# D2.1 Potential of landscape features to benefit biodiversity and ecosystem services 30/06/2026


Lead beneficiary: University of Helsinki

Authors: Irina Herzon, Fanny Keränen, Isabella Selin Norén, Lennart Fuchs, Johan Ekroos, Gilles Jean-Louis, Bartosz Bartkowski, Jolien Bracke, Francisco Moreira, Angela Lomba, Guy Pe’er, Irene Bouwma, Kaisa Västilä, Yanka Kazakova, Nikolay Mehandzhiyski







Funded by the European Union

LAFERIA receives funding from the European Union's Horizon Europe research and innovation programme under grant agreement No. 101181492.

D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

**Prepared under contract from the European Commission**
Grant agreement No. 101181492
EU Horizon Europe Research and Innovation Action

**Project acronym:** LAFERIA

**Project full title:** LAndscape FEatures Reintroduction in Intensive Agricultural land

**Project duration:** 1 January 2025 - 31 December 2028 (48 months)

**Project coordinator:** Francisco Moreira, BIOPOLIS

**Call:** Horizon-CL6-2024-BIODIV-01-7

**Deliverable title:** Potential of landscape features to benefit biodiversity and ecosystem services

**Deliverable Nº:** D2.1

**WP responsible:** WP2 (UH)

**Nature of the deliverable:** Report

**Dissemination level:** Public

**Lead partner:** UH

D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

**Recommended citation:** Herzon et al (2025) Potential of landscape features to biodiversity and ecosystem services. LAFERIA project deliverable 2.1. Available from CC

**Due date of deliverable:** 30.06.2026

**Actual submission date:** 30.06.2026

Deliverable status:

| Version | Status | Date | Author(s)/Contributor(s) and Reviewer(s) |
|---|---|---|---|
| 1.0 | Draft version, to be sent to the co-authors and contributing partners | 17.04.2026 | Irina Herzon, Johan Ekroos, Fanny Keränen, Lennart Fuchs, Isabella Selin Norén, |
| 1.1 | Version submitted to ExCom | 29.05.2026 | Irina Herzon, Fanny Keränen, Isabella Selin Norén, Lennart Fuchs, Johan Ekroos, Gilles Jean-Louis, Bartosz Bartkowski, Jolien Bracke, Francisco Moreira, Angela Lomba, Guy Pe'er, Irene Bouwma, Kaisa Västilä, Yanka Kazakova, Nikolay Mehandzhiyski |
| 1.2 | Final version | 25.06.2026 | Irina Herzon, Fanny Keränen, Isabella Selin Norén, Lennart Fuchs, Johan Ekroos, Gilles Jean-Louis, Bartosz Bartkowski, Jolien Bracke, Francisco Moreira, Angela Lomba, Guy Pe'er, Irene Bouwma, Kaisa Västilä, Yanka Kazakova, Nikolay Mehandzhiyski |

## Table of Contents

**Summary** ........ 6

**List of abbreviations** ........ 7

**1. Introduction** ........ 8

1.1. Background and scope ........ 8
1.2. Conceptual framework ........ 9
1.3. Aims and objectives ........ 11

**2. Evidence synthesis on LF benefits to biodiversity and ecosystem services** ........ 12

2.1. Methods of the Evidence Synthesis ........ 12
2.1.1. Definitions and classifications of key concepts ........ 13
2.1.2. Tier 1 sources: JRC-FP dataset and Conservation Evidence ........ 17
2.1.3. Tier 2 sources: reviews for gaps ........ 20
2.1.4. Tier 3 sources: primary papers ........ 20
2.1.5. Data extraction & reporting results ........ 21
2.1.6. Limitations and potential biases ........ 22
2.2. Evidence on biodiversity benefits ........ 23
2.2.1. Hedgerows ........ 24
2.2.2. Tree rows ........ 25
2.2.3. Individual trees ........ 26
2.2.4. Groups of trees/shrubs ........ 26
2.2.5. Herbaceous strips ........ 27
2.2.6. Herbaceous parcels ........ 31
2.2.7. Ditches ........ 33
2.2.8. Small wetlands ........ 34
2.2.9. Small ponds ........ 35
2.2.10. Riparian vegetation ........ 36
2.2.11. Terraces ........ 37
2.2.12. Stonewalls ........ 38
2.2.13. Rock outcrops ........ 39
2.2.14. Cultural features ........ 40
2.3. Evidence on ecosystem services ........ 41
2.3.1. Hedgerows ........ 42
2.3.2. Tree rows ........ 49
2.3.3. Individual trees ........ 51
2.3.4. Groups of trees/shrubs ........ 53
2.3.5. Herbaceous strips ........ 54
2.3.6. Herbaceous parcels ........ 64
2.3.7. Ditches ........ 65
2.3.8. Small wetlands ........ 67
2.3.9. Small ponds ........ 69
2.3.10. Riparian vegetation ........ 70
2.3.11. Terraces ........ 76
2.3.12. Stonewalls ........ 79
2.3.13. Cultural features ........ 80
2.4. Evidence of economic impacts of LFs ........ 82
2.5. Evidence synthesis for biodiversity and ecosystem services ........ 83
2.5.1. Integrated evidence on biodiversity and ecosystem services ........ 83
2.5.2. Context dependence of delivery of ecosystem services ........ 89

**3. The role of landscape features in a changing climate** ........ 95

3.1. Background ........ 95
3.2. Methodology ........ 95
3.3. Results ........ 95
3.3.1. Landscape features in general ........ 96
3.3.2. Grassy features ........ 98

3.3.3. Woody features ... 98
3.3.4. Wet features ... 99
3.3.5. Stony features ... 100
3.4. Knowledge gaps ... 100

**4. Natural Capital Accounting – from empirical evidence to monetary values ... 101**

4.1. Background ... 101
4.2. Methodology ... 101
4.3. Monetisation of ecosystem services ... 102
4.3.1. Purposes of monetising ecosystem services ... 102
4.3.2. Conceptual basis of monetising ecosystem services ... 102
4.4. Literature review: economic valuation of ecosystem services by landscape features 103
4.4.1. Types of economically valued landscape features and ecosystem services in literature 103
4.4.2. Patterns in the valuation of ES provided by LFs ... 104
4.4.3. The role of scale ... 105
4.4.4. Conclusions from literature review ... 105
4.5. Modelling of Natural Capital for biophysical and monetary valuation ... 106
4.5.1. Concept of NCM ... 106
4.5.2. Modelling of different ecosystem services and landscape features ... 107
4.5.3. From Natural Capital Modelling to Natural Capital Accounting ... 109
4.6. Examples from studies combining LFs and NCA or NCM ... 110
4.7. Discussion ... 111
4.7.1. Modelling of ecosystem services by LFs ... 111
4.7.2. Monetisation of ecosystem services by LFs ... 112

**5. Evidence from study areas ... 114**

**6. Crosscutting insights and patterns ... 120**

6.1. Key insights on biodiversity and ecosystem services ... 120
6.2. Climate change impacts ... 121
6.3. Key insights on economic valuation of ecosystem services ... 122
6.4. Strategic recommendations ... 122
6.5. Knowledge gaps and research needs ... 124

**7. Acknowledgements ... 126**

**8. References ... 127**

**9. Annexes ... 138**

## Summary

The LAFERIA project is identifying the key factors that can promote the (re)introduction of landscape features (LFs) in intensive agricultural areas and develop strategies to overcome key barriers to achieving the objective set in the EU Biodiversity Strategy 2030 of restoring a cover of 10% high diversity LFs on agricultural land and of planting 3 billion trees.

Applying a hierarchically structured evidence-synthesis design, this report integrates the currently fragmented knowledge on benefits of LFs for biodiversity and ecosystem services (ES) in high intensity agricultural landscapes. It also outlines approaches for understanding wider societal benefits of LFs through monetisation of ecosystem services.

Our results show strong academic evidence that all major types of landscape features in European farmland benefit biodiversity. Reviews consistently find positive effects of herbaceous strips and parcels, and ditches, while hedgerows and riparian vegetation show *taxa*-dependent outcomes. Many supported *taxa* provide key ES such as pollination and pest control, though increased biodiversity does not always translate into improved ES or crop benefits. Academic evidence is also strong and consistent that LFs enhance multiple ES without damage to production. Hedgerows, herbaceous strips, and riparian vegetation are the most multifunctional features, with little indication of negative ES impacts. There are considerable gaps in knowledge on the role of LFs in climate adaptation and on their vulnerability to climate change.

The main trade-off of establishing LFs, especially in intensive systems, is the loss of crop area. Especially complex features such as hedgerows and riparian vegetation are likely to create high opportunity costs for farmers relative to private benefits. This highlights the need to measure ES flows to wider beneficiaries and to value the ES in monetary terms. Comprehensive valuation can strengthen incentives and support restoration of LFs as providers of measurable societal benefits. Natural Capital Modelling can scale empirical evidence on benefits of LFs to landscape-level scenarios, estimating current and future ES supply under land-use or climate change. This could be integrated into Natural Capital Accounting with explicit consideration of landscape features.

Policy and investments should incentivise landscape features (re)introduction to improve biodiversity and ecosystem services, prioritise climate resilient designs, and address potential trade-offs. Decisionmakers should apply natural capital valuation cautiously, combining monetary metrics with other values to ensure that the full range of benefits provided by landscape features is recognised and sustained.

## List of abbreviations

| | |
|---|---|
| EU | European Union |
| LFs | Landscape features |
| ES | Ecosystem services |
| JRC | Joint Research Centre of the European Commission |
| JRC-FP) | JRC-Farming-Practices dataset |
| NCA | Natural Capital Accounting |
| NCM | Natural Capital Modelling |
| PICO | Population-Intervention-Comparator-Outcome |
| RP | Revealed Preference |
| SNA | System of National Accounts |
| SEEA EA | System of Environmental-Economic Accounting – Ecosystem Accounting |
| SP | Stated Preference |
| TEV | Total Economic Value |
| WTP | Willingness to pay |

# 1. Introduction

## 1.1. Background and scope

The LAFERIA project aims to identify the key factors that can promote the **(re)introduction of landscape features in intensive agricultural areas** and develop strategies to overcome key barriers to achieving the objective set in the EU Biodiversity Strategy 2030 of **restoring a cover of at least 10% high diversity landscape features on agricultural land and of planting 3 billion trees**.

**Landscape features** (LFs) are small fragments of natural or semi-natural vegetation in agricultural landscapes. There are attempts to develop a comprehensive and consistent definition of LFs for the European Union (Czúcz et al., 2022a). The EU's Biodiversity Strategy refers to a somewhat different list of elements than, for example, the EU Nature Restoration Regulation. The latter emphasizes that LFs must meet the definition of **high-diversity features**, meaning they provide ecosystem services and support biodiversity - and therefore must be subject to as little negative external disturbance as possible to provide safe habitats for nature.

LAFERIA project uses four LF classes describing functional attributes as suggested in Czúcz et al. (2022b), with an addition of culturally significant elements:

- **woody landscape features** present as narrow strips (treelines, hedgerows, riparian woody vegetation, <20 m wide) or as small patches (e.g. isolated trees, field copses, <0.5 ha);
- **grassy landscape features** covered by permanent herbaceous vegetation (e.g. field margins, buffer strips, small fragments of abandoned land);
- **wet landscape features** including ditches, streams, ponds, springs and the adjacent marsh vegetation (<5 m wide), as well as small wetlands (excluding concrete- or plastic-lined channels or ponds);
- **stony landscape features** such as rock or stone piles (e.g. stone walls, clearance cairns), terraced agricultural landscapes (including both vertical "steps" and flat "land block" parts), rocky outcrops; and,
- **cultural heritage landscape features** include historical mounds, hedge-stone wall combinations (e.g. knicks).

The EU Nature Restoration Regulation also includes trees in **small orchards or agroforestry areas with semi-natural vegetation**. This report excludes these because of the recently conducted synthesis by JRC (Joint Research Centre et al., 2025) and several EU projects focusing specifically on agroforestry, although some of the findings from the report will also be applicable for this type of LFs.

**Creation or reintroduction** refers to the establishment or restoration of LFs in agricultural areas where they have been completely or partially removed, or heavily modified. Specifically, LAFERIA focus on high intensive agricultural systems, defined as a geographical area, often associated to a specific biogeographic region, dominated by agricultural land uses and characterized by a set of landscape attributes, farming practices and management intensity (Moreira & Lomba, 2021).

**Ecosystem services** can be defined as the benefits that ecosystems provide to people (MA, 2005). The Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services (IPBES) introduced a more holistic framework of "Nature's

Contributions to People", defining NCP as "all contributions, beneficial or harmful, that individuals, communities, societies, nations, or humanity as a whole derive from nature". In this report, the term of **ecosystem services** is being used for beneficial contributions and **dis-services** for harmful contributions. Of several alternative classifications of ES, the Common International Classification of Ecosystem Services (CICES V5.2), developed by the European Environment Agency (Haines-Young & Potschin, 2023), is followed. We adopted a simplified version used in recent work of JRC that focuses on a limited number of provisioning, regulating and cultural ES (Joint Research Centre et al., 2025).

## 1.2. Conceptual framework

In interpreting effects of LFs on ES, we followed the Ecosystem Services Cascade model, which explicitly distinguishes between ecological structures on the one hand and their contribution to the well-being of people on the other (Potschin and Haines-Young, 2011) (Figure 1). This distinction is particularly pertinent to biotic regulating services, because impacts on biodiversity or habitat do not automatically translate into changes in ES delivery. A given LF may enhance biodiversity without necessarily resulting in a measurable service, or vice versa. For example, a hedgerow is a structure that provides habitat for organisms such as natural enemies of crop pests or pollinators (intermediate services; Figure 1). These organisms are service providers, and only when their presence and activity lead to tangible outcomes such as reduced pest pressure in adjacent fields or increased pollen deposition on crops can this be considered the actual delivery of an ES (final service). This may in turn translate into on-farm production benefits, such as increased yield and corresponding economic values, including a higher farm revenue. Other benefits may arise, for example, due to reduced pesticide applications and thus broader environmental, biodiversity and human health values. The process may also lead to undesirable impacts or dis-services. For example, an added LF may supress a crop growth through a changed microclimate or competition for resources (e.g. water). This may lead to production disadvantages and associated costs. Though establishing a LF within cropland inevitably leads to a respective reduction in cropping area, this is not considered a dis-service.

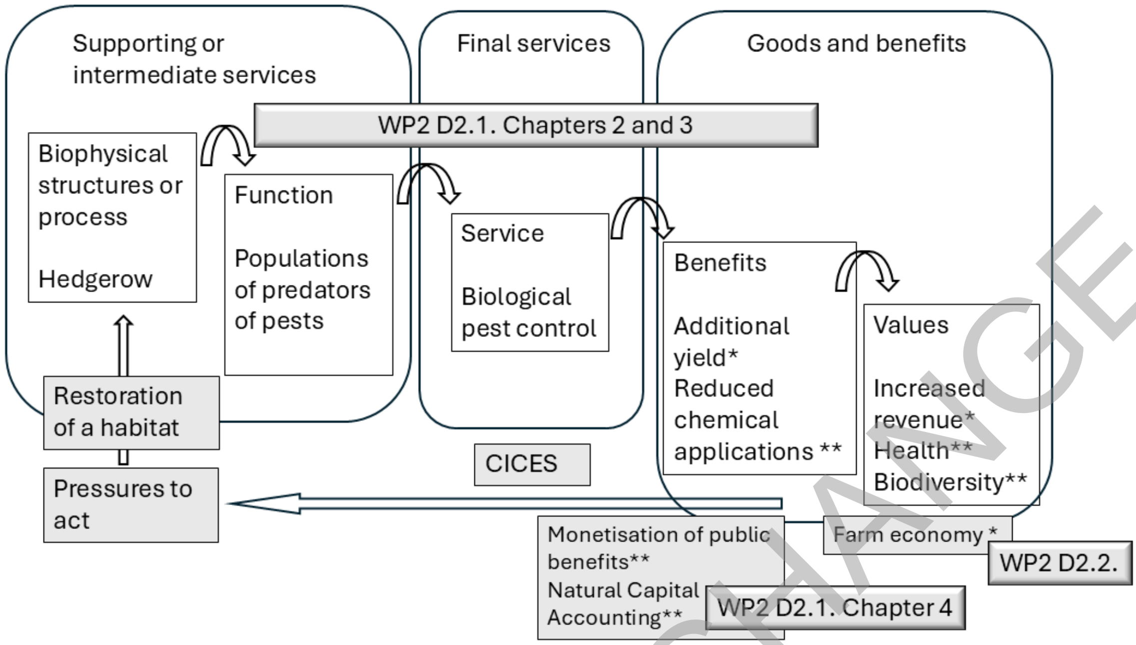



**Figure 1.** An example of the ecosystem services cascade adapted to the pest control ecosystem service. Modified from Potschin and Haines-Young, 2011.

This report works with all steps in the cascade (Figure 1), except the contributions of LFs to the farm economy (focus of Deliverable 2.3. of the LAFERIA project) and implementation part (i.e. pressures to act and restoration models; focus of WPs 3, 4 and 5). Some of the values of ES of public nature could be expressed in economic terms through techniques of elucidating economic values (see Chapter 4). The approach of valuation of ES flows - monetisation of ES - can subsequently inform the volumes of natural capital assets in the area. This is the foundation of Natural Capital Accounting.

The benefits of LFs are increasingly discussed in the context of climate change, highlighting the need to both mitigate impacts and strengthen the resilience of farming systems. Climate mitigation refers to practices that reduce greenhouse gas emissions or enhance the sinks, for example through carbon sequestration in LFs (see Chapter 2) (IPCC, 2021). Climate adaptation and resilience together describe the ability of systems to adjust to climate impacts, withstand disturbances, and recover while maintaining essential functions (see Chapter 3) (IPCC, 2014). Alongside rising average temperatures, climate change has already increased the frequency and intensity of extreme weather events, including droughts, heatwaves, wildfires and heavy precipitation, although the magnitude of these changes varies across regions. Such extremes can strongly affect agricultural production and LFs themselves through heat stress on crops and livestock, increased drought stress leading to higher irrigation demands, waterlogging of crops, nutrient runoff and erosion following heavy rainfall, and physical impacts such as soil sealing or seedbed disruption (Mittler et al., 2025). Furthermore, shifting geographic distribution and phenology of pests and their natural enemies has been disrupting established predator-pest relationships. Finally, climate change has resulted in stronger winds, causing for example an increased evapotranspiration. Such impacts are forecasted to increase in severity, demanding increased attention to resilience of agricultural systems and new investments in infrastructure, some of which can be based on natural solutions. In this context, **natural green and blue infrastructure,** including LF, may have considerable values for increasing resilience of landscapes (Ghofrani et al., 2017).

Because of the benefits of non-productive semi-natural LFs on agricultural land for biodiversity and ecosystem services (ES), they are included into the EU Biodiversity Strategy, Nature Restoration Regulation and are a target of several policy instruments in the Common Agricultural Policy. LFs are currently monitored in the EU targets using the LUCAS survey (European Commission, Eurostat, n.d.). LFs are potential components in the natural capital accounting work. The UN have established a framework for 'ecosystem accounting' named SEEA EA (System of Environmental-Economic Accounting – Ecosystem Accounting) to integrate ecosystems into national accounts to be compatible with the System of National Accounts (SNA). SEEA EA is a spatially-based integrated statistical framework to support that all nations can organise their data on ecosystems and ES in a similar way. It was developed to make the contributions of nature to the economy and people visible (Van Berkel et al., 2021).

## 1.3. Aims and objectives

**This report aims** to synthesise the available evidence on the benefits of LFs for biodiversity and ES, as well as the opportunities for monetising some of the ES flows, while exploring limitations inherent to the processes and opportunities specific to LFs. It also reviews evidence on the potential role of LFs in the context of climate change resilience and adaptation in agricultural landscapes.

The specific objectives of the work leading to this Deliverable were:

1) To integrate the currently fragmented knowledge on the multiple functions performed by various types of LFs in agricultural landscapes with a focus on their **benefits for biodiversity and ES**;
2) To investigate the functioning of the LFs in the view of resilience and adaptation of agriculture to **climate change**, as well as the impacts of climate change on ES delivery and structural integrity of LFs.
3) To connect benefits at farm and farmed landscape levels to wider societal benefits using the **Natural Capital Accounting (NCA** framework.

Work under objective 1 analyses the academic evidence in relation to the context of different agricultural systems and pedo-climatic regions, scale-dependence of recreation at field, farm, and landscape scales, interactive effects between LFs (e.g. optimal combinations), mechanisms through which LFs can contribute to different ES, as well as design principles for successful implementation and efficiency.

Work under objective 2 investigates the potential of the LFs in climate change adaptation and enhancing resilience of agricultural systems in face of the plausible changes.

Work under objective 3 critically reflects on possibilities and limitations of modelling ES for LFs (Natural Capital Modelling - NCM, and approaches of elucidating monetary values of the ES - Natural Capital Accounting). It illustrates these using examples in literature looking at LFs on farmland.

This deliverable employs different forms of evidence synthesis most appropriate for addressing each of its three objectives: systematic literature review, semi-structured review, and a review of grey literature (reports, outputs from other EU projects, technical and policy documents). It follows a hierarchically structured evidence-synthesis design by giving priority to high-quality evidence (e.g. peer-reviewed reviews and synthesis papers) and progressive inclusion of lower-tier sources (grey literature, expert opinion) to address the remaining knowledge gaps identified at each step. Such hierarchical gap-driven synthesis ensures both rigour and comprehensiveness (Müller-Bloch and Krantz, 2015). It prioritises already synthesized evidence over dispersed primary studies, thereby preventing potentially weaker evidence to overshadow more robust findings, while avoiding evidence inflation by consolidating results from the same underlying sources. At the same time, it allows the inclusion of valuable non-academic knowledge where appropriate. It is documented that reviewing of grey literature in addition to peer-reviewed literature may minimise publication bias in publishing positive effects, and that important evidence exists without translating into academic publications (Booth et al., 2016). When integrating evidence from different sources, we maintained transparency regarding the origin, quality, and strength of the evidence used.

