responses to changes in grazing intensity, and the way that grazing intensity affects diversity also varied with different taxa.

Two case studies (Pellaton et al., 2023; Wallis De Vries et al., 2007) looking at European countries also concluded that disturbance by low-level grazing was beneficial to biodiversity.

Wallis De Vries et al., (2007) tested the hypothesis that extensification of grazing management enhances fauna/wild animals' diversity. Over a period of three years, moderate and lenient grazing intensities were tested with commercial and traditional breeds across study sites in the UK, France, Germany and Italy. Animal diversity was recorded for birds, hares, butterflies, grasshoppers, and ground-dwelling arthropods. The study concluded that the effects of livestock breed on biodiversity are negligible and that, in the short term, a lenient grazing intensity is preferable from the viewpoint of butterfly and grasshopper diversity.

Pellaton et al. (2023) evaluated the effect of two grazing intensities and the effects of landscape complexity on plant functional traits in three grassland types in the Great Hungarian Plain. They found shorter-statured and earlier species in intensively grazed pastures and more disturbance-tolerant species with increasing boundary length. For pollination-related plant traits, there were more later flowering species in extensively grazed sites, while flowering duration and flower colour revealed complex relations between landscape complexity and grazing intensity. Generally, local low-intensity grazing (0.5 cattle/ha) supported specialist plant species, and increasing boundary length promotes generalist disturbance-tolerant species with traits beneficial for pollinators. They argued that policies should find a balance supporting extensive grazing according to local management history and preserving large, unfragmented pastures while promoting pollinators via diverse flower resources. In addition, policies should consider that local management and landscape-scale effects can be different in different regions; thus, no one policy will fit all regions and ecosystem types.

### **Urban agriculture:**

Urban agriculture is another type of agricultural system that can be deployed as a strategy to contribute to sustainability. Food supply from urban farming can be of many types, mainly perishable products, especially green vegetables, dairy products, poultry and pigs, mushrooms, ornamental plants, herbs, fish, but also root crops, legumes, nuts, and even cereals (FAOSTAT, 2024).

Urban agriculture is considered to be a major contributor to urban environmental sustainability with a variety of benefits, including the reduction of ghg emissions and urban heat island effects,

mitigation of flood risk and damage, enhancement of biodiversity, promotion of agro-tourism and improvement of agriculture-related urban land allocation (Ebissa et al., 2023). Several studies in the literature look at the impact of urbanisation on biodiversity and the role of urban agriculture in mitigating these effects. Urbanisation exerts pressure on biodiversity (Spotswood et al., 2021), while urban biodiversity comes mainly from urban green infrastructure and certain forms of urban farming can be an element of it (Contesse et al., 2018; Lin et al., 2017) as it can include both flora and fauna as well as edible (dietary) and non-edible (environmental) components (Zasada et al., 2020). However, while green infrastructure such as rooftop greenhouses offer potential benefits for food production, they do not provide opportunities for natural habitat creation and conserving biodiversity (Drottberger et al., 2023).

Results from a recent meta-analysis show that urban agriculture has, overall, a high potential for food production and that urban spaces can be as productive or even more productive than rural environments, with mean crop yields of urban agriculture being similar to or greater than conventional agricultural yields (Payen et al., 2022). While it was found that overall crop yields in various urban spaces were not found to be significantly different, even between indoor and green open-air spaces, results were dependent when looking at specific crop types. In general, however Payen et al. (2022). highlighted how indoor spaces did not impact crop yields differently than green spaces for “cabbages and other brassicas” nor for “vegetables, fresh [not elsewhere specified],” even though crops grown in controlled environments led to significantly higher yields than open-air agriculture for these crop categories. Notable crop and growing system combinations included tomatoes, which produced the highest yields in hydroponic greenhouses (average yields of 18 kg m<sup>-2</sup> cycle<sup>-1</sup> for tomatoes, which was more than thrice higher than tomato yields achieved in urban soil-based, open-air green spaces that were 5.3 kg m<sup>-2</sup> cycle<sup>-1</sup>) and “lettuce and chicory” which produced the highest yields in systems using vertical farming, hydroponic methods and a controlled environment with artificial light.

### 3.2.2 Impact of potential consumption changes on the demand for different agricultural practices

This section looks at the extent to which food consumption changes can support non-conventional agricultural practices. A review of the literature was conducted to answer this question and revealed that there are major gaps and few studies providing a quantitative assessment of this question. Linking consumption changes to the application of more sustainable agricultural practices is not a trivial task. The various agricultural practices considered in this study require a large quantity and variety of data. For instance, these include the physical and economic output for each agricultural practice, ideally at the national level and ideally with a crop-level resolution. Such data is not always available, or systematically collected, or it might be fragmented across countries and different scales.

Rieger et al. (2023) assessed the effect of dietary shift in the EU on production dynamics and on the environmental impact assuming the adoption of the EAT-Lancet dietary recommendations (Willett et al., 2019). Based on their agro-economic model, such a shift would result in decreasing production of animal-based products, and an increase in the production of fruits and vegetables. The economic consequences of the shift would decrease the prices of animal-based products while increasing the



prices of fruits and vegetables. These dietary changes would reduce agricultural GHG emissions. Overall, the agricultural sector could benefit from this type of dietary shift, though the results are mixed at country, regional and farm levels. Areas that are highly specialised in animal farming are likely to lose income, while in areas with higher shares of vegetable and fruit farms income gains are expected. Rööß et al. (2022) assessed the effect of a combined dietary shift (EAT-Lancet diet; Willet et al. 2019) and uptake of agroecological farming practices in the EU. The simultaneous shifts, together with a re-design of the supply chains to favour domestic consumption over export-oriented production, reduced waste generation, and increased productivity would meet all EU food system policy targets including GHG and ammonia emission reduction, lower fertiliser, pesticide and antimicrobial use, higher land areas protected for biodiversity or under organic production. The agroecological practices considered in the study were limited to organic production. Among the farming practices investigated, organic farming is the one receiving the most attention in the EU policies. It is the object of the Farm to Fork Strategy and Biodiversity strategies' targets, for which 25% of EU agricultural land should be organic by 2030, being around 9% in 2020 (EC, 2024g). This political attention, though still waiting to be backed by legal enforcement, implies more thorough and systematic data collection, at least at the national level, across EU countries.

On the other hand, data on other practices (e.g., urban farming, agroecological practices – other than organic farming) is not collected at the national or regional level. The lack of economic and physical output data on the rest of the practices hampers the possibility of understanding their potential deriving from a broader application across the EU.

Accordingly, we here focus on organic farming, as the only available option with sufficient data.

The share of land currently under organic farming is known and varies between the low (e.g., Malta with 0.6%, or Ireland with 1.7%) and high shares (e.g., Estonia with 22.4% and Austria with 25.3%) and the respective targets for 2030 vary (EC, 2023b). The share of organic land dedicated to the cultivation of crops can be derived from data available in Eurostat as the ratio between the organic agricultural production and the total agricultural production. However, this share does not correspond to the share of products (e.g., in tonnes) produced. Such kind of data is less systematically collected compared to land surface data. Furthermore, many data points are confidential and inaccessible via Eurostat, making estimating the market share of single organic products by country of production a hard task. The available data was summarised in Table 3 for crops and Table 4 for animal products for the year 2021. They present the ratio of organic products over the total production by food item or food group and by country (Eurostat, 2024a, 2024b).

Table 3. The share of organic over the total crop production generated in each country in 2021. \*For human consumption exclusively. NA = Not available (e.g., confidential), not significant, or not produced. Some figures are derived from estimated data. Source: Eurostat database (Eurostat, 2024a).

	Cereals	Roots	Oilseeds	Fresh vegetables	Permanent crops*	Citrus fruits	Grapes	Olives
Belgium	3%	NA	2%	4%	2%	NA	NA	NA
Bulgaria	0%	3%	0%	3%	8%	NA	6%	NA
Czechia	1%	0%	0%	1%	4%	NA	4%	NA
Denmark	NA	NA	NA	NA	NA	NA	NA	NA
Germany	NA	NA	NA	11%	NA	NA	NA	NA
Estonia	6%	2%	4%	1%	44%	NA	NA	NA
Ireland	0%	NA	0%	2%	2%	NA	NA	NA
Greece	NA	NA	NA	NA	NA	NA	NA	NA
Spain	1%	0%	1%	3%	7%	7%	8%	4%
France	2%	NA	NA	NA	NA	NA	NA	NA
Croatia	2%	0%	5%	1%	3%	0%	3%	5%
Italy	5%	3%	NA	6%	20%	18%	17%	46%
Cyprus	1%	1%	0%	1%	11%	2%	6%	40%
Latvia	3%	15%	0%	4%	23%	NA	NA	NA
Lithuania	3%	4%	1%	2%	11%	NA	NA	NA
Luxembourg	2%	0%	0%	NA	2%	NA	3%	NA
Hungary	1%	0%	1%	2%	4%	NA	1%	NA
Malta	NA	0%	NA	0%	0%	NA	0%	35%
Netherlands	1%	NA	NA	5%	NA	NA	NA	NA
Austria	NA	NA	NA	NA	NA	NA	NA	NA
Poland	1%	0%	0%	2%	4%	NA	14%	NA
Portugal	NA	NA	NA	NA	NA	NA	NA	NA
Romania	1%	0%	4%	0%	1%	NA	1%	NA
Slovenia	NA	NA	NA	NA	NA	NA	NA	NA
Slovakia	1%	0%	1%	0%	52%	NA	1%	NA
Finland	4%	1%	6%	2%	16%	NA	NA	NA
Sweden	6%	2%	6%	17%	9%	NA	6%	NA

Table 4. The share of organic products over the total animal production generated in each country in 2021. NA = Not available (e.g., confidential), not significant, or not produced. Some figures are derived from estimated data. Source: Eurostat database (Eurostat, 2024b).

	Bovine	Pig	Sheep	Goat	Poultry	Chicken	Cow milk	Sheep milk	Goat milk	Buffalo milk	Drinking milk
Belgium	0%	1%	8%	0%	1%	NA	3%	NA	8%	NA	NA
Bulgaria	0%	0%	NA	NA	0%	0%	1%	4%	3%	0%	0%
Czechia	4%	0%	NA	NA	0%	NA	1%	NA	NA	NA	0%
Denmark	0%	NA	0%	NA	NA	NA	13%	NA	NA	NA	32%
Germany	NA	NA	NA	NA	NA	NA	4%	NA	NA	NA	10%
Estonia	8%	1%	NA	NA	NA	NA	1%	NA	NA	NA	1%
Ireland	0%	0%	0%	NA	0%	NA	0%	NA	NA	NA	1%
Greece	0%	0%	0%	0%	0%	NA	4%	15%	13%	NA	2%
Spain	1%	1%	8%	5%	0%	0%	1%	1%	3%	NA	1%
France	2%	2%	3%	NA	2%	2%	5%	11%	4%	NA	12%
Croatia	2%	0%	56%	NA	0%	NA	0%	NA	NA	NA	0%
Italy	4%	1%	15%	64%	0%	NA	4%	10%	18%	NA	12%
Cyprus	0%	0%	0%	0%	0%	NA	1%	0%	3%	NA	1%
Latvia	10%	1%	43%	30%	0%	NA	10%	NA	NA	NA	1%
Lithuania	3%	NA	NA	NA	NA	NA	6%	NA	NA	NA	4%
Luxembourg	2%	1%	6%	NA	NA	NA	NA	NA	NA	NA	NA
Hungary	0%	1%	3%	NA	0%	NA	0%	NA	NA	NA	1%
Malta	0%	0%	0%	0%	0%	0%	0%	NA	NA	NA	NA
Netherlands	0%	2%	NA	NA	NA	0%	2%	NA	10%	NA	18%
Austria	NA	NA	NA	NA	NA	NA	21%	NA	NA	NA	19%
Poland	0%	0%	0%	0%	0%	NA	0%	NA	NA	NA	0%
Portugal	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Romania	0%	0%	0%	NA	0%	0%	3%	3%	0%	0%	3%
Slovenia	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Slovakia	1%	1%	NA	NA	NA	NA	2%	10%	NA	NA	1%
Finland	2%	1%	26%	NA	0%	0%	4%	NA	NA	NA	NA
Sweden	8%	5%	20%	0%	1%	1%	17%	NA	NA	NA	18%

A large variability is evident both across countries and across crops. For instance, for Ireland, Luxembourg, and Hungary, no organic crop goes beyond 4% of the total crop production (Table 3). Besides, it is evident that in mediterranean countries such as Spain, Croatia, Italy, Cyprus, and Malta, there is a relatively high share of organic production for typically mediterranean crops such as grapes and olives (Table 3). Data is more fragmented for animal products, but the shares appear to be generally lower compared to the crops (Table 4).

A direct consequence of the lack of data on production is reflected in a lack of data in terms of consumption, with only total consumption values being available, regardless of the underlying farming practice.

The consumption of plant-based alternatives to meat and seafood products has grown fivefold since 2011 and is likely to continue to grow further driven by a progressive shift from an omnivorous diet to vegan, vegetarian, and flexitarian diets (EC, 2023c).

Overall meat consumption is declining with poultry slightly compensating for the decline in beef and pigmeat (EC, 2023c). Consumption of pulses is expected to grow by 61% from the 2021-2023 levels to 2035 (2.8 million t), while sugar consumption is expected to decrease from 0.8 to 0.6 million t over the same period, driven by the shift towards low-sugar diets and declining population. The population decline and changing diets (e.g., shift to plant-based milk substitutes) will slightly drive down (-0.1%) dairy consumption over the same time frame, while cheese consumption is expected to grow (EC, 2023c). Olive oil consumption is expected to keep growing driven by the diffusion across EU countries of the mediterranean diet, while wine consumption is expected to keep declining (EC, 2023c).

At the EU level, per capita retail sales of organic food products reached 55.8 € in 2019 after a growth of 131.6% over the previous decade and a total market volume of 41.4 billion € (Willer et al., 2024). Germany (12.0 billion €), France (11.3 billion €), and Italy (3.6 billion €) were the top countries in terms of organic retail sales in the EU in 2019, while in the same year Denmark (344 €/head), Luxembourg (265 €/head), and Austria (216 €/head) were the top countries in terms of per capita sales (Willer et al., 2024).

At the EU level, organic food covered 3.2% of the food market in 2019, while in the same year it accounted for 12.1%, 9.3%, and 9.0% of the food market in Denmark, Austria, and Sweden, respectively (Willer et al. 2024). Market share data reach a higher resolution allowing distinguishing among food groups, as shown in Table 5. Data are however provided only for 12 EU countries out of 27.

Table 5. Organic shares for retail sales values (€) for selected products in 2019. \*Figures from 2018. \*\*Figures from 2017. Adapted from Table 65 in (Willer et al., 2023).

	Austria	Belgium*	Czech Republic*	Denmark	Finland	France	Germany	Italy	Netherlands*	Norway	Spain**	Sweden**
Baby food					23	26.9		4.8		33.1		
Beverages			0.4	14.4 (juice)		5.5		3		0.5		5.6
Bread & bakery products		4.5	0.4			5.3	7.4	4	2.6	2.1		3.5
Eggs	22.1	18.2		29.6	18.0	37.2	20.6	19.8	15.9	9.5	2.9	
Fish and fish products		0.6				3.1			1.3	1.5	0.6	12.9
Fresh vegetables	16.0				4.5	7.6	9.8	4.7		4.2	3.3	12.2
Fruit	11.0				7.5	8.8	7.5	6.6		2.2	1.7	18.4
Vegetables and fruit	13.6		1.3			8.2		7.7	5.8			
Meat and meat products	3.8		0.2	8 (beef)		3.2	2.9	2.9	4.7	0.5	1.2	2.9
Milk and dairy products	12.4		1.4			5.8	8.5	3.6	5.6	2.1		10.4
- Butter	1.2	4.7		16.8		7.4	4.5	2.8		3.1		
- Cheese	10.3			6.8		2.6	4.7	1.0		0.7		
- Milk	21.8	3.3		32.3	4.5	15.7	12.4	8.1		4.4		
- Yoghurt	23.7	10.1				9.1	8.2	6.1		0.7		

Organic eggs and milk appear to have reached a significant market share in many countries whereas organic bread and bakery products together with organic fish and meat, and related products appear to cover a reduced market share in most countries (Table 5).

