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BFAT

Biodiversity Framework for Agricultural Transition

Key methodological principles for an improved modelling of the impact of agricultural production on biodiversity

Methodology synthesis

March 2026

INTRODUCTION

The Biodiversity Framework for Agricultural Transition (BFAT) project is led by I Care, co-developed with CDC Biodiversité, and has been supported by Biosphères and Agrosolutions experts. It aims at improving the quantitative estimation of the environmental impact of agricultural practices.

This document synthetises key methodological principles of the BFAT methodology and database. It details the core methodological approach for estimating the impact of agricultural commodities according to the practices implemented in the field.

Additional papers associated with these methodological principles will further detail the specific methodology, data used and the preliminary results from case studies of commodities. 3 methodological notes are about to be released:

- Methodological notes for temporary crops – wheat case study
- Methodological notes for animal farming and grassland management – beef case study
- Methodological notes for perennial crops – palm case study

This note is intended for relevant stakeholders and can be circulated.

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1. The need for improved environmental assessment of agricultural commodities

The progressive overshoot of planetary boundaries highlights the urgent need to transform our economic systems. Of all the sectors, agriculture stands out as one of the most complex to redesign. To support decision towards a more sustainable agriculture, tools are needed to identify and quantify potential levers for reducing environmental pressures. In this context, environmental assessment methods such as Life Cycle Assessment (LCA), ecological footprinting, and other impact assessment frameworks play a key role.

However, although the environmental impacts of agricultural commodities are already widely calculated, current assessments rarely differentiate between farming practices, despite scientific evidence showing that several practices have positive ecological outcomes[1][2]. When LCAs do include management practices, they often do so in a simplified way, most commonly by adjusting yields[3]. Practices such as extended crop rotations, crop diversification, or the integration of legumes can reduce the demand for fertilizers, thereby lowering eutrophication potential and the emissions associated with nitrogen fertilizer production. Yet these positive effects are rarely standardized or consistently represented within current assessment frameworks. Soil quality, functional biodiversity, and overall agroecosystem health remain especially difficult to integrate into LCA. **There is a dual need for improved modelling of existing standardized indicators to document agroecological practices fairly, and for additional indicator(s) to characterize the ecological quality of agrosystems considering the practices implemented.**

The main challenge with the inclusion of practices in environmental impact assessments is the lack of standardized datasets documenting the extent to which biodiversity-friendly practices are implemented on a national scale. Data gaps, methodological inconsistencies, and the absence of global assessment frameworks limit the ability to make reliable cross-country comparisons. This is why **research initiatives that aim to synthesize existing literature, harmonize indicators, and quantify environmental impacts across a variety of agricultural practices are essential.** Such work is crucial not only to support the redesign of agricultural systems, but also to inform public policies and business decisions that seek to evaluate environmental performance and progress toward biodiversity conservation. In addition, this work aligns with the converging expectations of international initiatives and regulations for the disclosure of environmental impacts (such as SBTN, TNFD and the CSRD), all of which call for harmonized, science-based and comparable indicators to assess nature-related impacts, risks and dependencies.

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[1] Félix Teillard and others, 'What Does Life-Cycle Assessment of Agricultural Products Need for More Meaningful Inclusion of Biodiversity?', *Journal of Applied Ecology*, 53.5 (2016), pp. 1422–29, doi:10.1111/1365-2664.12683.

[2] Giulio Paolo Agnusdei and others, 'Life Cycle Thinking in Organic Agriculture: A Systematic Literature Review of Methodologies, Trends and Research Impact', *Journal of Agriculture and Food Research*, 24 (2025), p. 102361, doi:10.1016/j.jafr.2025.102361.

[3] Steve Harris and Venky Narayanaswamy, *A Literature Review of Life Cycle Assessment in Agriculture* (RIRDC, 2009).

2. Objectives and outputs

2.1. Objectives

In a context of rapidly emerging standards, companies are expected to commit to reducing their upstream environmental impacts. However, the tools currently available do not yet enable them to report on their efforts to support agroecological practices. BFAT aims to bridge this gap.

The overarching goal of BFAT is to enable economic actors to demonstrate the potentially reduced environmental impacts on ecosystems and biodiversity resulting from the agroecological practices implemented in their operations and/or supported in their supply chain. This means:

- **Improving & spreading knowledge on agroecological practices**, while considering the balance between yields, extensification, pressure reduction and potential gains in biodiversity;
- **Developing methodological approaches to quantify the effects of best practices** across the value chain **on diverse environmental flows;**
- **Providing new impact factors and state of nature indicators** adapted to low- and positive-impact practices, using various existing environmental metrics.

The results of BFAT are intended as indicators for piloting the agroecological transition. Therefore, BFAT may also be of interest to public authorities looking to pilot territorial management and agricultural development in line with sustainable territorial strategies.

2.2. Outputs

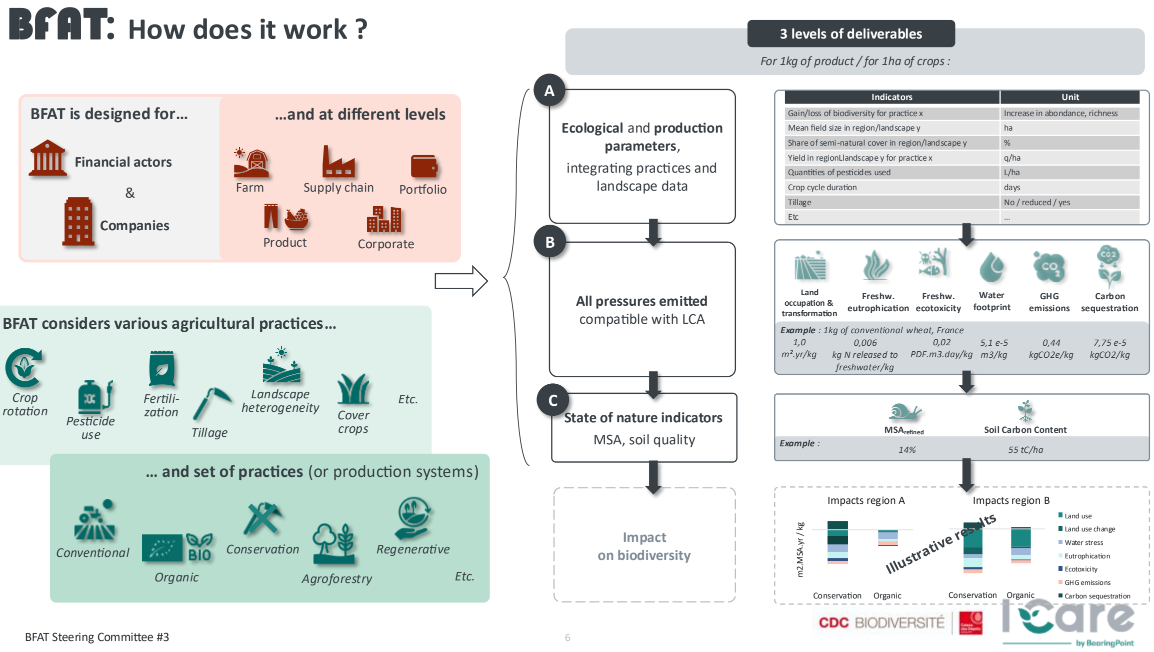
BFAT is primarily a methodology that provides a structured vision of the effects of agricultural practices on various indicators:

- **Production-related and ecological indicators** that describe specific production modes associated with the implementation of certain agricultural practices, compared to “no practice” or “standard practice”. These indicators relate to production parameters (e.g. yield, agricultural inputs used, mean field size, intensity of practices) or other ecological indicators (e.g. natural or semi-natural elements).
- **Pressure indicators** used in LCA. These indicators (see Table 1 – List of pressures and associated indicators considered in BFAT) are estimated based on geography, practices and commodities, and are calculated using the aforementioned ecological and production-related indicators, combined with effect factors. These outputs aim to be compatible with LCA approaches, as the aim of BFAT is to increase the granularity of life cycle assessment methods for food & agricultural products, taking into account production practices.
- **State-of-nature indicators**, such as ecosystem condition estimated with the MSA metric or soil carbon content (see State of nature - Ecosystem condition assessment). These indicators complement LCA approaches and are primarily

used in international frameworks (e.g. SBTN) or in biodiversity footprint assessments.

Indicators are provided either per unit of commodity produced (e.g. 1kg) or for a production area (e.g. in ha), taking into account the specific geographical, agricultural and ecological context.

These results can be used with several environmental assessment tools, including LCA (see Deliverable B) or biodiversity footprint assessment tools (such as the Global Biodiversity Score developed by CDC Biodiversité, or the Biodiversity Footprint Suite developed by I Care by BearingPoint). Biodiversity footprint assessment tools use pressure-impact relationships or state-of-nature indicators to estimate biodiversity impacts.



BFAT: How does it work ?

BFAT is designed for…

Financial actors

&

Companies

…and at different levels

Farm

Supply chain

Portfolio

Product

Corporate

BFAT considers various agricultural practices…

Crop rotation

Pesticide use

Fertili-zation

Tillage

Landscape heterogeneity

Cover crops

Etc.

… and set of practices (or production systems)

Conventional

Organic

Conservation

Agroforestry

Regenerative

Etc.