Overall, this deliverable is structured into three major parts. Chapter 2 synthesises evidence of LF benefits for biodiversity and ES. Chapter 3 collates evidence on the role of LFs in adaptation and resilience of agricultural systems to climate change, as well as the impacts of climate change on ES delivery and the structural integrity of LFs. Chapter 4 explores opportunities provided by NCM and NCA. Chapter 5 summarises evidence on the benefits of LFs available in the project study areas. Each chapter describes the background, methodological details, results and conclusions pertaining to its specific topic. The report is finalised with an integrated overview of the key findings of all chapters (chapter 6), outlining knowledge gaps and implications.

# 2. Evidence synthesis on landscape feature benefits to biodiversity and ecosystem services

## 2.1. Methods of the Evidence Synthesis

In the hierarchically structured evidence synthesis applied here, the main sources of evidence are organised in four tiers, with knowledge gaps identified in a tier subsequently addressed using sources from the next tier:

1) The recent JRC evidence library, which synthesises peer-reviewed reviews and meta-analyses on the effects of farming practices (Schievano et al., 2026) and the Conservation Evidence online database specifically for biodiversity (Conservation Evidence, n.d.):
2) Peer-reviewed reviews on landscape features (LFs) in agricultural landscapes in Europe or similar bioclimatic context for the gaps identified at stage 1;
3) Peer-reviewed primary studies for the LF-ES gaps after stage 2; and,
4) Relevant reports, including those produced by other EU projects, and practical evidence available from the project study areas.

At every stage, we kept transparency on the type of existing evidence in presenting results, so that the conclusions are clearly linked to the evidence tiers, that is, the source of evidence coming from reviews or individual original studies. We explicitly discuss contradictions in evidence arising from sources in different tiers. We also extracted knowledge gaps as they are described in literature and identified remaining evidence gaps for future research from reviewing evidence of all sources.

### 2.1.1. Definitions and classifications of key concepts

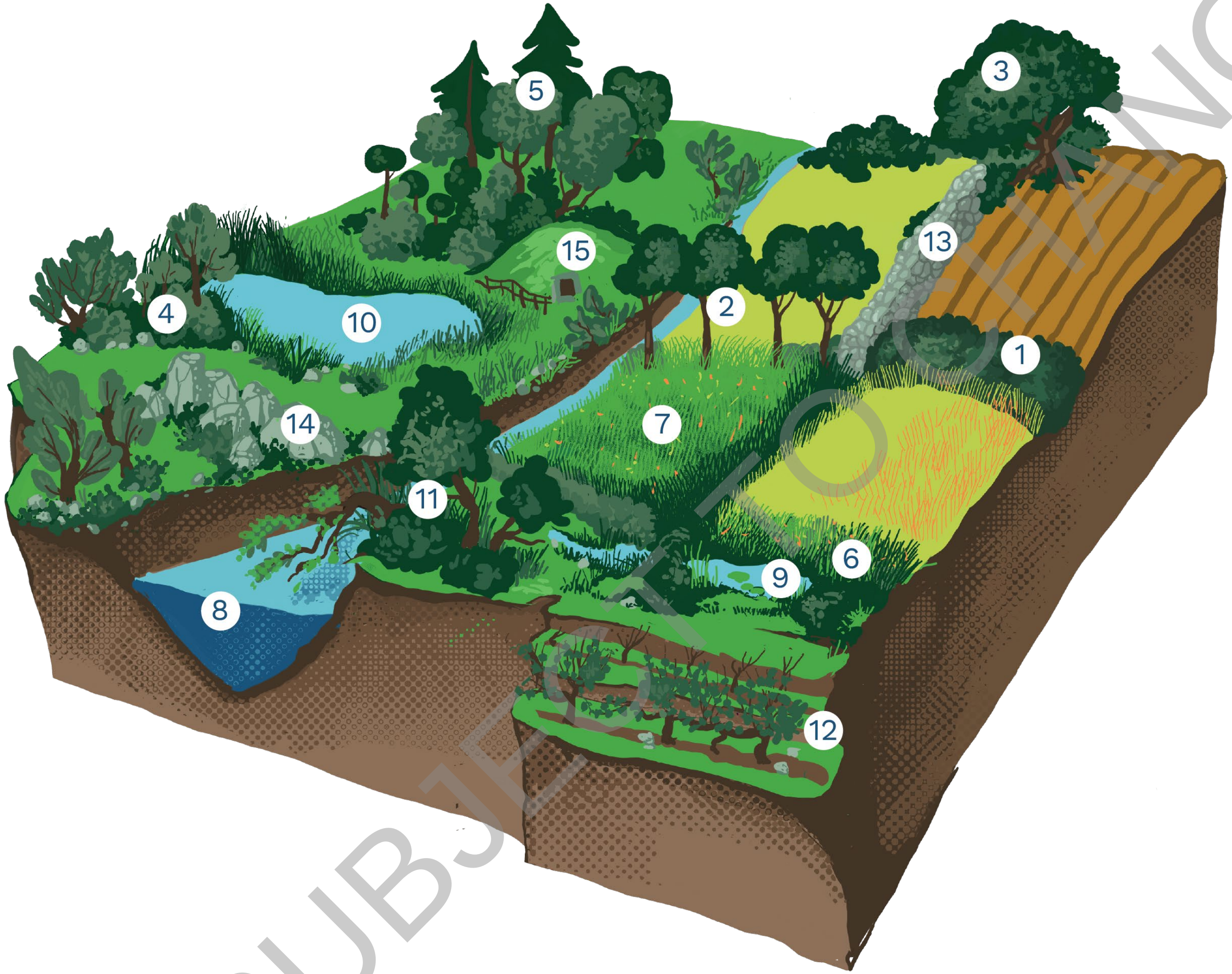
To structure the review analysis, classifications and definitions were established for landscape features, ecosystem services and agricultural systems. They were based on existing sources but adapted to the way of use of these concepts in the evidence synthesis.

The LF categories and definitions are based on the JRC classifications (Czúcz et al., 2022a) and the JRC-Farming-Practices dataset (hereafter, JRC-FP) (Schievano et al., 2026) (Table 1). Although cultural features are not inherently stony, they are classified under stony features. The key LFs included in this report with a typical placement in a landscape are presented in Figure 2.

**Table 1:** Landscape feature categories (LF type) as used in this report, based on and adapted from definitions provided by JRC classifications (Czúcz et al., 2022a) and the JRC-Farming-Practices dataset (Schievano et al., 2026).

| LF type | LF category | Definition |
|---|---|---|
| Woody | Hedgerows | Hedgerows are narrow linear areas covered in unfertilized perennial woody vegetation (shrubs and/or trees). Examples are shelterbelts and windbreaks around orchards or other crops. |
| | Tree rows | Tree rows are single lines of unfertilized trees, not clustered trees or shrubs. |
| | Individual trees | Isolated trees are non-productive trees occurring dispersed or scattered in croplands and/or grasslands, typically as legacies of historical vegetation and land uses. In many regions, old trees on farmland are regarded as having cultural or heritage values. |
| | Groups of trees/shrubs | Trees in group are small patchy areas of woody vegetation (including trees, shrubs and herbs). |
| | Small forests | Small forests are patches of woody vegetation (including trees and shrubs with underlying permanent vegetation) up to a maximum size of 0.5 ha. They can belong to ancient woodland remnants or to new plantations of non-native species. In some areas, they may also contain large stones or stone heaps. |
| Grassy | Herbaceous strips | Narrow linear strips of herbaceous vegetation. Often located at the field edges (either as permanent seminatural features or as sown along field edges as flower strips), within cropped areas (e.g. beetle banks), bordering roads and tracks (roadside verge), adjacent to forests (forest edges), as well as water courses (grass-dominated buffer strips). |

| | | |
|---|---|---|
| | Herbaceous parcels | Cropland with no productive use, including short-term fallow (1 year), medium-term fallow (>1–<6 years), and long-term set-aside (permanent non-productive grassland) that is never intentionally fertilized, sprayed, or tilled. It is called vegetated fallow or set-aside in literature. |
| Wet | Ditches | Ditches are human-made linear surface depressions, temporarily or permanently flooded. Ditches on farmland are typically created to lower the water table, for irrigation, drainage, and/or soil erosion prevention. They are commonly accompanied by grassy strips and often have a grassy slope towards water. We exclude ditches with fully artificial lining (e.g., concrete). |
| | Small wetlands | Small wetlands are small patchy transiently flooded surface depressions covered in wetland vegetation. This class includes the remnants of historical wetland or freshwater ecosystems, and human-made "constructed wetlands" created for treating wastewaters or as a refuge for species. Up to a maximum size of 0.5 ha. |
| | Small ponds | Ponds are small patchy surface depressions permanently covered by water and possibly a narrow strip of wetland vegetation. Up to a maximum size of 0.5 ha. |
| | Riparian vegetation | Riparian vegetation is a linear or patchy area of woody or herbaceous vegetation (including reedbed) bordering a watercourse or waterbody. The area can be submerged under water for a part of the year or be dry year-round. |
| Stony | Terraces | Terraces are human-made structures on sloping terrains created to permit or facilitate cultivation and to reduce the risk of erosion. Terraces consist of one or more "steps" (steep sections covered permanent woody or grassy vegetation or stone walls) and "land blocks" (flat sections that are used for agricultural production, separated by the steps). The specific size, appearance, choice of construction material (i.e., earth, stone or brick), age, land use/vegetation cover of terracing may differ across biogeographical areas. |
| | Stonewalls | Stonewalls are human-made rocky vertical surfaces with a variety of typologies. They have been used since prehistory as retaining walls and/or as field boundaries. |
| | Rock outcrops | Exposed bedrock or stones with minimal soil and vegetation, offering little or no productive use. |
| | Cultural features | Cultural features are local elements of cultural heritage. They often have (pre)historic origin and are often covered in semi-natural vegetation: e.g. sunken lanes, burial mounds, and cairns. |




LANDSCAPE FEATURES:

1. Hedgerow
2. Tree row
3. Individual tree
4. Group of trees/shrubs
5. Small forest
6. Herbaceous strip
7. Herbaceous parcel
8. Ditch
9. Small wetland
10. Small pond
11. Riparian vegetation
12. Terrace
13. Stonewall
14. Rock outcrop
15. Cultural feature

**Figure 2**. Representation of the landscape features included into the report. Pensoft.

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

For the ecosystem service (ES) categories, we adopted the classifications used in the JRC-FP dataset, with several adaptations reflecting the objectives of this report and review results (Table 2). Three services were added: *Biomass production*, *Water regulation* and *Cultural services*. Four services were also renamed: *Pests and diseases* → *Pest and disease control*, *Soil erosion* → *Soil erosion control*, *Nutrient leaching and runoff* → *Nutrient retention*, *Soil nutrients* → *Soil nutrient levels*. We formulated the ES definitions aligning with the reviewed literature, drawing on definitions in the JRC-FP dataset and from the widely adopted CICES ES list (Haines-Young & Potschin, 2023).

**Table 2:** Ecosystem service categories and their definitions as used in this study, drawing on definitions in the JRC-FP dataset and CICES (Haines-Young & Potschin, 2023).

| ES type | ES category | Definition |
|---|---|---|
| Provisioning services | Animal production | Productivity of livestock in terms of outputs (e.g., milk, meat, wool), as well as animal welfare (e.g. access to shade). |
| | Grassland production | Yield and quality of grassland adjacent to LF. |
| | Crop yield | Yield and quality of crops grown for food, feed, fibre or biomass production on crop fields adjacent to LF. |
| | Biomass production | Biomass production for use as a source of energy or fodder, originating from LF that are actively managed and harvested, such as woody vegetation or grassy strips. |
| Regulating services - biological | Pest and disease control | Natural pest and disease regulation, measured by indicators of service provision in crop fields (e.g., predation/parasitism rates, reduced pest pressure or crop damage) adjacent to LF. |
| | Pollination | Insect pollination services, measured by indicators of service provision in the crop field (e.g., crop visitation rates, pollinators occurring in the crop) adjacent to LF. |
| Regulating services - climate | Carbon sequestration | Capturing and storing carbon dioxide from the atmosphere in the biomass of LF and in the soil below and adjacent to them. |
| Regulating services – soil | Soil erosion control | Controlling or preventing soil or sediment loss by running water of wind on crop fields adjacent to LF. |
| | Nutrient retention | Reducing nutrient losses from crop fields adjacent to LF to the surrounding area by removing nutrients transported in soil water by various mechanisms. |
| | Soil nutrient levels | Influencing soil nutrient levels in crop fields adjacent to LF. |
| | Soil water retention | Influencing soil moisture levels in crop fields adjacent to LF. |
| Regulating services - water | Water quality | Improving water quality by influence on decomposing, filtering and sequestering pollutants, measured as the difference between |

| | | |
|---|---|---|
| | | water quality entering and exiting the LF area adjacent to a crop field. |
| | Water regulation | Influencing the ability to manage water flows in crop fields adjacent to LF. |
| Cultural services | Cultural services | Non-material benefits from LF, such as recreation, aesthetics, sense-of-place, inspiration, education, and cultural identity. |

In analysing LF evidence by agricultural systems in view of the focus of LAFERIA project, we used a modified version of the classes of High Intensity Agricultural Systems as given in table 1 of deliverable D1.1. (LAFERIAM12_20260309) (Table 3).

**Table 3.** Broader agricultural systems classifications considered in this study. Based on Table 5 of milestone report "M1. Map of agricultural systems selected and geographic boundaries of regional areas encompassing case study locations defined".

| Agricultural systems |
|---|
| **Arable land** |
| Irrigated arable land |
| Non-irrigated arable land |
| Rice fields |
| **Permanent crops** |
| Fruit trees and berry plantations |
| Vineyards |
| Olive groves |
| **Grasslands** |
| Permanent grasslands |
| **Mixed crop systems** |
| Complex agricultural landscapes |
| Croplands with significant natural vegetation |

### 2.1.2. Tier 1 sources: JRC-FP dataset and Conservation Evidence

As a starting point, we used the JRC-FP dataset, an evidence library on the effects of Farming Practices on the environment and climate developed by the Joint Research Centre (Schievano et al., 2026). The database compiles English-language, peer-reviewed systematic reviews and meta-analyses reporting pairwise comparisons between one or several LFs and a control without the LF according to the PICO framework (PICO = population, intervention, comparator, outcome). The dataset assesses agronomic, environmental, or climatic impacts (Joint Research Centre et al., 2025). Within this library, LF management is one of the main studied categories.

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

The file '10_pico_cat_results.csv' was downloaded from the JRC-FP dataset repository (Schievano et al., 2026). The dataset was filtered using the farming practices identifier (fpid) corresponding to "Landscape features." The JRC-FP landscape feature categories that could not be assigned to LUCAS categories, primarily those combining multiple feature types or broader landscape variables, such as 'Linear elements' or 'Percentage of natural area', were excluded from the review, because our analysis focused only on LFs that could be clearly assigned to the individual LF categories used in this study (see Table 1). When a review did not report results at the level of specific LFs but instead combined multiple features or assessed broader landscape complexity, it was not possible to attribute the effects to a single LF, and these cases were therefore not included. Because the JRC-FP dataset is global in scope, we excluded reviews conducted fully outside Europe or under climates dissimilar to those in Europe. This first step resulted in the inclusion of 26 reviews (Figure 3).

We modified some ES classes used in JRC-FP to better align with the ES Cascade model as formalized in CICES (Haines-Young & Potschin, 2023). Reviews originally categorized under "pollination" or "pests and diseases" were re-evaluated based on the indicators used in the study. When these reviews assessed only the abundance or richness of ecosystem service providers within the landscape feature (e.g., pollinator abundance in the field margin), rather than the actual provision of the service (e.g., pollinator abundance within the crop or crop visitation rates), they were recategorized under "biodiversity" in our dataset. The results from such cases are reported in both sub-chapters on biodiversity (2.3) and on the ES (2.4), with the categorization made explicit: under "biodiversity", reflecting the studied biodiversity metrics, and under "pollination" and "pest and disease control", reflecting the potential contribution to the ES service provision. In the ecosystem services chapter (2.4), we clarify which results represent actual measured service provision and which indicate only the potential for service provision.

For biodiversity, we also harvested evidence from Conservation Evidence, the largest online database assessing the effectiveness of conservation actions for biodiversity (Conservation Evidence, n.d.). Here, "effectiveness" refers to the expert-assessed benefit or harm of an action and the certainty of that assessment, based on the available evidence. We focused on the Farmland Conservation and the Mediterranean Farmland synopses (Dicks et al., 2020; Shackelford et al., 2017) to ensure that the evidence came from agricultural contexts. From these synopses, we extracted information on conservation actions related to LFs and biodiversity, resulting in a total of 16 conservation actions. The effectiveness of these actions is presented as additional information in chapter 2.2. For each conservation action, we included the expert-based assessment metrics by Conservation Evidence. These consist of three scores: effectiveness (the magnitude of the benefit), certainty (the strength of the supporting evidence), and harms (the extent of negative side effects). These scores are derived through a structured expert assessment process and are expressed as percentages. Based on the combination of these median scores, each action is assigned an overall effectiveness category (Beneficial, Likely to be beneficial, Trade-off between benefit and harms, Unknown effectiveness, Unlikely to be beneficial, Likely to be ineffective or harmful), enabling standardized comparison across conservation actions. In contrast, Conservation Evidence was not used to harvest evidence on ecosystem services, because the ecosystem service categories used in their evaluations do not align with those applied in our assessment. Consequently, the effectiveness scores and expert evaluations of conservation actions could not be incorporated. This is because individual studies grouped under a single conservation action span multiple ecosystem service categories in our framework. For example, the action category "pollination" in Conservation Evidence includes studies that, according to our classification logic, would fall partly under "biodiversity" and partly under "pollination", depending on the specific metrics assessed. This mismatch in categorization prevented systematic use of Conservation Evidence for ecosystem service evaluations.

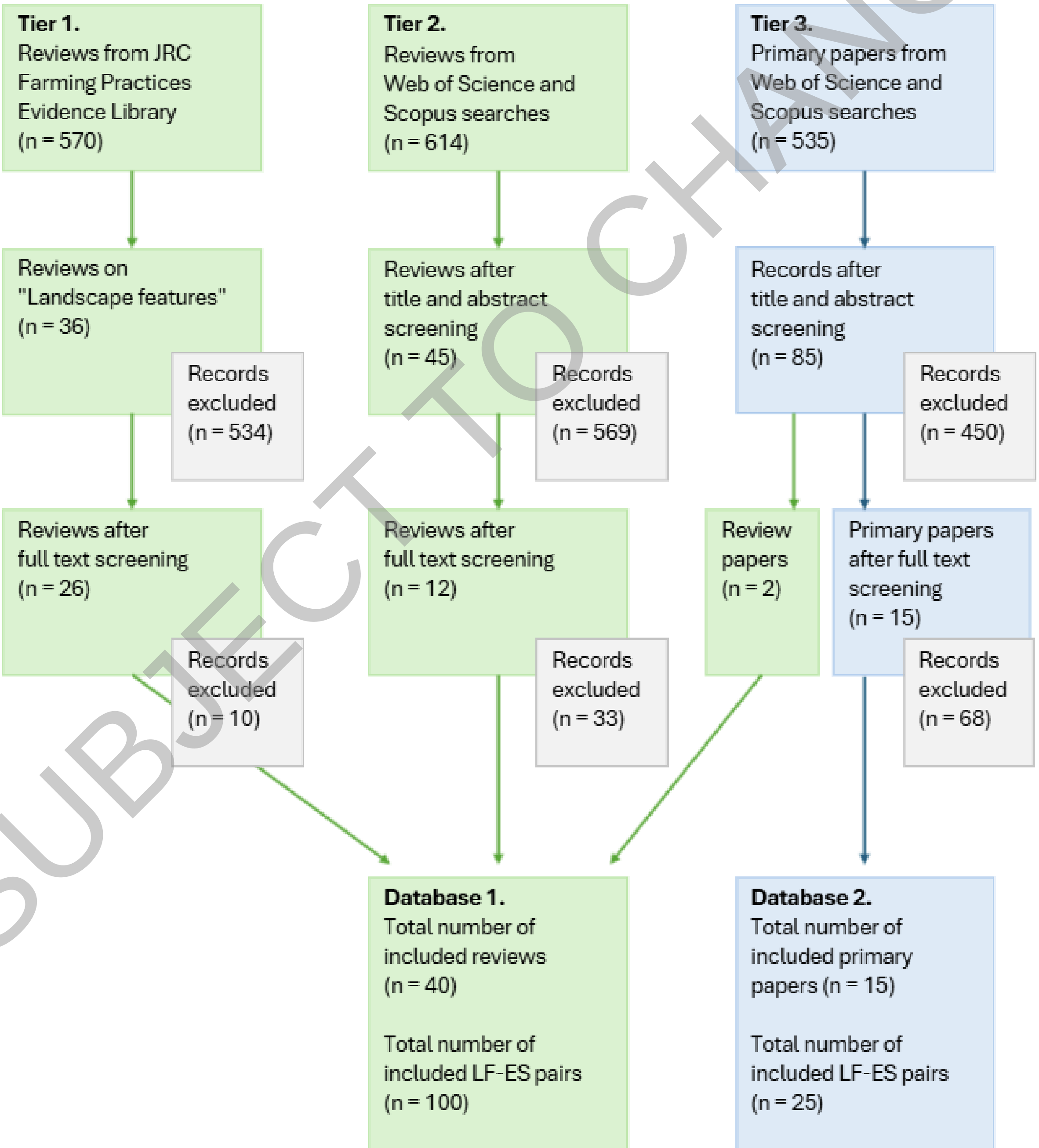



**Figure 3.** Diagram illustrating the workflow of inclusion of reviews and primary

studies through the evidence synthesis process (tiers 1, 2 and 3). Work with grey literature (tire 4) is not included because it was not conducted systematically.