It is evident that if on the production side, organic farming data appeared to be fragmented, on the consumption side this fragmentation is exacerbated, making it difficult to drive meaningful and sound investigations on the effects of changes in consumption (demand) on production (supply) even for organic farming, which is the most diffused among the agricultural practices considered in this study.

However, there is a growing interest in the organic product market, which is shown by the rapid growth of the number of organic operators registered in the EU. For instance, between 2012 and 2022 the number of organic agricultural producers grew from around 248 000 to 426 000 (+72%), while organic aquaculture producers grew from 363 to 660 (+82%). Food processors (+245%) and other operators (+239%) in the organic food market grew too over the same period, passing from 34 000 to 119 000 and from 7 000 to 25 000, respectively. The number of operators registered as organic products international trade operators grew remarkably over the same period with importers passing from around 1000 to around 7000 (+491%) and exporters growing from about 289 to 4352 (+1406%) (Willer et al., 2024).

Such a trend follows the growing production of organic crops that occurred over the decade 2012-2022. For instance, considering the tonnes of production, cereals grew by 111%, citrus fruits by 89%, dry pulses and protein crops by 106%, and root crops by 412% (Eurostat, 2024a). A similar trend was observed for organic livestock production between 2012 and 2020. Over this period, in terms of heads live bovines grew by 51%, live goats by 61%, live horses by 213%, live poultry by 176%, live sheep by 38%, and live swine by 79% (Eurostat, 2024c, 2024b)).

Prior to the European Green Deal, the limited availability of organic food was a major barrier to increasing demand among Europeans regularly consuming organic food. At the time, the lack of interest in and knowledge about production and processing and the lack of trust in stakeholders and certification procedures limited the demand from Europeans who occasionally consumed organic food (Jensen et al., 2011). It was expected stable increase in demand would be subject to the expansion of the production side, reduction of the gap between organic and conventional food prices, increasing availability of organic products on national markets and enhanced support for research and conversion of organic production systems by public authorities (Jensen et al. 2011).

The growth in demand for organic products appears to be the result of successful socially oriented marketing campaigns, rather than the consequence of deep and science-based decision making. The organic/conventional price gap was often too large for consumers to change their shopping habits (Falguera et al., 2012). The demand for organic foods seems to be more driven by perceived attributes, such as expected better taste and more animal-friendly production systems. Consumers with low concern for food prices tend to have a higher propensity to be buyers, suggesting that consumers of organic products are willing to pay a price premium for these products (Jensen et al., 2019). In this context, labelling might play a major role. Consumer perceptions of organic labelling schemes appear subjective and often not based on objective knowledge. Therefore, it was suggested to label organic products with well-known organic certification logos that consumers trust could promote the demand (Janssen and Hamm, 2012). The identification of technical, social, and economic factors highlights an important research gap in studies that investigate the quantitative benefits of approaches that aim to holistically combine these different aspects.

A complete overview of the current practices, their output and their future potential could enable linking changes in dietary patterns to changes in production patterns with a shift towards more sustainable farming practices and overall environmental benefits. However, studies in this sense are not available. Rather, Poux et al (2018) modelled a gradual dietary pattern shift in Europe by 2050 through the Ten Years For Agroecology (TYFA) modelling exercise. The TYFA scenario is based on an agroecological approach which abandons imports of plant proteins for feed and adopts healthier diets by 2050. On the production side, the model assumed a nitrogen management based on closed fertility cycles, i.e., the fixation of nitrogen by crops and the use of manure as fertilisers, thus excluding fossil fertilisers. Furthermore, the model assumes phasing out pesticides, matching the principle of organic farming. It assumes a high complexity and diversity in land management by improving crop rotation and spatial heterogeneity, increasing areas dedicated to semi-natural vegetation and vertical layering of crops through the introduction of woody elements. The model assumes maintaining the existing extensive permanent grassland coupled with a de-intensification of ruminant livestock systems and a general reduction of monogastric systems. These practices were modelled to be applied across the whole EU farmland. On the consumption side, the model assumed the adoption of healthier, more balanced diets according to nutritional recommendations which



included a reduction in animal products in favour of plant-based proteins, and an increased intake of fruit and vegetables. On the other hand, the model assumed a general prioritisation of food production instead of feed or other non-food uses (e.g., energy) and reductions in food waste.

Despite an induced decline in production by 35% compared to 2010 (in Kcal) the scenario:

- 1) Feeds Europeans with healthy food while maintaining export capacity;
- 2) Reduces Europe's global food footprint;
- 3) Results in a 40% reduction in agricultural GHG emissions;
- 4) Helps to restore biodiversity and to protect natural resources.

Modelling these potential scenarios provides quantitative results and an improved understanding of the potential that changing dietary patterns (the demand) can induce a general shift in farming practices (the supply) with overall environmental and social benefits. It must be noted that the model assumed crucial modification across the agro-economic system, including international trade, and that it relies on significant changes in the dietary composition.

Our investigation shows that there are significant data gaps that limit understanding of the effects of potential consumption changes on the demand for different agricultural practices, at least other than organic farming. The existence of data on organic products (production and consumption, although fragmentary) is linked with the existence of the certification system and commercial labelling system and related standards in the EU, which can guide consumers' choices and allows tracking and linking production to consumption.

Accordingly, for a better understanding, several data gaps should be filled. For instance, a complete quantification of the current EU production (ideally in both physical and monetary terms), should be conducted at least at the national level, regardless of the production scale. On the other hand, an investigation of the current consumption levels of products linked with the different sustainable farming practices should be conducted, again ideally in both physical and monetary terms. Lastly, an estimation of the potential expansion from the current situation should be conducted for such practices, even though there are no current policy targets.

Only by obtaining the complete picture of the current situation in terms of both production and consumption, as well as the potential production it would be possible to conduct a study to estimate the effects of changes in terms of consumption on production. In the context of the broader overarching agro-economic system, it would be important to account for the expected shift from a linear to a circular economy, which, in this case, would coincide with the cycling of the fundamental macronutrients (Sporchia and Caro, 2023).

### 3.2.3 Potential impacts of diet and consumption patterns in the EU on biodiversity

Conserving biodiversity requires immediate and widespread action to reduce the biodiversity footprint of food consumption. In recent years, a growing interest from consumers to know the origins and contents of foods has put dietary changes and alternative choices such as organic foods and diverse protein sources on the agenda of societal and political debates. Dietary choices are crucial to address environmental issues and while greenhouse gas emissions are important, often other

environmental impact categories are not considered in the assessment of the sustainability of different foods, diets and choices. For instance, diets lower in meat not only reduce greenhouse gas emissions but also reduce agricultural expansion and intensification thereby reducing biodiversity impacts.

Willett et al. (2019) quantitatively described a universal healthy reference diet to provide a basis for estimating the health and environmental effects of adopting an alternative diet to standard current diets, many of which are high in unhealthy foods. This healthy reference diet largely consists of vegetables, fruits, whole grains, legumes, nuts, and unsaturated oils, includes a low to moderate amount of seafood and poultry, and includes no or a low quantity of red meat, processed meat, added sugar, refined grains, and starchy vegetables.

Henry et al. (2019) quantified land-use change in locations important for biodiversity across taxa and found diets low in animal products reduce agricultural expansion and intensity in regions with high biodiversity. Reducing ruminant meat consumption alone however was not sufficient to reduce fertiliser and irrigation application in biodiverse locations. The results differed according to taxa, emphasising that land-use change effects on biodiversity will be taxon specific.

Leclère et al. (2020), using an ensemble of land-use and biodiversity models, assessed whether and how humanity can reverse the declines in terrestrial biodiversity caused by habitat conversion. They showed that immediate efforts could enable the provision of food for the growing human population while reversing the declining global terrestrial biodiversity trends caused by habitat conversion. They concluded that through further sustainable intensification and trade, reduced food waste and more plant-based human diets, more than two-thirds of future biodiversity losses could be avoided. The biodiversity trends from habitat conversion could be reversed by 2050 for almost all of the models.

A recent study estimated the biodiversity footprints of 151 popular local dishes from around the world comparing global and local based production (Cheng et al., 2024). They found that in some areas with very high agricultural pressure, such as India and Brazil, specific ingredients such as beef, legumes and rice, which were also used in popular dishes, led to high biodiversity footprints. Legume-based dishes originating in India had high biodiversity footprints simply because they were sourced from biodiversity hotspots localities, but overall, across such factors as locally or globally produced, feedlot or pasture livestock production, vegan and vegetarian dishes presented lower biodiversity footprints than dishes containing meat.

Walker et al. (2019) assessed the intake-related health impacts and the food-production related impacts on ecosystems and human health by applying LCA methods to habitual diet data of 1457 European adults and concluded that reducing red and processed meat would result in the highest impact as a result of reductions in greenhouse gas emissions and land-use change. Furthermore, they highlighted that these food items have particularly high impacts on biodiversity loss.

Martin & Brandão (2017) quantified the implications of dietary choices for Swedish food consumption on a broad range of environmental impact categories using LCA to provide insight into the impacts, and potential trade-offs, associated with certain food products and dietary choices. Scenarios were used to assess the implications of diets with reduced meat, increased Swedish food consumption, increased organic foods, vegan and semi-vegetarian diets. They found that increasing domestic food production and consumption may lead to lower negative impacts for all impact

categories by reducing imports, and increasing consumption of organic foods may also lead to a reduction in biodiversity damage potential. Organic production was found to increase the risk of eutrophication and total land use. These organic matter-rich substances used in fertilisers can contain high levels of nitrogen and phosphorus, which can leach into soil and groundwater, increasing eutrophication. Kozicka et al., (2023) used a global economic land use model to assess the food system-wide impacts of a global dietary shift towards plant-based animal product alternatives which are promoted as more sustainable. Replacing 50% of the main animal products (pork, chicken, beef and milk) with plant-based products showed a general reduction of the environmental impacts, including climate change and land-use change. Moreover, it reduced the decrease in biodiversity Intactness Index by 2050. This is an index estimating how the average abundance of native terrestrial species in a region compares with their abundances before pronounced human impacts were revealed.

Röös et al., (2015) estimated the biodiversity damage potential of different food items consumed in Sweden, using LCA. The data inputs were hectares of land occupied, classified by land type (annual crops, permanent crops, pastures and meadows) and biome (natural vegetation type, e.g. temperate broadleaf forest or tropical savannah). The biodiversity damage potential (BDP) was based on differences in species richness between agricultural and natural land use of the biome. They showed that the biodiversity damage potential for meat such as beef (0.0032 BDP/kg of food), pork (0.0018 BDP/kg of food) and chicken (0.0013 BDP/kg of food) was substantially higher than vegetables (0.0001 BDP/kg of food) but also than cheese (0.0009 BDP/kg of food). They conclude that diets following Nordic Nutrition Recommendations (NNR) (Blomhoff et al., 2023) with low carbohydrate and high fat have a significantly lower impact on biodiversity compared to both a diet corresponding to Nordic recommendations and the current average Swedish diet. Additional impacts such as climate and land-use change were assessed following the same trend of the biodiversity damage potential.

By using a LCA, Castañé & Antón, (2017) assessed and compared the nutritional quality and the environmental impact of two different food diets, a Mediterranean diet and a vegan diet. Among the impact categories investigated, they included the global warming potential, the impact due to land use and the regional biodiversity impact was much higher for the Mediterranean diet. In particular, the regional biodiversity impact of the Mediterranean diet was around three times higher than the vegan one. Moreover, they claimed that this could be underestimated as fish biodiversity was not assessed. However, the study concluded that a shift towards a mix of these two diets, where all nutrients are consumed at the recommended levels and where only the least environmental impacting livestock products are consumed, would maybe help reduce biodiversity loss.

The Mediterranean diet was analysed in a recent study, quantifying the potential environmental impacts of lower adherence to Mediterranean diet by Italians (Vinci et al., 2022). Italian food patterns were analysed and compared to the Mediterranean diet through an LCA. The environmental impacts were expressed in species/year, measuring the impact on biodiversity by the extinction rate. Results of this study showed that Italian food patterns ( $2.1 \times 10^{-6}$  species year<sup>-1</sup>), compared to the Mediterranean diet ( $9.0 \times 10^{-7}$  species year<sup>-1</sup>) have +133% greater environmental impacts. This was mainly due to the increased consumption of meat and meat products, both in terms of quantity and frequency, which require significant use of resources such as soil and water.

A very recent study focused on the options for reducing the food-related biodiversity footprint of Vienna (Matej et al., 2024). The biodiversity footprints of 24 food consumption patterns were calculated with a life-cycle-assessment approach applying country- and primary biomass-specific factors for vertebrate species loss. The biodiversity footprint was derived from a high-resolution global countryside species-area-relationship model. The model incorporated land-use intensity and spatially explicit information on Vienna's source regions. The study found that compared to the baseline food consumption in Vienna, diets with fewer animal products could reduce the footprint by 21%–43%. Decreasing the demand for primary biomass under alternative diets could also free up domestic cropland and allow for reducing imports and relocating production from abroad to Austria. This could reduce Vienna's biodiversity footprint additionally by 5%–21%, depending on diet and demand level. They concluded that substituting animal products with plant-based alternatives from area-efficient production systems located outside of biodiversity hotspots emerges as a promising strategy for Western cities to reduce their biodiversity footprint.

Scarborough et al. (2023) linked dietary data among vegans, vegetarians, fish-eaters and meat-eaters to food-level data on different environmental impacts. Among the impacts, they included the global warming potential, land-use change and the potential biodiversity loss from a review of 570 life-cycle assessments covering more than 38,000 farms in 119 countries. Results of this study showed a positive association in the potential biodiversity loss with amounts of animal-based food consumed. The dietary impacts of vegans were 34.3% of the dietary impacts of meat-eaters ( $\geq 100$  g total meat consumed per day) for biodiversity. As the other environmental impact categories followed the same trend of the potential biodiversity loss, the authors concluded that despite substantial variation due to where and how food is produced, the relationship between environmental impact and animal-based food consumption is clear.

Walker et al. (2021) developed a methodology to build a country-specific database of food items, nutrient contents and environmental impacts. Based on the month of the year and source country, it calculates impacts. This database was then used to develop a detailed and personalised, healthy, low-impact diet by using linear optimisation. They applied this methodology to several case studies to compare what low-impact diets would look like depending on country, season, sex, the inclusion of dietary supplements, and for different diet types and impact categories among which land-use related biodiversity loss. The study found that the lowest possible biodiversity loss can be achieved with a vegan diet, provided it includes supplements to meet nutrient requirements. Alternatively, an omnivorous diet that includes animal-derived products but excludes meat and disregards the minimum milk constraint (50 grams) also results in low biodiversity loss. Without supplementation, the diet with the least impact on biodiversity that meets all nutritional needs is one that includes significant amounts of fish.

By using the Finnish diet as a case study, Kytä et al. (2023) developed two life cycle impact assessment methods to assess the biodiversity impacts of five dietary scenarios: the current Finnish diet and four alternative scenarios that involve a gradual reduction in the intake of foods of animal origin. The results showed that biodiversity impacts vary depending on the assessment method used. Most of the impacts were found to be related to land-use change, and linked to the production of feed, which leads to the dietary impacts being reduced with intake of foods of animal origin. High land occupation impacts were associated with different food groups than those with high land-use change impacts, they were higher than the land-use change impacts for beverages, sugars and

sweets. Trade played a significant role in the biodiversity impacts of diets, with over 85% of impacts being linked to imported foods and feeds.