3 levels of deliverables

For 1kg of product / for 1ha of crops :

A — **Ecological** and **production parameters**, integrating practices and landscape data

Indicators	Unit
Gain/loss of biodiversity for practice x	Increase in abondance, richness
Mean field size in region/landscape y	ha
Share of semi-natural cover in region/landscape y	%
Yield in regionLlandscape y for practice x	q/ha
Quantities of pesticides used	L/ha
Crop cycle duration	days
Tillage	No / reduced / yes
Etc	…

B — **All pressures emitted** compatible with LCA

Land occupation & transformation	Freshw. eutrophication	Freshw. ecotoxicity	Water footprint	GHG emissions	Carbon sequestration
1,0 m².yr/kg	0,006 kg N released to freshwater/kg	0,02 PDF.m3.day/kg	5,1 e-5 m3/kg	0,44 kgCO2e/kg	7,75 e-5 kgCO2/kg

Example *: 1kg of conventional wheat, France*

C — **State of nature indicators** MSA, soil quality

	$MSA_{refined}$	Soil Carbon Content
Example *:*	*14%*	*55 tC/ha*

Impact on biodiversity

Figure 1 - BFAT - Scope & deliverables

3. Scope of BFAT

3.1. Commodities

BFAT aims at **covering all major agricultural commodities.** This includes **primary crops and livestock commodities** from the FAOSTAT classification[1], with **temporary crops** (such as cereals, pulses, roots and tubers, sugar crops, fibre, oil crops and vegetables), **perennial crops** (fruits and berries, nuts, oil seeds trees, other perennial crops), and **animal production** (egg, milk and meat and associated byproducts).

Due to the significant time and work required to collect data on each commodity, its production locations and production methods, **most relevant commodities** for the project's funders and those with the greatest environmental impact were addressed at first.

Additional raw materials derived from crop and livestock production, such as oil palm, leather, cashmere, wool and cheese, also play a significant role in global trade. This is why these commodities are also included in BFAT scope. Their environmental impacts are mainly evaluated by allocating the impacts of primary crops to derived products.

3.2. Geographical scale

BFAT has a global coverage, as the methodology is intended to be applicable to all countries and regions relevant to agricultural commodities. For each commodity, the main countries or regions of production are targeted first[2]. **The ambition is to capture the geographical variability of production practices, as well as variations in pressures and impacts depending on the location**. For instance, the impact of tillage on carbon removal varies significantly between tropical and temperate regions [3], and the rate of fertilizer application varies greatly between countries and farming practices[4].

The BFAT methodology enables assessments to be conducted at various geographical scales. **Impacts are at least calculated at the country level**. However, when sufficient data exist, or when the literature highlights meaningful differences within a single country, impacts can also be estimated for specific subnational regions.

The accuracy and specificity of results to the local context will **depend heavily on the availability and geographical granularity of the data**. While some regions and crops have extensive documentation on production practices and ecological conditions, others have been studied only partially. As a result, pressure indicators, impacts, as well as ecosystem condition in the field, may be inferred from global models adapted with assumptions, rather than being measured directly.

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[1] FAOSTAT production domain: FAOSTAT

[2] approx. >80% of the volumes documented worldwide on a yearly basis, considered FAOSTAT data

[3] Maren Oelbermann, R. Paul Voroney, and A. M. Gordon, 'Carbon Sequestration in Tropical and Temperate Agroforestry Systems: A Review with Examples from Costa Rica and Southern Canada', *Agriculture, Ecosystems & Environment*, 104.3 (2004), pp. 359–77, doi:10.1016/j.agee.2004.04.001.

[4] Cameron I. Ludemann and others, 'Global Data on Fertilizer Use by Crop and by Country', *Scientific Data*, 9.1 (2022), p. 501, doi:10.1038/s41597-022-01592-z; Pallab Mozumder and Robert P. Berrens, 'Inorganic Fertilizer Use and Biodiversity Risk: An Empirical Investigation', *Ecological Economics*, 62.3–4 (2007), pp. 538–43, doi:10.1016/j.ecolecon.2006.07.016.

3.3. Value chain perimeter

In theory, the selected indicators can cover the entire value chain. In practice, however, the perimeter of the value chain evaluated will vary depending on the commodity under consideration. **Our primary objective is to assess the effects of agricultural practices on plot and across the broader landscape (water basin or landscape)**, on both production and environmental indicators.

Nevertheless, certain stages in the value chain, both upstream and downstream, may generate significant impacts, and some agricultural practices can influence upstream processes (esp. fertilization). To address these effects, the impact assessment framework aligns with cradle-to-gate life cycle assessment (LCA) approaches, enabling the integration of relevant upstream impacts on a case-by-case basis.

In addition, impact allocation procedures are applied to raw commodities derived from agricultural production, such as cotton fibre, palm oil or wool. These procedures enable the upstream impacts to be systematically attributed to the specific raw commodity being assessed.

Impacts are attributed using economic allocations, as this better captures the incentive of a production for farmers, i.e. which products drive the activity and the subsequent impacts, and which products are by-products, given their relative market values. The economic allocation is also the most used type of allocation in attributional LCAs in the Agri-food sector[1]. Economic allocations are made for crops (grains and straws), as well as for animal farming systems (meat, milk, hides, etc.), including animal feed (economic allocation between soy oil and soybean cake for instance).

3.4. Evaluated metrics

3.4.1. Pressures

BFAT aims at assessing the environmental impacts of agricultural commodities, following the IPBES structure of biodiversity and ecosystems loss impacts drivers. The following pressures will be documented in BFAT.

Table 1 – List of pressures and associated indicators considered in BFAT

Driver (IPBES)	Pressure	Indicators
Land/sea use & use change	Land use	Land occupation (m² and m².yr) per land use type (incl. management intensity) and geography
	Land use change	Land use change per land use type (incl. management intensity) and geography (m²)
Direct exploitation	Water use	Surface water and groundwater consumption (m^3)
Climate change	Non-FLAG emissions	GHG emissions (kgCO2eq)
	FLAG LUC	GHG emissions (kgCO2eq)
	FLAG others	GHG emissions (kgCO2eq)
	Carbon removals	Carbon removals (kgCO2)

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[1] Daniela Dominguez Aldama and others, 'Allocation Methods in Life Cycle Assessments (LCAs) of Agri-Food Co-Products and Food Waste Valorization Systems: Systematic Review and Recommendations', *Journal of Cleaner Production*, 421 (2023), p. 138488, doi:10.1016/j.jclepro.2023.138488.

Pollution	Terrestrial acidification	Emissions of N (kg N emitted to air)
	Freshwater eutrophication	Emissions of N & P (kg P emitted and kg N emitted to freshwater)
	Freshwater ecotoxicity	CTUe (based on the emissions of ecotoxic substances)

These pressures were selected because they are material for the agricultural sector and are key pressures in LCA or other environmental impacts assessment methods. The methods used to estimate the selected indicators are briefly outlined in the 4.4 Pressure modelling and will be more deeply explained in future case studies.

Several biodiversity pressures were **excluded** from the present assessment due to methodological and data limitations, that are commonly highlighted in the literature.

Invasive alien species could not be covered because there is currently no widely accepted, operational life cycle impact assessment (LCIA) methodology to consistently link product systems to biological invasions.

Marine ecosystem degradation was not explicitly covered as it is only material for a limited number of specific commodities, and BFAT mainly covers inland agriculture.

Disturbance pressures, such as noise and light pollution, were not included as considered limited in case of farming practices, esp. compared to other sectors (e.g. energy production, manufacturing, construction, mining).

Finally, the **impact of solid waste** (e.g. plastic pollution) was excluded, as biodiversity-related impact pathways are not yet fully standardized within LCA, despite the existence of parallel and emerging frameworks.

These aspects represent important areas for future methodological development and potential integration.

3.4.2. State of nature

In the context of the DSPIR framework (“Driving Forces – Pressures – State – Impacts – Responses”), the state of nature is defined as the “State” component and **refers to “the quantity and quality of physical, chemical, and biological components of the natural and built environment**”[1]. To assess the state of nature in agroecosystems at the plot scale, we will focus on **ecosystem condition for the biological component and carbon removal for the chemical component**, as shaped by production modes and practices.

Ecosystem condition

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[1] Patricia Bradley and S. Yee, *Using the DPSIR Framework to Develop a Conceptual Model: Technical Support Document* (Narragansett, RI, 2015), doi:10.13140/RG.2.1.1870.7608.

More specifically, for the biological component, we use the TNFD definition[1] that defines “State of nature” as “the condition and extent of ecosystems, and species population size and extinction risk, including positive or negative changes”. The state of nature is a key factor in determining the quality of an agroecosystem, but it is often overlooked in life cycle assessments.

As illustrated in Figure 2, ecosystem condition is assessed with respect to three components of an ecosystem: its composition, structure, and functioning. As often in corporate assessments and methodologies, BFAT proposes to focus on **ecosystem composition** to quantify the state and change in state of ecosystems.

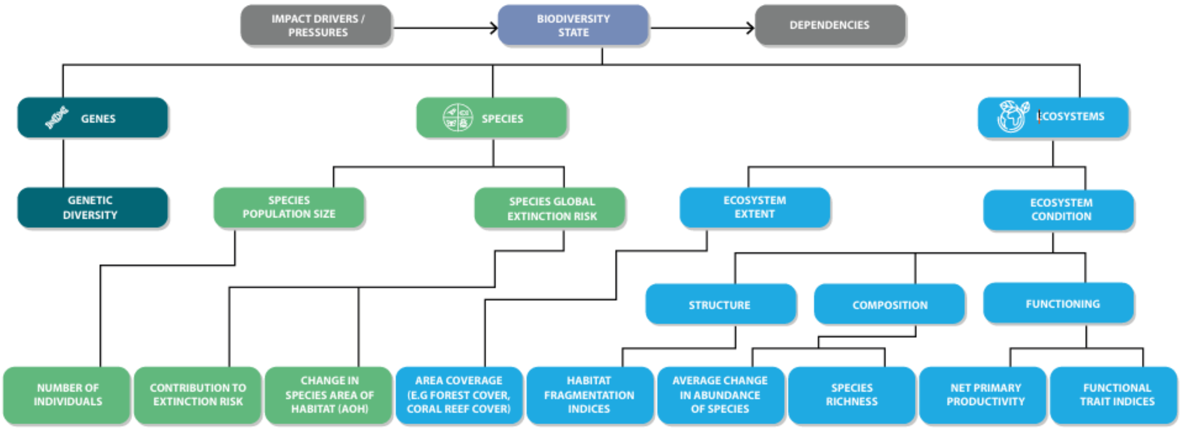

Figure 2 - Components of biodiversity and example measurement indicators[2]

Ecosystem condition was selected because it already has well-established metrics and is already widely used in corporate biodiversity assessment tools and LCIA approaches, enabling the integration and comparison of BFAT results with such tools. It also positively correlates with other ecosystem health indicators, such as their capacity to provide services[3].