### 2.1.3. Tier 2 sources: reviews for gaps

We have identified three initial gaps of relevance to the project in the JRC evidence:

1) LF categories that were insufficiently covered by JRC-FP dataset search terms: *Individual trees*, *Herbaceous parcels*, *Streams, Riparian vegetation*, *Floodplains* and *Cultural features* (1900-October 2025);
2) Three missing ecosystem services: *Biomass production*, *Water regulation*, and *Cultural services* (1900-January 2026); and,
3) Time span extended from May 2022 to June 2025.

First, we carried out targeted searches for **landscape features** included in the LUCAS LF categories but not represented in the JRC-FP database. A separate set of search terms was developed for this purpose (Annex 1). Second, we conducted new searches for the LFs already covered in the JRC-FP, but this time screened the results for three ES categories that were outside the JRC-FP scope but relevant for our assessment.

To ensure the inclusion of more recent literature from the time of JRC-FP evidence synthesis, we conducted an additional literature search covering the period May 2022–June 2025, corresponding to the interval after the most recent JRC-FP update. We applied the same search methodology and search terms as those used in the original JRC-FP dataset.

The resulting records were screened using inclusion criteria similar to those applied in the JRC-FP dataset (Schievano et al., 2026). However, the requirement for pairwise comparisons were adapted to also include qualitative review papers relevant to our topics of interest.

These searches resulted in the inclusion of 12 additional reviews.

### 2.1.4. Tier 3 sources: primary papers

For LF-ES combinations for which no reviews were available, we conducted an additional search to identify relevant primary research papers. The approach followed a similar procedure to the supplementary search for reviews. However, the search strings were more specific and structured as combinations of LF terms + ES terms + agriculture-related terms + geographical/climatic terms + “ecosystem servic*” (Annex 1). This approach aimed to retrieve studies addressing specific LF-ES combinations within a temperate or European agricultural contexts.

The inclusion criteria required that studies: (i) addressed one of the LF categories, (ii) examined the effect of the LF on an ecosystem service or biodiversity, (iii) were conducted in an agricultural context in Europe or under comparable climatic conditions, (iv) were published in English, and (v) were primary research papers. Papers on LF-ES combinations for which reviews were available were thereafter excluded. This resulted in the inclusion of 15 primary papers. In addition, the search disclosed two relevant review papers that were incorporated into the review database.

Tier 4 sources: grey literature and evidence from study areas

We used Google search to identify relevant reports and projects, checked references in them, and consulted the project partners. We also asked the leaders of the case study area for most valuable evidence sources pertaining to their respective areas, regions or countries. This evidence includes diverse sources: academic and grey literature as well as primary papers identified in the reviews.

We used this evidence as additional to that derived from the other tires, mostly as examples. The evidence for the project study areas is summarised in Chapter 5. The summaries and links from the projects and reports are collated in Annex 2.

### 2.1.5. Data extraction and reporting results

We built and published two datasets: one comprising evidence on ES and LFs extracted from review papers, and a second comprising evidence from primary research papers (Keränen et al., 2026). The database on review papers included columns for topics such as effects on ES, study context, ES delivery mechanisms, influential factors, knowledge gaps, climate change, economic performance, and design and management aspects. We screened the studies and extracted relevant information into the predefined columns, recording it either as direct quotations from the original papers or as concise summaries of the reported findings. In total, the review database includes 40 review papers: 26 from the JRC-FP dataset, 12 identified through supplemental searches for reviews, and 2 found during the search for primary papers. As many review papers addressed multiple LF–ES combinations, the number of extracted evidence entries exceeded the number of papers, resulting in a total of 100 LF–ES cases derived from the review literature. The database on primary papers follows a simplified structure compared to the review paper database, with fewer extracted details. It includes 15 primary research papers, which together yielded 25 individual LF–ES cases.

The most relevant data extracted from the two review databases were organised by LF–ES combinations in this synthesis report. For each combination, information was structured under the headings: “Context,” “Effectiveness and Mechanisms,” “Factors of Influence,” and “Design and Management.”

From Conservation Evidence (CE), 16 conservation actions were reviewed. We did not integrate this evidence into the databases derived from literature because CE is conducted through a different approach. First, CE provides expert assessments of conservation actions, only some of which focus on introducing a LF, while others on involving management interventions applied to an existing feature, making them conceptually distinct from the literature-based data. Second, unlike systematic reviews, which apply consistent criteria for the study design, scale, and metrics and then synthesise results across comparable studies, CE assesses only the overall effectiveness of each action based on a heterogeneous set of primary studies. Unlike the review papers in our database, which allow extraction of information on factors of influence, results by taxonomic groups, and contextual conditions, CE does not synthesise such aspects across studies. The underlying studies for each conservation action are presented as brief “key messages” and short summaries. This precludes a possibility for deriving more detailed information (e.g. taxon-specific effects) in a form compatible with our review-based data. We used

CE actions as review-type summaries of action effectiveness, while recognising that more detailed study-level information cannot be extracted in a compatible format. Finally, the CE evidence may rely on much of the same primary literature as the more recent JRC evidence synthesis, although the level of overlap is unclear. For these reasons, CE actions are presented only as complementary qualitative information in the report text.

### 2.1.6. Limitations and potential biases

This review methodology is subject to limitations and potential biases:

- The JRC-FP dataset used in Tier 1 strictly follows a PICO framework, which excludes study designs that do not conform to this structure. As a result, the evidence base is biased towards field-scale experimental studies, while underrepresenting approaches such as case studies and modelling scenarios, which are particularly relevant for services like water and climate regulation and for analysing scale dependence.
- The initial focus on review studies limits the availability of detailed contextual information that could be relevant for the report objectives.
- The review design focuses on synthesizing the effectiveness of individual LFs, thereby excluding studies that assess LFs in a more integrated or aggregated manner, which could be valuable for our objectives. In addition, the emphasis on small features introduces uncertainty, as feature size is often not consistently reported in literature.
- In the search for primary papers, the inclusion of the search term "ecosystem servic*" may have introduced selection bias by restricting the search to studies explicitly using the ecosystem services framework, following Rau et al. (2020). This likely excludes relevant studies that do not adopt this terminology (e.g. studies referring to yield without framing it as a provisioning ecosystem service). However, this choice was made as a pragmatic compromise to ensure feasibility of the rapid review while maintaining conceptual consistency across the evidence base.

## 2.2. Evidence on biodiversity benefits

The evidence on the effects of landscape features (LFs) on biodiversity is organised into separate chapters for each LF type. Table 4 summarizes both the amount of evidence and the direction of these effects.

**Table 4.** Summary of the effects of LFs on biodiversity. The direction of the effect is indicated by colour (green = positive, purple = negative, yellow = neutral, and blue = mixed across studies). The number in each cell indicates the number of papers on which the evidence is based.

| | Woody | | | | | Grassy | | Wet | | | | Stony | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Hedgerows | Tree rows | Individual trees | Groups of trees/shrubs | Small forests | Herbaceous strips | Herbaceous parcels | Ditches | Small wetlands | Small ponds | Riparian vegetation | Terraces | Stonewalls | Rock outcrops | Cultural features |
| Biodiversity | **2 (+)** | **1** | **1** | **1** | | **9 (±)** | **2** | **1** | | *1* | **3 (±)** | *1* | *3 (±)* | *1* | **1** |
| **Count of LF effects per effect direction** | | | | | | | | | | | | | | | |
| Positive | | **1** | **1** | **1** | | **7** | **2** | **1** | | *1* | **2** | *1* | *2* | *1* | **1** |
| Mixed | **2 (+)** | | | | | **2 (±)** | | | | | **1 (±)** | | *1 (±)* | | |

Evidence type:
**review paper**
*primary paper*

Direction of effect:
- Positive
- Negative
- Neutral
- Mixed

Mixed effect type:
(+) mix of positive and neutral effects
(-) mix of negative and neutral effects
(±) mix of positive and negative effects

### 2.2.1. Hedgerows

***Context***

Results are based on two reviews (Donkersley et al., 2023; Van Vooren et al., 2017) focusing on arable cropping systems in temperate regions of the global north. Predatory insects and pollinators were the functional groups assessed.

Additional evidence is included from two Conservation Evidence synopses (Dicks et al., 2020; Shackelford et al., 2017) covering two hedgerow-related interventions in farmland contexts. The first intervention in the Evidence synopsis is planting hedgerows (Shackelford et al., 2017). The second is managing hedgerows to benefit wildlife, includes practices such as reducing hedge cutting frequency, limiting pesticide and herbicide use, mowing vegetation beneath hedges, and using traditional methods such as coppicing and hedge laying to restore structure (Dicks et al., 2020).

*Effectivity and mechanism*

Hedgerows have a mix of positive and neutral effects on biodiversity. Predatory insect diversity increased significantly in the presence of hedgerows, whereas predatory insect density tended to increase, but not to a statistically significant extent (Van Vooren et al., 2017). For pollinators, species richness increased, while abundance gains from hedgerows were moderate (Donkersley et al., 2023). The main mechanism is habitat and resource provision: hedges form a three-dimensional network of twigs and branches that provide nesting and overwintering sites, and flowering hedgerows additionally provide food resources for pollinators (Donkersley et al., 2023).

In addition to the review results above, Conservation Evidence provides an assessment of two hedgerow-related interventions. Based on evidence from 20 studies, wildlife-friendly hedgerow management is assessed as *likely to be beneficial* for biodiversity. The action has an estimated effectiveness score of 70%, a certainty score of 50%, and no assessment of potential harms (Dicks et al., 2020). For planting hedgerows, evidence from three studies is insufficient to determine overall effectiveness, and it is categorized as *unknown* due to limited evidence. The action is assigned an effectiveness score of 20%, a certainty score of 10%, and no assessment of potential harms (Shackelford et al., 2017). In summary, wildlife-friendly hedgerow management is likely to yield biodiversity benefits, while evidence for hedgerow planting remains uncertain due to limited data.

*Factors of influence*

For pollinators, intervention size influenced pollinator responses: the greatest effect of hedgerow interventions was observed in small plots (~40 m²), where species richness showed a 0.041-fold increase per m² (Donkersley et al., 2023). Data were insufficient to evaluate pollinator responses to larger hedgerow plantings. When multiple pollinator conservation intervention types (i.e. wildflower strips, hedgerows, no-mow schemes) were analysed together, interventions of approximately 500 m² supported the highest overall pollinator species richness and abundance. Above

~500 m², intervention benefits increased only marginally, indicating diminishing returns with scale (Donkersley et al., 2023).

For predatory insects, explanatory variables were not tested because no data were available in the retained studies.

*Design and management*

Small-scale hedgerow interventions (~40 m²) deliver high pollinator benefits per unit area, though they are not expected to outperform larger hedgerows in total biodiversity support. Their limited spatial footprint and relatively low establishment and maintenance costs may increase feasibility and adoption by landowners, particularly where larger hedgerow creation is constrained (Donkersley et al., 2023).

*Knowledge gaps*

Not described.

### 2.2.2. Tree rows

*Context*

Results are based on a global review (England et al., 2020) focusing on grazed dairy or mixed dairy/beef systems. The landscape feature categorised here as tree rows corresponds to "shelterbelts" in that review, defined as "linear plantings of trees along pasture edges to provide shade and shelter" (England et al., 2020).

*Effectivity and mechanism*

Tree rows were found to increase invertebrate species richness, abundance, and composition with medium confidence. They also improved habitat function, again with medium confidence. These effects are due to two key mechanisms, both supported by medium-confidence evidence: first, tree rows provide habitat and structural complexity, supporting greater invertebrate diversity; second, they enhance habitat quality, which contributes to improved overall habitat function (England et al., 2020).

*Factors of influence*

Not described

*Design and management*

Not described.

*Knowledge gaps*

Not described.

### 2.2.3. Individual trees

*Context*

Results are based on the same global review of England et al. (2020) mentioned above. The landscape feature categorised here as *individual trees* corresponds to "pasture trees" in the review, defined as "scattered trees in existing pastures at varying densities; may be remnant trees, natural regeneration or planted" (England et al., 2020).

*Effectivity and mechanism*

Individual trees increase invertebrate species richness, abundance, and composition with medium confidence. There is medium confidence that this effect is driven by the habitat provision offered by trees, which supports greater invertebrate diversity (England et al., 2020).

*Factors of influence*

Not described.

*Design and management*

Not described.

*Knowledge gaps*

Not described.

### 2.2.4. Groups of trees/shrubs

*Context*

The underlying evidence is based on England et al. (2020). The landscape feature categorised here as groups of trees/shrubs corresponds to "remnant native vegetation" in the review, defined as "patches of natural woody vegetation existing as fragments within agricultural landscapes, not as large continuous areas" (England et al., 2020).

Additional evidence is included from a Conservation Evidence synopsis (Dicks et al., 2020) on the intervention *planting new hedges*. The action does not define the configuration of the plantings, which may consist of hedge islands or linear hedgerows. Because only two of the eight included studies examined hedgerows specifically, this action is categorized under *groups of trees/shrubs* in this report.

*Effectivity and mechanism*

Groups of trees increase vertebrate species richness, abundance, and composition with high confidence. There is also high confidence that this effect is driven by habitat provision. For invertebrates, groups of trees increase the species richness, abundance, and composition with low confidence. The mechanism is also considered to be habitat provision, but with low confidence (England et al., 2020).

In addition to the review results above, Conservation Evidence provides an assessment of an intervention of planting groups of shrubs. Based on eight studies, the overall effectiveness of planting new hedges is categorized as *unknown* due to limited evidence. The action is assigned an effectiveness score of 60%, a certainty score of 25%, and no assessment of potential harms (Dicks et al., 2020).

*Factors of influence*

Not described.

*Design and management*

Not described.

*Knowledge gaps*

Not described.

### 2.2.5. Herbaceous strips

*Context*

Results are based on nine reviews examining herbaceous strip interventions in agricultural landscapes, with a primary focus on Europe and North America and some global coverage (Christen & Dalgaard, 2013; Donkersley et al., 2023; Ferrarini et al., 2017; Haaland et al., 2011; Jachowicz et al., 2025; Lowe et al., 2021; Pérez-Sánchez et al., 2023; Scheper et al., 2013; Zamorano et al., 2020). The reviews cover cropland- and grassland-based agricultural systems and assess responses in pollinators, natural enemies and pests, insects more broadly, soil biota, birds, and small mammals.

Herbaceous strips include flower-dominated strips (Donkersley et al., 2023; Haaland et al., 2011; Jachowicz et al., 2025; Lowe et al., 2021; Pérez-Sánchez et al., 2023; Scheper et al., 2013; Zamorano et al., 2020), grass strips (Scheper et al., 2013), and herbaceous biomass buffer strips (Christen & Dalgaard 2013; Ferrarini et al., 2017). Although these strip types differ in vegetation structure and management, they share key functional characteristics, notably the provision of habitat and resources.

In addition to the review literature, evidence is supplemented by two Conservation Evidence synopses evaluating the biodiversity impacts of nine herbaceous strip interventions implemented in farmland settings (Dicks et al., 2020; Shackelford et al., 2017). These interventions include (*ad verbatim*): (1) planting nectar-flower or wildflower mixtures to provide forage for bees and other flower-visiting insects; (2) planting grass buffer strips or margins around arable or pasture fields using grass-rich seed mixtures; (3) planting wild-bird seed or cover mixtures -typically composed of cereal, maize, or sunflower- to provide food and cover for farmland birds; 4) creating uncultivated margins around intensive arable or pasture fields by allowing natural regeneration with mowing applied if needed; 5) creating beetle banks, defined as approximately 2-m-wide grassy ridges established across arable fields to provide overwintering habitat for predatory insects and potential habitat for birds and small mammals; (6) leaving cultivated but uncropped margins or plots to provide habitat for a wide range of farmland taxa; (7) establishing or maintaining vegetated buffer strips alongside rivers and streams, primarily to reduce runoff while also providing habitat; (8) planting nettle strips to enhance invertebrate diversity;

and (9) planting flowers, an action not further specified in the evidence base (Dicks et al., 2020; Shackelford et al., 2017).

*Effectivity and mechanism*

Biodiversity outcomes associated with herbaceous strips are mixed when considered across strip types, taxa, and landscape contexts. However, flower-dominated strips consistently support pollinators and other insect groups, whereas herbaceous biomass strips show more variable biodiversity responses.

Herbaceous strips dominated by flowering vegetation consistently show positive effects on biodiversity, particularly for pollinators. Across five reviews, both pollinator abundance and species richness increased significantly following the establishment of flower strips (Donkersley et al., 2023; Lowe et al., 2021; Pérez-Sánchez et al., 2023; Scheper et al., 2013; Zamorano et al., 2020). Evidence for insects more broadly indicates that wildflower strips generally support higher overall abundances than sown grass margins, although certain ground-dwelling taxa exhibit preferences for grass-dominated vegetation (Haaland et al., 2011). For natural enemies and pests, one review reported a clear increase in natural enemy abundance (+48% to +70%), while pest populations showed no significant response (Jachowicz et al., 2025). The positive effects of flower strips are primarily attributed to enhanced resource availability and habitat provision. Higher plant diversity increases habitat complexity and provides a more diverse supply of pollen, nectar, and alternative prey resources (Scheper et al., 2013; Jachowicz et al., 2025). Additionally, perennial flower strips offer overwintering sites, supporting the persistence of invertebrate populations across multiple years (Haaland et al., 2011; Jachowicz et al., 2025).

Grass strips also exhibit positive biodiversity effects. Increases in both pollinator species richness and abundance have been reported across bees (Apiformes), hoverflies (Syrphidae), and butterflies and moths (Lepidoptera), with no significant differences in effect magnitude among these taxonomic groups (Scheper et al., 2013). Similar to flower strips, these effects are primarily driven by the creation of ecological contrast, whereby grass strips enhance resource availability, especially relative to intensively managed cropland (Scheper et al., 2013).

Herbaceous biomass strips can also generate positive effects on biodiversity overall, although responses are mixed and strongly dependent on previous land use (Christen & Dalgaard 2013; Ferrarini et al., 2017).

Ferrarini et al. (2017) reported positive effects on both below- and aboveground biodiversity when herbaceous biomass buffers replaced cropland. In these cases, herbaceous biomass crops increased several soil biodiversity indicators, including arbuscular mycorrhizal fungi as well as the diversity and abundance of microarthropods, nematodes, and earthworms. Aboveground benefits were reported for arthropods, ground beetles, small mammals, and birds (Ferrarini et al., 2017). Similarly, Christen & Dalgaard (2013) reported that herbaceous biomass strips provide foraging resources for seed-eating birds, offer ground cover for small mammals, and support high plant diversity, likely due to low agrochemical inputs and the presence of untreated headlands. Beyond local resource provision, grassy biomass strips enhance biodiversity at the landscape scale by increasing habitat heterogeneity and creating ecotones between cropped fields and adjacent habitats.

These edge habitats are often particularly species-rich, as they support species from neighbouring habitats as well as species specialised to ecotonal conditions (Christen & Dalgaard 2013).

In contrast, biodiversity responses were negative or mixed when biomass strips replaced existing grasslands, highlighting the importance of previous land use (Ferrarini et al., 2017). In addition, some open-field bird species may be negatively affected by the conversion of open farmland to tall, dense bioenergy crops such as miscanthus (Christen & Dalgaard 2013).

Across the nine herbaceous strip interventions assessed by Conservation Evidence, six supported beneficial biodiversity outcomes. The strongest evidence relates to planting nectar flower or wildflower mixtures, which is classified as beneficial based on 104 studies, with an effectiveness score of 100% and a certainty score of 75%, and no assessment of harms (Dicks et al., 2020). Planting grass buffer strips or margins around arable or pasture fields is likewise considered beneficial, supported by 69 studies and associated with an effectiveness score of 90% and a certainty score of 65%, with no assessment of harms (Dicks et al., 2020). Planting wild bird seed or cover mixtures is also assessed as beneficial, supported by 49 studies and linked to an effectiveness score of 100% and a certainty score of 65%, with no assessment of harms (Dicks et al., 2020). Similarly, creating uncultivated margins around intensive arable or pasture fields is supported by 49 studies and classified as beneficial, with an effectiveness score of 100% and a certainty score of 63%, and harms not assessed (Dicks et al., 2020).

Creating beetle banks is categorised as likely to be beneficial, based on 24 studies, with an effectiveness score of 80% and a certainty score of 60%, and no assessment of harms (Dicks et al., 2020). Finally, leaving cultivated, uncropped margins or plots is also considered beneficial, supported by 20 studies and associated with an effectiveness score of 100% and a certainty score of 65%, with no assessment of harms (Dicks et al., 2020).

In contrast, the remaining three interventions are characterised by limited evidence and uncertain outcomes. Establishing or maintaining vegetated buffer strips along rivers and streams is supported by only seven studies, which was insufficient to determine overall effectiveness, resulting in the intervention being categorised as having unknown effectiveness, with an effectiveness score of 10% and a certainty score of 15%, and no assessment of harms (Dicks et al., 2020). Planting nettle strips is supported by a single study and is therefore also categorized as having unknown effectiveness, with an effectiveness score of 50% and a low certainty score of 10%, and no assessment of harms (Dicks et al., 2020). Similarly, planting flowers is categorised as having unknown effectiveness due to small evidence base of three studies, with an effectiveness score of 40%, a certainty score of 20%, and a harms score of 20% (Dicks et al., 2020).

Taken together, the Conservation Evidence assessments indicate that the majority of herbaceous strip interventions supported by substantial evidence bases consistently deliver positive biodiversity outcomes, whereas interventions with smaller evidence bases remain uncertain due to limited data.