Veerkamp et al. (2020) assessed the consequences for biodiversity and ecosystem services in Europe under four socio-environmental scenarios designed where food choices were considered. They warned that climate and land use change will continue to pose significant threats to biodiversity and some ecosystem services, even in the most optimistic scenario. Although targeted policies (e.g. on biodiversity conservation and sustainable land management) and behavioural change (e.g. reducing meat consumption) reduced the magnitude of biodiversity and ecosystem services loss, none of the socio-environmental scenarios modelled in this study achieved overall preservation of biodiversity and ecosystem services in Europe.

Keesing (2022) showed that the Planetary Health diet would reduce global extinctions by 30% compared to a traditional American diet (Read et al., 2022), largely because of a smaller footprint for pastureland to raise livestock for meat. Combining either of the biodiversity-friendly diets with less waste resulted in an even more dramatic effect on conservation, effectively preventing the extinction of dozens of species.

### 3.2.4 Transition toward consumption of products from alternative agricultural practices

Studies on specific alternative practices suggest that individually, provide some examples of successful transitions and highlight key factors of such transitions.

The rise of organic farming across the world has been supported by increased consumer demand for organic foods. Researchers and physicians have been emphasising consuming food grains, vegetables, milk products and fruits free of toxic substances, further creating a favourable outlook for organic farming globally. The trend is further accentuated by organic farming projects undertaken by several government and non-government organisations (NGOs) and companies as part of their corporate social responsibility to support farmers across developing economies. Organic food significantly strengthened its market foothold during the pandemic, especially across developed economies. This mainly happened because people gravitated toward healthier food choices due to an increased focus on boosting immune health as a precautionary measure. According to the "Organic Industry Survey" by the Organic Trade Association (OTA), organic sales exceeded \$63 billion, registering a 2% (1.4 billion) growth year over year between 2020 and 2021. Food sales reached \$57.5 billion, constituting over 90% of the total organic sales. From a range of factors that are transforming the organic seed industry demographics, favourable regulatory norms across numerous economies are one of the leading aspects favouring this business space. For instance, the European Regulation (EC) 834/2007 mandates that whenever external inputs are needed in organic seeds, these can only be organic inputs. Similarly, Regulation EC 889/2008 in its implementing rules states that seeds obtained from the organic production method only should be used in organic agriculture. Such regulatory regimes across different countries point towards an expansive scope for organic products, which consequently, will augment the organic seed market share in the coming years.

Concerning agroforestry practices, they enable landowners to generate income from the production of a wide range of conventional and specialty products while simultaneously protecting and conserving habitats, soil, water and other natural resources. Products produced through agroforestry

practices, including specialty or non-timber forest products, are produced from trees, within forests, or in various combinations with trees or shrubs, crops and/or animals. Many of these products have proven economic value but have been ignored by, or are unknown to, agricultural and forest landowners. Agroforestry enterprises often produce niche products for markets about which little is known. There is still not enough information about food preferences from agroforestry systems and the transition toward these products is unknown for the moment. However, it is ascertained that more explicit inclusion of agroforestry and the integration of agriculture and forestry agendas in global initiatives on climate change adaptation and mitigation can increase their effectiveness (Mbow et al., 2014).

Recently, a popular trend toward eating local, deemed being a locavore, evidenced by a growing social movement, has evolved (Osteen et al., 2012). While the benefits of buying food locally are debated due to the economics of comparative advantages, consumer groups support urban agriculture for a number of reasons, such as supporting local farmers; providing local, fresh food in inner city deserts; buying fresh food; knowing from where their food is coming; and respecting the environment (Peterson et al., 2015). Specifically, one study found that 66 percent of those surveyed welcomed more local food options because local food supports local economies (Scharber and Dancs, 2016). Many consumers also cite environmental impacts as a reason to buy local, evidenced by one study finding that environmental factors were an important reason to buy locally grown food for 61 percent of those surveyed (Reisman, 2012). Another popular reason is to reduce food insecurity.

Buying locally grown food can reduce food insecurity as local farms provide consumers who might not have previously had access to fresh produce or the opportunity to purchase it. Some urban farms make a point of targeting food insecurity, and having local farms allows a city to rely less heavily on external markets to feed its population. Despite the debate on realised benefits, consumers eat local food to feel good about it.

Grebitus et al. (2017) investigated whether young consumers perceive the health impacts and environmental benefits provided by urban agriculture, and what attitudes they hold towards this source of produce. Empirical results showed that both psychological and personal factors affect consumer intentions to participate in urban agriculture. Among others, subjective knowledge regarding urban agriculture and a generally favourable attitude towards urban farms increases the likelihood of buying and growing produce at urban farms.

### 3.2.5 How can changes in food consumption be implemented?

There is a broad scientific consensus that our current food system is unsustainable and a major driver of climate change, biodiversity loss and environmental degradation. The Farm to Fork strategy, announced by the European Commission in 2020, is an important step towards a much-needed overarching framework for governing the EU's food systems in a holistic manner. Many of its policy goals are based on the premise that consumers choose food through rational and reflective processes and that the 'well-informed, sovereign consumer' can always choose what to buy and eat. However, scientific evidence shows that food-related behaviours are often dominated by habits, routines and emotional processes, and that the food environment strongly shapes consumer



choices, concerns and priorities. A shift in consumer attitudes and behaviours could certainly contribute to making the whole food system more sustainable, but the above considerations suggest that policy interventions should address not only consumers but also food providers, producers, manufacturers, distributors and retailers (EC, 2023d).

The vast complexity of food systems and challenges related to their transformation covers the interactions between the economic, societal and natural environments. Therefore, shifting food consumption in a way that supports alternative agricultural practices has many potential entry points in order to address change at multiple scales. Studies on the transition and uptake of specific alternative practices (see section 3.2.4) highlight some key factors of transitions and suggest that a more holistic approach that promotes synergy could improve the uptake of such practices and transition to a more sustainable food system.

### *3.2.5.1 Agroecology as an integrated approach*

The agroecology approach includes value-driven social principles, while promoting a synergistic, multi-scale approach to address consumption changes as one part of a holistic revision of the food system to enable a global transformation.

Gliessman (2016) describes a multi-level transition pathway, beginning with an incremental transition of existing agricultural production systems towards the use of agroecological practices, resulting in the creation of an agroecological food production system. Incremental transition, characterised by the uptake of agroecological scientific principles by current agricultural systems to improve efficiency and ecosystem health, must then be supported by social and political changes to the broader food system to effectively support a food system transformation (HLPE, 2019).

Strengthening the relationship between consumers and producers is a key aspect of such a transformation. The promotion of local economies and short distribution networks such as farmers' markets and community-supported agriculture (CSA) helps strengthen the connection between producers and consumers while providing consumers with access to fresh, locally produced food and providing producers with fair compensation. These could include actions by multiple actors to ensure and promote access to local products, for example, the provision of infrastructure such as local markets, and the provision of space for them in popular locations to allow access to locally sourced food (UNEP, 2015).

Transformational change is accelerated and reinforced at the social level by embedding a new food system with social values including participation, localness, fairness, and justice (Gliessman, 2016). The application of values and principles can be applied in a holistic manner with respect to varying spatial scales and across systems from agricultural production systems to the broader food systems.

By providing a broad principled approach, agroecology can be applied to support consumption changes in a manner that is adaptable to different scales and specific situations. Synergy, one of the underlying principles of the agroecology approach, and the applicability of approaches to different situations is a key aspect to address and enable systemic change. In a similar manner, Harris (2023), in a recent overview of policy interventions, recommends a more holistic approach by applying

multiple interventions simultaneously to target different actors and leverage points, as a key factor in the implementation of solutions in order to achieve synergistic effects.

### *3.2.5.2 Potential Policy Interventions*

The scientific literature presents examples of policy interventions to overcome the barriers that are preventing consumers from adopting more sustainable and healthier diets. Information provisioning aligns with the agroecological principle of “co-creation of knowledge” and aims to improve sustainability-based decision-making by providing relevant information to consumers. Examples of such information-based interventions could include school education, ecolabeling and certification of products, and awareness campaigns.

Awareness-raising campaigns are an example of an approach to improve consumer knowledge regarding health, and sustainability of diet choices. They could target various practices, such as the Irish example on energy consumption (Pape et al., 2011), the Danish example focusing on CO<sub>2</sub> emissions (Scholl et al., 2010), or the Swedish campaign on food waste reduction (Röös et al., 2022).

A complementary entry point for information provisioning is education in school. Governments, for example, can integrate the teaching of sustainable consumption habits in national educational curricula to stimulate behavioural changes by increasing consumers’ knowledge (Jonkute and Staniskis, 2019). An example could be the integration of classes where the schools are instructed on preparing sustainable and appetizing meals (Tucker, 2018). Training teachers on sustainable consumption must be part of this kind of action, ensuring that they have all relevant skills and knowledge to boost the effectiveness of education-based actions.

Economic interventions represent another leverage point. The main economic measures are taxes and subsidies. They could target the final consumer by stimulating a change in consumption-related behaviour. A proposed example is reforming the Value Added Tax (VAT) to include an ecologically differentiated VAT based on the environmental impact generated by the production of a certain good across all economic sectors (Bahn-Walkowiak and Wilts, 2015; De Camillis and Goralczyk, 2013; Timmermans and Achten, 2018). Another example is introducing the most impactful food-related taxes, such as meat tax (Caro et al., 2017). An example of incentive-based action is the implementation of deposit refund schemes, acting on the post-consumption impacts (waste management), and stimulating a shift towards a circular economy (UNEP, 2015).

Another kind of action could be based on regulatory interventions. These aim to make use of legislation or agreements with trade partners to curb the environmental impact linked with food consumption. Trade agreements between countries can leverage the supply of more sustainable foods to consumers (Sporchia et al., 2021b, 2023). Although the origin of a food might be masked by trade dynamics (e.g., re-export and re-import), a broad (ideally general) implementation of trade agreement favouring sustainable sourcing would inevitably incur an overall (global) reduction in terms of impacts, regardless of where these impacts occur, ultimately obtaining significant benefits in terms of sustainable consumption (Sporchia et al., 2021a).

Multilateral environmental agreements (MEAs) are a specific kind of agreement that involves at least three countries (Harris, 2023). A classic example is the Montreal Protocol effectively resulted in the ban of ozone-depleting substances. The key advantage of MEAs is the reduction in risk of displacement, as long as enough countries are involved. A more recent example is the 2021 Glasgow Declaration on Forests and Land Use, in which 145 countries committed to “working collectively to halt and reverse forest loss and land degradation by 2030 while delivering sustainable development and promoting an inclusive rural transformation”.

A market-based action could be issuing bans or implementing quotas focusing on removing or reducing specific options from the market that are identified as unsustainable. EU’s REACH directive is an example of targeting chemicals. Due diligence, which is a process or effort to collect and analyse information before making a decision, is another example of legislation banning the least sustainable options. For instance, market-based actions may force companies to guarantee that no imported good (e.g., soybean or meat) is linked with deforestation (EC, 2024h). Imposing restrictions on the consumption of specific goods might face consumers’ resistance, as in the case of targeting meat (Röös et al., 2021), while encouraging a “less and better” approach should be more effective (Trewern et al., 2022). Another transversal action could be the implementation of restrictions on packaging size to discourage overconsumption, resulting in environmental as well as health benefits (Röös et al., 2021).

In terms of market-based possibilities, governments have a significant leveraging power to induce change in food consumption by acting on their purchasing practices in public procurement (Jonkute and Staniskis, 2019). An immediate example is action on school meals (Sporchia et al., 2024). However, trade-offs involving the economic side of sustainability should be kept in mind when proposing shifts from impactful foods such as pre-processed ones in favour to less impactful raw foodstuffs (Wolff et al., 2017).

Ultimately, control over the advertising of the most impactful foods should be implemented in order to curb demand and consumption (UNEP, 2015). Controls should be implemented to avoid marketing campaigns potentially fuelling overconsumption while campaigns encouraging more sustainable choices should be favoured (Beaton et al., 2012).

### 3.3 Discussion

This section focuses on the knowledge of the links between biodiversity-friendly farming and food consumption changes. A literature review was conducted to investigate the impacts of potential consumption changes on the demand for different agricultural practices with biodiversity benefits.

Organic agriculture has been identified as one of the solutions to reverse the ongoing loss of biodiversity in agricultural landscapes. While organic farming is not considered the only solution, it has demonstrated significant benefits for biodiversity. Increasing landscape complexity in agricultural lands has positive effects, especially on functional groups beneficial for agriculture (i.e., pollinators and natural enemies). The measured benefits of organic farming and plant diversification on the abundance and diversity of arthropod pollinators, predators, herbivores, and detritivores are promising, suggesting that increasing consumption of organic food can have important benefits in

terms of biodiversity. The European Commission has set out a comprehensive organic action plan for the European Union and aims to achieve the European Green Deal target of 25% of agricultural land under organic farming by 2030.

In Europe, extensive grazing adapted to local conditions and to the carrying capacity of the habitats has an important role in maintaining and restoring biodiversity. Agroforestry, especially silvopastoral agroforestry results in an increased species richness, functional and structural biodiversity and ecosystem services at the landscape level.

In combining livestock and crop production at farm level, also called mixed crop-livestock systems, the main benefit is the reduction of nutrient leakage from food production, which is one of the main pressures on biodiversity.

Rewarding these production systems and enhancing the market of their products can have important benefits for biodiversity. However, there are knowledge gaps about how and to what extent changes in consumption patterns can enable at European level their uptake.

Urban agriculture may substantially contribute to urban environmental sustainability, while it can enhance biodiversity and mitigate environmental impacts also beyond urban ecosystems by reducing land use outside for agriculture. Additionally, it can reduce transportation and introduce more green space and associated biodiversity in cities. An increase in economic profits has been also observed although the economic viability of some innovative technologies such as rooftop greenhouses can vary depending on factors such as yields and specific local conditions.

This study highlights how consumers' choices are therefore crucial to improve biodiversity. From this review emerges how consumption changes can be significant drivers of the reduction of biodiversity loss.

Firstly, consumption changes can drive the development and reinforcement of innovative agricultural practices that enhance biodiversity. For instance, increasing consumption of organic foods may lead to a reduction in biodiversity damage potential and support the development of organic farming.

Secondly, a large body of literature indicate that food consumption is among the largest drivers of biodiversity loss and diets lower in meat not only reduce greenhouse gas emissions but also reduce agricultural expansion and intensification thereby reducing biodiversity impacts. Some models have shown that more plant-based human diets are needed for reversing biodiversity decline (Leclère et al., 2020). The highest biodiversity-footprint dishes are meat dishes, especially beef, regardless of being locally or globally produced. Overall, substituting animal products with plant-based alternatives emerges as a promising strategy for reducing biodiversity footprint. Consumption changes can be implemented across Europe with a series of strategies and approaches for which some key examples have been presented in this working paper. While some countries have included sustainable consumption and production as a national sustainable development strategy priority, a coordinated action plan with coherent policies is increasingly needed to achieve large systemic effects. Such an action plan could target a thorough redesign of the food system. This could encompass suitable education measures, economic and social incentives, as well as rules for production, processing, retail, food services, transport, and final consumption.

Agroecology offers a comprehensive, holistic approach that can be applied broadly to support the transition to a biodiversity-friendly agri-food system. It is a set of principles designed to promote the synergistic implementation of environmental, socioeconomic, cultural, and governance changes within food systems. While the broad and dynamic nature of agroecology makes it adaptable to almost any context, there is limited empirical evidence quantifying the impacts of the fully integrated system. Any application of a subset of agroecological practices or principles can be considered a case study. However, the strength of agroecology lies in its participatory, transdisciplinary nature and the synergistic effects of multiple principles. Therefore, more studies are needed to quantify its performance when applied in a more comprehensive and holistic manner.

## 4 Conclusion

The overarching sustainability challenge we face is that the human economy must fit within the limits of the planet. This working paper focuses on food consumption, a major subcomponent of humanity's overall footprint. On the environmental side, we also look more closely at biodiversity, a key component of the environment that directly contributes to the life-supporting function of the environment while enhancing the resilience of our food systems to disturbance. In the context of climate change, extreme weather events, and increasing environmental stressors, such resilience may be vital to maintain ecosystem functions.