However, it should be noted that, to fully capture the state of nature in an ecosystem, it is necessary to complement ecosystem-level assessments with assessments at species and gene levels, as specified in the TNFD definition and recommended in all disclosure frameworks. Although robust and consensual indicators for the gene component of biodiversity are currently lacking, an evaluation of biodiversity at the species level could

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[1] Taskforce on Nature-related Financial Disclosures TNFD, ‘Recommendations of the Taskforce on Nature-Related Financial Disclosures’, September 2023 <https://tnfd.global/publication/recommendations-of-the-taskforce-on-nature-related-financial-disclosures/>.

[2] ALIGN, *Recommendations for a Standard on Corporate Biodiversity Measurement and Valuation* (European Commission, 2022) <https://ec.europa.eu/environment/biodiversity/business/assets/pdf/2022/Align_Report_301122.pdf> [accessed 8 December 2022].

[3] A. C. Smith and others, ‘How Natural Capital Delivers Ecosystem Services: A Typology Derived from a Systematic Review’, *Ecosystem Services*, 26 (2017), pp. 111–26, doi:10.1016/j.ecoser.2017.06.006.

be carried out using the Red List Index (RLI) or extinction risk metrics for instance[1][2][3][4]. This is however not yet included within the scope of BFAT.

MSA definition

Of the various existing metrics of ecosystem composition, the **Mean Species Abundance (MSA)** metric was prioritized in BFAT, to track average changes in abundance of species in agroecosystems as a result of agricultural production and farming practices. Results in species richness can also be computed.

The MSA describes the integrity of ecosystems relative to a pristine, undisturbed state. It was developed with the GLOBIO3 modelling framework to assess scenarios of human-induced changes in biodiversity[5] (more details on the GLOBIO model below). The MSA is defined as "the average abundance of originally occurring species relative to their abundance in the original, pristine or state as the basis". MSA values range from 0% – in a fully artificialised ecosystem – to 100% – in a pristine one. It can be calculated based on ecological measurements *via* the following formula:

$$MSA = \frac{1}{N_{reference\ species}} \sum_{i=1}^{N_{reference\ species}} Min\ (\frac{A_{observed}(i)}{A_{pristine}(i)}, 100\%)$$

With

- MSA: the ecosystem condition, expressed in MSA, of the ecosystem assessed
- $N_{reference\ species}$: the total number of species in the reference (pristine) ecosystem
- $A_{observed}(i)$ the abundance of species i in the ecosystem assessed
- $A_{pristine}(i)$ the abundance of species i in the reference (pristine) ecosystem

Although MSA can be measured at plot scale, this method is difficult implement due to the extensive ecological surveys required. A more practical approach is to focus on a limited number of reference species, from different taxa, and extrapolate their variations in abundance, and the subsequently calculated MSA, to the entire populations of the ecosystem. It is also often the case that the reference ecosystem is not a pristine ecosystem (as few pre-industrial ecological surveys are available), but another, natural ecosystem which is considered undisturbed and that is sampled at the same time as the assessed ecosystem.

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[1] WWF, *Bending the Curve: The Restorative Power of Planet-Based Diets* (2020) <https://files.worldwildlife.org/wwfcmsprod/files/Publication/file/7b5iok5vqz_Bending_the_Curve__The_Restorative_Power_of_Planet_Based_Diets_FULL_REPORT_FINAL.pdf.pdf>.

[2] David Leclère and others, 'Bending the Curve of Terrestrial Biodiversity Needs an Integrated Strategy', *Nature*, 585.7826 (2020), pp. 551–56, doi:10.1038/s41586-020-2705-y.

[3] Walter Willett and others, 'Food in the Anthropocene: The EAT–Lancet Commission on Healthy Diets from Sustainable Food Systems', *The Lancet*, 393.10170 (2019), pp. 447–92, doi:10.1016/S0140-6736(18)31788-4.

[4] Diego García-Vega and others, 'A Safe Agricultural Space for Biodiversity', *Frontiers in Sustainable Food Systems*, 8 (2024), doi:10.3389/fsufs.2024.1328800.

[5] Rob Alkemade and others, 'GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss', *Ecosystems*, 12.3 (2009), pp. 374–90, doi:10.1007/s10021-009-9229-5.

These methods underpin many assessments of MSA levels in ecosystems, including those on which the GLOBIO model is based[1][2]. **MSA is defined using GLOBIO's definition**, whereby it is calculated based on changes in abundance of **6 taxa** (vascular plants, insects, reptiles, mammals, amphibians, birds), which are derived from assessments of keystone species. In practice, the inclusion of species and taxa is conditioned by the existence of robust ecological data.

Per se, the MSA does not capture the specific dynamics of farmland species. Instead, it focuses on species occurring in primary vegetation or pre-industrial era records[3], as it is also the case for other metrics of ecosystem condition (e.g. Biodiversity Intactness Index). Nevertheless, the BFAT approach to using and adjusting MSA estimates compensates for this limitation by taking these farmland species into account (details in section 4.3).

Landscape condition

BFAT includes effects of agroecological practices outside the plot, through landscape biodiversity responses (incl. habitat connectivity, edge effects, or spatial heterogeneity).

According to Sirami et al. 2019[4], in order to capture the overall effects of landscape complexity and agricultural practices on biodiversity, considering the landscape as a **1 km² represents the best compromise between highly mobile taxa**, e.g. birds, **and taxa with more limited dispersal abilities**, e.g. plants or spiders. This spatial extent of 1x1 km is also used in Hass et al. 2018[5] and Priyadarshana et al. 2024[6]. Moreover, effect of overall spatial heterogeneity significantly increases biodiversity for invertebrates, pollinators and predators at different spatial scales (from <0.5km radius area to >1km radius area) (Priyadarshana et al. 2024)[7].

Crop heterogeneity, mean field size and semi-natural cover[8] in the landscape are major drivers of multitrophic diversity in agricultural landscapes[9] (see Figure 3). There are several metrics for measuring crop heterogeneity, so as Mean Field Size (MFS) or number of crops within the landscape. Several exploratory approaches have been tested in BFAT to account for such effects.

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[1] Aafke M. Schipper and others, *The GLOBIO Model. A Technical Description of Version 3.5.*, Text (2016) <https://www.pbl.nl/en/publications/globio-35-technical-model-description> [accessed 20 November 2023].

[2] Alkemade and others, 'GLOBIO3', 2009.

[3] Alkemade and others, 'GLOBIO3', 2009; Leclère and others, 'Bending the Curve of Terrestrial Biodiversity Needs an Integrated Strategy'.

[4] Clélia Sirami and others, 'Increasing Crop Heterogeneity Enhances Multitrophic Diversity across Agricultural Regions', *Proceedings of the National Academy of Sciences*, 116.33 (2019), pp. 16442–47, doi:10.1073/pnas.1906419116. Sirami and others, 'Increasing Crop Heterogeneity Enhances Multitrophic Diversity across Agricultural Regions'.

[5] Annika L. Hass and others, 'Landscape Configurational Heterogeneity by Small-Scale Agriculture, Not Crop Diversity, Maintains Pollinators and Plant Reproduction in Western Europe', *Proceedings of the Royal Society B: Biological Sciences*, 285.1872 (2018), p. 20172242, doi:10.1098/rspb.2017.2242.

[6] Tharaka S. Priyadarshana and others, 'Crop and Landscape Heterogeneity Increase Biodiversity in Agricultural Landscapes: A Global Review and Meta-analysis', *Ecology Letters*, 27.3 (2024), p. e14412, doi:10.1111/ele.14412.

[7] Priyadarshana and others, 'Crop and Landscape Heterogeneity Increase Biodiversity in Agricultural Landscapes'.

[8] includes woodlands, open lands (e.g. shrubland, grassy margins) and wetland cover (including ponds, rivers, ditches) in the landscape, from Sirami (2019)

[9] Sirami and others, 'Increasing Crop Heterogeneity Enhances Multitrophic Diversity across Agricultural Regions'.

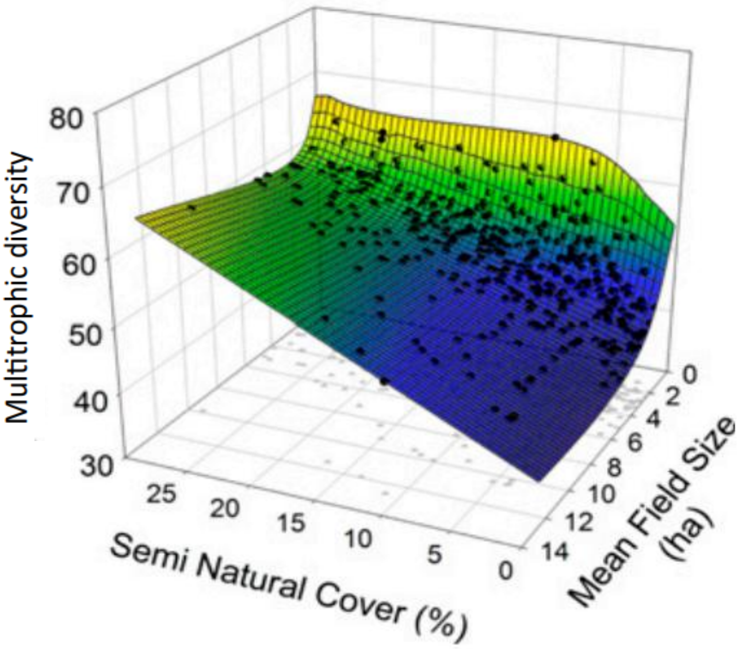

Figure 3 - Effect of Mean Field Size and Semi-Natural Cover on Multitrophic Diversity (from Sirami et al, 2019)

Carbon stocks

Soils and biomass act as major terrestrial carbon sinks and jointly underpin global climate mitigation efforts. Soils alone store up to three times more carbon than the atmosphere, while vegetation, especially perennial systems and agroforestry, adds substantial above and below-ground biomass stocks (Saritha et al., 2025)[1]. Agricultural practices, such as cover cropping, agroforestry, organic fertilization, and reduced tillage, have been shown to increase carbon storage across whole ecosystems, including biomass and soil, strengthening agricultural resilience and climate benefits (Villat & Nicholas, 2024)[2]. As companies move toward net -zero commitments, quantifying carbon removal across biomass and soils becomes essential for companies aiming to measure progress, secure high quality carbon removals, and assess the climate performance impacts of shifting from conventional to regenerative agriculture. Emerging carbon markets and MRV frameworks increasingly emphasize **permanence**, **transparency**, and **practice based- verification**, further reinforcing the strategic importance of robust carbon removal assessments.