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

*Factors of influence*

Landscape context and ecological contrast emerge as key determinants of the effectiveness of herbaceous strips. Two reviews show that, for pollinators, the strongest positive responses occur when high-quality flower strips are embedded within low-quality surrounding habitats, whereas additional benefits are minimal when both the strip and the surrounding habitat are of high quality (Pérez-Sánchez et al., 2023; Scheper et al., 2013). Moderate positive effects are nevertheless observed when both the strip and surrounding habitat are of low quality, indicating that even relatively small enhancements can be beneficial in degraded landscapes (Pérez-Sánchez et al., 2023; Scheper et al., 2013).

Agricultural system type further influences outcomes, particularly for grass-dominated strips. Effects on pollinator species richness and abundance are stronger and statistically significant in cropland systems, while responses in grasslands are weaker and often non-significant (Scheper et al., 2013). Similarly, previous land use strongly influences outcomes for herbaceous biomass strips: establishment on intensively managed cropland generally results in positive biodiversity responses, whereas conversion of existing grasslands leads to more variable or negative effects (Ferrarini et al., 2017). Christen and Dalgaard (2013) further conclude that in intensively managed landscapes -where modern land management has reduced or removed ecotones through sharp contrasts between land uses- the introduction of grass buffers can restore more gradual habitat transitions and thereby enhance biodiversity.

Vegetation diversity and strip characteristics also play important roles. For pollinators, evidence on the effects of sown floral diversity is mixed: some studies report increased pollinator abundance and species richness with higher numbers of flowering plant species (Scheper et al., 2013), while others find no significant relationship (Lowe et al., 2021). Similarly, effects of strip size vary among reviews, with one reporting no measurable effect (Lowe et al., 2021) and another showing markedly stronger biodiversity gains in plots larger than 500 m² (Donkersley et al., 2023). Strip maturity further modifies responses, as older plantings tend to support higher pollinator abundance and richness (Lowe et al., 2021).

For natural enemies, vegetation diversity within herbaceous strips emerges as a particularly strong driver of effectiveness. Flower strips containing more than two sown species increased natural enemy abundance by approximately 70%, and each additional plant species increased abundance by more than 4% (Jachowicz et al., 2025). However, response strength differs among functional groups, with foliar predators showing the most consistent positive responses (Jachowicz et al., 2025), highlighting the importance of community composition. For insects more broadly, flower abundance and plant diversity are important factors. In addition, strip age and successional stage play a crucial role: with age, strips generally support higher insect diversity, their better physiological condition, and provide improved overwintering habitat (Haaland et al., 2011).

*Design and management*

Herbaceous strips are most effective when implemented in structurally simple, intensively managed agricultural landscapes, where they create strong ecological contrast and provide resources that are otherwise limiting (Pérez-Sánchez et al., 2023; Christen & Dalgaard, 2013). In more structurally complex landscapes, where background habitat quality is higher, herbaceous strips need to be designed more carefully to supply complementary resources and enhance structural diversity in order to generate additional biodiversity benefits (Pérez-Sánchez et al., 2023).

Vegetation composition is a key design consideration. Because different insect groups favour different margin types, combining wildflower-only mixtures, grass-wildflower mixtures, and pollen- and nectar-rich mixes can support a broader range of insect taxa (Haaland et al., 2011). For grass-dominated biomass strips, miscanthus and reed canary grass have been identified as particularly beneficial for seed-eating birds (Christen & Dalgaard, 2013), while switchgrass (*Panicum virgatum*) buffers have been associated with increased abundances of arthropods, ground beetles, small mammals, and birds (Ferrarini et al., 2017).

Temporal aspects of management further affect biodiversity outcomes. Maintaining strips of different ages can be beneficial, as variation in successional stage influences vegetation structure, floral composition, and the availability of overwintering habitat (Haaland et al., 2011). For herbaceous biomass strips, late-winter harvesting is identified as an optimal management practice to support biodiversity outcomes (Ferrarini et al., 2017).

*Knowledge gaps*

- There is uncertainty about how populations in wildflower strips depend on surrounding habitats, highlighting the need for baseline data in nearby areas and before strip establishment (Haaland et al., 2011).
- More research is needed on how field-edge plantings affect different pollinator taxa. Longer-duration studies of mature plantings are also needed (Lowe et al., 2021).
- There is insufficient evidence on whether observed increases in pollinators reflect true population growth rather than short-term behavioural changes (Scheper et al., 2013).
- The effect of flower strips on pests is underrepresented in literature (Jachowicz et al., 2025).
- Research on flower strip effectiveness is limited by underrepresentation of certain habitat contrast levels - especially where control sites have high habitat quality - and by uneven coverage of pollinator taxa. Future studies should address these gaps by testing a wider range of habitat contrasts, including diverse land-use contexts, flower strip designs and management practices, reporting results for different pollinator groups and considering landscape effects across spatial scales (Pérez-Sánchez et al., 2023).

### 2.2.6. Herbaceous parcels

*Context*

Results are based on two reviews focused on Europe and North America (Staggenborg et al., 2022; Van Buskirk et al., 2004). The agricultural systems included both croplands and grasslands, and the functional groups included birds (Staggenborg et al., 2022; Van Buskirk et al., 2004), insects, spiders and harvestmen, and plants (Van Buskirk et al., 2004). The studied LFs were set-aside land (Van

Buskirk et al., 2004) and arable or grassland fallows (Staggenborg et al., 2022). In both reviews, these features are defined as fields taken out of production for at least one growing season, managed with low or no inputs, and with vegetation established either through natural regeneration or sowing. These features are categorised here as herbaceous parcels.

Additional evidence is drawn from a Conservation Evidence synopsis by Dicks et al. (2020) evaluating the intervention of providing or retaining set-aside areas in farmland. In the action synthesis, set-aside areas are defined as rotational or non-rotational areas that are either sown with fallow crops or left to regenerate naturally, and which are not ploughed or harrowed except for sowing.

*Effectivity and mechanism*

Herbaceous parcels have consistently positive effects on biodiversity. For birds, insects, spiders and harvestmen, and plants, species richness and densities were significantly higher on set-aside land than on adjacent conventionally managed farmland (Van Buskirk et al., 2004). Similarly, Staggenborg et al. (2022) found that fallows generally enhance farmland bird abundance, with long-term arable fallows (2–25 years) producing the strongest positive effects for birds, while rotational arable fallows (≤1 year) had weaker positive impacts. Grassland fallows resulted in moderately increased bird abundance and substantially enhanced species richness. Overall, fallows support farmland biodiversity by offering food, shelter, and habitat across multiple taxa.

In addition to the review results above, Conservation Evidence provides an assessment of one action related to set-aside areas. Based on evidence from 54 studies, providing or retaining set-aside areas in farmland is assessed as beneficial for biodiversity. The action has an effectiveness score of 90%, a certainty score of 70%, and no assessment of potential harms (Dicks et al., 2020).

*Factors of influence*

The effectiveness of herbaceous parcels in promoting biodiversity is shaped by landscape context, management practices, and species traits. At the local scale, the ecological contrast between fallows and adjacent farmland modulates their biodiversity value: fallows embedded in intensively managed cropland systems generate the largest relative gains, whereas those in grassland or low-intensity systems contribute less because baseline habitat quality is already higher (Staggenborg et al., 2022). At the regional scale, landscapes with less intensive agriculture and strong agri-environmental policies achieve the highest absolute gains (Van Buskirk et al., 2004).

Additional influencing factors included the method of vegetation establishment and parcel characteristics. Naturally regenerated set-asides supported 59% higher species richness than sown covers, and larger plots were associated with higher species richness and densities (Van Buskirk et al., 2004). Parcel age also mattered: species densities of insects, spiders and harvestmen, and plants generally increased over time (Van Buskirk et al., 2004). However, Van Buskirk et al. (2004) found that bird species richness sometimes declined in older plots, suggesting that a combination of rotational and long-term fallows may best support diverse communities. This contrasts with the findings of Staggenborg et al. (2022), who reported that long-term fallows had the strongest positive effects on farmland birds. Responses also differed among bird guilds (Staggenborg et al., 2022). Long-term arable fallows benefited insectivores, granivores, and mixed feeders by providing abundant food, nesting sites, and structural diversity. In contrast, rotational arable fallows mainly supported mixed feeders and open-habitat species, reflecting limited insect availability and nesting cover.

*Design and management*

To maximize biodiversity benefits, farmers should be encouraged to enrol relatively large parcels, as larger set-aside areas support higher species richness across multiple taxa (Van Buskirk et al., 2004). Landscapes should incorporate a heterogeneous mix of rotational and long-term set-aside lands, since fallows of different ages provide resources for different species (Van Buskirk et al., 2004). In long-term fallows, vegetation can become overly dense over time, indicating the need for periodic management to maintain habitat suitability (Staggenborg et al., 2022). At broader spatial scales, the coordinated regional implementation of set-aside programs -ensuring sufficient land area and connectivity- can further enhance conservation outcomes by providing abundant and accessible source habitats for colonization by a wide range of taxa (Van Buskirk et al., 2004).

*Knowledge gaps*

- Further research is needed to improve the effectiveness of fallows in grassland systems (Staggenborg et al., 2022).
- There is limited understanding of how set-aside land functions at the landscape scale and how specific target species respond to them (Van Buskirk et al., 2004).

### 2.2.7. Ditches

*Context*

Results are based on a global review by Dollinger et al. (2015); the agricultural systems of the primary studies were not reported. Additional evidence is drawn from a Conservation Evidence synopsis by Dicks et al. (2020) assessing the intervention of managing ditches to benefit wildlife. Wildlife-friendly ditch management practices include, for example, mowing, grazing, and dredging techniques, as well as differences in timing and frequency.

*Effectivity and mechanism*

Ditches can support high species richness across multiple groups, including plants, invertebrates, insects, amphibians, birds, and mammals. This effect is primarily driven by habitat provision: ditches combine wetland and stream characteristics, with beds, sidewalls, and margins creating a stratification of microhabitats ranging from aquatic to terrestrial. Habitat connectivity further contributes to their positive effects, as ditches serve as ecological corridors in intensively farmed landscapes, facilitating movement and dispersal of diverse species from amphibians to mammals (Dollinger et al., 2015).

In addition to the review results above, Conservation Evidence provides an assessment of one action related to ditches. Based on evidence from 11 studies, managing ditches to benefit wildlife is assessed as likely to be beneficial for biodiversity. The action has an effectiveness score of 40%, a certainty score of 45%, and no assessment of potential harms (Dicks et al., 2020).

*Factors of influence*

There is no analysis of factors influencing ditch biodiversity, but several aspects have been discussed. Vegetation strongly shapes habitat availability and shelter: plant type, cover, and permanence influence species richness, while trees along ditch margins particularly enhance bird diversity. Perennial baseflows support higher invertebrate diversity, whereas intense or highly variable flows can degrade habitats. Seasonal drying may limit predatory fish, thereby favouring amphibians such as frogs and newts. Nutrient loads can stimulate plant and microbial growth but may also lead to eutrophication, negatively affecting some species. Finally, the physical structure of ditches - including beds, sidewalls, and margins - creates a mosaic of aquatic, wetland, and terrestrial microhabitats, with habitat stratification determined by vegetation type and persistence, and water-level fluctuations (Dollinger et al., 2015).

*Design and management*

Ditches support biodiversity most effectively when they offer diverse microhabitats and sheltered ecological corridors. This is typically achieved in highly vegetated ditches with a relatively permanent base flow and low water velocity, which together maintain habitat complexity and stability for multiple taxa (Dollinger et al., 2015).

*Knowledge gaps*

There is limited understanding of how different ditch characteristics and maintenance practices influence biodiversity (Dollinger et al., 2015).

### 2.2.8. Small wetlands

*Context*

The results are based on a Conservation Evidence synopsis by Dicks et al. (2020) on the intervention of creating scrapes and pools. Scrapes and pools are small waterbodies added to wetlands or wet grasslands in agricultural landscapes to increase habitat heterogeneity (Conservation Evidence, n.d.-a; Dicks et al., 2020).

*Effectivity and mechanism*

Based on evidence from six studies from Sweden, the UK, and Ireland, the overall effectiveness of creating scrapes and pools is categorized as unknown due to limited evidence. The intervention is assigned an effectiveness score of 100%, but a low certainty score of 28%, and potential harms were not assessed.

*Factors of influence*

Not described.

*Design and management*

Not described.

*Knowledge gaps*

Not described.

### 2.2.9. Small ponds

*Context*

The results are based on a single primary study conducted in France, in a landscape where natural wetlands had been almost entirely. The study examined how artificial farm ponds contribute to the Odonata (dragonflies and damselflies) species richness and assemblage patterns in an agricultural system dominated by maize and wheat cultivation, cattle pastures, and duck farming (Ruggiero et al., 2008).

*Effectivity and mechanism*

Small artificial ponds had a positive effect on Odonata species composition and richness. A set of 37 artificial ponds collectively supported around 40% of the regional Odonata species pool, despite individual ponds hosting only a limited number of species. This high contribution occurred in a landscape lacking natural freshwater habitats, showing that artificial ponds can functionally substitute lost wetlands. The positive effect was driven by the diversity of pond sizes, habitat structures, and land-use contexts, which together created a mosaic of aquatic habitats suitable for species with different ecological requirements. At the landscape scale, the variety among ponds enabled a broader assemblage to persist than any individual pond could support (Ruggiero et al., 2008).

*Factors of influence*

Two pond-level variables primarily shaped Odonata species richness. Larger ponds supported higher Odonata species richness, whereas the presence of ducks had a negative effect on species richness. This negative effect may result from direct predation on larvae and adults. The authors also examined a potential indirect effect of waterbirds through their impact on macrophyte abundance, assuming that reduced macrophyte cover could negatively affect Odonata. This mechanism was not supported by the data, as neither macrophyte presence nor relative abundance was significantly correlated with Odonata species richness. Other potential indirect effects of waterbird activity (e.g., sediment disturbance, vegetation degradation, or nutrient enrichment) were not examined.

At the landscape level, the variation of ponds with different habitat characteristics contributed to Odonata diversity. However, most of the measured habitat variables (such as vegetation cover, shading, water depth, transparency, or distance to the nearest pond) did not significantly predict species richness at the individual pond scale (Ruggiero et al., 2008).

*Design and management*

Not described.

*Knowledge gaps*

Not described.

### 2.2.10. Riparian vegetation

*Context*

Results are based on three reviews focused on global grazed dairy or mixed dairy/beef systems (England et al., 2020), grasslands and croplands (Ferrarini et al., 2017), and temperate European agricultural systems more broadly (Christen & Dalgaard 2013). The LFs examined are riparian plantings (England et al., 2020), short-rotation woody bioenergy crops (Ferrarini et al., 2017) and woody biomass buffers (Christen & Dalgaard, 2013), categorized in this report under riparian vegetation.

Additional evidence is included from a Conservation Evidence synopsis by Shackelford et al. (2017) on the intervention of restoring habitat along watercourses. The restoration methods included under this intervention are not clearly defined and vary across the included studies. While most of the 24 assessed studies focused on vegetation restoration, others applied different approaches, including physical modifications such as adding gravel to channel margins or restoring river flow. Therefore, this intervention does not exclusively represent riparian vegetation restoration, and findings should therefore be interpreted with caution.

*Effectivity and mechanism*

Riparian plantings increase stream macroinvertebrate species richness, abundance, and community composition with medium confidence (England et al., 2020). These effects are primarily attributed to improvements in habitat quality, including greater substrate diversity, enhanced channel structure, and increased bank stability (England et al., 2020).

Woody biomass buffers also have positive effects on biodiversity across both terrestrial and aquatic systems (Christen & Dalgaard 2013). Habitat quality for aquatic organisms is improved through shading by trees and shrubs, as well as through the input of leaf litter and woody debris. Soil biodiversity benefits from tree root systems, which shape trophic interactions within soil food webs (Christen & Dalgaard 2013).

Aboveground biodiversity also increased in woody bioenergy crops compared to adjacent annual cropping systems, particularly for arthropods, ground beetles, birds (Ferrarini et al., 2017), and small mammals (Christen & Dalgaard 2013; Ferrarini et al., 2017). Woody biomass buffers provide nesting and foraging resources for various taxa (Christen & Dalgaard 2013; Ferrarini et al., 2017). However, overall biodiversity responses were mixed in Ferrarini et al. (2017), as outcomes strongly depended on previous land use: positive effects were observed when woody buffers replaced cropland, whereas negative or mixed effects were reported when they replaced existing grasslands.

In addition to the review results above, Conservation Evidence provides an assessment of one action related to riparian vegetation. Based on 24 studies, the intervention of restoring habitat along watercourses is assessed as likely to be beneficial for biodiversity, with an estimated effectiveness score of 60%, a certainty of 50%, and potential harms of 15% (Shackelford et al., 2017).

*Factors of influence*

Ferrarini et al. (2017) found that conversion of intensively managed cropland to bioenergy buffers generally has positive effects both on belowground and aboveground biodiversity. In contrast, conversion of grasslands was associated with negative or mixed impacts on both below- and above-ground biodiversity.

*Design and management*

Increased abundances of arthropods, ground beetles, small mammals, and birds have been reported particularly in willow-based systems (Ferrarini et al., 2017). Additionally, harvesting bioenergy buffers in late winter is identified as an optimal management practice to support biodiversity outcomes (Ferrarini et al., 2017).

*Knowledge gaps*

Not described.

### 2.2.11. Terraces

*Context*

The results are based on one primary study conducted in Germany, which examined the effects of terracing on plant diversity within a viticultural production system (Wersebeckmann et al., 2024). Terraces leave a substantial proportion of the area (more than 50%) as embankments that remain uncultivated and are managed extensively, with no application of fertilisers or pesticides and no tractor traffic.

*Effectivity and mechanism*

Vinyard terracing was found to have a positive effect on plant diversity. Species richness was about 1.5 times higher in terraced vineyards than in vertically oriented vineyards, with the highest diversity found on terrace embankments exposed to intermediate levels of disturbance. Terrace embankments were also characterised by lower nutrient levels, a condition rarely found in modern agricultural systems and favourable for the development of richer plant communities.

In addition, terracing generated a fine-scale environmental gradient between embankments and inter-rows, resulting from differences in management, nutrient inputs, and disturbance. This gradient produced distinct habitat conditions over short distances, allowing a wider variety of plant species to coexist than in vineyards without terracing (Wersebeckmann et al., 2024).

*Factors of influence*

Plant diversity on terraces was primarily shaped by disturbance intensity, vegetation management practices, and nutrient conditions. Embankments with intermediate disturbance supported higher species richness than abandoned sites with very low disturbance or vertically oriented vineyards with high disturbance. Vegetation management influenced species composition across vineyard compartments: tillage in open and terrace inter-rows favoured annual and ruderal species, whereas vehicular disturbance and frequent mulching in vegetated inter-rows promoted grasses and reduced herb cover.

Nutrient conditions also played a key role: embankments had lower nitrogen levels than inter-rows, partly because fertilisation and mineralisation processes occurred mainly in inter-rows and nutrients were subsequently washed downslope. These lower nutrient levels supported more diverse perennial plant communities, whereas higher nutrient availability in inter-rows led to grass dominance, reducing overall diversity (Wersebeckmann et al., 2024).

*Design and management*

To increase plant diversity on terrace embankments, the study recommends targeted cutting regimes that include biomass removal and careful adjustment of mowing timing and height. Leaving temporary uncut patches can help provide refuges for invertebrates. Reintroducing target species and regional seed mixtures may be necessary especially on newly constructed embankments. Overall, terraced vineyards have strong potential to support species-rich plant communities and habitat heterogeneity if embankments are managed to maintain low nutrient levels and appropriate disturbance regimes (Wersebeckmann et al., 2024).

*Knowledge gaps*

Not described.

### 2.2.12. Stonewalls

*Context*

The evidence is based on three primary studies examining the effect of stonewalls on different bird species in vineyard landscapes in Italy (Assandri et al., 2017; Assandri et al., 2018) and Portugal (Santos et al., 2023).

*Effectivity and mechanism*

Stonewalls elicited species-specific responses. Positive breeding-season effects were found for great tit and blackcap, whose abundance increased where stonewalls occurred, and great tits were also observed using cavities in the walls for nesting (Assandri et al., 2017). Eurasian robin, dunnock, wren, and rock bunting likewise showed increased abundance in vineyards with stonewalls (Assandri et al., 2017).

Traditional dry-stone walls also increased the occurrence of common redstart by providing nesting cavities and foraging opportunities, whereas modern cemented walls offered no comparable benefits (Assandri et al., 2018). Abundance of Black wheatear during breeding season (February-July) was higher in terraced vineyards containing stonewalls, with nests occurring exclusively in wall crevices (Santos et al., 2023).

In contrast, mixed seasonal responses were found for blackbird, chaffinch, and serin: these species showed a reduced breeding-season abundance in vineyards bordered by stonewalls but positive responses in winter (Assandri et al., 2017). The authors suggest that stonewalls on sloping valley sides may be associated with milder winter microclimates; however, this mechanism was not directly linked to the observed species-specific patterns (Assandri et al., 2017).

*Factors of influence*

Not described.

*Design and management*

Stonewalls support biodiversity most effectively when maintained as traditional dry-stone structures that preserve nesting cavities, irregular surfaces, and adjacent marginal vegetation (Assandri et al., 2018; Santos et al., 2023). Replacing them with cemented walls removes nesting sites and reduces associated habitat heterogeneity (Assandri et al., 2018). Retaining structurally complex stonewalls is particularly important for species that nest exclusively in crevices, such as Black wheatear (Santos et al., 2023). In addition, avoiding the clearing or smoothing of stonewall margins maintains weedy edge vegetation that provides resources for a range of species (Assandri et al., 2017; Assandri et al., 2018).

*Knowledge gaps*

Not described.

### 2.2.13. Rock outcrops

*Context*

The results are based on one primary study conducted in evergreen oak wood-pastures (montados) in Portugal, which assessed how small rock outcrops influence plant diversity and functional diversity (Chozas et al., 2022).