From an overarching perspective, the size of the European economy's ecological footprint is slightly decreasing over time, however it is still 1.5 times the size of the region's biocapacity. Food consumption is the largest component of Europe's<sup>2</sup> ecological footprint category, making up nearly a third (29-31%) of the total ecological footprint. While total food consumption and food-related consumption of biocapacity has been decreasing, the resulting negative impacts of overconsumption appear to be increasing according to the consumption footprint. Within food consumption, the greatest environmental and biodiversity impact across multiple indicators (including pressure type indicators and impact type indicators including both midpoint and endpoint impacts) occurs within the production stage of the supply chain and is associated with animal-based food products. Europe depends on international trade for its food and outsources significant environmental impacts to overseas production, with over 21% of its food biocapacity sourced from countries outside Europe.

Identifying consumption patterns and impacts related to agri-food systems can provide a quantitative starting point for monitoring and improvement. Further, it allows the identification of high-pressure or high-impact food system stages, consumption categories, and food types for targeted intervention.

Complementary to approaches that target categorical hotspots, the concept of circularity provides a set of cross-cutting principles that have great potential to reduce pressure and impact on the environment through the efficient use of resources, nutrient cycling and reduction of waste. These circular practices closely align and at least partly intrinsic elements of biodiversity-friendly

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<sup>2</sup> Europe used here refers specifically to the EEA-32 countries.

production systems and practices such as organic farming and agroecology. This means that coherent promotion of circular measures and biodiversity-friendly farming is important to achieve synergies, which can increase the success of the implementation of these practices. While there is uncertainty in the ability and degree to which a highly optimised and circular food system can be achieved, models of such optimised scenarios predict that the application of circular practices in combination with dietary shifts, have the potential to reduce land use by 71% and GHG emissions by 29% while self-sufficiently producing enough food to maintain current dietary protein levels.

Environmentally- and biodiversity-friendly practices such as agroecology, organic agriculture, agroforestry, urban farming and mixed crop-livestock systems offer significant benefits to the environment in terms of improved soil health, increased biodiversity and reduction of pollution and greenhouse gas emissions. From an environmental perspective, they support a shift in perspective of food systems to a more holistic view that is inclusive of multiple types of land cover, multifunctionality, and consideration for the function of biodiversity and nature.

Understanding how consumption can help drive changes in production towards environmentally- and biodiversity-friendly agri-food systems may be an approach with different leverage points. These leverage points can be aligned with multiple policy goals, introducing co-benefits to consumers and to society. While we know that food consumption and especially meat consumption is a major driver of biodiversity loss, more research is needed to understand how different consumption changes can directly or indirectly support biodiversity-friendly production practices.

Data exists to show an increased uptake of organic food products in Europe in the recent years. However, more data and research on the current and potential output of other sustainable production systems and practices are needed to establish evidence-based linkages between consumption and production.

Clear evidence exists that shifting current diets toward more plant-based foods is more resource efficient, reduces environmental impacts, and reduces impacts on biodiversity. However, shifting consumption in support of a sustainable and biodiversity-friendly food production system and the food system overall that operates within the capacity of our ecosystems is a much more complex challenge. Tools exist by which to monitor the size and type of impacts of our food systems, both globally and for the EU.

From the perspective of food systems transformation, the literature highlights that a successful transition requires an approach that targets not only the environmental sustainability of food production systems, but also the socioeconomic, cultural, and governance context within which the food system is embedded. International case studies suggest that several levers exist to effectively support the transition of current agri-food systems to a more sustainable and biodiversity-friendly one.

This working paper provides empirical evidence supporting biodiversity-friendly agricultural practices, while it acknowledges the challenges in the empirical assessment of synergistic effects of more holistic approaches.



The current evidence from implemented policy interventions highlights the need for a transdisciplinary and holistic approach to simultaneously target key aspects of socioeconomic, cultural, and political contexts. Ultimately a food systems transformation cannot be addressed through a single practice, set of practices, or intervention due to the dynamic and multi-scale landscape of food systems sub-components.

Agroecology, as a set of consolidated principles, is an example of an approach that addresses the challenges of a complex system in the sense that it can be applied by multiple actors and adapted to different sub-systems and scales. At its base, it is a scientific approach that addresses environmental aspects of food production. It includes, for example, the practices identified in this working paper, with quantified positive impacts on biodiversity. More importantly, agroecology incorporates principles that address social, economic, and governance facets of food systems transformation, thus making it a holistic approach. The evolution of agroecology into a social movement provides additional evidence of its broader societal uptake, a crucial aspect of food systems transformation.

Tracking and monitoring progress are crucial components of transformation. To effectively support a food systems transformation, a suitable monitoring framework must align with the key elements and principles of the transformation. For example, this framework should cover all relevant stages of the food system, incorporate both incremental and holistic measures, and be adaptable to the specific contexts and scales at which transition efforts are applied and outcomes are expected.

## 5 References

- Aleksić, N., Šušteršič, V., 2020. Analysis of application of aquaponic system as a model of the circular economy: A review. *Reciklaža i održivi razvoj* 13, 73–86. <https://doi.org/10.5937/ror2001073A>
- Allan, E., Bossdorf, O., Dormann, C.F., Prati, D., Gossner, M.M., Tschardt, T., Blüthgen, N., Bellach, M., Birkhofer, K., Boch, S., Böhm, S., Börschig, C., Chatzinotas, A., Christ, S., Daniel, R., Diekötter, T., Fischer, C., Friedl, T., Glaser, K., Hallmann, C., Hodac, L., Hölzel, N., Jung, K., Klein, A.M., Klaus, V.H., Kleinebecker, T., Krauss, J., Lange, M., Morris, E.K., Müller, J., Nacke, H., Pašalić, E., Rillig, M.C., Rothenwöhrer, C., Schall, P., Scherber, C., Schulze, W., Socher, S.A., Steckel, J., Steffan-Dewenter, I., Türke, M., Weiner, C.N., Werner, M., Westphal, C., Wolters, V., Wubet, T., Gockel, S., Gorke, M., Hemp, A., Renner, S.C., Schöning, I., Pfeiffer, S., König-Ries, B., Buscot, F., Linsenmair, K.E., Schulze, E.-D., Weisser, W.W., Fischer, M., 2014. Interannual variation in land-use intensity enhances grassland multidiversity. *Proc. Natl. Acad. Sci. U.S.A.* 111, 308–313. <https://doi.org/10.1073/pnas.1312213111>
- Amoah-Antwi, C., Kwiatkowska-Malina, J., Thornton, S.F., Fenton, O., Malina, G., Szara, E., 2020. Restoration of soil quality using biochar and brown coal waste: A review. *Science of The Total Environment* 722, 137852. <https://doi.org/10.1016/j.scitotenv.2020.137852>
- Antonelli, M., Tamea, S., Yang, H., 2017. Intra-EU agricultural trade, virtual water flows and policy implications. *Science of The Total Environment* 587–588, 439–448. <https://doi.org/10.1016/j.scitotenv.2017.02.105>
- Bahn-Walkowiak, B., Wilts, H., 2015. Reforming the EU VAT system to support the transition to a low-carbon and resource efficient economy. *Carbon pricing—Design, experiences and issues*. Cheltenham, Elgar 111–126.
- Bajan, B., Łukasiewicz, J., Mrówczyńska-Kamińska, A., Čechura, L., 2022. Emission intensities of the food production system in the European Union countries. *Journal of Cleaner Production* 363, 132298. <https://doi.org/10.1016/j.jclepro.2022.132298>
- Barros, M.V., Salvador, R., de Francisco, A.C., Piekarski, C.M., 2020. Mapping of research lines on circular economy practices in agriculture: From waste to energy. *Renewable and Sustainable Energy Reviews* 131, 109958. <https://doi.org/10.1016/j.rser.2020.109958>
- Bas-Bellver, C., Barrera, C., Betoret, N., Seguí, L., 2020. Turning Agri-Food Cooperative Vegetable Residues into Functional Powdered Ingredients for the Food Industry. *Sustainability* 12, 1284. <https://doi.org/10.3390/su12041284>
- Baue, B., 2019. Compared to what? A three-tiered typology of sustainable development performance indicators from incremental to contextual to transformational. *UNRISD Working Paper*.
- Beaton, C., Perera, O., Arden-Clarke, C., Farah, A.Z., Polsterer, N., 2012. Global Outlook on Sustainable Consumption and Production (SCP) Policies: Taking Action Together. Executive Summary.
- Belfrage, K., Björklund, J., Salomonsson, L., 2005. The Effects of Farm Size and Organic Farming on Diversity of Birds, Pollinators, and Plants in a Swedish Landscape. *AMBIO: A Journal of the Human Environment* 34, 582–588. <https://doi.org/10.1579/0044-7447-34.8.582>
- Bianchi, F.J.J.A., Mikos, V., Brussaard, L., Delbaere, B., Pulleman, M.M., 2013. Opportunities and limitations for functional agrobiodiversity in the European context. *Environmental Science & Policy* 27, 223–231. <https://doi.org/10.1016/j.envsci.2012.12.014>

- Billen, G., Aguilera, E., Einarsson, R., Garnier, J., Gingrich, S., Grizzetti, B., Lassaletta, L., Le Noë, J., Sanz-Cobena, A., 2021. Reshaping the European agro-food system and closing its nitrogen cycle: The potential of combining dietary change, agroecology, and circularity. *One Earth* 4, 839–850. <https://doi.org/10.1016/j.oneear.2021.05.008>
- Blomhoff, R., Andersen, R., Arnesen, E.K., Christensen, J.J., Eneroth, H., Erkkola, M., Gudaviciene, I., Halldórsson, Þ.I., Höyer-Lund, A., Lemming, E.W., 2023. Nordic Nutrition Recommendations 2023: integrating environmental aspects. Nordic Council of Ministers.
- Bonnet, C., Bouamra-Mechemache, Z., Corre, T., 2018. An Environmental Tax Towards More Sustainable Food: Empirical Evidence of the Consumption of Animal Products in France. *Ecological Economics* 147, 48–61. <https://doi.org/10.1016/j.ecolecon.2017.12.032>
- Buckwell, A., Matthews, A., Baldock, D., Mathijs, E., 2017. CAP-thinking out of the box: further modernisation of the CAP—why, what and how?
- Calisto Friant, M., Vermeulen, W.J.V., Salomone, R., 2021. Analysing European Union circular economy policies: words versus actions. *Sustainable Production and Consumption* 27, 337–353. <https://doi.org/10.1016/j.spc.2020.11.001>
- Caro, D., Frederiksen, P., Thomsen, M., Pedersen, A.B., 2017. Toward a more consistent combined approach of reduction targets and climate policy regulations: The illustrative case of a meat tax in Denmark. *Environmental Science & Policy* 76, 78–81. <https://doi.org/10.1016/j.envsci.2017.06.013>
- Castañé, S., Antón, A., 2017. Assessment of the nutritional quality and environmental impact of two food diets: A Mediterranean and a vegan diet. *Journal of Cleaner Production* 167, 929–937. <https://doi.org/10.1016/j.jclepro.2017.04.121>
- Castellani, V., Fusi, A., Sala, S., 2018. Consumer Footprint. Basket of Products indicator on Food [WWW Document]. JRC Publications Repository. <https://doi.org/10.2760/668763>
- Castle, S.E., Miller, D.C., Merten, N., Ordonez, P.J., Baylis, K., 2022. Evidence for the impacts of agroforestry on ecosystem services and human well-being in high-income countries: a systematic map. *Environ Evid* 11, 10. <https://doi.org/10.1186/s13750-022-00260-4>
- Cheng, E.M.Y., Cheng, C.M.L., Choo, J., Yan, Y., Carrasco, L.R., 2024. Biodiversity footprints of 151 popular dishes from around the world. *PLoS ONE* 19, e0296492. <https://doi.org/10.1371/journal.pone.0296492>
- Christie, A.P., Amano, T., Martin, P.A., Petrovan, S.O., Shackelford, G.E., Simmons, B.I., Smith, R.K., Williams, D.R., Wordley, C.F.R., Sutherland, W.J., 2021. The challenge of biased evidence in conservation. *Conservation Biology* 35, 249–262. <https://doi.org/10.1111/cobi.13577>
- Clark, M.A., Springmann, M., Hill, J., Tilman, D., 2019. Multiple health and environmental impacts of foods. *Proc. Natl. Acad. Sci. U.S.A.* 116, 23357–23362. <https://doi.org/10.1073/pnas.1906908116>
- Coelho, F.C., Coelho, E.M., Egerer, M., 2018. Local food: benefits and failings due to modern agriculture. *Scientia Agricola*. <https://doi.org/10.1590/1678-992x-2015-0439>
- Cohen, M., Godron, M., Cretin-Pablo, R., Pujos, R., 2023. Plant biodiversity in Mediterranean orchards is related to historical land use: perspectives for biodiversity-friendly olive production. *Reg Environ Change* 23, 70. <https://doi.org/10.1007/s10113-023-02067-6>
- Consumption Footprint Europe: top-down approach [WWW Document], 2024. URL <https://www.eea.europa.eu/en/analysis/indicators/europes-consumption-footprint> (accessed 4.30.24).
- Contesse, M., Van Vliet, B.J.M., Lenhart, J., 2018. Is urban agriculture urban green space? A comparison of policy arrangements for urban green space and urban agriculture in Santiago de Chile. *Land Use Policy* 71, 566–577. <https://doi.org/10.1016/j.landusepol.2017.11.006>

- Crenna, E., Sinkko, T., Sala, S., 2019. Biodiversity impacts due to food consumption in Europe. *Journal of Cleaner Production* 227, 378–391. <https://doi.org/10.1016/j.jclepro.2019.04.054>
- Crippa, M., Solazzo, E., Guizzardi, D., Monforti-Ferrario, F., Tubiello, F.N., Leip, A., 2021. Food systems are responsible for a third of global anthropogenic GHG emissions. *Nat Food* 2, 198–209. <https://doi.org/10.1038/s43016-021-00225-9>
- Čuček, L., Klemeš, J.J., Kravanja, Z., 2015. Overview of environmental footprints, in: *Assessing and Measuring Environmental Impact and Sustainability*. Elsevier, pp. 131–193. <https://doi.org/10.1016/B978-0-12-799968-5.00005-1>
- Cunningham, R., Barclay, K., Jacobs, B., Sharpe, S., McClean, N., 2022. Circular economy opportunities for fisheries and aquaculture in Australia [WWW Document]. URL <https://www.frdc.com.au/project/2020-078> (accessed 5.17.24).
- Damiani, M., Sinkko, T., Caldeira, C., Tosches, D., Robuchon, M., Sala, S., 2023. Critical review of methods and models for biodiversity impact assessment and their applicability in the LCA context. *Environmental Impact Assessment Review* 101, 107134. <https://doi.org/10.1016/j.eiar.2023.107134>
- Danso-Abbeam, G., Dagunga, G., Ehiakpor, D.S., Ogundeji, A.A., Setsoafia, E.D., Awuni, J.A., 2021. Crop–livestock diversification in the mixed farming systems: implication on food security in Northern Ghana. *Agric & Food Secur* 10, 35. <https://doi.org/10.1186/s40066-021-00319-4>
- Dao, H.T., Sharma, N.K., Swick, R.A., Moss, A.F., 2023. Feeding recycled food waste improved feed efficiency in laying hens from 24 to 43 weeks of age. *Sci Rep* 13, 8261. <https://doi.org/10.1038/s41598-023-34878-2>
- Davison, C.W., Rahbek, C., Morueta-Holme, N., 2021. Land-use change and biodiversity: Challenges for assembling evidence on the greatest threat to nature. *Global Change Biology* 27, 5414–5429. <https://doi.org/10.1111/gcb.15846>
- De Camillis, C., Goralczyk, M., 2013. Towards stronger measures for sustainable consumption and production policies: proposal of a new fiscal framework based on a life cycle approach. *Int J Life Cycle Assess* 18, 263–272. <https://doi.org/10.1007/s11367-012-0460-5>
- Del Borghi, A., Moreschi, L., Gallo, M., 2020. Circular economy approach to reduce water–energy–food nexus. *Current Opinion in Environmental Science & Health, Environmental Monitoring Assessment: Water-energy-food nexus* 13, 23–28. <https://doi.org/10.1016/j.coesh.2019.10.002>
- Dolganova, I., Mikosch, N., Berger, M., Núñez, M., Müller-Frank, A., Finkbeiner, M., 2019. The Water Footprint of European Agricultural Imports: Hotspots in the Context of Water Scarcity. *Resources* 8, 141. <https://doi.org/10.3390/resources8030141>
- Donner, M., Gohier, R., de Vries, H., 2020. A new circular business model typology for creating value from agro-waste. *Science of The Total Environment* 716, 137065. <https://doi.org/10.1016/j.scitotenv.2020.137065>
- Dora, M., Biswas, S., Choudhary, S., Nayak, R., Irani, Z., 2021. A system-wide interdisciplinary conceptual framework for food loss and waste mitigation strategies in the supply chain. *Industrial Marketing Management* 93, 492–508. <https://doi.org/10.1016/j.indmarman.2020.10.013>
- Drottberger, A., Zhang, Y., Yong, J.W.H., Dubois, M.-C., 2023. Urban farming with rooftop greenhouses: A systematic literature review. *Renewable and Sustainable Energy Reviews* 188, 113884. <https://doi.org/10.1016/j.rser.2023.113884>
- Dubey, A., 2024. The new encyclopaedia Britannica, 15th ed. ed. Encyclopaedia britannica, Chicago Paris.
- Duquennoy, C., Martinez, J., 2022. European Union’s policymaking on sustainable waste management and circularity in agroecosystems: The potential for innovative interactions