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[1] Saritha, Dr. J. D., Nalini, Dr. N., Ramakrishna, N., Teja, Dr. C., & Kumar, Dr. M. P. Soil Carbon Sequestration in Agro-ecosystems: Mechanisms, Management, and the Role in Climate Change Mitigation. International Journal of Environment, Agriculture and Biotechnology (2025). https://doi.org/10.22161/ijeab.106.5

[2] Villat, Jessica, et Kimberly A. Nicholas. « Quantifying Soil Carbon Sequestration from Regenerative Agricultural Practices in Crops and Vineyards ». *Frontiers in Sustainable Food Systems* 7 (2024). https://doi.org/10.3389/fsufs.2023.1234108.

The objective of this assessment is to estimate the annual carbon removal flux in croplands and grasslands, expressed as kgCO2/ha.yr, by comparing soil organic carbon (SOC) stocks under reference management (conventional agriculture) and the equilibrium SOC stock under agroecological practices. As carbon removals is a pressure indicator, soil organic carbon is a state indicator.

Soil Organic Carbon (SOC) refers to the carbon contained in organic compounds within the soil, arising from decomposed plant and animal remains, soil microorganisms, and root secretions. SOC is fundamental for soil fertility, structure, and water retention, and serves as a key indicator of soil health. The build-up and stabilization of SOC are essential for carbon removal, helping to mitigate climate change by capturing atmospheric carbon within terrestrial ecosystems.

3.5. Practices and production modes

To estimate the effects of agricultural practices, the different production systems are first defined theoretically. Two levels are distinguished: **individual practices**, and sets of practices, which constitute **production modes**. The resulting impacts are therefore estimated based on this description.

3.5.1. Selecting and defining agricultural practices

Agricultural practices refer to the **techniques and methods used by farmers** to cultivate crops and raise livestock. The agricultural practice is the **most granular** level of description of agricultural production in BFAT.

These practices can be oriented towards different goals, such as increasing productivity and/or promoting ecological balance and human health. BFAT considers both perspectives by **estimating the effects of practices on both production and environmental metrics**. This approach enables to explore the potential trade-offs between enhancing agricultural productivity – essential to limiting anthropized land occupation — and maintaining the compatibility of agricultural practices with ecosystem and human health.

After a first literature review at the project framing phase, we compiled a list of relevant practices to consider. These were either identified as having a significant effect on environmental pressures or the state of nature, or as being widely adopted by farmers (Table 2 - Practices covered in BFAT). This list is not exhaustive, but it does cover the most relevant practices to explore.

Table 2 - Practices covered in BFAT (see Appendix for definitions)

Crop type	Category	Subcategory
Temporary crops (incl. annual crops)	**Crop rotation**	Short rotation
		Long rotation
		Monocropping
		Crop-livestock farming
		Legumes in rotation
		Meadow in rotation
		Intercropping
	Cover crop	Cover crop
	Soil tillage	No-tillage
		Reduced tillage
		Tillage
Temporary crops (incl. annual crops) **Perennial crops**	**Pesticides use**	Mean use of pesticides
		No pesticides use
	Water use	Rainfed
		Improved irrigation type
Temporay crops (incl. annual crops) **Pasture** **Perennial crops**	**Fertilisation**	Mineral fertilization (synthetic N and mineral P)
		Organic + mineral fertilization
		Organic fertilization
		No fertilization
	Landscape	No ecological infrastructures
		Presence of semi-natural habitats (Onfield unproductive biodiversity zones or buffer areas around field)
Pasture	**Pasture management**	High, Medium, Low grazing intensity
		Rotational grazing
		Mowing intensity
Perennial crops	**Monocropping**	
	Agroforestry	

3.5.2. Production modes

Production modes encompass **consistent sets of farming practices** that may be formalized through labels (such as organic agriculture) or defined by conceptual frameworks (such as regenerative agriculture or conservation farming). **The production mode often constitutes the only information available to downstream companies regarding the production systems of their sourcing.**

The table below shows the main production modes considered in BFAT database, considering it can be completed or adjusted depending on commodities covered.

Table 3 – Production modes covered in BFAT

Production mode	**Definition**	**Source**
Intensive cropland	High external input agriculture, conventional agriculture, mostly with a degree of regional specialization, irrigation-based agriculture, drainage-based agriculture.	Schipper et al, 2016[1]
Organic	Organic production is a sustainable management system that is based on the following general principles: • Respect for nature's systems and cycles and the sustainment and enhancement of the state of the soil, the water and the air, of the health of plants and animals, and of the balance between them; • The preservation of natural landscape elements, such as natural heritage sites; • The responsible use of energy and natural resources, such as water, soil, organic matter and air;	EU Regulation, 2018[2]

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[1] Schipper and others, *The GLOBIO Model. A Technical Description of Version 3.5.*

[2] **REGULATION (EU) 2018/848 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 30 May 2018** on organic production and labelling of organic products and repealing Council Regulation (EC) No 834/2007. ELI: http://data.europa.eu/eli/reg/2018/848/2025-03-25 [accessed 22 January 2026].

	- The production of a wide variety of high-quality food and other agricultural and aquaculture products that respond to consumers' demand for goods that are produced by the use of processes that do not harm the environment, human health, plant health or animal health and welfare.	
Conservation agriculture	Conservation agriculture is based on three principles: - Permanent soil organic cover ((at least 30 percent) with crop residues and/or cover crops) - Minimum mechanical soil disturbance ((i.e. no tillage) through direct seed and/or fertilizer placement.) - Species diversification (through varied crop sequences and associations involving at least three different crops.)	FAO, 2019[1]
Agroforestry	Crop production underneath tree canopy. Trees may be harvested and planted or seeded on occasion.	Van 't Veen et al. 2025[2]
Regenerative[3] intermediate	Include essential practices that qualify a system as regenerative agriculture: reduced-tillage or no-tillage, cover crops, long rotations, and crop-livestock farming	FAO, 2022[4][5]
Regenerative advanced	"Advanced" goes beyond, incorporating the above practices and no pesticide use, legumes in rotation, adaptative multi-paddock grazing, improved water management	

3.6. Temporal perimeter

The impacts of agricultural commodities are typically estimated over a crop cycle, using statistical data on production and environmental indicators. If no information is available on cycle duration, a one-year period is used.

However, when evaluating the effects of agricultural practices, it is important to consider the **temporal dynamics** inherent in these systems. Two distinct temporal scales can be distinguished:

- **Implementation temporality** refers to the period following the adoption of new farming practices. Immediately after implementation, changes in management (e.g., no-tillage, diversified rotations, reduced chemical inputs) may not produce detectable effects on environmental or ecological indicators. Short-term responses can be modest or variable across components of the agroecosystem.[6]
- **Long-term effects** emerge gradually as ecological processes – such as soil organic matter accumulation, biodiversity recovery, pest and disease regulation, and nutrient cycling – respond to sustained practice changes. Long-term studies have shown that the benefits of diversified and sustainable practices (such as intercropping, organic amendments, cover crops) on biodiversity, soil quality,

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[1] FAO, 'Conservation Agriculture', 2019 <https://www.fao.org/conservation-agriculture/en/> [accessed 22 January 2026].

[2] Hanneke Van 't Veen and others, 'A Global Assessment of Plant and Animal Community Responses to Forest Management Over Time', *Global Change Biology*, 31.6 (2025), p. e70279, doi:10.1111/gcb.70279.

[3] "Regenerative agriculture," as a production mode whose concept is recognised as blur, is detailed in our approach in two levels. Intermediate and advanced.

[4] FAO, 'Regenerative Agriculture', 2022 <https://www.fao.org/family-farming/detail/fr/c/1512632/> [accessed 6 September 2024].

[5] Chesapeake Bay Foundation, 'Regenerative Agriculture', 2022 <https://www.cbf.org/document-library/cbf-reports/farm-forward-report.pdf> [accessed 6 September 2024].

[6] Adam S. Davis and others, 'Increasing Cropping System Diversity Balances Productivity, Profitability and Environmental Health', *PLoS ONE*, 7.10 (2012), p. e47149, doi:10.1371/journal.pone.0047149.

ecosystem functions and even economic outcomes can increase significantly over decades.[1]

Impact assessment focuses on the effects of practices at a "stable" stage – that is, once the practices are assumed to have been sufficiently established and their effects fully manifested. This means that the dynamics of pressure changes or biodiversity recovery following the adoption of new practices are not explicitly modelled within the annual impact estimates. Consequently, the **temporal dimension of ecological recovery or lagged responses should be investigated in parallel and through dedicated long-term studies.**

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[1] Estelle Raveloaritiana and Thomas Cherico Wanger, 'Long-Term Agricultural Diversification Increases Financial Profitability, Biodiversity, and Ecosystem Services: A Second-Order Meta-Analysis', *Nature Communications*, 17.1 (2026), p. 1016, doi:10.1038/s41467-025-67757-7.