*Effectivity and mechanism*

Rock outcrops had a clear positive effect on biodiversity. They increased overall plant species richness and functional diversity, with a large share of the recorded species occurring exclusively on outcrops. The plant communities on outcrops differed markedly from those in the surrounding matrix, with shrubs, trees, and climbers forming an important component of outcrop vegetation (Chozas et al., 2022).

These positive effects are linked to the structural and microhabitat variation provided by rock outcrops. Because rocks limit mechanical clearing and grazing, they allow the development of woody vegetation and provide refuges for species sensitive to light, heat, or disturbance. This microhabitat heterogeneity also benefits other organisms such as lichens, beetles, birds, reptiles, and mammals by supplying habitat and food resources (Chozas et al., 2022).

*Factors of influence*

Plant distribution within outcrops is mainly shaped by light availability and disturbance. Shading from shrubs and small trees favours species adapted to low-light conditions, while the reduced grazing pressure inside rocky patches allows species lacking disturbance-tolerance traits to persist (Chozas et al., 2022).

*Design and management*

Protecting existing rock outcrops can enhance biodiversity in oak wood-pastures by increasing structural and habitat heterogeneity. The authors recommend preserving, enlarging, or reshaping natural outcrops, and even creating artificial ones where rocks are available, as low-cost, biodiversity-friendly nature-based solutions (Chozas et al., 2022). Because rocky outcrops serve as refuges for mesic-adapted species and potentially as stepping stones facilitating movement between habitat patches, they should also be integrated into protected-area networks designed to promote environmental heterogeneity at the landscape scale (Chozas et al., 2022). Under climate-change scenarios, the micro-climatic variation within outcrops may become increasingly important, as these features can act as refugia for specialist species (Chozas et al., 2022).

*Knowledge gaps*

Not described.

### 2.2.14. Cultural features

*Context*

The results are based on a global qualitative review in which the agricultural context of the underlying studies is not specified (Zgłobicki et al., 2021). The studied landscape feature is sunken lanes, categorized here under cultural features.

*Effectivity and mechanism*

Sunken lanes can act as biodiversity hotspots in otherwise simplified agricultural landscapes. They provide refuges for plants and animals, connect distant habitats, and create varied microhabitats due to gradients in topography, moisture, and light (Zgłobicki et al., 2021).

*Factors of influence*

Not described.

*Design and management*

Not described.

*Knowledge gaps*

More research is needed on the management and restoration of sunken lanes to maintain their role as biodiversity hotspots in agricultural landscapes (Zgłobicki et al., 2021).

## 2.3. Evidence on ecosystem services

The evidence on the effects of landscape features (LFs) on ES is organised into separate chapters for each LF type, where the ES for which evidence was found are listed. Table 5 summarizes both the amount of evidence and the direction of these effects.

**Table 5.** Summary of the effects of LFs on ES. The direction of the effect is indicated by colour (green = positive, purple = negative, yellow = neutral, and blue = mixed effect [any combination of the previous effects]). The number in each cell indicates the number of papers on which the evidence is based.

| | Woody | | | | | Grassy | | Wet | | | | Stony | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Hedgerows | Tree rows | Individual trees | Groups of trees/shrubs | Small forests | Herbaceous strips | Herbaceous parcels | Ditches | Small wetlands | Small ponds | Riparian vegetation | Terraces | Stonewalls | Rock outcrops | Cultural features |
| **Provisioning services** | | | | | | | | | | | | | | | |
| Animal production | | **1** | **1** | | | | | | | | | | | | |
| Grassland production | | **1 (±)** | **1 (±)** | | | | | | | | | | | | |
| Crop yield | **1 (±)** | | | | | **3** | | | | | | **1 (±)** | | | |
| Biomass production | *1* | | | | | **2** | | | | | **2** | | | | |
| **Regulating services - biological** | | | | | | | | | | | | | | | |
| Pest and disease control | **1** | | | | | **3** | *1 (+)* | | | | | | | | |
| Pollination | **1 (+)** | | | | | **3 (+)** | | | | | *1* | | | | |
| **Regulating services - climate** | | | | | | | | | | | | | | | |
| Carbon sequestration | **3** | **1** | **1** | | | **2** | *1* | | | | **1** | | | | |
| **Regulating services - soil** | | | | | | | | | | | | | | | |
| Soil erosion control | **3** | | | **1** | | **6** | | **1** | | | **1** | **4** | | | **1** |
| Nutrient retention | **2** | | | | | **6** | | | | | **5** | | | | |
| Soil nutrient levels | | | | | | | | | | | | **1** | | | |
| Soil water retention | | | | | | | | | | | | **2** | | | |
| **Regulating services - water** | | | | | | | | | | | | | | | |
| Water quality | | | | | | *1* | *1* | **5** | **3** | **1** | **5** | | | | **1** |
| Water regulation | *1* | | | | | | | | | *1* | *1* | | | | |
| **Cultural services** | | | | | | | | | | | | | | | |
| Cultural services | *3* | *2* | | *1* | | *1* | | | | *1* | **1** | | *2* | | **1** |

| **Count of LF effects per effect direction** | Hedgerows | Tree rows | Individual trees | Groups of trees/shrubs | Small forests | Herbaceous strips | Herbaceous parcels | Ditches | Small wetlands | Small ponds | Riparian vegetation | Terraces | Stonewalls | Rock outcrops | Cultural features |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Positive | **8** / *5* | **2** / *2* | **2** | **1** | | **19** / *1* | | **6** | **3** | **1** / *2* | **15** / *2* | **7** | *2* | | **1** |
| Negative | | | | | | | | | | | | | | | **2** |
| Neutral | **1** | | | *1* | | **5** / *1* | *2* | | | | | | | | |
| Mixed | **2 (±)** | **1 (±)** | **1 (±)** | | | **1 (+)** | *1 (+)* | | | | | **1 (±)** | | | |

Evidence type:
**review paper**
*primary paper*

Direction of effect:
Positive
Negative
Neutral
Mixed

Mixed effect type:
(+) mix of positive and neutral effects
(-) mix of negative and neutral effects
(±) mix of positive and negative effects

### 2.3.1. Hedgerows

**Crop yield**

*Context*

Results are based on one review (Van Vooren et al., 2017) conducted in temperate arable systems, including maize, cereals, legumes, and mustard as crops. The review synthesised evidence on how hedgerows and grass strips bordering arable fields affect a range of ES, including crop yield.

*Effectivity and mechanism*

Hedgerows have a mixed, but overall positive, effect on crop yield when the full influence zone is considered. Yield is reduced close to the hedgerow due to competition, particularly at low distance-to-height ratios (D/H; a measure describing crop distance from the hedgerow relative to hedgerow height). In the zone from D/H = 0.0 to 2.1, relative crop yield is lower than in open fields, reaching a minimum at D/H = 0.5, where yield is 68% of the control. As distance increases (higher D/H), the effect becomes positive. Beyond D/H = 2.1, yield exceeds hedgerow-free controls, rising to 109% at D/H = 5.0. These yield gains are attributed to the shelter effects and improved microclimatic conditions provided by the hedgerow. When the entire influence zone is considered (up to D/H = 20.4), the net relative crop yield is 103%, indicating that yield reductions near the hedgerow are outweighed by yield increases further into the field, resulting in an overall positive effect (Van Vooren et al., 2017).

*Factors of influence*

The key factor influencing crop yield is the distance-to-height ratio (D/H), which describes how far crops are from the hedgerow compared to the hedgerow's height. When crops are close to tall hedgerows (low D/H), yields are reduced. As the distance increases relative to hedgerow height (higher D/H), yields gradually increase. Tall or wide hedgerows, as well as narrow parcels, increase the area under negative influence and reduce overall parcel-level yield (Van Vooren et al., 2017).

*Design and management*

Tall or wide hedgerows, as well as narrow parcels, increase the area under negative influence and reduce overall parcel-level yield (Van Vooren et al., 2017). To minimise crop yield losses, while maintaining benefits, low and narrow hedgerows are preferred over tall or wide structures. Optimising hedgerow height and width in relation to parcel dimensions reduces the area exposed to competitive effects near the field boundary (Van Vooren et al., 2017).

**Biomass production**

Here, biomass production refers to the additional woody biomass potentially available from LFs in agricultural landscapes.

D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

*Context*

The results are based on a single primary modelling study that assessed the implementation of multifunctional hedgerow windbreaks in European landscapes (Englund et al., 2021).

*Effectivity and mechanism*

The model estimates that establishing hedgerow windbreaks on an additional 1.6-2.1% of the current annual crop area in the EU27 + the UK would result in an increase in biomass production of 18-24 Mt DM per year. The windbreaks are therefore positive for biomass production (Englund et al., 2021).

*Factors of influence*

Not described.

*Design and management*

The modelling approach assumes that only half of the windbreak is harvested at a time to ensure the windbreak functionality (Englund et al., 2021).

**Pest and disease control**

*Context*

Results are based on one review (Albrecht et al., 2020) focused on North America, Europe, and New Zealand. The review assessed how hedgerows influence natural pest control services in adjacent arable and perennial crops. Additional findings are included from Van Vooren et al. (2017), a review that did not evaluate pest control services directly but examined predatory insect communities, indicating the potential for service provision (see Figure 1).

*Effectivity and mechanism*

Hedgerows showed no consistent or statistically significant effect on pest control in adjacent crops (neutral effect). Although mean pest control values were slightly higher next to hedgerows, the effect size was small and highly variable across studies (Albrecht et al., 2020).

One review (Van Vooren et al., 2017) assessed predatory insect communities in parcels with and without hedgerows. Predatory insect diversity increased significantly in hedgerow systems, while predator density showed a positive but statistically non-significant trend. These findings describe the status of the ecosystem service providers, rather than the provision of pest control service itself. Although higher predator diversity may indicate a greater potential for natural pest control, the review did not include data on pest populations, and therefore provides no evidence that the observed increases in predator diversity translated into improved pest control (Van Vooren et al., 2017).

*Factors of influence*

Pest control services declined exponentially with increasing distance from the field edge, but the rate of decline did not differ between fields with and without adjacent hedgerows. No hedgerow-specific characteristics (e.g. floral diversity, age) could be evaluated, as the underlying studies lacked sufficient descriptive data (Albrecht et al., 2020)

*Design and management*

Not described.

**Pollination**

*Context*

Results are based on one review (Albrecht et al., 2020) focused on North America, Europe, and New Zealand. The study assessed how hedgerows influence pollination services in adjacent arable and perennial crops, with the most commonly studied being oilseed rape (*Brassica napus*), strawberry (*Fragaria ananassa*), apple (*Malus domestic*a), and pumpkin (*Cucurbita pepo*).

Additional findings are included from Donkersley et al. (2023), a review that did not evaluate pollination services directly but analysed pollinator communities, indicating the potential for service provision.

*Effectivity and mechanism*

Evidence from Albrecht et al. (2020) indicates a mix of positive and neutral effects of hedgerows on pollination services. At the whole-field scale, pollination levels in fields adjacent to hedgerows did not differ significantly from those in fields without hedgerows. By contrast, pollination was enhanced in crop areas immediately adjacent to hedgerows but declined sharply with increasing distance into the crop. These findings suggest that hedgerows generate localized gains in pollination, which do not scale up to consistent improvements across entire fields.

Donkersley et al. (2023) found that hedgerows increased pollinator species richness, whereas abundance responses were moderate. These findings describe the status of the ecosystem service providers, but not the provision of pollination service itself. The main mechanism underlying positive biodiversity outcomes is habitat and resource provision: hedgerows form a three-dimensional structure of twigs and branches that provides nesting, foraging, and overwintering sites for a wide range of taxa (Donkersley et al., 2023).

*Factors of influence*

In Albrecht et al. (2020), pollination services declined exponentially with distance from hedgerows, while no distance-decay pattern was detected in control fields. Additionally, Donkersley et al., (2023) identified intervention size as an important factor: the strongest effects were observed in small plots (~40 $m^2$), where pollinator species richness increased by 0.041-fold per $m^2$.

*Design and management*

Because pollination declines strongly with increasing distance from floral vegetation, the results indicate that a dense network of relatively small, well-distributed plantings is more effective than a limited number of large features (Albrecht et al., 2020). In addition, reduced cutting frequency is important for maintaining a high availability and diversity of floral resources in hedgerows (Albrecht et al., 2020). Donkersley et al. (2023) suggest that small-scale hedgerow interventions (~40 $m^2$) can enhance pollinator ecological metrics, with modest size and lower cost supporting feasibility and adoption by landowners.

**Carbon sequestration**

*Context*

Results are based on three reviews (Drexler et al., 2021; Van Vooren et al., 2017; Zheng et al., 2020), two of which focus on temperate regions, while the third has a global scope. All three included mostly primary studies conducted outside Europe. The reviews examined cropland systems, including both arable crops and grassland.

*Effectivity and mechanism*

Hedgerows have a positive effect on carbon sequestration in biomass and soil compared to cropland without hedgerows. One review reported a non-significant positive effect when hedgerows are established on grasslands (Drexler et al., 2021). Reported increases in soil organic matter (SOM) within or near hedgerows range from an average of 6 to 37% for carbon and 46% for organic matter, depending on the measurement location and soil depth of measurement (Drexler et al., 2021; Van Vooren et al., 2017; Zheng et al., 2020). Drexler et al. (2021) state that the levels of carbon sequestration are comparable to those of forests. The sequestration is driven by biomass accumulation in woody biomass and increased organic matter inputs to the soil from dead organic matter, in which deep soil carbon inputs from trees with extensive root systems play a large role (Drexler et al., 2021).

*Factors of influence*

Among the many potential influencing factors assessed, only distance to the field edge -that is, proximity to trees- was found to have a significant effect on soil carbon stocks (Van Vooren et al., 2017). Non-significant variables were age, soil texture, climate, height, plant composition, and land use. One review showed that there was no increase in soil organic carbon when hedgerows were added on grassland (Drexler et al., 2021).

*Design and management*

Not described.

**Soil erosion control**

*Context*

Results are based on three reviews, of which two were global and focused on temperate regions, with a large share of the included studies conducted outside Europe (Van Vooren et al., 2017; Xiong et al., 2018), and one review from China (Jia et al., 2019). The reviews examined hedgerows in cropland systems, including arable crops, grassland, and orchards.

*Effectivity and mechanism*

Hedgerows have a significant positive effect on reducing soil erosion, including both soil loss and runoff, when compared to cropland without hedgerows. One of the three reviews, however, reported a non-significant positive effect on runoff (Xiong et al., 2018). On average, hedgerows reduce the loss of soil particles by 20–90% relative to hedgerow-free fields (Jia et al., 2019; Van Vooren et al., 2017; Xiong et al., 2018). For reducing runoff water, they are generally less effective than grass buffer strips, likely because grass buffer strips promote a high soil cover and higher infiltration. Hedgerows can intercept runoff water, disperse and retain runoff, enhance soil infiltration, and thereby reduce the downslope transport of sediment and water (Xiong et al., 2018).

*Factors of influence*

A wide range of potential influencing factors was considered, but the evidence remains limited. No significant effects could be detected due to the small number of available studies and the lack of information on factors such as slope, soil characteristics, soil cover, and species composition (Van Vooren et al., 2017; Xiong et al., 2018).

*Design and management*

Not described.

**Nutrient retention**

*Context*

Results are based on two reviews. Van Vooren et al. (2017) focused on hedgerows in temperate regions and arable crops (maize, cereals, legumes, and mustard). Zheng et al. (2020) conducted a global review, with most studies from the US and China. In addition, the agricultural systems considered in this review are not specified clearly.

*Effectivity and mechanism*

Hedgerows have a strong positive effect on nutrient retention. Van Vooren et al. (2017) reported that hedgerows intercept approximately 69% of nitrogen and 67% of phosphorus from surface runoff, and 34% of nitrogen and 32% of phosphorus from subsurface flow. Positive influence on nutrient retention is also shown in soil nutrient contents. Significant increases have been observed in total nitrogen, total phosphorus, available nitrogen, available phosphorus, and available potassium, whereas total soil potassium shows no response (Zheng et al., 2020).

Hedgerows contribute to nutrient retention by intercepting runoff carrying dissolved nutrients or nutrients bound to soil particles, dispersing and retaining this flow, which enhances infiltration and reduces further downslope transport (Van Vooren et al., 2017). In addition, hedgerows improve nutrient cycling by taking up nutrients from deeper soil layers and returning them to the surface through leaf litter, thereby enhancing nutrient use efficiency and nutrient availability in upper soil layers. Nitrogen removal from runoff occurs primarily through denitrification rather than plant uptake, while phosphorus losses are mainly reduced through increased infiltration facilitated by hedgerows (Zheng et al., 2020).

*Factors of influence*

Wider hedgerows increase nutrient interception (in the range 1-10 m width), with nitrogen interception being 42% in a 2 m-wide hedgerow and 72% in a 5 m-wide hedgerow (Van Vooren et al., 2017). No other factors were identified because of limited data, but general drivers such as input concentrations, slope, hydraulic parameters and soil characteristics, as well as hedgerow-specific factors such as soil cover and species composition, are likely to affect infiltration and nutrient uptake. It is suggested that removing decaying plant material may help to reduce phosphorus (P) losses (Van Vooren et al., 2017; Zheng et al., 2020).

*Design and management*

Not described.

**Water regulation**

*Context*

The results are based on a single primary study (Albaladejo-García et al., 2025), which modelled scenarios of hedgerow implementation in orange and broccoli production systems in Spain.

*Effectivity and mechanism*

The model approach estimates that planting of hedgerows would have a positive effect on water infiltration and storage with an increased water storage of about 28% in mm of water per area. The land planted with hedgerows would be better at retaining water during heavy rainfall and to slow down runoff, which is relevant for climate change resilience (Albaladejo-García et al., 2025).

*Factors of influence*

Not described.

*Design and management*

Not described.

**Cultural services**

*Context*

Results are based on three primary papers. One focuses on Mediterranean irrigated agriculture in Spain (Albaladejo-García et al., 2025), the second on terraced viticultural landscapes in Italy (Assandri et al., 2018), and the third examines mosaic lowland farmland in the Netherlands and Germany (van Zanten et al., 2016).

*Effectivity and mechanism*

Hedgerows have a positive effect on cultural services across all three studies. In Mediterranean irrigated agriculture, establishing perimeter hedgerows increased the Recreation Index, indicating improved opportunities for recreational enjoyment in irrigated farm landscapes (Albaladejo-García et al., 2025). In the Italian terraced vineyards, hedgerows formed part of the landscape's traditional traits together with dry-stone walls and tree rows that were associated with higher aesthetic value and perceived as elements of local cultural heritage (Assandri et al., 2018). In the Dutch and German agricultural landscapes, visitors evaluated green linear elements positively, but their relative importance differed, being a dominant driver of preference in the Dutch landscape but less important in the German case (van Zanten et al., 2016).

*Factors of influence*

Differences in the visual appearance of LFs across regions likely influence preferences and partly explain the variation observed between Dutch and German agricultural landscapes (van Zanten et al., 2016).

*Design and management*

In Mediterranean irrigated agriculture, establishing perimeter hedgerows is recommended as a best management practice because it enhances landscape greenness, vegetation diversity, and structure, thereby increasing recreational cultural services without major negative effects on farm viability (Albaladejo-García et al., 2025). In the Italian terraced vineyards, conserving hedgerows alongside other traditional features helps maintain the landscape's aesthetic and cultural-heritage values (Assandri et al., 2018). In the Dutch and German lowlands, ensuring the maintenance or restoration of hedgerows supports scenic quality in mosaic farmland and can be integrated into landscape-scale planning aimed at improving visitor experience and recreational appeal (van Zanten et al., 2016).

**Knowledge gaps**

- There is a lack of studies evaluating multiple ES simultaneously, rather than optimizing hedgerows for a single service. Identifying potential trade-offs between services and determining designs that enable the combined delivery of diverse ES is highlighted (Van Vooren et al., 2017).
- In many cases, it remains unclear which explanatory factors influence hedgerow effectiveness, due to the limited number of studies and incomplete reporting of key variables in original studies (Xiong et al., 2018; Van Vooren et al., 2017).
- Data on belowground biomass and carbon sequestration in hedgerows remain scarce, particularly below 30 cm soil depth. Future research should include deeper soil layers to better quantify the overall carbon sequestration capacity of hedgerows (Drexler et al., 2021; Van Vooren et al., 2017).

### 2.3.2. Tree rows

**Animal production**

*Context*

Results are based on a global review (England et al., 2020) synthesising ES of woody systems in grazed dairy and mixed dairy/beef systems. The landscape feature categorised here as tree rows corresponds to “shelterbelts” in the review, defined as “linear plantings of trees along pasture edges to provide shade and shelter” (England et al., 2020).

*Effectivity and mechanism*

Tree rows had a positive effect on animal production. The mechanism was shade provision, which reduces heat stress in grazing cattle and improves stock productivity. Shade leads to more favourable feeding behaviour and higher dry-matter intake, resulting in better weight gain, improved performance, and healthier animals. These outcomes are supported by high confidence evidence, with several studies consistently demonstrating strong links between shade from tree rows, reduced heat stress, and improved stock productivity (England et al., 2020).

*Factors of influence*

Not described.

*Principles for implementation*

Not described.

**Grassland production**

Context

Results are based on the same global review by England et al. (2020). The landscape feature categorised here as tree rows corresponds to “shelterbelts” in the review, defined as “linear plantings of trees along pasture edges to provide shade and shelter” (England et al., 2020).

*Effectivity and mechanism*

Tree rows have mixed effects on grassland production. Evidence for improved pasture quality is of medium confidence: shade from tree rows can increase the crude protein content of forage, likely by enhancing organic matter degradation and nitrogen recycling under shaded conditions. Effects on pasture yield are inconsistent and of low confidence. Yield responses vary across zones: production commonly decreases in the competition zone near the tree row, where water and nutrients are limited, but can increase in the sheltered zone, where reduced wind exposure, lower evapotranspiration, and a more favourable microclimate support pasture growth (England et al., 2020).