- between science and decision-making. *Front. Sustain. Food Syst.* 6.  
<https://doi.org/10.3389/fsufs.2022.937802>
- Earth Overshoot Day [WWW Document], 2024. Earth Overshoot Day. URL  
<https://overshoot.footprintnetwork.org/> (accessed 4.3.24).
- Ebissa, G., Yeshitela, K., Desta, H., Fetene, A., 2023. Urban agriculture and environmental sustainability. *Environ Dev Sustain.* <https://doi.org/10.1007/s10668-023-03208-x>
- EC, 2024a. Food Waste - European Commission [WWW Document]. URL  
[https://food.ec.europa.eu/food-safety/food-waste\\_en](https://food.ec.europa.eu/food-safety/food-waste_en) (accessed 4.16.24).
- EC, 2024b. EU actions against food waste [WWW Document]. URL  
[https://food.ec.europa.eu/safety/food-waste/eu-actions-against-food-waste\\_en](https://food.ec.europa.eu/safety/food-waste/eu-actions-against-food-waste_en) (accessed 4.24.24).
- EC, 2024c. Food waste reduction targets [WWW Document]. URL  
[https://food.ec.europa.eu/safety/food-waste/eu-actions-against-food-waste/food-waste-reduction-targets\\_en](https://food.ec.europa.eu/safety/food-waste/eu-actions-against-food-waste/food-waste-reduction-targets_en) (accessed 4.29.24).
- EC, 2024d. Date marking and food waste prevention [WWW Document]. URL  
[https://food.ec.europa.eu/safety/food-waste/eu-actions-against-food-waste/date-marking-and-food-waste-prevention\\_en](https://food.ec.europa.eu/safety/food-waste/eu-actions-against-food-waste/date-marking-and-food-waste-prevention_en) (accessed 5.16.24).
- EC, 2024e. Bioeconomy strategy [WWW Document]. URL [https://research-and-innovation.ec.europa.eu/research-area/environment/bioeconomy/bioeconomy-strategy\\_en](https://research-and-innovation.ec.europa.eu/research-area/environment/bioeconomy/bioeconomy-strategy_en) (accessed 5.16.24).
- EC, 2024f. Discarding in fisheries [WWW Document]. URL [https://oceans-and-fisheries.ec.europa.eu/fisheries/rules/discarding-fisheries\\_en](https://oceans-and-fisheries.ec.europa.eu/fisheries/rules/discarding-fisheries_en) (accessed 4.26.24).
- EC, 2024g. Organic Farming Systems Impact: Biodiversity [WWW Document]. URL  
[https://wikis.ec.europa.eu/display/IMAP/Organic+systems\\_Impacts?preview=/44157512/121444454/Organic%20farming%20systems\\_Biodiversity.pdf](https://wikis.ec.europa.eu/display/IMAP/Organic+systems_Impacts?preview=/44157512/121444454/Organic%20farming%20systems_Biodiversity.pdf)
- EC, 2024h. Deforestation Regulation implementation - European Commission [WWW Document]. URL [https://green-business.ec.europa.eu/deforestation-regulation-implementation\\_en](https://green-business.ec.europa.eu/deforestation-regulation-implementation_en) (accessed 12.4.24).
- EC, 2023a. The common agricultural policy: 2023-27 [WWW Document]. Agriculture and rural development. URL [https://agriculture.ec.europa.eu/common-agricultural-policy/cap-overview/cap-2023-27\\_en](https://agriculture.ec.europa.eu/common-agricultural-policy/cap-overview/cap-2023-27_en) (accessed 7.3.23).
- EC, 2023b. Organic farming in the EU: a decade of growth - European Commission. DG Agriculture and Rural Development, Brussels.
- EC, 2023c. EU agricultural outlook for markets, 2023–2035. European Commission, DG Agriculture and Rural Development, Brussels.
- EC, 2023d. Towards sustainable food consumption: promoting healthy, affordable and sustainable food consumption choices. Publications Office, LU.
- EC, 2022a. Food waste and food waste prevention — estimates [WWW Document]. European Commission. URL [https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Food\\_waste\\_and\\_food\\_waste\\_prevention\\_-\\_estimates](https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Food_waste_and_food_waste_prevention_-_estimates) (accessed 12.5.22).
- EC, 2022b. Circular economy action plan [WWW Document]. European Commission. URL [https://environment.ec.europa.eu/strategy/circular-economy-action-plan\\_en](https://environment.ec.europa.eu/strategy/circular-economy-action-plan_en) (accessed 6.29.22).
- EC, 2022c. Common agricultural policy for 2023-2027: 28 CAP strategic plans at a glance. European Commission.

- EC, 2022d. Common fisheries policy (CFP) [WWW Document]. European Commission. URL [https://ec.europa.eu/oceans-and-fisheries/policy/common-fisheries-policy-cfp\\_en](https://ec.europa.eu/oceans-and-fisheries/policy/common-fisheries-policy-cfp_en) (accessed 5.15.22).
- EC, 2021a. A European Green Deal [WWW Document]. European Commission - European Commission. URL [https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal\\_en](https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en) (accessed 5.17.22).
- EC, 2021b. COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS Pathway to a Healthy Planet for All EU Action Plan: “Towards Zero Pollution for Air, Water and Soil.”
- EC, 2021c. Treating urban waste water: new data shows improvement across Europe [WWW Document]. URL [https://environment.ec.europa.eu/news/treating-urban-waste-water-new-data-shows-improvement-across-europe-2021-11-19\\_en](https://environment.ec.europa.eu/news/treating-urban-waste-water-new-data-shows-improvement-across-europe-2021-11-19_en) (accessed 6.6.24).
- EC, 2021d. Common agricultural policy [WWW Document]. European Commission. URL [https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy\\_en](https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy_en) (accessed 12.13.21).
- EC, 2020a. A new Circular Economy Action Plan - For a cleaner and more competitive Europe.
- EC, 2020b. Farm to fork strategy: for a fair, healthy and environmentally-friendly food system. European Commission.
- EC, 2020c. COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS EU Biodiversity Strategy for 2030 Bringing nature back into our lives (No. COM (2020) 380 final).
- EC, 2018. Market study on date marking and other information provided on food labels and food waste prevention. Publications Office of the European Union, Luxembourg.
- EC, 2015. Energy use in the EU food sector: state of play and opportunities for improvement. Publications Office of the European Union.
- EC, 2013. Communication from the Commission to the European Parliament and the Council - Building the Single Market for Green Products - Facilitating better information on the environmental performance of products and organizations. COM 196.
- EC-JRC, 2023. Consumption footprint and domestic footprint: assessing the environmental impacts of EU consumption and production: life cycle assessment to support the European Green Deal. Publications Office, LU.
- EC-JRC, 2012. Life cycle indicators basket-of-products: development of life cycle based macro level monitoring indicators for resources, products and waste for the EU 27. European Commission, Joint Research Centre. Institute for Environment and Sustainability, Luxembourg.
- EEA, 2023a. The benefits to biodiversity of a strong circular economy (Briefing).
- EEA, 2023b. From source to sea — The untold story of marine litter [WWW Document]. European Environment Agency. URL <https://www.eea.europa.eu/publications/european-marine-litter-assessment/from-source-to-sea-the> (accessed 10.20.23).
- EEA, 2022a. Water abstraction by source and economic sector in Europe [WWW Document]. URL <https://www.eea.europa.eu/ims/water-abstraction-by-source-and> (accessed 3.26.24).
- EEA, 2022b. Transforming Europe’s food system — Assessing the EU policy mix. European Environment Agency, Copenhagen.
- EEA, 1999. Environmental indicators: Typology and overview. European Environmental Agency.



- Ellen MacArthur Foundation, 2024. Food and the circular economy [WWW Document]. URL <https://www.ellenmacarthurfoundation.org/food-and-the-circular-economy-deep-dive> (accessed 4.23.24).
- Environmental Footprint [WWW Document], 2021. European Platform on LCA | EPLCA: Environmental Footprint. URL <https://eplca.jrc.ec.europa.eu/EnvironmentalFootprint.html> (accessed 4.21.24).
- EP, 2024. New EU rules to improve urban wastewater treatment and reuse | News | European Parliament [WWW Document]. URL <https://www.europarl.europa.eu/news/en/press-room/20240408IPR20307/new-eu-rules-to-improve-urban-wastewater-treatment-and-reuse> (accessed 7.17.24).
- Eroldoğan, O.T., Glencross, B., Novoveska, L., Gaudêncio, S.P., Rinkevich, B., Varese, G.C., de Fátima Carvalho, M., Tasdemir, D., Safarik, I., Nielsen, S.L., Rebours, C., Lada, L.B., Robbens, J., Strode, E., Haznedaroğlu, B.Z., Kotta, J., Evliyaoğlu, E., Oliveira, J., Girão, M., Vasquez, M.I., Čabarkapa, I., Rakita, S., Klun, K., Rotter, A., 2023. From the sea to aquafeed: A perspective overview. *Reviews in Aquaculture* 15, 1028–1057. <https://doi.org/10.1111/raq.12740>
- ESTAT, 2024. Extra-EU trade in agricultural goods [WWW Document]. URL [https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Extra-EU\\_trade\\_in\\_agricultural\\_goods](https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Extra-EU_trade_in_agricultural_goods) (accessed 5.31.24).
- Estrada-Carmona, N., Sánchez, A.C., Remans, R., Jones, S.K., 2022. Complex agricultural landscapes host more biodiversity than simple ones: A global meta-analysis. *Proc. Natl. Acad. Sci. U.S.A.* 119, e2203385119. <https://doi.org/10.1073/pnas.2203385119>
- EU, 2024. Directive (EU) 2024/825 of the European Parliament and of the Council of 28 February 2024 amending Directives 2005/29/EC and 2011/83/EU as regards empowering consumers for the green transition through better protection against unfair practices and through better information, OJ L, 2024/825, 6.3.2024.
- EU, 2008. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives.
- EUBA, 2023. The Bioeconomy Blueprint: Building a circular and resilient Europe [WWW Document]. EU Bioeconomy Alliance. URL <https://www.bioeconomyalliance.eu/news/the-bioeconomy-blueprint-building-a-circular-and-resilient-europe/> (accessed 5.22.24).
- European Parliament, 2024a. Food waste reduction: what EU actions are there? [WWW Document]. Topics | European Parliament. URL <https://www.europarl.europa.eu/topics/en/article/20240318STO19401/food-waste-reduction-what-eu-actions-are-there> (accessed 5.31.24).
- European Parliament, 2024b. Textiles and food waste reduction: New EU rules to support circular economy [WWW Document]. URL <https://www.europarl.europa.eu/news/en/press-room/20240212IPR17625/textiles-and-food-waste-reduction-new-eu-rules-to-support-circular-economy> (accessed 5.28.24).
- Eurostat, 2024a. Organic crop production by crops. [https://doi.org/10.2908/ORG\\_CROPPRO](https://doi.org/10.2908/ORG_CROPPRO)
- Eurostat, 2024b. Organic production of animal products. [https://doi.org/10.2908/org\\_aprod](https://doi.org/10.2908/org_aprod)
- Eurostat, 2024c. Organic livestock. [https://doi.org/10.2908/ORG\\_LSTSPEC](https://doi.org/10.2908/ORG_LSTSPEC)
- Eurostat, 2023a. Guidance note on ecosystem extent accounts.
- Eurostat, 2023b. Daily consumption of fruit and vegetables by sex, age and body mass index. [https://doi.org/10.2908/HLTH\\_EHIS\\_FV3M](https://doi.org/10.2908/HLTH_EHIS_FV3M)
- Falguera, V., Aliguer, N., Falguera, M., 2012. An integrated approach to current trends in food consumption: Moving toward functional and organic products? *Food Control* 26, 274–281. <https://doi.org/10.1016/j.foodcont.2012.01.051>

- FAO, 2022. Urban and peri-urban agriculture sourcebook. FAO. <https://doi.org/10.4060/cb9722en>
- FAO (Ed.), 1991. Guidelines: land evaluation for extensive grazing, FAO soils bulletin. Food and Agriculture Organization of the United Nations, Rome.
- FAOSTAT, 2024. Food Balance Sheet (FBS) [WWW Document]. URL <https://www.fao.org/faostat/en/#data/FBS>
- Feistner, A.T., 2004. Biodiversity: An Introduction by Kevin J. Gaston & John I. Spicer (2004), xv+ 191 pp., Blackwell Publishing, Oxford, UK. ISBN 1 4051 1857 1 (pbk), \pounds 19.99. Oryx 38, 465–465.
- FODAFO, 2024. Footprint Data Foundation [WWW Document]. URL <https://www.fodafo.org/index.html> (accessed 4.1.24).
- Galli, A., Antonelli, M., Wambersie, L., Bach-Faig, A., Bartolini, F., Caro, D., Iha, K., Lin, D., Mancini, M.S., Sonnino, R., Vanham, D., Wackernagel, M., 2023. EU-27 ecological footprint was primarily driven by food consumption and exceeded regional biocapacity from 2004 to 2014. *Nat Food* 4, 810–822. <https://doi.org/10.1038/s43016-023-00843-5>
- Galli, A., Caro, D., Antonelli, M., 2022. Measuring the sustainability of food systems: The rationale for Footprint Indicators, in: *Routledge Handbook of Sustainable Diets*. Routledge.
- Galli, A., Moreno Pires, S., Iha, K., Alves, A.A., Lin, D., Mancini, M.S., Teles, F., 2020. Sustainable food transition in Portugal: Assessing the Footprint of dietary choices and gaps in national and local food policies. *Science of The Total Environment* 749, 141307. <https://doi.org/10.1016/j.scitotenv.2020.141307>
- Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B., Cosby, B.J., 2003. The Nitrogen Cascade. *BioScience* 53, 341. [https://doi.org/10.1641/0006-3568\(2003\)053\[0341:TNC\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2003)053[0341:TNC]2.0.CO;2)
- García-Herrero, L., Gibin, D., Damiani, M., Sanyé-Mengual, E., Sala, S., 2023. What is the water footprint of EU food consumption? A comparison of water footprint assessment methods. *Journal of Cleaner Production* 415, 137807. <https://doi.org/10.1016/j.jclepro.2023.137807>
- Garske, B., Heyl, K., Ekardt, F., Weber, L.M., Gradzka, W., 2020a. Challenges of Food Waste Governance: An Assessment of European Legislation on Food Waste and Recommendations for Improvement by Economic Instruments. *Land* 9, 231. <https://doi.org/10.3390/land9070231>
- Garske, B., Stubenrauch, J., Ekardt, F., 2020b. Sustainable phosphorus management in European agricultural and environmental law. *Review of European, Comparative & International Environmental Law* 29, 107–117. <https://doi.org/10.1111/reel.12318>
- Geueke, B., Groh, K., Muncke, J., 2018. Food packaging in the circular economy: Overview of chemical safety aspects for commonly used materials. *Journal of Cleaner Production* 193, 491–505. <https://doi.org/10.1016/j.jclepro.2018.05.005>
- Gilman, E., 2015. Status of international monitoring and management of abandoned, lost and discarded fishing gear and ghost fishing. *Marine Policy* 60, 225–239. <https://doi.org/10.1016/j.marpol.2015.06.016>
- Giosuè, A., Recanati, F., Calabrese, I., Dembska, K., Castaldi, S., Gagliardi, F., Vitale, M., Vaccaro, O., Antonelli, M., Riccardi, G., 2022. Good for the heart, good for the Earth: proposal of a dietary pattern able to optimize cardiovascular disease prevention and mitigate climate change. *Nutrition, Metabolism and Cardiovascular Diseases* 32, 2772–2781. <https://doi.org/10.1016/j.numecd.2022.08.001>
- Giudice, F., Caferra, R., Morone, P., 2020. COVID-19, the Food System and the Circular Economy: Challenges and Opportunities. *Sustainability* 12, 7939. <https://doi.org/10.3390/su12197939>
- Gliessman, S., 2016. Transforming food systems with agroecology. *Agroecology and Sustainable Food Systems* 40, 187–189. <https://doi.org/10.1080/21683565.2015.1130765>