4. General methodological approach

The paragraphs below outline the primary approach to impact assessment and explain how the effects of practices are incorporated into the assessment. This approach is generic for all commodities.

4.1. Literature review and data collection

In order to capture and differentiate the effects of agricultural practices on agronomic and environmental indicators, data can be obtained from two sources:

- In most cases, **data is collected from scientific literature** (e.g. research articles, technical journals).
- **In the context of field projects**, certain data can be obtained **directly from the project managers**. This data is distinguished from literature-based data in order to enable comparison.

As production modes are combinations of multiple practices, assessing their effects requires estimating the combined effects of these practices, capturing their synergies and counteracting effects. In practice, the approach that can be carried out will depend on the availability of data:

- First, in all cases, **data collection is prioritised at the production mode level** in order to directly capture the combined effects of practices. Such data is more likely to be available for **widely spread production modes that are defined consistently in terms of practices** in multiple contexts (geographies, commodities). This is the case, for example, for organic farming which is defined by clear and consistent certification standards – such as the prohibition of synthetic fertilizer and pesticides – regardless of local context.
- Second, when data at the production-mode level are not available, the **production mode is instead treated as a combination of individual practices**, with data collected at the practice level. This situation typically occurs when there is substantial heterogeneity in practices within a given production mode "nomenclature", as is the case for regenerative agriculture.
  The main challenge with this approach is to estimate the combined effects of practices.[1] Indeed, **interactions between practices may generate synergies, trade-offs, or non-linear effects that cannot be inferred simply by summing individual impacts**. For example, combining multiple practices, such as reduced tillage, organic amendments, and diversified rotations, can lead to synergistic effects on soil organic carbon that would not be detectable when practices are evaluated in isolation[2]. Nevertheless, these **interactions remain only partially documented and are often not synthesised comprehensively in the scientific**

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[1] Alexander Wezel and others, 'Multiple Agroecological Practices Use and Climate Change Mitigation. A Review', *Agronomy for Sustainable Development*, 45.5 (2025), p. 58, doi:10.1007/s13593-025-01048-9.

[2] Raül López I Losada and others, 'Synergistic Effects of Multiple "Good Agricultural Practices" for Promoting Organic Carbon in Soils: A Systematic Review of Long-Term Experiments', *Ambio*, 54.11 (2025), pp. 1715–28, doi:10.1007/s13280-025-02188-8.

literature, which limits the robustness of aggregated impact estimates and underscores the need for more integrative research.

4.2. Assessing Robustness: Transparency and Confidence Criteria

State of nature and pressures estimates **should be interpreted as indicative trends rather than precise measurements of biodiversity outcomes**. The methodology reflects a deliberate trade-off between scientific robustness and operational feasibility, aiming to improve comparability across agricultural systems.

To remain transparent about uncertainty, robustness score is computed in order to evaluate the reliability of the estimates. This score is based on five key criteria:

- **The method used to estimate production mode effects**. Higher uncertainty is assumed when production mode effects are inferred by aggregating individual practices than when they are directly assessed in empirical studies.
- **The relevance of the evidence to the system assessed**. The studies used to estimate the practice effect should be consistent with the system of interest in terms of geography, commodity, practices and production modes.
- **Consistency with the GLOBIO framework**. The extent to which the biodiversity metrics and reference states used in the literature are compatible with GLOBIO's definition of Mean Species Abundance.
- **Type and number of studies available**. Meta-analyses are given greater weight than individual studies, as they synthesise multiple empirical results and generally provide more robust estimates.
- **Taxonomic coverage**. Whether a sufficient number of taxonomic groups are covered to adequately represent biodiversity responses, in line with the minimum requirements defined during the framing of the literature review.

The scoring methodology is currently being defined. The score will range from 1 to 5, with 1 being the lowest.

4.3. State of nature - Ecosystem condition assessment

4.3.1. Using GLOBIO's land use classes as a reference

The GLOBIO model is a set of **quantitative relationships that show how biodiversity responds to human pressures.** Specifically, they link the level of human pressure applied to an ecosystem (e.g. land use, climate change, eutrophication, infrastructure, and different forms of encroachment and fragmentation) to the condition of the ecosystem, expressed in MSA. These relationships can be used to estimate the remaining biodiversity in an ecosystem, when direct field measurement is not available. For each of these pressures, the **relationships were derived from meta-analyses**.[1][2]

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[1] Schipper and others, *The GLOBIO Model. A Technical Description of Version 3.5.*

[2] Jan H. Janse, Michel Bakkenes, and J. Meijer, *Globio-Aquatic* (2016) <https://www.globio.info/globio-aquatic-technical-model-description>.

In GLOBIO, the state of biodiversity in croplands is considered to be mostly influenced by the land use and management practices[1]. Therefore, **assessments of the ecosystem condition on plot in BFAT rely on GLOBIO's relationships for land use**,[2] which correspond to global, averaged levels of MSA for a number of land use classes.

However, **GLOBIO's land use classes have little granularity**. As shown in Table 4, GLOBIO 3.5 only distinguishes two management intensities for pasture ("man-made pasture" and "moderately to intensively use pasture"), three for cropland ("irrigated", "intensive" and "extensive"), and one for agroforestry in tropical biomes. GLOBIO 4 has an even smaller number of classes[3]. For instance, the category "intensive agriculture" in GLOBIO is only described qualitatively and remains very loosely defined. GLOBIO's definition refers broadly to high-input, high-yield systems, without allowing for more granularity within this category. This limited level of detail makes it difficult to compare farming practices in a meaningful way. **BFAT aims at refining these land use classes by linking them more closely to implemented farming practices.**

Table 4 - MSA$_{LU}$ values assigned to GLOBIO's agricultural land use classes[4]

Cropland / grassland type	Land use class	MSA Level	Description
Cropland	Irrigated	5%	High external input irrigation-based agriculture, conventional agriculture, mostly with a degree of regional specialization, drainage-based agriculture
	Intensive	10%	High external input agriculture, conventional agriculture, mostly with a degree of regional specialization, irrigation-based agriculture, drainage-based agriculture.
	Extensive	30%	Low input agriculture. Subsistence and traditional farming, extensive farming, and low external input agriculture
Pasture and rangeland	Pasture – man made	30%	Natural ecosystems that have been converted to grasslands for livestock grazing.
	Pasture – moderately to intensively used	60%	Livestock grazing. Grasslands where wildlife is replaced by grazing
	Natural grassland	100%	Primary vegetation (grass- or scrublands). Grassland or scrubland-dominated vegetation (for example, steppe, tundra, or savannah)
Agroforestry[5]	Agroforestry	50%	Crop production underneath tree canopy. Trees may be harvested and planted or seeded on occasion. There is a heterogenous vegetal structure that is multi-layered. There are several tree species.

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[1] Rob Alkemade and others, 'Assessing the Impacts of Livestock Production on Biodiversity in Rangeland Ecosystems', *Proceedings of the National Academy of Sciences*, 110.52 (2013), pp. 20900–05, doi:10.1073/pnas.1011013108.

[2] Alkemade and others, 'GLOBIO3', 2009.

[3] Aafke M. Schipper and others, 'Projecting Terrestrial Biodiversity Intactness with GLOBIO 4', *Global Change Biology*, 26.2 (2020), pp. 760–71 (p. 4).

[4] Rob Alkemade and others, 'GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss', *Ecosystems*, 12.3 (2009), pp. 374–90, doi:10.1007/s10021-009-9229-5; Alkemade and others, 'Assessing the Impacts of Livestock Production on Biodiversity in Rangeland Ecosystems'.

[5] Van 't Veen and others, 'A Global Assessment of Plant and Animal Community Responses to Forest Management Over Time'.

Cropland / grassland type	Land use class	MSA Level	Description
Plantation[1]	Perennial tree crop	30%	Tree planting plantations for crop production, with homogeneous vegetal structure and a single layer vertical structure. There is mostly one planted species either native or non-native.

4.3.2. Refining GLOBIO's land use classes to better capture diverse practices

4.3.2.1. General approach

To develop additional, more granular land use classes, estimates of MSA levels on agricultural fields would **ideally be derived from extensive meta-analyses**, pooling results from ecological studies to statistically quantify how different farming practices affect mean species abundance (MSA). This approach has been used in previous work, such as the meta-analyses by Durand[2] and by Kuipers[3].

However, it faces several **major limitations in practice**. There are few ecological studies, and even fewer that compare on-field biodiversity with a pristine reference state, and collecting and analysing the data requires substantial time and resources.

Due to these constraints, **an alternative method was developed** to refine and estimate MSA at field level, as shaped by the agricultural practices implemented. This approach relies on applying a correction factor ($\Delta practices$) to GLOBIO's reference land use classes to capture changes in biodiversity **resulting from specific farming practices**, compared to the reference agricultural system:

$$MSA_{refined} = MSA_{standard} \times \Delta practices$$

Where:

- $MSA_{refined}$ is the MSA level on the assessed plot, where practices are implemented
- $MSA_{standard}$ is the MSA level on the reference plot, as defined by GLOBIO
- $\Delta practices$ is the change in biodiversity (in %) observed on a plot where practices are implemented, compared to the reference plot

To ensure consistency and comparability with the other land use classes of GLOBIO 3.5, $MSA_{refined}$ is **capped by the MSA value of other relevant land use class**. Therefore, this approach is mainly a way to differentiate, within a range of GLOBIO classes, more granular type of land use management.

For example, the ecosystem condition of annual crops ecosystems is considered between 10%, the reference state with intensive practices, and 30% MSA which is the maximum potential level corresponding to "extensive cropland" defined as subsistence, low-input agriculture. As BFAT focuses on commodity crops, exceeding this level is considered not

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[1] Van 't Veen and others, ' Assessment of Plant and Animal Community Responses to Forest Management Over Time'.

[2] Margaux Durand, *Améliorer les mesures d'impacts des entreprises sur la biodiversité*, n.d.