*Factors of influence*

Not described.

*Principles for implementation*

Not described.

**Carbon sequestration**

*Context*

Results are based on England et al., (2020). The landscape feature categorised here as tree rows corresponds to “shelterbelts” in the review, defined as “linear plantings of trees along pasture edges to provide shade and shelter” (England et al., 2020).

*Effectivity and mechanism*

The review reports high confidence that tree rows sequester carbon in the tree biomass and increase organic soil inputs, which together lead to greater carbon sequestration in both biomass and soil. Conversely, the review reports low confidence for the possibility of reduced carbon sequestration, which could occur due to lower biomass production in the understorey, including grassland vegetation. There was no statistical analysis included in the review to support this. Previous studies have shown that tree rows -characterized by a higher proportion of edge trees per planted area- tend to exhibit greater biomass growth rates than block plantings. This difference is attributed to reduced intra-specific competition for light, water, and nutrients (England et al., 2020).

*Factors of influence*

Factors such as climate, soil type, tree species and age, and previous land use may influence carbon sequestration, however this is not statistically confirmed in this review (England et al., 2020).

*Design and management*

Not described.

**Cultural services**

*Context*

Results are based on two primary papers. One study focuses on terraced viticultural landscapes in Italy (Assandri et al., 2018), while the other examines mosaic lowland farmland in the Netherlands and Germany (van Zanten et al., 2016).

*Effectivity and mechanism*

Tree rows had a positive effect on cultural services in both contexts. In Italian terraced vineyards, they formed part of the landscape’s traditional traits -alongside dry-stone walls and hedges- that were associated with higher aesthetic value and perceived as part of the cultural heritage (Assandri et al., 2018). In the Dutch and German agricultural landscapes, visitors evaluated green linear elements positively in a choice experiment, but their relative importance differed, being a dominant driver of preference in the Dutch landscape but less important in the German case (van Zanten et al., 2016).

*Factors of influence*

Differences in the visual appearance of LFs across regions likely influence preferences and partly explain the variation observed between Dutch and German agricultural landscapes (van Zanten et al., 2016).

*Design and management*

Cultural services can be strengthened by maintaining tree rows as part of broader landscape management. In terraced vineyards, conserving tree rows alongside other traditional features supports both aesthetic and heritage values (Assandri et al., 2018). In lowland agricultural landscapes, the presence of tree lines and other green linear elements helps sustain scenic attractiveness and recreational appeal, making them valuable components in landscape-scale planning (van Zanten et al., 2016).

**Knowledge gaps**

- Research should further examine the role of tree row characteristics (e.g., age and species composition) in ecosystem service delivery (England et al., 2020), and
- There is a need for more comparative studies that assess how LFs contribute to aesthetic value across different landscape contexts (van Zanten et al., 2016).

### 2.3.3. Individual trees

**Animal production**

*Context*

Results are based on a global review (England et al., 2020) synthesising ES of woody systems in grazed dairy and mixed dairy/beef systems. The landscape feature categorised here as individual trees corresponds to "pasture trees" in the review, defined as "scattered trees" in existing pastures at varying densities; originating from remnant trees, natural regeneration, or planting" (England et al., 2020).

*Effectivity and mechanism*

Individual trees had a positive effect on animal production. Evidence of medium confidence links the presence of pasture trees to increased milk production. The primary mechanism is shade provision, which reduces heat stress and enables better feeding behaviour and higher dry-matter intake. Together, these effects contribute to improved milk yield (England et al., 2020).

*Factors of influence*

Not described.

*Design and management*

Not described.

**Grassland production**

*Context*

The evidence is based on a global meta-analysis by Rivest et al. (2013) examining pasture yield under individual trees in grazing systems. The landscape feature categorised here as individual trees corresponds to "scattered trees" in the review, defined as mature trees occurring at low densities within grazing landscapes (Rivest et al., 2013).

*Effectivity and mechanism*

Individual trees had a mixed effect on grassland production. Across all functional groups combined, the net effect on pasture yield was close to zero, but species-specific responses varied markedly. Deciduous and evergreen oak trees exhibited near-zero, non-significant effects; eucalyptus trees had a strongly negative and significant effect on pasture yield. In contrast, nitrogen-fixing species such as *Acacia* spp. showed a positive, near-significant ($p = 0.076$) effect on pasture production.

These contrasting outcomes arise from the balance of facilitation and competition: positive effects can occur through reduced evapotranspiration, improved soil nutrient availability, and hydraulic lift, while negative effects result from competition for water, nutrients, and light (Rivest et al., 2013).

*Factors of influence*

Annual precipitation significantly shaped tree-pasture interactions for two functional groups. For *Eucalyptus* spp., drier conditions intensified negative impacts on pasture yield ($r^2 = 0.35$, $P = 0.0005$), while nitrogen-fixing trees became increasingly beneficial as conditions became drier ($r^2 = 0.61$, $P = 0.0026$) (Rivest et al., 2013).

*Design and management*

Not described.

**Carbon sequestration**

*Context*

Results are based on one global review (England et al., 2020) that focused on grazed dairy or mixed dairy/beef systems. The landscape feature categorised here as individual trees corresponds to "pasture trees" in the review, defined as "scattered trees in existing pastures at varying densities; may be remnant trees, natural regeneration or planted" (England et al., 2020).

*Effectivity and mechanism*

The review reports medium confidence that the growth of individual trees in pastures enhances carbon sequestration via assimilation in tree biomass. There was no statistical analysis done in the review to support this. Previous studies have shown that line planting or individual planting of trees results in greater biomass production compared to the same number of trees planted in blocks (England et al., 2020).

*Factors of influence*

Factors such as the age of the trees, climate, soil type, tree species, and previous land use may influence carbon sequestration outcomes, however this is not statistically confirmed in this review (England et al., 2020).

*Design and management*

Not described.

**Knowledge gaps**

See Tree rows.

### 2.3.4. Groups of trees/shrubs

**Soil erosion control**

*Context*

Results are based on one global review (England et al., 2020) that focused on grazed dairy or mixed dairy/beef systems. The landscape feature categorised here as groups of trees/shrubs corresponds to "remnant native vegetation" in the review, defined as "patches of natural woody vegetation existing as fragments within agricultural landscapes, not as large continuous areas." (England et al., 2020).

*Effectivity and mechanism*

The review concludes with medium confidence that remnant native vegetation composed of trees and shrubs contributes to reduced soil erosion. However, there was no statistical analysis conducted in the review to support this. The woody vegetation is expected to enhance local soil stability through rooting and increased soil organic matter, thereby reducing slope erosion. Additionally, trees slow runoff, promoting greater water infiltration, further contributing to erosion control (England et al., 2020).

*Factors of influence*

The age of the plantings was discussed but not analysed. Young plantings, with low canopy cover and underdeveloped root systems, may be ineffective at stabilizing soil and reducing erosion in a pasture setting (England et al., 2020).

*Design and management*

Not described.

**Cultural services**

*Context*

The results are based on one primary paper in traditional olive-grove systems in southern Spain (Rodríguez-Entrena et al., 2017). Woodland islets (small groups of trees/shrubs within olive orchards) were, amongst others, tested for their contribution to aesthetic value and recreational demand.

*Effectivity and mechanism*

Woodland islets showed no effect on cultural services, assessed through respondents' willingness to pay for their presence, which was not significant. This lack of effect is attributed to low visual salience: woodland islets do not stand out against olive trees and introduce only minor chromatic contrast in the viewed scenes (Rodríguez-Entrena et al., 2017).

*Factors of influence*

Visual prominence appears to be the key factor: respondents consistently prioritised LFs with strong visual salience, such as stonewalls. Although audience segments differed in price sensitivity and general aesthetic preferences, none expressed significant willingness to pay for woodland islets alone (Rodríguez-Entrena et al., 2017).

*Design and management*

Not described.

**Knowledge gaps**

See Tree rows.

### 2.3.5. Herbaceous strips

**Crop yield**

*Context*

The results are based on three global reviews (Albrecht et al., 2020; Lowe et al., 2021; Zamorano et al., 2020) with most studies conducted in Europe and North America. The reviews focused on arable and permanent crop systems growing a range of crops, such as cereals, oilseeds, blueberries, and strawberries. All three reviews examined field-margin flower strips, categorized here as herbaceous strips.

*Effectivity and mechanism*

All three reviews reached the same conclusion: field-margin flower strips did not significantly affect crop yield in adjacent fields, resulting in an overall neutral effect (Albrecht et al., 2020; Lowe et al., 2021; Zamorano et al., 2020).

*Factors of influence*

None of the examined variables modified this overall finding. Across the reviews, crop yield was not significantly influenced by distance from the field edge, flowering-plant species richness, strip size, time since establishment, or landscape simplification (Albrecht et al., 2020; Lowe et al., 2021; Zamorano et al., 2020).

*Design and management*

Not described.

**Biomass production**

*Context*

Results are based on two narrative reviews focused on grassy biomass buffers. One focuses on temperate European agricultural systems (Christen & Dalgaard 2013), while the other examines grasslands and croplands with an unclear geographical scope (Ferrarini et al., 2017).

*Effectivity and mechanism*

Both reviews report a high potential of grassy buffers for producing biomass for fuel, feed, or fibre. Annual dry matter yields range from 2 to 12 tons per hectare, depending on species (Christen & Dalgaard, 2013; Ferrarini et al., 2017). Even though the buffers did not receive direct fertilization, nutrient inputs from surrounding agricultural areas via runoff still enable them to achieve acceptable yields (Christen & Dalgaard et al., 2013).

*Factors of influence*

Influential factors include previous land use and nutrient inputs from surrounding land (Christen & Dalgaard, 2013; Ferrarini et al., 2017).

*Design and management*

Both reviews identify mixtures based on common ryegrass (*Lolium perenne* L.), reed canary grass (*Phalaris arundinacea* L.), or grass-clover mixtures as suitable species choice for biomass production in a buffer. Reed canary grass is particularly advantageous because it tolerates wet soils, forms dense root mats and stiff stems that help reduce runoff, and can be grazed early in the season. To protect water quality, an unharvested strip of vegetation should be maintained directly adjacent to the watercourse (Christen & Dalgaard, 2013).

**Pest and disease control**

*Context*

The results are based on three reviews (Albrecht et al., 2020; Crowther et al., 2023; Van Vooren et al., 2017) focusing on arable and permanent crop systems in temperate regions and globally (Albrecht et al., 2020; Crowther et al., 2023). Two of the reviews examined field margin flower strips (Albrecht et al., 2020; Crowther et al., 2023), while the third examined perennial grass strips established along field borders (Van Vooren et al., 2017).

Additional findings are included from Jachowicz et al. (2025), a review that did not evaluate pest control services directly, but examined the status of natural enemy and pest populations in flower strips, indicating the potential for service provision.

*Effectivity and mechanism*

Both floral field margins and grass strips had a positive effect on pest control (Albrecht et al., 2020; Crowther et al., 2023; Van Vooren et al., 2017). Van Vooren et al. (2017) found that both predator density and predator diversity were higher on parcels with grass strips, and that aphid density was significantly reduced compared with parcels without grass strips. Albrecht et al. (2020) found that flower strips enhanced natural pest control services by 16% on average compared with control fields without flower strips. Crowther et al. (2023) further demonstrated that floral field margins significantly reduced pest abundance ($z = 4.53$, $P < 0.001$) and crop damage ($z = 3.63$, $P < 0.001$) compared with grassy margins.

These positive effects arise because herbaceous strips support diverse and abundant natural enemy communities. Floral margins enhance pest control by supplying nectar and pollen to adult parasitoids and by creating stable and structurally complex microhabitats that favour predatory and parasitic arthropods (Crowther et al., 2023). They also supply shelter, nesting and oviposition sites, overwintering habitats, and food resources that help maintain natural enemy populations throughout the season (Albrecht et al., 2020). Grass strips offer similar advantages, including shelter for overwintering and aestivation, alternative food resources during non-predatory life stages, and favourable microhabitats for reproduction. These resources allow predator populations to build up within the strip and subsequently colonise nearby arable fields, where they contribute to pest suppression (Van Vooren et al., 2017).

In addition, one review (Jachowicz et al., 2025) showed a strong increase in natural enemy abundance in flower strips (+48–70%), while pest abundances did not change significantly. According to the “exporter” hypothesis, increased natural enemy abundance at the field margin can support biological pest control, with flower strips functioning as reservoirs from which natural enemies can disperse into the crop following pest outbreaks (Jachowicz et al., 2025). However, these findings only describe the status of natural enemies and pests within the flower strip itself, and the review did not provide evidence of the actual provision of pest control services in adjacent crops. The mechanisms described by Jachowicz et al. (2025) align closely with those identified in the three other reviews: higher plant diversity increases habitat complexity and resource availability -including pollen, nectar, and alternative prey- while perennial flower strips additionally offer overwintering sites that support the persistence of invertebrate populations across years.

*Factors of influence*

Crowther et al. (2023) found that the type of floral field margin influenced pest responses. Naturally regenerated margins reduced pest abundance more effectively than sown margins, while both types increased natural enemy abundance and diversity relative to grass margins. Responses also varied among natural enemy groups: ground-dwelling species benefitted more strongly from floral margins than aerial species, and parasitoid diversity increased more strongly than predator diversity (Crowther et al, 2023).

Van Vooren et al. (2017) suggested that a spatial gradient in predator and pest responses relative to distance from the grass strip is likely, but the dataset was too limited to model this pattern. Given the importance of the spatial distribution of predators and pests for optimising pest-management decisions, the review emphasised the need for more detailed field-scale studies.

Jachowicz et al. (2025) reported that natural enemy responses were strongly influenced by floral diversity within the strip. More diverse flower strips (> 2 sown species) increased natural enemy abundance by approximately 70%, and each additional sown species raised abundance by over 4%. Response strength varied among natural enemy groups, with foliar predators showing the most consistent positive effects (Jachowicz et al., 2025).

*Design and management*

Not described.

**Pollination**

*Context*

The results are based on three global reviews with greater prominence of Europe and North America (Albrecht et al., 2020; Lowe et al., 2021; Zamorano et al., 2020). The reviews focused on arable systems cultivating a wide range of crops, including cereals, oilseeds, berries, fruits, and vegetables. The studied LFs were flower strips and floral field margins, categorised here under herbaceous strips.

Additional findings are included from Donkersley et al. (2023), Pérez-Sánchez et al. (2023), and Scheper et al. (2013), three reviews that did not evaluate pollination services directly but examined the status of pollinating insects in flower strips and grass strips, indicating the potential for service provision.

*Effectivity and mechanism*

Across reviews, there is no consistent evidence that herbaceous strips enhance pollination services at the field scale within crop interiors. Measures such as pollinator abundance, richness, and visitation rates in crop interiors did not differ significantly between fields with herbaceous strips and control fields (Albrecht et al., 2020; Lowe et al., 2021; Zamorano et al., 2020).

However, Albrecht et al. (2020) provide evidence for a mix of positive and neutral effects of flower strips on crop pollination. While the review found no significant overall effect when pollination was averaged across all sites and distances, it identified a clear, spatially restricted positive effect near flower strips. Pollination services were enhanced in crop areas directly adjacent to floral plantings but declined exponentially with increasing distance into the field, while no comparable pattern was observed in control fields. Modelled responses indicated that pollination dropped to less than 50 % of edge values at 10 m and to around 20 % at 20 m from the strip. This steep decline likely reflects the limited foraging range of many pollinators, particularly small solitary bees, and explains why the localised edge effect does not translate into higher pollination when measured across whole fields.

The other two reviews emphasise neutrality. Lowe et al. (2021) and Zamorano et al. (2020) both conclude that although floral margins increase pollinator abundance and richness at field edges, these positive effects do not extend into adjacent crops. Importantly, Lowe et al. (2021) also show that field margin plantings do not reduce pollination services. This finding addresses concerns among some farmers that floral field margins might draw pollinators away from the crop, and thereby lower crop pollination. If such displacements were occurring, reduced crop visitation rates or lower pollinator abundance and richness within crops would be expected; however, no evidence for these negative effects was found (Lowe et al., 2021).

Three biodiversity-metric focused reviews provide additional insight into how flower strips affect pollinator communities. Scheper et al. (2013) and Pérez-Sánchez et al. (2023) reported strong positive effects of flower strips on pollinator abundance and species richness. In addition to flower strips, Scheper et al. (2013) also reported positive effects of grass strips on pollinator abundance and species richness. Donkersley et al. (2023) reported strong increases in pollinator communities in large herbaceous interventions ($\geq 500$ m²), but only modest responses in small plots. These findings describe pollinator community responses rather than direct evidence of pollination service provision in adjacent crops.

Overall, the evidence indicates mixed effects of herbaceous strips on pollination: positive localized, distance-dependent effects occur adjacent to plantings, and neutral effects when pollination is assessed across whole fields.

*Factors of influence*

Two of the three reviews assessing pollination services found that time since establishment may influence pollination responses, although the evidence is limited. Albrecht et al. (2020) report that the effectiveness of flower strips increased with time since establishment, with the strongest gains occurring during the first two to three years after sowing. Pollination services appeared to continue increasing beyond three years, although evidence for older plantings remains limited due to a lack of long-term studies. Lowe et al. (2021) found that pollinator abundance and richness at field edges increased as plantings matured, although this did not translate into consistent increases in pollination within the crop. Zamorano et al. (2020) did not analyse the effects of flower-strip age, but note that pollinators may require several seasons to colonise new habitats.

Albrecht et al. (2020) also found that higher flowering-plant species richness tended to enhance pollination services in adjacent crops. Diverse plantings likely support complementary floral resources for different pollinator taxa and ensure continuity of forage availability throughout the season. In contrast, Lowe et al. (2021) did not detect consistent effects of planting richness on pollination metrics.

Evidence from the reviews focused on biodiversity metrics highlights additional factors shaping pollinator community responses within flower strips. Intervention size strongly influenced outcomes: large herbaceous plots ($\geq 500$ m²) produced substantial gains in pollinator communities, with species richness increasing by up to 12.7-fold and abundance by up to 7.4-fold. Small plots ($< 500$ m²) generated only modest responses, including a non-significant 0.15-fold increase in richness and a 1.4-fold increase in abundance (Donkersley et al., 2023).

Landscape context and ecological contrast also played a major role, with the strongest responses occurring when high-quality flower strips were embedded in low-quality surrounding habitats, while effects were weak or non-significant in high-quality landscapes (Pérez-Sánchez et al., 2023). Similarly, Scheper et al. (2013) found that the positive effects of both flower strips and grass strips were strongest in structurally simple, resource-poor landscapes dominated by arable fields, where herbaceous enhancements create pronounced ecological contrasts. Floral diversity within the strip further influenced outcomes, with higher numbers of sown species increasing pollinator abundance (Scheper et al., 2013).

*Design and management*

The spatial patterns identified by Albrecht et al. (2020) provide important design insights. Because pollination services decline exponentially with distance from floral plantings, a dense network of relatively small, well-distributed herbaceous strips is likely to be more effective than a few large plantings. In addition, older, perennial strips with high flowering plant diversity are more likely to deliver measurable pollination benefits in adjacent crops (Albrecht et al., 2020).

**Carbon sequestration**

*Context*

Results are based on two reviews: a global narrative review on grassy bioenergy buffers by Ferrarini et al. (2017) focused on croplands and grasslands, and a systematic meta-analysis on grass strips by Van Vooren et al. (2017) focused on temperate regions and arable crops (maize, cereals, legumes, and mustard).

*Effectivity and mechanism*

Grass strips have a positive effect on carbon sequestration. Van Vooren et al. (2017) reported that soil carbon stock in the upper 30 cm was 37% higher in the grass strip compared to the adjacent parcel with arable land, with carbon sequestration occurring primarily through root biomass in these upper soil layers. Ferrarini et al. (2017) report a potential negative effect when grassland is replaced by grassy bioenergy buffers in the short term, but positive in the long term (>3 years).

*Factors of influence*

Sampling depth was a significant explanatory variable, with an increased soil carbon stocks down to 62 cm soil depth. Grass strip width and age had no significant effect on carbon sequestration (Van Vooren et al., 2017).

*Design and management*

Increasing the width of grass strips results in only modest additional gains in soil carbon stocks but substantially enhances their capacity for nutrient interception and erosion reduction. Consequently, if water quality improvement and/or erosion reduction are high priorities, wider strips are preferred over narrower ones (Van Vooren et al., 2017).

**Soil erosion control**

*Context*

The results are based on six reviews: four global reviews (Dorioz et al., 2006; Van Vooren et al., 2017; Liu et al., 2008; Xiong et al., 2018), one European review (Maetens et al., 2012) and one narrative review where the geographical scope is not stated (Ferrarini et al., 2019). The reviews include primary studies focused on grass strips, grassy buffer strips, grassed waterways or grassy bioenergy buffers integrated into systems with arable croplands, grasslands, orchards, and vineyards.

*Effectivity and mechanism*

Grass buffer strips have a positive effect on soil erosion, reducing both soil loss and runoff compared to cropland without buffer strips (Maetens et al., 2012; Van Vooren et al, 2017; Xiong et al., 2018). Although one review reported a statistically nonsignificant effect on runoff reduction (Maetens et al., 2012) and another did not formally test effects statistically (Dorioz et al., 2006), the overall body of evidence strongly supports the effectiveness of grass buffer strips in reducing erosion. with averages ranging from 50-90% and runoff reductions between 40-100%. The large variation indicates that effectiveness may depend strongly on site-specific factors.

Grass strips achieve reduced soil erosion by increasing soil surface roughness, which slows surface water flow. This promotes greater infiltration into the soil and enhances soil deposition and filtration by the vegetation. Effectiveness is generally enhanced when water flows at a velocity that allows infiltration into the soil, which is influenced by rainfall intensity and upslope land use. Grass buffers perform best under diffuse, sheet-like runoff, rather than when their flow is concentrated in streams or gullies (Dorioz et al., 2016; Xiong et al., 2018).