- Global Footprint Network, 2020. Circular Companies [WWW Document].  
<https://overshoot.footprintnetwork.org/portfolio/circular-companies/>.
- Grant, H., 2018. Legal plastic content in animal feed could harm human health, experts warn. *The Guardian*.
- Grebitus, C., Printezis, I., Printezis, A., 2017. Relationship between consumer behavior and success of urban agriculture. *Ecological Economics* 136, 189–200.
- GTAP10, 2020. GTAP Data Bases: GTAP 10 Data Base [WWW Document]. URL  
<https://www.gtap.agecon.purdue.edu/databases/v10/index.aspx>
- Guillen, J., Natale, F., Carvalho, N., Casey, J., Hofherr, J., Druon, J.-N., Fiore, G., Gibin, M., Zanzi, A., Martinsohn, J.Th., 2019. Global seafood consumption footprint. *Ambio* 48, 111–122.  
<https://doi.org/10.1007/s13280-018-1060-9>
- Günther, J., Manshoven, S., Paleari, S., Fuchs, G., Carré, A., Fischer-Bogason, R., 2023. ETC/CE Report 2023/7 Circular Economy and Biodiversity.
- Habibullah, M.S., Din, B.H., Tan, S.-H., Zahid, H., 2022. Impact of climate change on biodiversity loss: global evidence. *Environ Sci Pollut Res* 29, 1073–1086.  
<https://doi.org/10.1007/s11356-021-15702-8>
- Hamam, M., Chinnici, G., Di Vita, G., Pappalardo, G., Pecorino, B., Maesano, G., D’Amico, M., 2021. Circular Economy Models in Agro-Food Systems: A Review. *Sustainability* 13, 3453.  
<https://doi.org/10.3390/su13063453>
- Hammond, G.P., 2006. People, planet and prosperity: The determinants of humanity’s environmental footprint. *Natural Resources Forum* 30, 27–36.  
<https://doi.org/10.1111/j.1477-8947.2006.00155.x>
- Harris, M., 2023. Policy interventions to encourage sustainable consumption. JNCC Report 747 (Guidance report). JNCC, Peterborough.
- Hasan, Z., Lateef, M., 2024. Transforming food waste into animal feeds: an in-depth overview of conversion technologies and environmental benefits. *Environ Sci Pollut Res* 31, 17951–17963. <https://doi.org/10.1007/s11356-023-30152-0>
- Hayes, M., Gallagher, M., 2019. Processing and recovery of valuable components from pelagic blood-water waste streams: A review and recommendations. *Journal of Cleaner Production* 215, 410–422. <https://doi.org/10.1016/j.jclepro.2019.01.028>
- Hegwood, M., Burgess, M.G., Costigliolo, E.M., Smith, P., Bajželj, B., Saunders, H., Davis, S.J., 2023. Rebound effects could offset more than half of avoided food loss and waste. *Nat Food* 4, 585–595. <https://doi.org/10.1038/s43016-023-00792-z>
- Henry, R.C., Alexander, P., Rabin, S., Anthoni, P., Rounsevell, M.D.A., Arneeth, A., 2019. The role of global dietary transitions for safeguarding biodiversity. *Global Environmental Change* 58, 101956. <https://doi.org/10.1016/j.gloenvcha.2019.101956>
- HLPE, 2019. Agroecological and other innovative approaches for sustainable agriculture and food systems that enhance food security and nutrition. A report by the High Level Panel of Experts on Food Security and Nutrition of the Committee on World Food Security. Rome.
- HLPE, 2014. Food losses and waste in the context of sustainable food systems, A report by the High Level Panel of Experts on Food Security and Nutrition of the Committee on World Food Security. Rome.
- Hochkirch, A., Samways, M.J., Gerlach, J., Böhm, M., Williams, P., Cardoso, P., Cumberlidge, N., Stephenson, P.J., Seddon, M.B., Clausnitzer, V., Borges, P.A.V., Mueller, G.M., Pearce-Kelly, P., Raimondo, D.C., Danielczak, A., Dijkstra, K.B., 2021. A strategy for the next decade to address data deficiency in neglected biodiversity. *Conservation Biology* 35, 502–509.  
<https://doi.org/10.1111/cobi.13589>

- Hoekstra, A.Y., Mekonnen, M.M., 2012. The water footprint of humanity. *Proc. Natl. Acad. Sci. U.S.A.* 109, 3232–3237. <https://doi.org/10.1073/pnas.1109936109>
- Hoekstra, A.Y., Wiedmann, T.O., 2014. Humanity's unsustainable environmental footprint. *Science* 344, 1114–1117. <https://doi.org/10.1126/science.1248365>
- Hoeven, D. van der, 2019. Circular agriculture, the model of the future [WWW Document]. Bio Based Press. URL <https://www.biobasedpress.eu/2019/01/circular-agriculture-the-model-of-the-future/> (accessed 5.13.24).
- Holden, N.M., White, E.P., Lange, Matthew.C., Oldfield, T.L., 2018. Review of the sustainability of food systems and transition using the Internet of Food. *npj Sci Food* 2, 18. <https://doi.org/10.1038/s41538-018-0027-3>
- Hole, D.G., Perkins, A.J., Wilson, J.D., Alexander, I.H., Grice, P.V., Evans, A.D., 2005. Does organic farming benefit biodiversity? *Biological Conservation* 122, 113–130. <https://doi.org/10.1016/j.biocon.2004.07.018>
- Hortal, J., De Bello, F., Diniz-Filho, J.A.F., Lewinsohn, T.M., Lobo, J.M., Ladle, R.J., 2015. Seven Shortfalls that Beset Large-Scale Knowledge of Biodiversity. *Annu. Rev. Ecol. Evol. Syst.* 46, 523–549. <https://doi.org/10.1146/annurev-ecolsys-112414-054400>
- Houšková, B., Bušo, R., Makovníková, J., 2021. Contribution of Good Agricultural Practices to Soil Biodiversity. *OJE* 11, 75–85. <https://doi.org/10.4236/oje.2021.111007>
- Hunter Jr, M.L., Gibbs, J.P., 2006. *Fundamentals of conservation biology*. John Wiley & Sons.
- IPBES, 2019. Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES Secretariat, Bonn, Germany. <https://doi.org/10.5281/ZENODO.3831673>
- IPCC, 2023. IPCC, 2023: Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, H. Lee and J. Romero (eds.)]. IPCC, Geneva, Switzerland. Intergovernmental Panel on Climate Change (IPCC). <https://doi.org/10.59327/IPCC/AR6-9789291691647>
- ISO, 2006. Environmental management-life cycle assessment-principles and framework. International Organization for Standardization.
- Janssen, M., Hamm, U., 2012. Product labelling in the market for organic food: Consumer preferences and willingness-to-pay for different organic certification logos. *Food quality and preference* 25, 9–22.
- Jensen, J.D., Christensen, T., Denver, S., Ditlevsen, K., Lassen, J., Teuber, R., 2019. Heterogeneity in consumers' perceptions and demand for local (organic) food products. *Food Quality and Preference* 73, 255–265. <https://doi.org/10.1016/j.foodqual.2018.11.002>
- Jensen, K.O., Denver, S., Zanolli, R., 2011. Actual and potential development of consumer demand on the organic food market in Europe. *NJAS - Wageningen Journal of Life Sciences, Improving Production Efficiency, Quality and Safety in Organic and "Low-Input" Food Supply Chains* 58, 79–84. <https://doi.org/10.1016/j.njas.2011.01.005>
- Jonkute, G., Staniskis, J.K., 2019. THE ROLE OF DIFFERENT STAKEHOLDERS IN IMPLEMENTING SUSTAINABLE CONSUMPTION AND PRODUCTION IN LITHUANIA. *Environ. Eng. Manag. J.* 18, 617–632. <https://doi.org/10.30638/eemj.2019.057>
- Jurgilevich, A., Birge, T., Kentala-Lehtonen, J., Korhonen-Kurki, K., Pietikäinen, J., Saikku, L., Schösler, H., 2016. Transition towards Circular Economy in the Food System. *Sustainability* 8, 69. <https://doi.org/10.3390/su8010069>
- Kardung, M., Cingiz, K., Costenoble, O., Delahaye, R., Heijman, W., Lovrić, M., van Leeuwen, M., M'Barek, R., van Meijl, H., Piotrowski, S., Ronzon, T., Sauer, J., Verhoog, D., Verkerk, P.J.,

- Vrachioli, M., Wesseler, J.H.H., Zhu, B.X., 2021. Development of the Circular Bioeconomy: Drivers and Indicators. *Sustainability* 13, 413. <https://doi.org/10.3390/su13010413>
- Keegan, D., Kretschmer B., Elbersen B., Panoutsou C., 2013. Cascading use: a systematic approach to biomass beyond the energy sector. *Biofuel, Bioproducts and Biorefining* 7, 193–206.
- Keesing, F., 2022. Diet for a small footprint. *Proc. Natl. Acad. Sci. U.S.A.* 119, e2204241119. <https://doi.org/10.1073/pnas.2204241119>
- Kirchherr, J., Reike, D., Hekkert, M., 2017. Conceptualizing the circular economy: an analysis of 114 definitions. *Resources, Conservation and Recycling* 127, 221–232. <https://doi.org/10.1016/j.resconrec.2017.09.005>
- Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., i Canals, L.M., Saad, R., de Souza, D.M., Müller-Wenk, R., 2013. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *Int J Life Cycle Assess* 18, 1188–1202. <https://doi.org/10.1007/s11367-013-0579-z>
- Köninger, J., Lugato, E., Panagos, P., Kochupillai, M., Orgiazzi, A., Briones, M.J.I., 2021. Manure management and soil biodiversity: towards more sustainable food systems in the EU. *Agricultural Systems* 194, 103251. <https://doi.org/10.1016/j.agsy.2021.103251>
- Koslowski, M., Moran, D.D., Tisserant, A., Verones, F., Wood, R., 2020. Quantifying Europe's biodiversity footprints and the role of urbanization and income. *Glob. Sustain.* 3, e1. <https://doi.org/10.1017/sus.2019.23>
- Kozicka, M., Havlík, P., Valin, H., Wollenberg, E., Deppermann, A., Leclère, D., Lauri, P., Moses, R., Boere, E., Frank, S., Davis, C., Park, E., Gurwick, N., 2023. Feeding climate and biodiversity goals with novel plant-based meat and milk alternatives. *Nat Commun* 14, 5316. <https://doi.org/10.1038/s41467-023-40899-2>
- Kraak, V.I., Aschemann-Witzel, J., 2024. The Future of Plant-Based Diets: Aligning Healthy Marketplace Choices with Equitable, Resilient, and Sustainable Food Systems. *Annual Review of Public Health* 45, 253–275. <https://doi.org/10.1146/annurev-publhealth-060722-032021>
- Kyttä, V., Hyvönen, T., Saarinen, M., 2023. Land-use-driven biodiversity impacts of diets—a comparison of two assessment methods in a Finnish case study. *Int J Life Cycle Assess* 28, 1104–1116. <https://doi.org/10.1007/s11367-023-02201-w>
- Leach, A.M., Galloway, J.N., Bleeker, A., Erisman, J.W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. *Environmental Development* 1, 40–66. <https://doi.org/10.1016/j.envdev.2011.12.005>
- Leclère, D., Obersteiner, M., Barrett, M., Butchart, S.H.M., Chaudhary, A., De Palma, A., DeClerck, F.A.J., Di Marco, M., Doelman, J.C., Dürauer, M., Freeman, R., Harfoot, M., Hasegawa, T., Hellweg, S., Hilbers, J.P., Hill, S.L.L., Humpenöder, F., Jennings, N., Krisztin, T., Mace, G.M., Ohashi, H., Popp, A., Purvis, A., Schipper, A.M., Tabeau, A., Valin, H., Van Meijl, H., Van Zeist, W.-J., Visconti, P., Alkemade, R., Almond, R., Bunting, G., Burgess, N.D., Cornell, S.E., Di Fulvio, F., Ferrier, S., Fritz, S., Fujimori, S., Grooten, M., Harwood, T., Havlík, P., Herrero, M., Hoskins, A.J., Jung, M., Kram, T., Lotze-Campen, H., Matsui, T., Meyer, C., Nel, D., Newbold, T., Schmidt-Traub, G., Stehfest, E., Strassburg, B.B.N., Van Vuuren, D.P., Ware, C., Watson, J.E.M., Wu, W., Young, L., 2020. Bending the curve of terrestrial biodiversity needs an integrated strategy. *Nature* 585, 551–556. <https://doi.org/10.1038/s41586-020-2705-y>
- Leip, A., Caldeira, C., Corrado, S., Hutchings, N.J., Lesschen, J.P., Schaap, M., De Vries, W., Westhoek, H., Van Grinsven, H.Jm., 2022. Halving nitrogen waste in the European Union food systems requires both dietary shifts and farm level actions. *Global Food Security* 35, 100648. <https://doi.org/10.1016/j.gfs.2022.100648>