[3] Koen J. J. Kuipers and others, 'Land Use Diversification May Mitigate On-site Land Use Impacts on Mammal Populations and Assemblages', *Global Change Biology*, 29.22 (2023), pp. 6234–47, doi:10.1111/gcb.16932.

feasible. For grazed pastures, the maximum is set at 60% MSA, corresponding to extensive grassland.

4.3.2.2. Selecting $MSA_{standard}$

In line with data availability in the literature, the reference system refers to one of GLOBIO land use classes (Table 4), depending on the system studied. For example, for annual crops, intensive cropland is considered as the reference state and $MSA_{standard}$ is set at 10% MSA.

4.3.2.3. Estimating $\Delta practices$ based on the literature

A **specific** $\Delta practices$ is estimated for each production mode (or practice), area of production, and commodity based **on a targeted literature review of ecological studies.** The estimation of $\Delta practices$ for a specific practice (or production mode) falls into three main steps:

1. Identify research papers of interest and extract their results

The papers and data selected in the review must address the practice (or production mode) studied and be consistent with the region and commodity of interest. The extent to which the commodity and region covered by the paper match the system under study will depend on the availability of data (see). Publications are selected based on the following criteria:

- **Commodity**: Commodities can be defined by a variety, a species (e.g. wheat) or a broader category (e.g. cereals). It has to be consistent with the system studied.
- **Geography**: The geography covered by the study may be defined in different ways: a region (e.g. Europe), climate zone (e.g. temperate latitudes), or biome (e.g. temperate forest). It has to be consistent with the system studied.
- **Practices assessed and reference system used for comparison**: Any study included must clearly describe the practice implemented and the reference state—which must be coherent with the reference ecosystem condition considered in computation.

  The practice (or production mode) assessed must match the one of interest. Practices presented in the studies can have several terminologies. Based on all terminologies identified in the literature, CDC Biodiversité and I Care propose a single nomenclature of practices or production mode (Table 2 - Practices covered in BFAT).

Additional parameters regarding the results of ecological studies, especially **biodiversity responses parameters,** are extracted:

- **Taxa**: The same six taxonomic groups as in GLOBIO are considered: plants, invertebrates, reptiles, mammals, amphibians, and birds. However, many studies report results for taxa that do not directly match these categories.

  A preliminary reclassification step is therefore applied to align taxa reported in the literature with GLOBIO’s taxonomic framework. GLOBIO’s MSA values are derived from a meta-analysis covering these six groups. Taxa documented in the literature

(e.g. insects, arthropods, microorganisms, nematodes, bacteria, fungi) are reassigned accordingly. For example, nematodes, arthropods, bacteria, fungi and other microorganisms are grouped under the invertebrate taxon. Among these, only arthropods and nematodes are invertebrates, but to simplify and as a first approach, we have grouped them together. To overcome the limitations of the scope of species accounted for in the MSA definition, in the BFAT context, we aim at capturing variations in biodiversity also considering non-pristine species, i.e. including farmland communities.
Therefore, we consider some priority taxa to include in our literature review in the Table 5 below.

Table 5 - Priority taxa for evaluating the effects of practices

Practices / Taxa	Plants	Invertebrates	Birds	Reptiles	Mammals	Amphibians
Pesticides use	🟩	🟩	🟩			
Polyculture / mixed farming	🟩	🟩	🟩			
Crop rotation		🟩	🟩			
Soil-tillage		🟩	🟩			
Fertilization		🟩	🟩			
Presence of semi-natural habitats	🟩	🟩	🟩			
Cover crop	🟩	🟩	🟩			
Irrigation						
Intercropping		🟩	🟩			
Mowing	🟩	🟩	🟩			
Grazing	🟩	🟩	🟩		🟩	
Agroforestry with timber, fruit or nut trees	🟩	🟩	🟩	🟩	🟩	

🟩	Priority research area
⬜	Low priority

- **Biodiversity metrics**: Biodiversity metrics retained for the literature review are metrics related with ecosystem composition. It includes **species abundance or richness** variation metrics (see 3.4.2 State of nature).

For each selected publication, the measured or estimated quantitative biodiversity responses of the production mode or agricultural practice is extracted, relative to its reference production system. These values are stored in a database developed and maintained by CDC Biodiversité and I Care (see Table 6 below). The first line means that in Puissant et al (2021), it was documented an increase in invertebrates' abundance of +96% thanks to conservation practices compared to intensive practices, across a significant diversity of geographies and commodities.

Table 6 - Example of datapoints extracted from literature on different taxa & practices

Practices assessed	Standard practices compared	Commodity	References	Country	Metric	Plants	Invertebrates	Reptiles	Mammals	Amphibians	Birds	Not mentionned. Consider all taxas
Conservation	Intensive cropland	All	Puissant et al.(2021)	All	Abundance	N/A	96%	N/A	N/A	N/A	N/A	
Organic farming	Intensive cropland	All	Puissant at al.(2021)	All	Abundance	N/A	33%	N/A	N/A	N/A	N/A	
Organic farming	Intensive cropland	All	Janne Bengtsson (2005)	All	Richness	N/A	N/A	N/A	N/A	N/A	N/A	30%
Organic farming	Intensive cropland	All	Janne Bengtsson (2005)	All	Abundance	N/A	N/A	N/A	N/A	N/A	N/A	50%
Organic farming	Intensive cropland	All	Stein-Bachinger et al. (2020)	N/A	Richness	78%	19%	N/A	N/A	N/A	35%	
Organic farming	Intensive cropland	All	Stein-Bachinger et al. (2020)	N/A	Abundance	148%	46%	N/A	N/A	N/A	24%	

2. Aggregate biodiversity responses per taxon and practice

Once all quantitative biodiversity responses have been compiled, multiple studies may examine the effects of the same practice on the same taxonomic group (in the relevant geography and for the crop of interest). The next step is therefore to calculate the average of these results to estimate the mean variation in MSA for this taxon (ΔMSA^{p}_{taxon}) due to the practice p:

$$\Delta MSA^{p}_{taxon_i} = \sum_{i}^{n} \frac{d^{p}_{i,taxon}}{n^{p}_{taxon}}$$

Where:

- $d^{p}_{i,taxon}$ is a datapoint extracted from the literature that estimate the effect of the practice p on taxon i.
- n^{p}_{taxon} the number of datapoints available regarding practice p and the taxon i
- p the practice under study

3. Aggregate biodiversity responses per taxon into the overall biodiversity response $\Delta practices$

The taxon-specific biodiversity responses (ΔMSA^{p}_{taxon}) are then combined into a single estimate of the response of biodiversity to each practice. Three alternative approaches are proposed:

- (a) **Conservative**: for taxa not covered by a publication, it is assumed that the practice has no effect on their abundance, i.e. that there are as many individuals of this taxon with and without the practice. Therefore, the average is calculated across all 6 GLOBIO taxa, divided by 6, as shown on the equation below;

- (b) **Mean**: this value represents the average between the Conservative and Optimistic approaches;
- (c) **Optimistic**: for taxa not covered by a publication, it is assumed that their variation is equal to that of documented taxa. Consequently, the average is computed by dividing the values by the number of documented taxa. The formula is equivalent to that below with a change in denominator.

Equation: Conservative method to aggregate taxon-specific results into a $\Delta practices$

$$\Delta practice\_p = \frac{\Delta MSA^{p}{}_{plants} + \Delta MSA^{p}{}_{invertebrates} + \Delta MSA^{p}{}_{reptiles} + \Delta MSA^{p}{}_{mammals} + \Delta MSA^{p}{}_{amphibians} + \Delta MSA^{p}{}_{birds}}{6}$$

Where:

- ΔMSA_{taxon}^{p} *refers to the aggregated variation in MSA for each taxon relative to specific practice p*
- *p refers to the studied practice*

4.3.2.4. Estimating the effect of combined practices

As described in section 4.1 Literature review and data collection, **when empirical data are not available to directly estimate the biodiversity effects of a production mode using the approach outlined above, these effects may be estimated by combining the effects of single practices**. For example, in the case of conservation agriculture, the effects of practices such as extended crop rotations, no-tillage, cover crops, and reduced pesticide use are combined to estimate an overall biodiversity response.

At present, **there is no harmonised framework in the scientific literature for estimating the combined effects of multiple agricultural practices on biodiversity**. Existing studies typically assess these combined effects on a case-by-case basis. Alternative methodological approaches have been proposed, including the statistical review of practice effects developed by Cozim-Melges et al[1], which can be used to prioritise practices according to their relative importance and potential biodiversity impacts, and the mathematical frameworks proposed by Lindner et al. (2021)[2], which move beyond multiplicative assumptions by applying fuzzy-logic approaches to land-use intensity.

Several approaches are being explored. At this stage, collection of biodiversity response data at the production mode level are preferred, rather than at the level of individual practices, in order to reduce the uncertainties associated with aggregating multiple practice-level effects. Yet, as such data are scares, an alternative way through single practices effect has been developed to estimate biodiversity impact for a given production mode, by multiplying the effects of its individual practices using the following formula:

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[1] Felipe Cozim-Melges and others, 'Farming Practices to Enhance Biodiversity across Biomes: A Systematic Review', *Npj Biodiversity*, 3.1 (2024), p. 1, doi:10.1038/s44185-023-00034-2.

[2] Jan Lindner and others, 'Moving beyond Land Use Intensity Types: Assessing Biodiversity Impacts Using Fuzzy Thinking', *The International Journal of Life Cycle Assessment*, 26 (2021), pp. 1–19, doi:10.1007/s11367-021-01899-w.

$$\Delta_{practices} = \prod_{i=1}^{n} [1 + \Delta practice_i]$$

Where:

$\Delta practice_i$ is the gain or loss in biodiversity due to the implementation of an agricultural practice i compared to a baseline practice (corresponding to conventional mode);

n is the total number of practices associated with the production mode

The team also remains open to methodological recommendations and to participating in research initiatives addressing the aggregation of biodiversity effects across multiple agricultural practices.