*Factors of influence*

One review identified buffer slope and buffer width as significant determinants of buffer effectiveness (Liu et al., 2008). Together, they explained a substantial part of the variation in sediment trapping (29% and 23%, respectively). Increasing slope reduces erosion-control efficiency, whereas increasing width improves the efficiency. Vegetation cover is also critical: at least 40% cover is required for significant sediment retention (up to 80%), while 60–70% coverage may represent a saturation threshold (Dorioz et al., 2016). Several additional variables were tested or mentioned, but their effects were either non-significant or could not be statistically assessed due to data limitations. These included slope length, species composition, buffer age, soil characteristics, and hydraulic parameters such as runoff velocity and volume.

*Design and management*

The size and shape of grass buffer strips should be adapted to both the source (e.g., slope, erosion risk) and the sink (e.g., water contamination risk) (Dorioz et al., 2016). Proper placement relative to slope and runoff source area is crucial for maximizing retention efficiency, with an optimum buffer slope estimated at 9% (Liu et al., 2008). Effective buffers are typically 5–10 meters wide, with at least 6 meters recommended on slopes between 1–10%. They should consist of dense, well-rooted vegetation, mowed two to three times per year with cuttings removed to improve flow. Taller grass does not increase effectiveness; shorter grass or stiff-stemmed grass is preferable as it resists lodging and maintains runoff dispersion. Plant species composition appears to be of lesser importance (Christen & Dalgaard, 2013; Dorioz et al., 2006).

**Nutrient retention**

*Context*

The results are based on six reviews: two reviews included studies from temperate zones (Van Vooren et al., 2017; Valkama et al., 2019), while four had a global focus including two non-systematic reviews (Christen & Dalgaard, 2013; Dorioz et al., 2006; Ferrarini et al., 2017; Mayer et al., 2007). In three studies agricultural context, being cropland or grassland, was reported. The feature types were grass (buffer) strips, riparian vegetation, and grassy bioenergy buffers.

*Effectivity and mechanism*

Grass strips have a significantly positive effect on nutrient retention compared to land without buffers. Grass strips were found to intercept up to approximately 80% of nitrogen and 50–78% of phosphorus from surface flow. (Dorioz et al., 2006; Ferrarini et al., 2017; Van Vooren et al., 2017; Valkama et al., 2019). Grass strips slow down and spread flowing water, and thereby help to trap soil particles to which nutrients are bound. Dissolved nutrients infiltrate into the ground, where they bind to soil particles. Nutrients are also taken up by vegetation and stored in biomass, while part of the nitrogen is lost by denitrification (Christen & Dalgaard, 2013; Valkama et al., 2019; Dorioz et al., 2006). Ferrarini et al. (2017) report that buffer strips combining grassy and woody vegetation are more efficient at nutrient retention than grassy ones alone.

*Factors of influence*

The effectiveness of buffer strips in reducing nutrient runoff is influenced by factors such as strip width (Ferrarini et al., 2017; Van Vooren et al., 2017), age, and nitrogen concentration (Valkama et al., 2019). Beyond these specific results, the variables that influence soil erosion are also relevant for nutrient retention. However, their effects were either non-significant or could not be statistically assessed due to data limitations. These include slope length, species composition, soil characteristics, and hydraulic parameters such as runoff velocity and volume (Valkama et al., 2019).

*Design and management*

The design principles for effective nutrient retention are similar to those for soil-erosion control, as both dissolved nutrients and soil or sediment particles benefit from the ability of grass strips to slow and disperse runoff. In contrast to erosion control, however, the removal of cut biomass is particularly important for nutrient management, as harvesting directly exports nutrients from the system rather than merely redistributing them. For effective nutrient removal, fast-growing plant species are therefore preferred.

Pesticide removal, in contrast, is enhanced by the presence of dead organic matter, which increases sorption capacity. Because phosphorus stored in plant biomass is rapidly released during frost, timing is crucial: organic matter should preferably be removed outside of the spraying and frost periods, if possible. Drainage strongly reduces buffer effectiveness, as sufficient residence time of the soil water is required for nutrient removal processes to take place (e.g. plant uptake, denitrification). Mitigation options include rerouting drainage water through buffers and establishing ditches or wetlands.

Recommended buffer widths for efficient nutrient removal vary largely across studies (Christen & Dalgaard, 2013). Most filtering occurs within widths below 10 m; however, wider strips further reduce the risk of nitrogen losses (Valkama et al., 2019).

**Water quality**

*Context*

The results are based on a single primary study (Bullock et al., 2021), which focused on a farm-scale experiment integrating flower strips in a system with pasture and seed crops in England. In this synthesis, the flower strips are categorised under herbaceous strips.

*Effectivity and mechanism*

There was no effect of flower strips on water quality, measured as sediment, nitrogen and phosphorus concentrations, within the duration of the study (Bullock et al., 2021).

*Factors of influence*

Not described.

*Design and management*

Not described.

**Cultural services**

*Context*

The results are based on one primary study conducted in England evaluating how residents and visitors perceived different agri-environmental options on a mixed arable farm. The farm was dominated by arable crops (winter wheat, winter oilseed rape, spring barley, spring beans), with smaller areas of sheep and cattle pasture (Bullock et al., 2021).

*Effectivity and mechanism*

Floral field margins had a positive effect on cultural ES. Both farm visitors and local residents showed a strong preference for fields with flower-rich margins over fields without margins or with grass-only margins. Respondents preferred the appearance of flower margins, expressed a desire to see more of them on farms, and stated they would prefer food to be grown under such conditions (Bullock et al., 2021).

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

*Factors of influence*

Preferences for flower-rich field margins were shaped by several factors. Audience differences played a role: visitors responded positively to all enhanced habitats, while residents were more selective but still showed a clear preference for flower-rich margins. Familiarity with the landscape also influenced responses. Residents tended to view long-established features such as hedgerows and meadows as ordinary components of the farmed landscape and therefore less distinctive, whereas newly created flower margins stood out more and elicited stronger positive reactions. Expectations further shaped perceptions. Both visitors and residents generally did not expect to see flower margins on farms, despite expressing a clear preference for them when presented (Bullock et al., 2021).

*Design and management*

The study suggests that relying on a single agri-environmental measure is unlikely to maximise cultural service benefits. Instead, deploying multiple measures can enhance public appreciation of farmed landscapes. The authors recommend a diversified management strategy, in which farmers apply a suite of agri-environmental options, recognising that no single option will deliver all ES across all contexts (Bullock et al., 2021).

**Knowledge gaps**

- There is a scarcity of long-term studies examining how floral plantings affect the provision of pest control and pollination, as well as yield (Albrecht et al., 2020; Lowe et al., 2021).
- Existing research is also taxonomically and functionally biased, with limited data on pollinator community composition and realized floral richness or flower cover (Lowe et al., 2021).
- Most work focuses on pollinator conservation rather than pollination services or yield and its quality, leaving major gaps in functional outcomes (Zamorano et al., 2020).
- Crop damage, yield, and yield quality are rarely measured directly, limiting robust assessment of pest-control effectiveness (Crowther et al., 2023).
- Better prediction of pest-control outcomes requires better understanding of predator movement, life cycles, and how parcel- and landscape-level conditions shape their activity (Van Vooren et al., 2017).
- Current conclusions from yield simulations require validation through field-level implementation and and additional modelling to test sensitivity to assumptions and landscape configuration. Moreover, management changes are unlikely to be adopted without farmer involvement; future strategies should therefore be co-developed with farmers (Colbach et al., 2018).
- Key knowledge gaps regarding soil erosion and nutrient retention include the long-term effectiveness of grass strips and their resilience to extreme weather events, particularly in relation to phosphorus removal (Dorioz et al., 2006).
- Additional research is needed to identify other, currently insufficiently understood factors controlling overall effectiveness (Maetens et al., 2012; Mayer et al., 2007; Xiong et al., 2018).

- Also see knowledge gaps for productive buffer strips under section 'Riparian vegetation'.

### 2.3.6. Herbaceous parcels

**Pest and disease control**

*Context*

The results are based on a single primary study (Elek et al., 2020) on an agricultural system with winter wheat in Hungary. The study focused on set-aside areas, categorized here under herbaceous parcels.

*Effectivity and mechanism*

Aphid abundance was lower in cereal fields adjacent to set-aside fields compared to control fields without set-aside areas. In contrast, the abundance of the studied natural enemy groups, carabids and spiders, remained unchanged. The reduction in aphids may be explained by the activity of other natural enemy groups, such as parasitoid wasps, the increased availability of resources and habitats in the set-aside fields for both pests and natural enemies. In addition, set-aside fields may act as partial barriers that reduce pest colonisation and establishment in neighboring crop fields (Elek et al., 2020).

*Factors of influence*

The study focused on the near-final ripening phase of winter wheat, when natural enemy activity is at its peak. During this period, intensive movement of natural enemies between fields can lead to an equilibrium in habitat use, resulting in similar abundances in croplands and adjacent set-aside fields. The authors suggest that such spillovers can mask potential differences in habitat utilization, explaining the lack of response observed in carabid and spider abundance (Elek et al., 2020). However, this pattern was not confirmed by longer-term experiments.

*Design and management*

Not described.

**Carbon sequestration**

*Context*

The results are based on a single primary study (Bullock et al., 2021), which focused on a farm-scale experiment integrating fallows into a system with pasture and seed crops in England.

*Effectivity and mechanism*

There was no significant difference between fallows and the grassland or cropland reference fields (Bullock et al., 2021).

*Factors of influence*

Not described.

*Design and management*

Not described.

**Water quality**

*Context*

The results are based on a single primary study (Bullock et al., 2021), which focused on a farm-scale experiment where fallows were integrated in a system with pasture and seed crops in England.

*Effectivity and mechanism*

There was no effect on water quality in the form of sediment, nitrogen and phosphorus concentrations within the timeframe of the study (Bullock et al., 2021).

*Factors of influence*

Not described.

*Design and management*

Not described.

**Knowledge gaps**

Not described.

### 2.3.7. Ditches

**Soil erosion control**

*Context*

Results are based on one narrative review (Dollinger et al., 2015) focused on agricultural ditches in agricultural landscapes worldwide.

*Effectivity and mechanism*

The authors conclude that ditch networks play a key role in preventing soil erosion in agricultural plots by intercepting surface runoff. Situated along field edges, ditches primarily function by slowing down and managing surface runoff, which is a key driver of soil-particle detachment and transport. By capturing and directing runoff from individual fields, ditches limit flow acceleration and redistribution to downslope areas, thereby reducing erosion risks for lower-lying plots (Dollinger et al., 2015).

*Factors of influence*

No specific factors have been identified as directly responsible for efficient reduction of soil erosion. However, general characteristics that influence the ESprovided by ditches include vegetative cover, ditch morphology, slope orientation, and connectivity features such as piped sections and weirs. Additional elements that can affect ditch performance include soil and sediment properties, litter composition, biological activity and biofilms, as well as the overall configuration of the ditch network (Dollinger et al., 2015).

*Design and management*

Soil erosion control by ditches is particularly important in hilly agricultural areas experiencing high rainfall intensity (Dollinger et al., 2015).

**Water quality**

*Context*

The results are based on five globally oriented reviews (Bai et al., 2025, Dollinger et al., 2015, Shen et al., 2021, Shen et al., 2024, Stehle et al., 2024). The agricultural context is mostly unspecified, although in some reviews vineyards, arable land, and fruit production are mentioned. The assessed features include agricultural ditches and 'ecological' ditches, sometimes described as vegetated treatment systems (VTS). Two of the reviews also include ditches which are not (fully) vegetated, while Shen et al. (2024) focused on ditches and wetlands together.

*Effectivity and mechanism*

Ditches can reduce total nitrogen and phosphorus in incoming water ranging between 3-92% (Dollinger et al., 2015), with averages reported between 40-65% (Bai et al., 2025, Shen et al., 2021, Shen et al., 2024). Pesticide reductions in ditches have been reported to vary considerably, ranging from 3–99% (Dollinger et al., 2015) to often above 70% for ditches (and wetlands) in other studies (Stehle et al., 2011). The nutrient removal occurs through plant uptake, sedimentation, adsorption, and volatilization, while microbial processes such as denitrification and nitrification further transform and remove nitrogen compounds. Pesticide retention is mainly driven by sorption, with additional contributions from sedimentation, degradation, and plant uptake.

*Factors of influence*

A wide range of factors influence the removal rates of nutrients and other pollutants in ditches. Vegetated ditches are generally much more effective than those with little or no vegetation (Shen et al, 2021; Stehle et al., 2011). Temperature affects performance, with higher temperatures improving nitrogen removal, while low temperatures reduce overall efficiency (Bai et al., 2025; Shen et al, 2021). Influent concentration also matters as higher concentrations can increase removal rates, and certain nutrient ranges provide optimal efficiency. Additionally, longer hydraulic retention times (HRT) generally improve pollutant removal by increasing time available for physical, chemical, and biological processes to occur (Bai et al., 2025; Shen et al, 2021).

*Design and management*

Soil ditches fully lined with vegetation are generally preferred over concrete ditches. However, in areas prone to heavy rainfall and high erosion risk, partially vegetated ditches may be more appropriate, as dense vegetation can impede water flow and thus reduce retention time (Bai et al., 2025; Shen et al, 2021). Species composition may be less critical, but including cold-tolerant plants and maintaining dense vegetation cover helps ensure year-round purification (Bai et al., 2025; Shen et al, 2021). Regular mowing and removal of plant material, particularly before winter, prevents vegetation from becoming a secondary pollutant source (Shen et al., 2024).

For pesticide removal, ditches with dense vegetation, a thick litter layer ((i.e. accumulated dead plant material), and fine-textured sediment are recommended (Dollinger et al., 2015). Pollutant removal can be further enhanced by extending hydraulic retention time through careful ditch design and the integration of additional interception measures (Bai et al., 2025, Shen et al., 2024). To maximise effectiveness, ditches should be located close to pollution sources to intercept contaminants early, before negative downstream effects occur (Bei et al., 2025; Stehle et al., 2011). Ditches may be a more space-efficient option to improve water quality than buffer strips, especially in intensive agricultural areas (Stehle et al., 2011).

**Knowledge gaps**

The detailed processes underlying pollutant removal in ditches, as well as the factors influencing their performance, remain insufficiently understood. This highlights the need for a more comprehensive assessment of the drivers of ditch effectiveness across different climatic and geographical contexts (Shen et al., 2024; Stehle et al., 2011). In addition, the effect of management practices such as burning and mowing on removal mechanisms and ecosystem service delivery requires further study to enable optimized design and management and a better understanding of trade-offs among services (Dollinger et al., 2015).

### 2.3.8. Small wetlands

**Water quality**

*Context*

Results are based on three reviews (Carstensen et al., 2020; Land et al., 2016; Stehle et al., 2011), mainly focused on North America and Europe. The type of agricultural system is unspecified or broadly described, such as arable crops and fruit production. The reviews focus on (surface) constructed wetlands or restored wetlands. However, it is often unclear to what extent findings are specifically applicable to small wetlands, as studies often include larger wetlands. To date, no meta-analyses have focused exclusively on small wetlands.

*Effectivity and mechanism*

All reviews show significant reductions in water pollutants achieved by small wetlands. Average nitrogen removal efficiencies range from approximately 37 to 41% (Carstensen et al., 2020; Land et al., 2016), while phosphorus removal typically ranges from 33 to 46% (Carstensen et al., 2020; Land et al., 2016). Reported pesticide removal rates for systems including ditches and wetlands generally exceed 70% (Stehle et al., 2011). When both land take and pollutant-removal capacity are considered, wetlands may in some cases represent a more cost-effective option for improving water quality than buffer zones (Stehle et al., 2011).

Nitrogen removal is achieved primarily through denitrification, a microbial process in which dissolved nitrogen is converted to atmospheric nitrogen. Nitrogen removal also takes place through sedimentation and sediment accumulation, plant uptake, and volatilization (Carstensen et al., 2020; Land et al., 2016). Phosphorus removal is driven by similar processes, notably sedimentation and plant uptake, but also by sorption and precipitation, whereby phosphorus binds to soil or sediment surfaces or forms insoluble mineral compounds (Land et al., 2016). Pesticide removal processes are less well understood but are thought to involve dilution, adsorption to plants and sediments, as well as degradation and decomposition processes (Stehle et al., 2011).

*Factors of influence*

Multiple factors determine the water purification capacity of small wetlands, although only a limited number show significant effects across reviews. For nutrient removal, key determinants include hydraulic loading rate (water inflow per unit wetland area), wetland size, and mean annual air temperature, with warmer conditions generally enhancing removal efficiency. Pesticide removal is additionally influenced by the physicochemical properties of the compounds involved. In general, removal efficiency also depends on influent pollutant concentrations, with higher inputs often leading to higher apparent removal rates (Land et al., 2016; Stehle et al., 2011).

*Design and management*

The residence time of polluted water in small wetlands strongly influences purification efficiency, as longer retention allows more complete pollutant removal. Careful design is therefore essential to ensure optimal water inlet and outlet rates and appropriate wetland size. Additionally, small wetlands should be located as close as possible to the pollution source or combined with upstream measures, as ecosystem damage may already occur during transport (Stehle et al., 2011).

Restored wetlands on former farmland are significantly less efficient than other wetlands at removing total phosphorus (Land et al., 2016), most likely due to long-term phosphorus buildup.

**Knowledge gaps**

It remains unclear whether wetlands maintain their effectiveness in water purification as they age, since most studies have focused on newly established wetlands. In addition, the influence of varying water inflow rates on wetland performance requires further investigation (Land et al., 2016).

### 2.3.9. Small ponds

**Water quality**

*Context*

The results are based on one review focused on agricultural ditches and ponds (Shen et al., 2021), drawing on primary studies from North America and Europe. The type of agricultural system is unspecified.

*Effectivity and mechanism*

Ponds and ditches reduced the total nitrogen in water by an average of about 39%. Ponds reduce nitrogen through sedimentation, plant uptake, microbial processes, and volatilization (Shen et al., 2021).

*Factors of influence*

Nitrogen-removal efficiency is influenced by factors such as sediment characteristics, vegetation, and water inflow and temperature. Low temperatures reduce removal efficiency, while increased vegetation density and higher influent nitrogen concentrations generally enhance removal. In contrast, vegetation type and water depth have little impact on the removal rate (Shen et al., 2021).

*Design and management*

Partly concrete or natural ditches and ponds are recommended for effective purification, with partly concrete ditches potentially performing better at rapidly draining water. The choice between natural or selected vegetation does not significantly affect nitrogen-removal efficiency (Shen et al., 2021).

**Water regulation**

*Context*

The results are based on a single primary study (Staccione et al., 2021) where natural pond implementation scenarios were explored in an arable and fruit-growing region of Italy.

*Effectivity and mechanism*

The study shows that ponds improve water balance by increasing water availability for irrigation and mitigate water scarcity during dry season (Staccione et al., 2021).

*Factors of influence*

Not described.

*Design and management*

The study focuses on green ponds, which can deliver additional ES next to water storage. However, the evaporation from green ponds is likely to be higher than from conventional ponds. The vegetation may also increase water infiltration and hence cause more water loss than a regular pond (Staccione et al., 2021).

**Cultural services**

*Context*

The results are based on one primary study from England evaluating how different agri-environmental options on a mixed arable farm were perceived by residents and visitors. The farm was dominated by arable crops (winter wheat, winter oilseed rape, spring barley, spring beans), with smaller areas of sheep and cattle pasture (Bullock et al., 2021).

*Effectivity and mechanism*

Small artificially created ponds had a positive effect on cultural ES. The ponds were described as providing aesthetic appeal, and both visitors and residents included ponds among the agri-environmental features they valued on the farm (Bullock et al., 2021).

*Factors of influence*

Not described.

*Design and management*

The study suggests that cultural service benefits cannot be achieved through a single agri-environmental measure, and a combination of different management options is needed to improve how farmed landscapes are perceived. The authors recommend a diversified management strategy, in which farmers adopt several agri-environmental actions to provide different ecosystem service benefits (Bullock et al., 2021).

**Knowledge gaps**

Not described.

## 2.3.10.Riparian vegetation

**Biomass production**

*Context*

Results are based on two narrative reviews focused on woody biomass buffers. One review focuses on temperate European agricultural systems (Christen & Dalgaard 2013), while the other focuses on grasslands and croplands with an unspecified geographical scope (Ferrarini et al., 2017).

*Effectivity and mechanism*

Both reviews report a high potential of woody buffers for producing biomass for fuel, feed, or fibre. Annual dry matter yields reported are 5-17 tons per hectare depending on species and production system type, such as short-rotation coppice or short-rotation forestry) (Christen & Dalgaard, 2013; Ferrarini et al., 2017). Due to the high water and nutrient availability in a buffer zone in an agricultural area, the growth rate of woody buffers is relatively high (Ferrarini et al., 2017).

Factors of influence

The choice of woody species and the type of short-rotation biomass system strongly influence production, which is further shaped by previous land use and nutrient inputs from surrounding land (Christen & Dalgaard, 2013; Ferrarini et al., 2017).

*Design and management*

Short Rotation Forestry (SRF) systems are harvested every 12–30 years, while Short Rotation Coppice (SRC) systems using willow or poplar hybrids can be harvested much more frequently (1-10 years), which may be more compatible with farm-level implementation. To protect water quality, an unharvested strip of vegetation should be maintained next to the watercourse (Christen & Dalgaard, 2013). Harvesting in late winter is recommended for various ES (Ferrarini et al., 2017).

**Pollination**

*Context*

The results are based on a single primary study (Affek et al., 2021) conducted in Poland. The agricultural system of the study was not defined.