- Leip, A., Weiss, F., Lesschen, J.P., Westhoek, H., 2014. The nitrogen footprint of food products in the European Union. *J. Agric. Sci.* 152, 20–33. <https://doi.org/10.1017/S0021859613000786>
- Lichtenberg, E.M., Kennedy, C.M., Kremen, C., Batáry, P., Berendse, F., Bommarco, R., Bosque-Pérez, N.A., Carvalheiro, L.G., Snyder, W.E., Williams, N.M., Winfree, R., Klatt, B.K., Åström, S., Benjamin, F., Brittain, C., Chaplin-Kramer, R., Clough, Y., Danforth, B., Diekötter, T., Eigenbrode, S.D., Ekroos, J., Elle, E., Freitas, B.M., Fukuda, Y., Gaines-Day, H.R., Grab, H., Gratton, C., Holzschuh, A., Isaacs, R., Isaia, M., Jha, S., Jonason, D., Jones, V.P., Klein, A., Krauss, J., Letourneau, D.K., Macfadyen, S., Mallinger, R.E., Martin, E.A., Martinez, E., Memmott, J., Morandin, L., Neame, L., Otieno, M., Park, M.G., Pfiffner, L., Pocock, M.J.O., Ponce, C., Potts, S.G., Poveda, K., Ramos, M., Rosenheim, J.A., Rundlöf, M., Sardiñas, H., Saunders, M.E., Schon, N.L., Sciligo, A.R., Sidhu, C.S., Steffan-Dewenter, I., Tschamntke, T., Veselý, M., Weisser, W.W., Wilson, J.K., Crowder, D.W., 2017. A global synthesis of the effects of diversified farming systems on arthropod diversity within fields and across agricultural landscapes. *Global Change Biology* 23, 4946–4957. <https://doi.org/10.1111/gcb.13714>
- Lin, B.B., Philpott, S.M., Jha, S., Liere, H., 2017. Urban Agriculture as a Productive Green Infrastructure for Environmental and Social Well-Being, in: Tan, P.Y., Jim, C.Y. (Eds.), *Greening Cities, Advances in 21st Century Human Settlements*. Springer Singapore, Singapore, pp. 155–179. [https://doi.org/10.1007/978-981-10-4113-6\\_8](https://doi.org/10.1007/978-981-10-4113-6_8)
- Lin, D., Hanscom, L., Murthy, A., Galli, A., Evans, M., Neill, E., Mancini, M., Martindill, J., Medouar, F.-Z., Huang, S., Wackernagel, M., 2018. Ecological Footprint Accounting for Countries: Updates and Results of the National Footprint Accounts, 2012–2018. *Resources* 7, 58. <https://doi.org/10.3390/resources7030058>
- Mancini, M.S., Galli, A., Niccolucci, V., Lin, D., Bastianoni, S., Wackernagel, M., Marchettini, N., 2016. Ecological Footprint: Refining the carbon Footprint calculation. *Ecological Indicators* 61, Part 2, 390–403. <https://doi.org/10.1016/j.ecolind.2015.09.040>
- Manfredi, S., Allacker, K., Pelletier, N., Chomkamsri, K., de Souza, D.M., 2012. Product environmental footprint (PEF) guide.
- Marques, A., Verones, F., Kok, M.T., Huijbregts, M.A., Pereira, H.M., 2017. How to quantify biodiversity footprints of consumption? A review of multi-regional input–output analysis and life cycle assessment. *Current Opinion in Environmental Sustainability* 29, 75–81. <https://doi.org/10.1016/j.cosust.2018.01.005>
- Martin, M., Brandão, M., 2017. Evaluating the Environmental Consequences of Swedish Food Consumption and Dietary Choices. *Sustainability* 9, 2227. <https://doi.org/10.3390/su9122227>
- Maskell, L.C., Radbourne, A., Norton, L.R., Reinsch, S., Alison, J., Bowles, L., Geudens, K., Robinson, D.A., 2023. Functional Agro-Biodiversity: An Evaluation of Current Approaches and Outcomes. *Land* 12, 2078. <https://doi.org/10.3390/land12112078>
- Mason, R., McCann, P., Connors, H., Hiron, S., White, O., forthcoming. Emerging opportunities from social innovation to enhance the transition to sustainable farming systems.
- Matej, S., Kaufmann, L., Semenchuk, P., Dullinger, S., Essl, F., Haberl, H., Kalt, G., Kastner, T., Lauk, C., Krausmann, F., Erb, K.-H., 2024. Options for reducing a city's global biodiversity footprint – The case of food consumption in Vienna. *Journal of Cleaner Production* 437, 140712. <https://doi.org/10.1016/j.jclepro.2024.140712>
- Mattas, K., Raptou, E., Alayidi, A., Yener, G., Baourakis, G., 2023. Assessing the Interlinkage between Biodiversity and Diet through the Mediterranean Diet Case. *Advances in Nutrition* 14, 570–582. <https://doi.org/10.1016/j.advnut.2023.03.011>



- Maxim, L., Spangenberg, J.H., O'Connor, M., 2009. An analysis of risks for biodiversity under the DPSIR framework. *Ecological Economics* 69, 12–23.  
<https://doi.org/10.1016/j.ecolecon.2009.03.017>
- Mbow, C., Van Noordwijk, M., Luedeling, E., Neufeldt, H., Minang, P.A., Kowero, G., 2014. Agroforestry solutions to address food security and climate change challenges in Africa. *Current Opinion in Environmental Sustainability* 6, 61–67.
- Melovic, B., Cirovic, D., Dudic, B., Vulic, T.B., Gregus, M., 2020. The Analysis of Marketing Factors Influencing Consumers' Preferences and Acceptance of Organic Food Products—Recommendations for the Optimization of the Offer in a Developing Market. *Foods* 9, 259.  
<https://doi.org/10.3390/foods9030259>
- Mhatre, P., Panchal, R., Singh, A., Bibyan, S., 2021. A systematic literature review on the circular economy initiatives in the European Union. *Sustainable Production and Consumption* 26, 187–202. <https://doi.org/10.1016/j.spc.2020.09.008>
- Moll, S., Skovgaard, M., Schepelmann, P., 2005. Sustainable use and management of natural resources, 2005th ed. European Environment Agency, Copenhagen.
- Mosquera-Losada, R., Amador-García, A., Muñoz-Ferreiro, N., Santiago-Freijanes, J.J., Ferreiro-Domínguez, N., Romero-Franco, R., Rigueiro-Rodríguez, A., 2017. Sustainable use of sewage sludge in acid soils within a circular economy perspective. *CATENA* 149, 341–348.  
<https://doi.org/10.1016/j.catena.2016.10.007>
- Muradin, M., Joachimiak-Lechman, K., Foltynowicz, Z., 2018. Evaluation of Eco-Efficiency of Two Alternative Agricultural Biogas Plants. *Applied Sciences* 8, 2083.  
<https://doi.org/10.3390/app8112083>
- 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>
- Muscio, A., Sisto, R., 2020. Are Agri-Food Systems Really Switching to a Circular Economy Model? Implications for European Research and Innovation Policy. *Sustainability* 12, 5554.  
<https://doi.org/10.3390/su12145554>
- Noss, R.F., 1990. Indicators for Monitoring Biodiversity: A Hierarchical Approach. *Conservation Biology* 4, 355–364. <https://doi.org/10.1111/j.1523-1739.1990.tb00309.x>
- Notarnicola, B., Tassielli, G., Renzulli, P.A., Castellani, V., Sala, S., 2017. Environmental impacts of food consumption in Europe. *Journal of Cleaner Production* 140, 753–765.  
<https://doi.org/10.1016/j.jclepro.2016.06.080>
- OECD, 2021. Biodiversity, natural capital and the economy: A policy guide for finance, economic and environment ministries (OECD Environment Policy Papers No. 26), OECD Environment Policy Papers. <https://doi.org/10.1787/1a1ae114-en>
- OECD, C., 2003. OECD environmental indicators: Development, measurement and use. Reference Paper), OECD Environmental Performance and Information Division, <http://www.oecd.org/env>.
- Oliver, R.Y., Meyer, C., Ranipeta, A., Winner, K., Jetz, W., 2021. Global and national trends, gaps, and opportunities in documenting and monitoring species distributions. *PLoS Biol* 19, e3001336. <https://doi.org/10.1371/journal.pbio.3001336>
- Ometto, A.R., Ramos, P.A.R., Lombardi, G., 2007. The benefits of a Brazilian agro-industrial symbiosis system and the strategies to make it happen. *Journal of Cleaner Production*, Approaching zero emissions 15, 1253–1258. <https://doi.org/10.1016/j.jclepro.2006.07.021>

- Ostandie, N., Giffard, B., Bonnard, O., Joubard, B., Richart-Cervera, S., Thiéry, D., Rusch, A., 2021. Multi-community effects of organic and conventional farming practices in vineyards. *Sci Rep* 11, 11979. <https://doi.org/10.1038/s41598-021-91095-5>
- Osteen, C., Gottlieb, J., Vasavada, U., 2012. Agricultural resources and environmental indicators. *USDA-ERS Economic Information Bulletin*.
- Pape, J., Rau, H., Fahy, F., Davies, A., 2011. Developing Policies and Instruments for Sustainable Household Consumption: Irish Experiences and Futures. *J Consum Policy* 34, 25–42. <https://doi.org/10.1007/s10603-010-9151-4>
- Payen, F.T., Evans, D.L., Falagán, N., Hardman, C.A., Kourmpetli, S., Liu, L., Marshall, R., Mead, B.R., Davies, J.A.C., 2022. How Much Food Can We Grow in Urban Areas? Food Production and Crop Yields of Urban Agriculture: A Meta-Analysis. *Earth's Future* 10, e2022EF002748. <https://doi.org/10.1029/2022EF002748>
- Pearman, P.B., Broennimann, O., Aavik, T., Albayrak, T., Alves, P.C., Aravanopoulos, F.A., Bertola, L.D., Biedrzycka, A., Buzan, E., Cubric-Curik, V., Djan, M., Fedorca, A., Fuentes-Pardo, A.P., Fussi, B., Godoy, J.A., Gugerli, F., Hoban, S., Holderegger, R., Hvilsom, C., Iacolina, L., Kalamujic Stroil, B., Klinga, P., Konopiński, M.K., Kopatz, A., Laikre, L., Lopes-Fernandes, M., McMahon, B.J., Mergeay, J., Neophytou, C., Pálsson, S., Paz-Vinas, I., Posledovich, D., Primmer, C.R., Raeymaekers, J.A.M., Rinkevich, B., Rolečková, B., Ruņģis, D., Schuerz, L., Segelbacher, G., Kavčič Sonnenschein, K., Stefanovic, M., Thurfjell, H., Träger, S., Tsvetkov, I.N., Velickovic, N., Vergeer, P., Vernesi, C., Vilà, C., Westergren, M., Zachos, F.E., Guisan, A., Bruford, M., 2024. Monitoring of species' genetic diversity in Europe varies greatly and overlooks potential climate change impacts. *Nat Ecol Evol* 8, 267–281. <https://doi.org/10.1038/s41559-023-02260-0>
- Pellaton, R., Csecserits, A., Szitár, K., Rédei, T., Batáry, P., Báldi, A., 2023. Grazing and boundaries favour weedy plants with functional traits beneficial for pollinators. *Global Ecology and Conservation* 48, e02717. <https://doi.org/10.1016/j.gecco.2023.e02717>
- Peterson, H.H., Taylor, M.R., Baudouin, Q., 2015. Preferences of locavores favoring community supported agriculture in the United States and France. *Ecological Economics* 119, 64–73.
- Petruzzello, M., Gold, M.A., 2016. The new encyclopaedia Britannica, 15th ed. ed. Encyclopaedia britannica, Chicago Paris.
- Plieninger, T., Hartel, T., Martín-López, B., Beaufoy, G., Bergmeier, E., Kirby, K., Montero, M.J., Moreno, G., Oteros-Rozas, E., Van Uytvanck, J., 2015. Wood-pastures of Europe: Geographic coverage, social–ecological values, conservation management, and policy implications. *Biological Conservation* 190, 70–79. <https://doi.org/10.1016/j.biocon.2015.05.014>
- Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. *Science* 360, 987–992. <https://doi.org/10.1126/science.aag0216>
- Poux, X., Aubert, P.-M., 2018. An agroecological Europe in 2050: multifunctional agriculture for healthy eating. Findings from the Ten Years For Agroecology (TYFA) modelling exercise, Iddri-AScA, Study 9, 18.
- Read, Q.D., Hondula, K.L., Muth, M.K., 2022. Biodiversity effects of food system sustainability actions from farm to fork. *Proc. Natl. Acad. Sci. U.S.A.* 119, e2113884119. <https://doi.org/10.1073/pnas.2113884119>
- Reisman, A., 2012. A greenhouse in the city: The uses and roles of community-oriented urban greenhouses (Master's Thesis). Tufts University.
- Rieger, J., Freund, F., Offermann, F., Geibel, I., Gocht, A., 2023. From fork to farm: Impacts of more sustainable diets in the EU -27 on the agricultural sector. *J Agricultural Economics* 74, 764–784. <https://doi.org/10.1111/1477-9552.12530>

- Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S.I., Lambin, E., Lenton, T., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R., Fabry, V., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009. Planetary Boundaries: Exploring the Safe Operating Space for Humanity. *Ecology and Society* 14. <https://doi.org/10.5751/ES-03180-140232>
- Röös, E., Karlsson, H., Witthöft, C., Sundberg, C., 2015. Evaluating the sustainability of diets—combining environmental and nutritional aspects. *Environmental Science & Policy* 47, 157–166. <https://doi.org/10.1016/j.envsci.2014.12.001>
- Röös, E., Larsson, J., Resare Sahlin, K., Jonell, M., Lindahl, T., André, E., Säll, S., Harring, N., Persson, M., 2021. Policy options for sustainable food consumption: Review and recommendations for Sweden.
- Röös, E., Mayer, A., Muller, A., Kalt, G., Ferguson, S., Erb, K.-H., Hart, R., Matej, S., Kaufmann, L., Pfeifer, C., 2022. Agroecological practices in combination with healthy diets can help meet EU food system policy targets. *Science of The Total Environment* 847, 157612.
- Sala, S., Benini, L., Beylot, A., Castellani, V., Cerutti, A., Corrado, S., Crenna, E., Diaconu, E., Sanyé-Mengual, E., Secchi, M., 2019. Consumption and Consumer Footprint: methodology and results. *Indicators and Assessment of the Environmental Impact of European Consumption*. Luxembourg.
- Sala, S., De, L.V., Sanyé, M.E., 2023. Food consumption and waste: environmental impacts from a supply chain perspective [WWW Document]. JRC Publications Repository. URL <https://publications.jrc.ec.europa.eu/repository/handle/JRC129245> (accessed 1.24.24).
- Sanchez Lopez, J., Patinha Caldeira, C., De Laurentiis, V., Sala, S., Avraamides, M., 2020. Brief on food waste in the European Union. Joint Research Centre (JRC), Italy.
- Sandström, V., Valin, H., Krisztin, T., Havlík, P., Herrero, M., Kastner, T., 2018. The role of trade in the greenhouse gas footprints of EU diets. *Global Food Security* 19, 48–55. <https://doi.org/10.1016/j.gfs.2018.08.007>
- Santos, M., Cajaiba, R.L., Bastos, R., Gonzalez, D., Petrescu Bakış, A.-L., Ferreira, D., Leote, P., Barreto Da Silva, W., Cabral, J.A., Gonçalves, B., Mosquera-Losada, M.R., 2022. Why Do Agroforestry Systems Enhance Biodiversity? Evidence From Habitat Amount Hypothesis Predictions. *Front. Ecol. Evol.* 9, 630151. <https://doi.org/10.3389/fevo.2021.630151>
- Sanyé-Mengual, E., Biganzoli, F., Valente, A., Pfister, S., Sala, S., 2023. What are the main environmental impacts and products contributing to the biodiversity footprint of EU consumption? A comparison of life cycle impact assessment methods and models. *Int J Life Cycle Assess* 28, 1194–1210. <https://doi.org/10.1007/s11367-023-02169-7>
- SAPEA, 2023. Towards sustainable food consumption: promoting healthy, affordable and sustainable food consumption choices. Publications Office, LU.
- Scarborough, P., Clark, M., Cobiac, L., Papier, K., Knuppel, A., Lynch, J., Harrington, R., Key, T., Springmann, M., 2023. Vegans, vegetarians, fish-eaters and meat-eaters in the UK show discrepant environmental impacts. *Nat Food* 4, 565–574. <https://doi.org/10.1038/s43016-023-00795-w>
- Scharber, H., Dancs, A., 2016. Do locavores have a dilemma? Economic discourse and the local food critique. *Agric Hum Values* 33, 121–133. <https://doi.org/10.1007/s10460-015-9598-7>
- Scholl, G., Rubik, F., Kalimo, H., Biedenkopf, K., Söbech, Ó., 2010. Policies to promote sustainable consumption: Innovative approaches in Europe. *Natural Resources Forum* 34, 39–50. <https://doi.org/10.1111/j.1477-8947.2010.01294.x>
- Selvan, T., Panmei, L., Murasing, K.K., Guleria, V., Ramesh, K.R., Bhardwaj, D.R., Thakur, C.L., Kumar, D., Sharma, P., Digvijaysinh Umedsinh, R., Kayalvizhi, D., Deshmukh, H.K., 2023.