4.3.3. Current limitations

The team has identified the following limitations:

- **Capping of maximum MSA values**: The approach constrains MSA values within predefined GLOBIO ranges. While this ensures consistency, it may underestimate biodiversity levels achievable under certain agroecological or low-intensity systems.
- **Limited availability of relevant ecological studies**: There are relatively few studies that quantify on-plot biodiversity responses to specific agricultural practices, and even fewer that provide results comparable to a conventional reference system consistent with GLOBIO definitions.
- **Heterogeneity across studies**: Studies differ widely in terms of geography, crops, taxa, sampling methods, and biodiversity metrics, which limits comparability and increases uncertainty when aggregating results.
- **Geographical and taxonomic biases:** The available literature is unevenly distributed, with strong biases toward certain regions (e.g. Europe) and taxa (e.g. plants, insects, birds), potentially limiting the representativeness of the results at global scale.
- **Reclassification of taxa into GLOBIO categories**: Aligning diverse taxa reported in the literature with GLOBIO's six taxonomic groups requires simplifications that may hide important ecological differences between groups (e.g. microorganisms vs. macro-invertebrates).
- **Assumptions on undocumented taxa**: The conservative, mean, and optimistic aggregation approaches rely on strong assumptions regarding taxa not covered by the literature, which can substantially influence Δpractices estimates.
- **Higher uncertainty for production modes derived from practices**: When production mode effects are inferred by aggregating individual practices, uncertainties are higher than when production modes are directly assessed in empirical studies.
- **Asymmetry between biodiversity losses and gains**: The framework assumes that biodiversity losses due to a practice are not necessarily symmetric with potential gains when the practice is halted, but this asymmetry is difficult to quantify and not explicitly modelled.

- **Uncertainty not fully propagated**: While robustness scores are estimated to qualify results, it does not quantify properly uncertainties.

4.4. Pressure modelling

This section provides a first overview of the pressures considered in BFAT, their definition, perimeter and some elements about data used and approach to distinguish the effects of practices. Pressure and impacts are estimated for 1 kg of commodity.

4.4.1. Land use

Land use refers to the area of land occupied by agricultural production activities — including cropland and grazing (in m^2.yr) — and how this occupation alters ecosystems and biodiversity (in %MSArefined, see above)[1]. Land use is monitored with land surface metrics, *m^2 or m^2·year*, typically expressed as total occupied area over a given period. **The perimeter of the pressure evaluated** mainly focuses on **production areas** (land under cultivation or grazing), without accounting for indirect climate effects (i.e. w/o projecting potential yield variation due to climate change). According to the value chain perimeter evaluated, upstream or downstream land use can be accounted for (see 3.3 Value chain perimeter). To estimate land use pressures, **yield and area data** from global datasets (e.g., FAOSTAT) for crop-specific land requirements can be used, on top of **national agricultural surveys** and remote sensing products such as Land Cover maps or any other **scientific literature** on yields by practice type.

Finally, to compute land use impacts, we combine land occupation with ecosystem condition (estimated based on the **land management intensity and practices implemented).**

4.4.2. Land use change

Land use change refers to the conversion of ecosystems into agricultural production areas. Natural ecosystems (forests, grasslands, wetlands) can be distinguished from non-natural ecosystems converted. The IPBES Global Assessment highlights that such changes are one of the largest direct drivers of biodiversity loss globally, as they result in habitat loss and fragmentation. Land use change is monitored with *m^2 per ecosystem change*, tracking newly converted areas from into agriculture. Land use change can be estimated with **Satellite land cover change data** (e.g., Copernicus Land Monitoring Service), **national land conversion statistics** (such as Orbae database[2]) and agricultural censuses or **peer-reviewed studies** on dynamics of land conversion per crop and region. To evaluate the effects of production mode or agricultural practices on land use change, we rely on studies comparing conversion rates under different production systems (e.g., industrial vs agroecological) or standards or chain of custody from certifications.

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[1] Peter J. Stephenson and Giulia Carbone, *Guidelines for Planning and Monitoring Corporate Biodiversity Performance* (IUCN, International Union for Conservation of Nature, 2021), doi:10.2305/IUCN.CH.2021.05.en.

[2] Reinhard J and others, 'Orbae Methodology for Jurisdictional Direct Land Use Change, Version 2.1.', AdAstra Sustainability, Geneva, Switzerland., 2025.

4.4.3. Climate change

4.4.3.1. Carbon emissions

Greenhouse gases (GHG) are emitted from agricultural activities such as soil management, animal husbandry, machinery that contribute to **climate change**. GHG emissions are reported with $tCO_2\text{-}eq$ — expressed per unit of agricultural output or area. The **evaluation perimeter** includes **direct emissions** and potentially upstream/downstream activities if within the defined value chain (see Value chain perimeter). To assess GHG emissions we use several data sources such as **FAOSTAT Emissions database**, **Life Cycle Assessment (LCA) datasets** (e.g., Ecoinvent, Agribalyse), national GHG inventory reports, depending on best-available data. Emissions can be adjusted according to the practices implemented based on IPCC or LCA literature showing how practices (reduced tillage, feed changes) alter emission factors.

Currently, land use change emissions (LUC) and non-LUC emissions are distinguished. A distinction will gradually be made between three types of emissions accordingly with SBTi FLAG guidance[1] :

- **FLAG LUC emissions:** Emissions originating specifically from **land use change** such as deforestation, conversion of natural ecosystems, or afforestation/reforestation.

- **FLAG Non-LUC emissions:** Emissions from **agricultural and land management activities** that do not involve land use change (e.g., enteric fermentation, manure, fertilizers, soil carbon, forestry operations). They represent **biogenic fluxes** associated with agriculture, forestry and land management within the FLAG perimeter.

- **Non-FLAG emissions:** All emissions outside the land, agriculture, and forestry scope, i.e., fossil fuel based emissions from energy use and industrial processes.

4.4.3.2. Carbon removal

The estimation method follows the GHG Protocol Land Sector and Removals Standard[2] for croplands and for grasslands. At this stage, only removals from agricultural soils (croplands and grasslands) are accounted for, but biomass related removals will progressively be integrated to obtain a complete assessment of land- sector removals. Under the SBTi FLAG framework, these removals can be recognized as Scope 3 emission reductions and contribute to validated science based- targets.

For croplands, carbon removal flows are estimated by calculating the difference in carbon stock between lands managed under reference practices (conventional agriculture in this

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[1] CM Anderson and others, *Forest, Land and Agriculture Science-Based Target-Setting Guidance* (World Wildlife Fund, 2022) <https://files.sciencebasedtargets.org/production/files/SBTiFLAGGuidance.pdf> [accessed 13 February 2026].

[2] GHG Protocol, *Land Sector and Removals Standard - Version 1.0: Agriculture and CO2 Removal Technologies* (2026) <https://ghgprotocol.org/sites/default/files/2026-01/Land-Sector-and-Removals-Standard.pdf> [accessed 17 February 2026].

study) and the **equilibrium stock** under agroecological practices. This assessment follows the methodological framework provided in the *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*[1]. Specifically, the calculation follows Equation 2.25 of the 2006 IPCC Guidelines for National Greenhouse Gas Inventories[2].

The following formula was applied:

$$Carbon\ removal\ flows = [[SOC_{REF} \times FLU \times F_{best\ practice} - SOC_{REF} \times FLU] \times 3{,}67]$$

Where:

- Carbon removal flows in kg CO2/ha.year
- SOC_{REF} is the reference Soil Organic Carbon stock (kgC/ha) as provided by the IPCC.
- FLU is the stock change factor associated with Land Use or land-use change type provided by the IPCC (unitless).
- $F_{best\ practice}$ is the adjustment factor reflecting agroecological or regenerative practices. Values were extracted from a 2023 global meta-analysis synthesizing findings from 230 first-order meta-analyses comprising over 25 000 primary studies[3]. To calculate $F_{best\ practice}$ and remain methodologically conservative, we avoid aggregating the carbon benefits of multiple regenerative practices. For each set of practices, the value corresponds to the maximum observed variation (x%) to capture the full potential impact. Following IPCC logic, we divide this variation by the IPCC default transition period of **20 years** to which we add one**.** Then we have $F_{best\ practice}$ = $(\frac{x\%}{\mathbf{20}} + 1)$ (unitless).
- 3,67 is the conversion factor from C to CO$_2$

For grasslands, carbon removal values are derived directly from flows reported in the scientific literature[4], themselves based on stock assessments.

For livestock systems, where feed is composed of multiple components, the carbon removal flow is calculated as a weighted average:

$$Carbon\ removalflow = \sum(\%area_i \times Removal_i)$$

where *i* denotes each feed component which are distinguished according to whether it comes from grassland (temporary or permanent) or croplands. This approach ensures that the contribution of each land-use type to overall removals is proportionally represented.

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[1] IPCC, '2019 Refinement of the Good Practice Guidance for Land Use, Land-Use Change and Forestry (Volume 4, Chap 05 Cropland)', 2019.

[2] IPCC Volume 4 – Agriculture, Forestry and Other Land Use. Chapter 2 – Generic Methodologies Applicable to Multiple Land-Use Categories. < https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_02_Ch2_Generic.pdf> [accessed 5 February 2026]

[3] Damien Beillouin and others, 'A Global Meta-Analysis of Soil Organic Carbon in the Anthropocene', *Nature Communications*, 14.1 (2023), pp. 1–10, doi:10.1038/s41467-023-39338-z.

[4] Jean-Baptiste Dolle and others, 'Contribution de l'élevage Bovin Aux Émissions de GES et Au Stockage de Carbone Selon Les Systèmes de Production', *Fourrages*, no. 215 (2013), pp. 181–91.

The proposed approach is based on modeled estimates. Meaning it represents potential carbon removal and has not been directly verified through on-the-ground measurements (MRV). The estimated values reflect average flows across multiple years and geographic contexts, meaning that actual carbon removal at the plot level may vary substantially. The forthcoming GHG Protocol Land Sector and Removals Guidance is expected to provide improved methodologies for assessing and reporting carbon removal, addressing the limitations of the current approach, which stem from the inherent complexity of carbon dynamics in soils and biomass.

4.4.4. Pollution

4.4.4.1. Nutrient pollution

Nutrient pollution is the release of excess nitrogen (N) and phosphorus (P) into water bodies — driving eutrophication and harmful algal blooms — and is a recognized form of **pollution affecting biodiversity**. The pressure is reported with *kg N or kg P emitted into freshwater* (or nutrient load per hectare) and concerns primarily **direct field runoff**, extended to upstream/downstream if relevant (see Value chain perimeter). To estimate nutrient emissions several sources such as **LCA models and databases**, **field-level monitoring studies** (SALCA-P) or region-specific nutrient budgets can be used. To estimate variation by practices, the literature shows how precision application, buffer strips, or organic amendments change nutrient leaching profiles.

4.4.4.2. Ecotoxicity

Ecotoxicity refers to the harm caused by chemical substances — especially **pesticides** — to non-target organisms.

Only freshwater ecotoxicity, i.e. the effect of chemical run-offs on freshwater ecosystems outside the plot are considered here. Indeed, the main direct effects of substances on terrestrial biodiversity are considered captured within the land use pressure through indicators of management intensity, in line with GLOBIO's approach. Assessments of terrestrial ecotoxicity are also less robust in general, as impacts are not as well characterised by science and methodologies are less operational in LCA approaches[1].

The ecotoxicity is monitored with CTUe, computed through quantities (*kg) of active substances applied* (e.g., USEtox). Depending on the evaluation perimeter, additional upstream or downstream emissions can be included (see Value chain perimeter). To estimate ecotoxic emissions several sources such as pesticides usage **database** (e.g., PestChemGrids[2]) or agricultural chemical use statistics can be used. **Practice effects can be differentiated based on** studies on integrated pest management (IPM) or other literature identifying practices reducing the need of pesticides.

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[1] Emmanuel Maillard and Peter Fantke, *Status, next Steps and Documentation of Terrestrial Ecotoxicity Characterization*, OLCA-PEST (2020) <https://data.europa.eu/doi/10.2903/j.efsa.2012.2668> [accessed 13 February 2026].

[2] Federico Maggi and others, 'PEST-CHEMGRIDS, Global Gridded Maps of the Top 20 Crop-Specific Pesticide Application Rates from 2015 to 2025', *Scientific Data*, 6.1 (2019), p. 170, doi:10.1038/s41597-019-0169-4.

4.4.5. Water use

Water use for agricultural purposes alters local and regional water cycles by appropriating fractions of green water (from precipitation) and withdrawing blue water through irrigation and capillary rise. This affects the availability of water resources and hydrological flows, thereby contributing to water stress and impacting freshwater biodiversity.

The pressure related to water use is estimated based on the **annual consumption** of water (m³) required to produce agricultural commodities. As a preliminary approach, only the consumption of **blue water** (i.e. irrigation and capillary rise) is considered. According to the perimeter evaluated (see Value chain perimeter section), upstream and downstream water use can also be included.

Databases such as water footprint datasets (e.g., Mialyk et al. (2024)[1]), national irrigation statistics or the FAO's AQUASTAT database[2] can be used. These country- or region-specific estimates are then refined according to the practices applied in the field and the type of irrigation used.

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[1] Oleksandr Mialyk and others, 'Water Footprints and Crop Water Use of 175 Individual Crops for 1990–2019 Simulated with a Global Crop Model', *Scientific Data*, 11.1 (2024), p. 206, doi:10.1038/s41597-024-03051-3.

[2] 'AQUASTAT - Système d'information Mondial de La FAO Sur l'eau et l'agriculture', n.d. <https://www.fao.org/aquastat/fr/> [accessed 13 February 2026].

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ACRONYMS

FAO: Food and Agriculture Organization of the United Nations

MSA: Mean Species Abundance

LC(I)A: Life Cycle (Impact) Assessment

PDF: Potentially Disappeared Fraction of species

SBTN: Science Based Targets Network

TNFD: Taskforce on Nature-related Financial Disclosures

APPENDIX – LIST OF PRACTICES COVERED IN BFAT AND DEFINITIONS

Crop type	Category	Subcategory	Definition	Source
Temporary crops (incl. annual crops)	**Crop rotation**	Short rotation	Is considered as short rotation all rotation planned on less than 3 years.	Villenave et al. 2022[1]
		Long rotation	Is considered as long rotation all rotation planned on more than 3 years.	
		Monocropping	Single crop variety grown repeatedly on the same land	
		Crop-livestock farming	Crop-livestock farming is an agricultural production system which combines one or more crops (intended for sale and/or feeding of animals) and at least one type of livestock.	Martin et al. 2016[2]
		Legumes in rotation	A practice within crop rotation that involves alternating legumes with other crops in a specific area. Legumes play a vital role in this process by reducing weeds and enriching the soil through their partnership with nitrogen-fixing bacteria.	FAO 2021[3]
		Meadow in rotation	A practice within crop rotation that involves meadow occupation for at least one year (temporary) to more than 3 years (permanent). Meadow is associated with substantial soil carbon removal, higher N-use efficiency and higher crude protein production, compared with the arable cropping systems.	Castelli et al. 2017[4]
		Intercropping	Compilation of diverse farming areas settings such as strip cropping, multicrop field or pixel cropping that involve increasing the amount of crops in the field area.	Cozim-Melges et al. 2024[5]
	Cover crop	Cover crop	Usually cover crops, with and without leguminous plants, are grown as living mulch and incorporated into the soil as nutrient and structural contribution.	(Cozim-Melges, 2024)
	Soil tillage	No-tillage	No tilling applied to the agricultural area.	(Cozim-Melges, 2024)
		Reduced tillage	Use of only minimum secondary tillage and no primary tillage and without turning over the soil.	
		Tillage	Regular ploughing with full soil conversion and leaving <30% crop residue cover	
Temporary crops (incl. annual crops) Perennial crops	**Pesticides use**	Mean use of pesticides	Conventional use of synthetic pesticide, per sub-national area for a given crop (Treatment Frequency Index) following the recommended dose NB: Ecotoxicity of biocontrol products are not considered in this first version of BFAT. Indeed, the computation for treatment frequency index differ for those products.	Agreste 2019[6]
		No pesticides use	No synthetic pesticides	
	Water use	Rainfed	Rainfed agriculture	Schipper et al. 2016 Darouich et al. 2013[7]
		Improved irrigation type	Switch to irrigation systems to use water more effectively (drip irrigation etc..). Drip irrigation is a localized water application technique considered to be one of the most water-efficient forms of irrigation to date.	Guo et al. 2024[8]
Temporay crops (incl. annual crops) Pasture Perennial crops	**Fertilisation**	Mineral fertilization (synthetic N and mineral P)	Addition of nitrogen and/or phosphorus and/or potassium, in various mineral forms such as: $NaNO_3$, NH_4NO_3, $(NH_4)_3PO_4$…	Villenave et al. 2022[9]
		Organic + mineral fertilization	Any type of organic fertilizer combined with the application of synthetic, mineral nitrogen sources usually applied in conventional agriculture. The main N source is the organic fertilizer.	(Cozim-Melges, 2024)
		Organic fertilization	Input of organic matter in various forms: manure, slurry, compost, crop residues, sludge, etc.	Villenave et al. 2022[10]
		No fertilization	Agricultural area without application of any type of fertilizer.	(Cozim-Melges, 2024)
	Landscape	No ecological infrastructures	Absence of semi-natural habitats (unproductive biodiversity zone and buffer areas) around or within the field	(Cozim-Melges, 2024)
		Presence of semi-natural habitats (Onfield unproductive biodiversity zones or buffer areas around field)	Set of practices involving implementing semi-natural habitats within the field (vegetation strips to provide foraging or safe areas for local and/or agriculturally relevant species) and of field margins (buffer areas, natural reserves, etc.).	Sirami et al. 2019[11]
Pasture	**Pasture management**	High, Medium, Low grazing intensity	Grazing intensity is expressed as the cattle stocking rate (number of livestock units on farm divided by the area of grazed grasslands).	Farrugia et al.[12] (2006)Wang and Tang[13] (2019)
		Rotational grazing	Rotational grazing is the practice of containing and moving animals through pasture to improve soil, plant, and animal health.	Rodale Institute[14] (2026)
		Mowing intensity	In opposition to grazed pasture, there is no cattle on mown pasture.	Klimek[15] (2007)
Perennial crops	**Monocropping**		Tree planting plantations for crop production	Van 't Veen et al. 2025[16]
	Agroforestry		Crop production underneath tree canopy. Trees may be harvested and planted or seeded on occasion	Van 't Veen et al. 2025[17]

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[1] C. Villenave and others, ‘Impact des pratiques agricoles sur l’état biologique du sol : SIPANEMA, un outil d’aide à la décision basé sur les nématodes’, *Etude et Gestion des Sols*, 2022 <https://horizon.documentation.ird.fr/exl-doc/pleins_textes/2022-08/010085057.pdf>.

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[7] Hanaa M. Darouich and others, ‘Drip vs. Surface Irrigation: A Comparison Focussing on Water Saving and Economic Returns Using Multicriteria Analysis Applied to Cotton’, *Biosystems Engineering*, 122 (2014), pp. 74–90, doi:10.1016/j.biosystemseng.2014.03.010.

[8] Guo, Hui, and Sien Li. 2024. "A Review of Drip Irrigation’s Effect on Water, Carbon Fluxes, and Crop Growth in Farmland" *Water* 16, no. 15: 2206. https://doi.org/10.3390/w16152206

[9] Villenave and others, ‘Impact des pratiques agricoles sur l’état biologique du sol : SIPANEMA, un outil d’aide à la décision basé sur les nématodes’.

[10] Villenave and others, ‘Impact des pratiques agricoles sur l’état biologique du sol : SIPANEMA, un outil d’aide à la décision basé sur les nématodes’.

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