- Circular economy in agriculture: unleashing the potential of integrated organic farming for food security and sustainable development. *Front. Sustain. Food Syst.* 7. <https://doi.org/10.3389/fsufs.2023.1170380>
- Shurson, G.C., 2020. “What a Waste”—Can We Improve Sustainability of Food Animal Production Systems by Recycling Food Waste Streams into Animal Feed in an Era of Health, Climate, and Economic Crises? *Sustainability* 12, 7071. <https://doi.org/10.3390/su12177071>
- SITRA, 2022. Tackling root causes: Halting biodiversity loss through the circular economy.
- Sokal, R.R., 1963. The principles and practice of numerical taxonomy. *Taxon* 190–199.
- Sporchia, F., Antonelli, M., Aguilar-Martínez, A., Bach-Faig, A., Caro, D., Davis, K.F., Sonnino, R., Galli, A., 2024. Zero hunger: future challenges and the way forward towards the achievement of sustainable development goal 2. *Sustain Earth Reviews* 7, 10. <https://doi.org/10.1186/s42055-024-00078-7>
- Sporchia, F., Caro, D., 2023. Exploring the potential of circular solutions to replace inorganic fertilizers in the European Union. *Science of The Total Environment* 892, 164636. <https://doi.org/10.1016/j.scitotenv.2023.164636>
- Sporchia, F., Galli, A., Kastner, T., Pulselli, F.M., Caro, D., 2023. The environmental footprints of the feeds used by the EU chicken meat industry. *Science of The Total Environment* 886, 163960. <https://doi.org/10.1016/j.scitotenv.2023.163960>
- Sporchia, F., Kebreab, E., Caro, D., 2021a. Assessing the multiple resource use associated with pig feed consumption in the European Union. *Science of The Total Environment* 759, 144306. <https://doi.org/10.1016/j.scitotenv.2020.144306>
- Sporchia, F., Thomsen, M., Caro, D., 2021b. Drivers and trade-offs of multiple environmental stressors from global rice. *Sustainable Production and Consumption* 26, 16–32. <https://doi.org/10.1016/j.spc.2020.09.009>
- Spotswood, E.N., Beller, E.E., Grossinger, R., Grenier, J.L., Heller, N.E., Aronson, M.F.J., 2021. The Biological Deserts Fallacy: Cities in Their Landscapes Contribute More than We Think to Regional Biodiversity. *BioScience* 71, 148–160. <https://doi.org/10.1093/biosci/biaa155>
- Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., Vries, W. de, Wit, C.A. de, Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. *Science* 347, 1259855. <https://doi.org/10.1126/science.1259855>
- Strazza, C., Magrassi, F., Gallo, M., Del Borghi, A., 2015. Life Cycle Assessment from food to food: A case study of circular economy from cruise ships to aquaculture. *Sustainable Production and Consumption, Sustainability issues in the food–energy–water nexus* 2, 40–51. <https://doi.org/10.1016/j.spc.2015.06.004>
- Szilágyi, A., Mészáros, F., Kun, R., Sárospataki, M., 2021. Pollinator Communities in Some Selected Hungarian Conventional, Organic and Permaculture Horticultures, in: *The 1st International Electronic Conference on Biological Diversity, Ecology and Evolution*. Presented at the BDEE 2021, MDPI, p. 13. <https://doi.org/10.3390/BDEE2021-09492>
- Tan, R.R., 2019. Data challenges in optimizing biochar-based carbon sequestration. *Renewable and Sustainable Energy Reviews* 104, 174–177. <https://doi.org/10.1016/j.rser.2019.01.032>
- Teigiserova, D.A., Hamelin, L., Thomsen, M., 2020. Towards transparent valorization of food surplus, waste and loss: Clarifying definitions, food waste hierarchy, and role in the circular economy. *Science of The Total Environment* 706, 136033. <https://doi.org/10.1016/j.scitotenv.2019.136033>
- Thapa Karki, S., Bennett, A.C.T., Mishra, J.L., 2021. Reducing food waste and food insecurity in the UK: The architecture of surplus food distribution supply chain in addressing the sustainable

- development goals (Goal 2 and Goal 12.3) at a city level. *Industrial Marketing Management* 93, 563–577. <https://doi.org/10.1016/j.indmarman.2020.09.019>
- Timmermans, B., Achten, W.M.J., 2018. From value-added tax to a damage and value-added tax partially based on life cycle assessment: principles and feasibility. *Int J Life Cycle Assess* 23, 2217–2247. <https://doi.org/10.1007/s11367-018-1439-7>
- Torralba, M., Fagerholm, N., Burgess, P.J., Moreno, G., Plieninger, T., 2016. Do European agroforestry systems enhance biodiversity and ecosystem services? A meta-analysis. *Agriculture, Ecosystems & Environment* 230, 150–161. <https://doi.org/10.1016/j.agee.2016.06.002>
- Trewern, J., Chenoweth, J., Christie, I., 2022. “Does it change the nature of food and capitalism?” Exploring expert perspectives on public policies for a transition to ‘less and better’ meat and dairy. *Environmental Science & Policy* 128, 110–120.
- Trokanas, N., Cecelja, F., Yu, M., Raafat, T., 2014. Optimising Environmental Performance of Symbiotic Networks Using Semantics, in: Klemeš, J.J., Varbanov, P.S., Liew, P.Y. (Eds.), *Computer Aided Chemical Engineering, 24 European Symposium on Computer Aided Process Engineering*. Elsevier, pp. 847–852. <https://doi.org/10.1016/B978-0-444-63456-6.50142-3>
- Tscharntke, T., Grass, I., Wanger, T.C., Westphal, C., Batáry, P., 2021. Beyond organic farming – harnessing biodiversity-friendly landscapes. *Trends in Ecology & Evolution* 36, 919–930. <https://doi.org/10.1016/j.tree.2021.06.010>
- Tuck, S.L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L.A., Bengtsson, J., 2014. Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology* 51, 746–755. <https://doi.org/10.1111/1365-2664.12219>
- Tucker, C., 2018. Using environmental imperatives to reduce meat consumption: perspectives from New Zealand. *Kōtuitui: New Zealand Journal of Social Sciences Online* 13, 99–110. <https://doi.org/10.1080/1177083X.2018.1452763>
- UCPH, 2023. Most Europeans are reducing their meat consumption, EU-funded survey finds [WWW Document]. URL <https://food.ku.dk/english/news/2023/most-europeans-are-reducing-their-meat-consumption-eu-funded-survey-finds/> (accessed 6.4.24).
- UN DESA, 2021. Circular agriculture for sustainable rural development.
- UNEP, 2015. Uncovering pathways to an inclusive green economy: summary for policymakers. United Nations Environment Programme.
- UNFCCC, 2015. Paris Agreement.
- United Nations (Ed.), 2000. Classifications of expenditure according to purpose: classification of the functions of government, ST ESA STAT SER M. United Nations, New York.
- Urban, D.L., O’Neill, R.V., Shugart Jr, H.H., 1987. A hierarchical perspective can help scientists understand spatial patterns. *BioScience* 37, 119–127.
- van Selm, B., Frehner, A., de Boer, I.J.M., van Hal, O., Hijbeek, R., van Ittersum, M.K., Talsma, E.F., Lesschen, J.P., Hendriks, C.M.J., Herrero, M., van Zanten, H.H.E., 2022. Circularity in animal production requires a change in the EAT-Lancet diet in Europe. *Nat Food* 3, 66–73. <https://doi.org/10.1038/s43016-021-00425-3>
- van Zanten, H.H.E., Simon, W., van Selm, B., Wacker, J., Maindl, T.I., Frehner, A., Hijbeek, R., van Ittersum, M.K., Herrero, M., 2023. Circularity in Europe strengthens the sustainability of the global food system. *Nat Food* 4, 320–330. <https://doi.org/10.1038/s43016-023-00734-9>
- van Zanten, H.H.E., van Ittersum, M.K., de Boer, I.J.M., 2019. The role of farm animals in a circular food system. *Global Food Security* 21, 18–22. <https://doi.org/10.1016/j.gfs.2019.06.003>
- Vanham, D., Bruckner, M., Schwarzmüller, F., Schyns, J., Kastner, T., 2023. Multi-model assessment identifies livestock grazing as a major contributor to variation in European



- Union land and water footprints. *Nat Food* 4, 575–584. <https://doi.org/10.1038/s43016-023-00797-8>
- Vanham, D., Leip, A., Galli, A., Kastner, T., Bruckner, M., Uwizeye, A., Van Dijk, K., Ercin, E., Dalin, C., Brandão, M., Bastianoni, S., Fang, K., Leach, A., Chapagain, A., Van Der Velde, M., Sala, S., Pant, R., Mancini, L., Monforti-Ferrario, F., Carmona-Garcia, G., Marques, A., Weiss, F., Hoekstra, A.Y., 2019. Environmental footprint family to address local to planetary sustainability and deliver on the SDGs. *Science of The Total Environment* 693, 133642. <https://doi.org/10.1016/j.scitotenv.2019.133642>
- Vanham, D., Mekonnen, M.M., Hoekstra, A.Y., 2013. The water footprint of the EU for different diets. *Ecological Indicators* 32, 1–8. <https://doi.org/10.1016/j.ecolind.2013.02.020>
- 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. *Reg Environ Change* 20, 103. <https://doi.org/10.1007/s10113-020-01685-8>
- Velasco-Muñoz, J.F., Mendoza, J.M.F., Aznar-Sánchez, J.A., Gallego-Schmid, A., 2021. Circular economy implementation in the agricultural sector: Definition, strategies and indicators. *Resources, Conservation and Recycling* 170, 105618. <https://doi.org/10.1016/j.resconrec.2021.105618>
- Vieira Veríssimo, N., Ussemame Mussagy, C., Alves Oshiro, A., Nóbrega Mendonça, C.M., Carvalho Santos-Ebinuma, V. de, Pessoa, A., Souza Oliveira, R.P. de, Brandão Pereira, J.F., 2021. From green to blue economy: Marine biorefineries for a sustainable ocean-based economy. *Green Chemistry* 23, 9377–9400. <https://doi.org/10.1039/D1GC03191K>
- Vinci, G., Maddaloni, L., Prencipe, S.A., Ruggeri, M., Di Loreto, M.V., 2022. A Comparison of the Mediterranean Diet and Current Food Patterns in Italy: A Life Cycle Thinking Approach for a Sustainable Consumption. *IJERPH* 19, 12274. <https://doi.org/10.3390/ijerph191912274>
- Vlajic, J.V., Cunningham, E., Hsiao, H.-I., Smyth, B., Walker, T., 2021. Mapping Facets of Circularity: Going Beyond Reduce, Reuse, Recycle in Agri-Food Supply Chains, in: Mor, R.S., Panghal, A., Kumar, V. (Eds.), *Challenges and Opportunities of Circular Economy in Agri-Food Sector: Rethinking Waste*. Springer, Singapore, pp. 15–40. [https://doi.org/10.1007/978-981-16-3791-9\\_2](https://doi.org/10.1007/978-981-16-3791-9_2)
- Wackernagel, M., Rees, W., 1996. *Our Ecological Footprint: Reducing Human Impact on the Earth*. New Society Publishers, Gabriola Island, British Columbia, Canada.
- Wackernagel, M., Schulz, N.B., Deumling, D., Linares, A.C., Jenkins, M., Kapos, V., Monfreda, C., Loh, J., Myers, N., Norgaard, R., others, 2002. Tracking the ecological overshoot of the human economy. *Proceedings of the national Academy of Sciences* 99, 9266–9271.
- Walker, C., Gibney, E.R., Mathers, J.C., Hellweg, S., 2019. Comparing environmental and personal health impacts of individual food choices. *Science of The Total Environment* 685, 609–620. <https://doi.org/10.1016/j.scitotenv.2019.05.404>
- Walker, C., Pfister, S., Hellweg, S., 2021. Methodology and optimization tool for a personalized low environmental impact and healthful diet specific to country and season. *J of Industrial Ecology* 25, 1147–1160. <https://doi.org/10.1111/jiec.13131>
- Wallis De Vries, M.F., Parkinson, A.E., Dulphy, J.P., Sayer, M., Diana, E., 2007. Effects of livestock breed and grazing intensity on biodiversity and production in grazing systems. 4. Effects on animal diversity. *Grass and Forage Science* 62, 185–197. <https://doi.org/10.1111/j.1365-2494.2007.00568.x>
- Wang, C., Tang, Y., 2019. A global meta-analyses of the response of multi-taxa diversity to grazing intensity in grasslands. *Environ. Res. Lett.* 14, 114003. <https://doi.org/10.1088/1748-9326/ab4932>



- Wezel, A., Herren, B.G., Kerr, R.B., Barrios, E., Gonçalves, A.L.R., Sinclair, F., 2020. Agroecological principles and elements and their implications for transitioning to sustainable food systems. A review. *Agron. Sustain. Dev.* 40, 40. <https://doi.org/10.1007/s13593-020-00646-z>
- Willer, H., Schlatter, B., Travnicek, J., Schaak, D., 2023. Organic farming and market development in Europe and the European Union. Research Institute of Organic Agriculture FiBL and IFOAM-Organics International.
- Willer, H., Trávníček, J., Schlatter, B., 2024. The world of organic agriculture. Statistics and emerging trends 2024.
- Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., Jonell, M., Clark, M., Gordon, L.J., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J.A., De Vries, W., Majele Sibanda, L., Afshin, A., Chaudhary, A., Herrero, M., Agustina, R., Branca, F., Lartey, A., Fan, S., Crona, B., Fox, E., Bignet, V., Troell, M., Lindahl, T., Singh, S., Cornell, S.E., Srinath Reddy, K., Narain, S., Nishtar, S., Murray, C.J.L., 2019. Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. *The Lancet* 393, 447–492. [https://doi.org/10.1016/S0140-6736\(18\)31788-4](https://doi.org/10.1016/S0140-6736(18)31788-4)
- Winquist, E., Rikkonen, P., Pyysiäinen, J., Varho, V., 2019. Is biogas an energy or a sustainability product? - Business opportunities in the Finnish biogas branch. *Journal of Cleaner Production* 233, 1344–1354. <https://doi.org/10.1016/j.jclepro.2019.06.181>
- Winqvist, C., Ahnström, J., Bengtsson, J., 2012. Effects of organic farming on biodiversity and ecosystem services: taking landscape complexity into account. *Annals of the New York Academy of Sciences* 1249, 191–203. <https://doi.org/10.1111/j.1749-6632.2011.06413.x>
- Wolff, F., Schönherr, N., Heyen, D.A., 2017. Effects and success factors of sustainable consumption policy instruments: a comparative assessment across Europe. *Journal of Environmental Policy & Planning* 19, 457–472. <https://doi.org/10.1080/1523908X.2016.1254035>
- Yuan, G.N., Marquez, G.P.B., Deng, H., Lu, A., Fabella, M., Salonga, R.B., Ashardiono, F., Cartagena, J.A., 2022. A review on urban agriculture: technology, socio-economy, and policy. *Heliyon* 8, e11583. <https://doi.org/10.1016/j.heliyon.2022.e11583>
- Zarbà, C., Chinnici, G., La Via, G., Bracco, S., Pecorino, B., D’Amico, M., 2021. Regulatory Elements on the Circular Economy: Driving into the Agri-Food System. *Sustainability* 13, 8350. <https://doi.org/10.3390/su13158350>
- Zasada, I., Weltin, M., Zoll, F., Benninger, S.L., 2020. Home gardening practice in Pune (India), the role of communities, urban environment and the contribution to urban sustainability. *Urban Ecosyst* 23, 403–417. <https://doi.org/10.1007/s11252-019-00921-2>

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