*Effectivity and mechanism*

The studied small riparian hardwood ash-elm forest patches provide moderate pollination services, playing an important role especially in early spring when open areas offer few food resources. Early-blooming plants in the forest undergrowth, including *Glechoma hederacea*, *Ajuga reptans*, *Pulmonaria obscura*, *Viola reichenbachiana*, willows, maples, *Prunus padus*, and *Frangula alnus*, were frequently visited, and the highest numbers of *Apoidea* and bumblebees were recorded in April–May. Overall, the authors conclude that these small forest patches have greater potential to provide pollination services and support a diverse community of wild pollinators than previously reported for riparian and other broadleaved temperate forests.

*Factors of influence*

Not described.

*Design and management*

Not described.

**Carbon sequestration**

*Context*

Results are based on a review focused on temperate European agricultural systems (Christen & Dalgaard, 2013). The studied landscape feature was woody biomass buffers, including short-rotation forestry, coppice systems, and unmanaged woody strips, here categorised under riparian forests.

*Effectivity and mechanism*

Woody biomass buffers contribute to soil carbon sequestration by increasing plant material inputs to the soil, thereby influencing soil microbial activity and soil aggregation, both of which enhance carbon storage. In the short term, a decrease in soil carbon sequestration may occur following establishment. A long-term positive effect is more certain when buffers are established on former grasslands (Christen & Dalgaard, 2013).

*Factors of influence*

The original land use, whether cropland or grassland, influences the amount of carbon sequestered, as well as the age of the buffer, with sequestration increasing over time (Christen & Dalgaard 2013).

*Design and management*

To optimize carbon sequestration, the woody biomass buffer should be harvested in late winter when assimilates have been translocated to the root system and leaf litter can be incorporated into the soil, thereby enhancing soil organic-matter inputs (Christen & Dalgaard, 2013). Also see 'Nutrient retention'.

**Soil erosion control**

*Context*

The results are based on a single review focused on woody riparian vegetation buffers (Liu et al., 2008), with primary studies conducted in croplands of North America and Europe.

*Effectivity and mechanism*

Riparian buffers demonstrated strong potential for sediment removal from surface water, with efficiencies ranging from 45% to 100%. Riparian buffers, particularly grass strips, act as filters by increasing surface roughness, which slows runoff and reduces flow volume, allowing more water to infiltrate the soil. This promotes sediment and organic matter deposition, facilitates pollutant adsorption onto soil and plant surfaces, and enables plant uptake of soluble pollutants. Larger soil particles settle quickly as flow velocity decreases, while finer particles, such as clay, require more time and lower flow energy to be captured (Liu et al., 2008).

*Factors of influence*

Riparian buffer efficiency is primarily determined by slope and width. Increasing slope leads to decreasing efficiency while increasing width improves efficiency (Liu et al., 2008).

*Design and management*

A 10 m wide buffer on a 9% slope was found to maximize the sediment-trapping effectiveness of vegetated buffers, independent of the buffer-to-field area ratio.

Steeper slopes reduce efficiency, as they shorten residence time and disrupt consistent laminar (sheet-like) flow (Liu et al., 2008).

**Nutrient retention**

*Context*

The presented reviews all focus on woody vegetation in the form of woody buffer strips and riparian forests. Samples for measuring nutrient contents to estimate the amount of nutrients retained are taken either in surface runoff, soil water or submerged in wet riparian forests. To avoid repetition 'Nutrient retention' and the service 'Water quality' are therefore both reported here.

Nine reviews were identified in total for the two services (Christen & Dalgaard, England et al., 2020; Ferrarini et al., 2017; Lyu et al., 2021; Mayer et al., 2007; Tsai et al., 2022; Valkama et al., 2019; Wang et al., 2023; Zhang et al., 2010), including five systematic meta-analyses, and one additional relevant paper (Lind et al., 2019). Five reviews had a global focus, while the others focused solely or primarily on North America, Europe, and China. The studies focused on a grasslands or cropland focus, or the type of agriculture was not stated.

*Effectivity and mechanism*

A systematic qualitative review highlighted strong confidence in the ability of woody riparian buffers to reduce runoff of sediment and nutrients and thereby improve water quality (England et al., 2020). This is supported by several meta-analyses showing that woody buffer strips -sometimes in combination with grasses- are generally highly effective. Reported median sediment removal is 86%, with values ranging from 45 to 100% (Zhang et al., 2010). Nutrient-loss reductions typically range from 30 to 99% (Christen & Dalgaard, 2013).

For nitrogen, a mean reduction of 68% (range 2–99%) has been reported (Zhang et al., 2010), with a 60–79% reduction for surface runoff and 59–75% reduction for groundwater. Tree buffer strips alone achieve lower nitrogen reductions in surface runoff (approximately 25%) compared with combined grassy and woody buffers but remain effective for groundwater nitrogen removal (around 60%) (Lyu et al., 2021; Valkama et al., 2019). In contrast, Ferrarini et al. (2017) reported higher average efficiencies for woody buffers than for grassy buffers (approximately 90% versus 70%).

For phosphorus, mean reductions reported are 52–87% (range 20–100%) (Tsai et al., 2022; Zhang et al., 2010). Pesticide removal was also substantial, with a median of 88% (range 4–99%) (Zhang et al., 2010).

Woody buffer strips reduce nutrient and sediment runoff and leaching thereby improve water quality through several processes and mechanisms. They slow runoff by reducing flow velocity, promote infiltration into the soil, and capture sediment and associated nutrients through sedimentation and adsorption. Wider buffers enhance retention by allowing finer particles to settle as flow energy decreases (Christen & Dalgaard, 2013; England et al., 2020; Lind et al., 2019; Zhang et a., 2010).

Nutrient retention processes in buffers include plant uptake, physical retention, microbial immobilization and breakdown, and chemical transformation such as denitrification under anaerobic conditions (Mayer et al., 2007; Valkama et al., 2019; Zhang et a., 2010;). The deep rooting system of the woody vegetation contributes with a large area where denitrification can occur (Ferrarini et al., 2017). Nitrogen retention in surface runoff is primarily driven by plant uptake, whereas retention in groundwater depends mainly on denitrification.

Buffers also intercept pesticides through adsorption of hydrophobic compounds and infiltration of water-soluble forms. Subsurface nutrient removal may exceed surface removal due to greater root area and conditions that enhance denitrification (Zhang et a., 2010).

*Factors of influence*

The wider the strip, the higher the effectiveness of the buffer. Groundwater nutrient retention continues to increase until around 15 m (Lyu et al., 2021). Over time, however, the buffer efficiency can decrease (Valkama et al., 2019), likely due to the accumulation of nutrients. Initial pollutant concentrations and fertilization levels in nearby fields also influence buffer performance, providing insight into which types of land use benefit the most from the implementation of buffers.

Environmental factors play a major role: heavy rainfall increases runoff and can reduce effectiveness (Wang et al., 2023), while colder temperatures generally lower performance, suggesting that recommended buffer widths may differ between warm and cold climates (Lind et al., 2019). Topography and soil characteristics also influence buffer performance, with steeper slopes reducing removal efficiency and soils that promote infiltration enhancing it (Lyu et al., 2021).

For sediment retention, a vegetated buffer with a slope of around 10% appears optimal, as it facilitates runoff and laminar (sheet) flow. Phosphorus retention is primarily governed by buffer width in clay soils, while slope is more influential in sandy soils; overall, clay soils tend to show the highest removal efficiency (Tsai et al., 2022). Finally, management practices, such as biomass removal, are important for reducing nutrient leaching and maintaining buffer efficiency (Lind et al., 2019).

*Design and management*

- Buffers should be implemented in areas with a high risk of nutrient losses, such as cropland, rather than grassland. Buffer width recommendations vary, where less than 10 m provides basic filtering, with 3 m already having meaningful effects, but widths of up to 50 m may be needed for full protection depending on slope, watercourse size, and the size of the pollution source. A slope of 10% may be optimal (Zhang et al., 2010).
- Plant selection should prioritize native species adapted to local conditions and cold temperatures. Species choice should consider growth potential and rooting behaviour to maximize nutrient uptake (Lind et al., 2019). Common buffer species such as poplar (*Populus* spp.) and willow (*Salix* spp.) are effective at taking up a wide range of pollutants and immobilizing them in woody tissues.
- Woody buffer strips can also produce biomass for energy (Christen & Dalgaard, 2013). When used for this purpose, it is recommended to harvest during winter

for increased nutrient recycling (Ferrarini et al., 2017) and to prevent soil compaction as this can reduce the water infiltration (Mayer et al., 2007).

- To reduce the risk of nitrogen leaching as the buffer ages, aboveground biomass can be harvested (Valkama et al., 2019). Excessive litter removal should be avoided because some organic residue is needed for the denitrification process and breakdown of pesticides (Mayer et al., 2007).
- Drainage must be carefully managed, ensuring that water flows through the buffer rather than bypassing it via drains. (Christen & Dalgaard, 2013)

**Water quality**

See 'Nutrient retention' above.

**Water regulation**

*Context*

Results are based on a literature review focusing on riparian forests located within landscapes of grazed shrubland, pastures, and woody crops in Greece (Masiero et al., 2024).

*Effectivity and mechanism*

It is concluded that riparian forests improve the runoff retention by 15%, hence reducing flood risk (Masiero et al., 2024).

*Factors of influence*

Not described.

*Design and management*

Not described.

**Cultural services**

*Context*

Results are based on a review of temperate European agricultural systems (Christen & Dalgaard, 2013). The studied landscape feature was woody biomass buffers, including short-rotation forestry, coppice systems, and unmanaged woody strips, here categorised under riparian vegetation.

*Effectivity and mechanism*

Woody biomass buffers can positively contribute to cultural ES. Christen & Dalgaard (2013) note that such features enhance landscape aesthetics, creating structured, park-like mosaics with neat boundaries that many people perceive as peaceful, well cared for, and attractive. These elements resemble natural structural LFs and contribute to the visual quality of agricultural areas.

Woody buffers also influence the perception of good land stewardship: their presence signals skilful management and environmental responsibility, something the authors highlight as important in farming communities.

Additionally, increases in wildlife diversity along woody buffers can enhance opportunities for hunting and fishing, which can raise the value of sporting rights and support recreational use of farmland (Christen & Dalgaard, 2013).

*Factors of influence*

Not described.

*Design and management*

Woody buffers that resemble natural landscape elements, that are neither wild nor intensively managed, and are maintained subtly through low-intensity practices, tend to be most attractive to people (Christen & Dalgaard, 2013).

**Knowledge gaps**

- Despite extensive research, the factors determining buffer effectiveness remain insufficiently understood, as reflected in the variability of reported outcomes. This suggests that additional drivers, such as hydrological processes that have not yet been thoroughly examined, warrant closer investigation (Mayer et al., 2007; Tsai et al., 2022).
- Further research is also needed to optimize buffer design at the basin, watershed, and landscape scales, where modelling approaches may play an important role (Lyu et al., 2021; Zhang et al., 2010).
- The effects of vegetation harvesting on buffer performance also require further study (Lyu et al., 2021).
- Specific knowledge gaps remain for productive woody buffers, including their economic viability and life cycle assessment (Dalgaard and Christen, 2013; Ferrarini et al., 2017).
- In addition, more insight is needed into mechanisms during their establishment phase, impacts on soil health and the soil's capacity to deliver ES, and the role of woody buffers in enhancing landscape heterogeneity and supporting multiple ES (Ferrarini et al., 2017).

### 2.3.11. Terraces

**Crop yield**

*Context*

Results are based on one global review (Wei et al., 2016) focused on terrace cropping in general.

*Effectivity and mechanism*

Terracing shows a mixed effect on crop yield. On average, terraces increase yields (mean effect size = 1.94), but the variability across studies is substantial (coefficient of variation = 59%). Positive yield responses were associated with improvements in soil moisture retention, water–fertilizer interactions, and nutrient accumulation, all of which increase production potential. However, negative or neutral outcomes also occur. Newly constructed or poorly designed terraces can temporarily reduce yields because terrace construction often involve removal of fertile topsoil, soil compaction, or exposure of poor-quality subsoil, which lowers productivity until soil structure recovers. Terraces built on unsuitable materials may experience reduced soil quality and poor crop performance (Wei et al., 2016).

*Factors of influence*

Although not quantitatively analysed, several factors were discussed as likely drivers of yield variability of terracing. Key factors include terrace size, construction material, age, vegetation cover, climatic conditions, and management practices (Wei et al., 2016).

*Design and management*

To maximize the productivity of terraces, their construction should be integrated with complementary water recycling techniques and soil conservation measures, such as maintaining vegetation cover on terrace beds and stabilizing the vertical section using measures such as vegetation cover or stones (Wei et al., 2016).

**Soil erosion control**

*Context*

The results are based on four reviews of regular terraces and contour bunds (which can be considered a type of terrace): two global studies that include European research (Wei et al., 2016; Xiong et al., 2019), one study from China (Chen et al., 2017), and one entirely Europe-based study (Maetens et al., 2012). These studies cover a variety of agricultural land-uses, including cropland, grassland, and woody crops.

*Effectivity and mechanism*

Both contour bunds and terraces are highly effective at reducing runoff and soil erosion, with average reduction rates reported between 39% and 90%, depending on the study, terrace type, and specific metric (Wei et al., 2016; Xiong et al., 2019; Chen et al., 2017). The European study reported a runoff water reduction of 31% for contour bunds and 55% for terraces, and a soil loss reduction of 50% for contour bunds and 55% for terraces (Maetens et al., 2012). Reduced runoff also enhances soil-water availability for crops, promoting vegetation cover that further stabilises soil and limits erosion. In addition, terrace ridges and banks temporarily store runoff and promote infiltration, further reducing downslope water flow and sediment transport (Ziong et al., 2018).

*Factors of influence*

Many factors may influence the effectiveness of terraces; however, due to data limitations, not all could be analysed or shown to be of significant influence. The factors found to have a significant effect were land-use type, slope, and terrace type. In general, terracing has a stronger erosion control effect in croplands than in grasslands and tree crops, both systems with more bare soil. On slopes of 26°–35°

or 11°–15°, the terraces are reported to have the largest effect according to Chen et al. (2017) and Wei et al. (2016), respectively.

*Design and management*

The optimal design of terraces requires further research and depends on local topography and land use. Key design factors include terrace type, slope, and the ratio between the slope and height of the vertical section (Chen et al., 2017; Xiong et al., 2018; Wei et al., 2016).

Soil erodibility is generally higher in croplands than in permanent crops such as grassland, suggesting that terrace implementation should be prioritized in cropland areas (Wei et al., 2016). To reduce soil erosion, it is also recommended to maintain high vegetation cover on and around the terraces (Xiong et al., 2018).

**Soil nutrient levels**

*Context*

Results are based on one global review, which includes studies from Europe, focused on terrace cropping in general (Wei et al., 2016).

*Effectivity and mechanism*

When excluding extreme values, terracing was associated with an average increase of approximately 23% in soil nutrient retention. Terracing achieves increased nutrient retention by reducing water erosion, since most nutrients are either dissolved in water or bound to soil particles. This effect is particularly strong on bare slopes that are subject to irrigation and fertilization (Wei et al., 2016).

*Factors of influence*

There are no statistically tested factors that influence soil nutrients with terracing. However, terrace design (size, construction material), and land use were mentioned as potential factors affecting nutrient levels (Wei et al., 2016).

*Design and management*

See 'Soil erosion'.

**Soil water retention**

*Context*

The results are based on two reviews: one global and partly European (Wei et al., 2020), and one Chinese study (Chen et al., 2020), focused on different types of terraces and conducted in areas with various land uses.

*Effectivity and mechanism*

Both reviews reported increased soil moisture with terracing, by an average of 13% and 20%, respectively (Chen et al., 2020; Wei et al., 2016). Terracing conserves water by reshaping slope microtopography into small micro-watersheds that redirect and concentrate water flows, thereby enhancing rainwater harvesting at the field-scale. At the same time, terracing increases soil surface roughness and reduces runoff connectivity, which together improve infiltration, soil moisture, and overall water-holding capacity (Wei et al., 2016).

*Factors of influence*

Factors that significantly influence soil moisture in terraced systems include terrace type, climate, slope gradient, and land use. Among terrace types, systems with level ditches provide the greatest water conservation benefits. In general, the water storage benefits of terracing increase on slopes steeper than 15° but decrease with increasing soil depth. Terraced tree crops and grasslands show roughly twice the increase in soil moisture compared to bare land and cropland (Chen et al., 2020).

*Design and management*

See 'Soil erosion'. Additionally, for water storage, terracing is especially beneficial when using level ditches on slopes steeper than 15° (Chen et al., 2020).

**Knowledge gaps**

- There is a need for more knowledge on the influence of additional variables on the effectiveness of terraces in reducing erosion, such as age, size and management (Xiong et al., 2018; Maetens et al., 2012).
- Furthermore, more knowledge is needed regarding design, construction and maintenance alternatives of terraces adapted to different contexts (Wei et al., 2016).

### 2.3.12. Stonewalls

**Cultural services**

*Context*

Results are based on two primary papers, with focus on olive-groves in Spain (Rodríguez-Entrena et al., 2017) and terraced vineyards in Italy (Assandri et al., 2018).

*Effectivity and mechanism*

Stonewalls had a positive effect on cultural services in both contexts. In the olive-grove study, stonewalls were positively valued by respondents, who expressed a clear willingness to pay for their presence. Landscape scenarios combining green cover with stonewalls generated the highest increases in recreational demand. Stonewalls enhanced perceived landscape attractiveness along visitor routes, which increased stated willingness to pay and the probability of visiting, resulting in higher parking-fee revenues, greater visitor spending, and measurable multiplier effects at the municipal scale (Rodríguez-Entrena et al., 2017).

In the Italian terraced vineyards, landscapes characterised by dry-stone walls and other traditional features were perceived as having higher aesthetic and cultural-heritage value. Higher aesthetic value coincided with higher density of the flagship bird species, indicating that these traditional stonewall terraces simultaneously support cultural services and biodiversity (Assandri et al., 2018).

*Factors of influence*

In the Spanish olive-grove landscape, visitors preferred stone walls most strongly when combined with green cover, as this pairing created the most visible improvement in scenic quality (Rodríguez-Entrena et al., 2017).

In the Italian terraced vineyards, the cultural value of dry-stone walls was strengthened by a broader traditional landscape identity. Observers born in the region were more likely to perceive these as traditional, which increased their aesthetic and heritage value (Assandri et al., 2018).

*Design and management*

Cultural services can be enhanced by maintaining and restoring stone walls as part of wider landscape improvements. In olive groves, combining stone walls with other clearly visible elements, such as green cover, maximises recreational appeal (Rodríguez-Entrena et al., 2017). In terraced vineyards, conserving dry-stone walls helps preserve the traditional character of the landscape while also enhancing both aesthetic and cultural-heritage values (Assandri et al., 2018).

### 2.3.13. Cultural features

**Soil erosion control**

*Context*

One global qualitative review was found on sunken lanes as a measure of cultural features, in which soil erosion is mentioned. The agricultural context is not specified (Zgłobicki et al., 2021).

*Effectivity and mechanism*

Sunken lanes are created by human activity in combination with soil erosion and can therefore function as a source and a transport route of soil sediments. Sunken lanes are highly susceptible to erosion forms such as piping, and gully erosion (Zgłobicki et al., 2021).

*Factors of influence*

Erosion risk in sunken lanes is influenced by rock type, degree of weathering, topography, human activities, and land use (Zgłobicki et al., 2021).

*Design and management*

To prevent soil erosion and runoff in sunken lanes, surface water should be kept from entering them, for example by using grass and shrub buffer strips with deep roots along the lane banks and by adding ditches to redirect water away from the lane. Furthermore, sunken lanes are best preserved by continued use as lightly to moderately trafficked roads. Their shape can be maintained using openwork concrete slabs, while older lanes benefit from a hardened surface and side drainage (Zgłobicki et al., 2021).

**Water quality**

*Context*

One global qualitative review was found on sunken lanes where water quality is mentioned. The agricultural context is not specified (Zgłobicki et al., 2021).

*Effectivity and mechanism*

Due to the role of sunken lanes in soil erosion, they can also indirectly cause that runoff containing sediment, nutrients, pesticide and other pollutants are transported to water courses and reduce the water quality (Zgłobicki et al., 2021).

*Factors of influence*

See 'Nutrient retention'.

*Design and management*

See "Nutrient retention'.

**Cultural services**

*Context*

Results are based on a global qualitative review on sunken lanes. The agricultural context is not specified (Zgłobicki et al., 2021).

*Effectivity and mechanism*

Sunken lanes contribute positively to cultural ES through their heritage value, scenic beauty, and recreational use. Sunken lanes are considered part of the long-standing heritage of many regions and provide information on past human movement and historical route networks. They also provide opportunities for recreation (e.g. hiking and biking) and education. These values give sunken lanes a high touristic potential and strengthen cultural connections between people and the landscape (Zgłobicki et al., 2021).

*Factors of influence*

The cultural value of sunken lanes depends on their physical condition and accessibility. Bank collapse, excessive erosion, or gully development can reduce traffic and recreational use, leading to a loss of cultural and heritage functions (Zgłobicki et al., 2021).

*Design and management*

The conservation of sunken lanes for cultural services requires maintaining their stability and accessibility. This includes reducing runoff from adjacent land, managing vegetated bank shoulders, and reinforcing banks with suitable vegetation. Continued moderate use as a road and removal of excessive sediment deposits may help preserve their characteristic form (Zgłobicki et al., 2021).

**Knowledge gaps**

- Current understanding of the ecosystem service benefits provided by sunken lanes is limited, and further work is needed to quantify their contributions to biodiversity, landscape connectivity, recreation, and climate-related functions.
- Additional research is also required on management, restoration, and recreation strategies to ensure their long-term persistence in highly modified landscapes (Zgłobicki et al., 2021).

## 2.4. Evidence of economic impacts of LFs

Though Deliverable 2.2. will provide a systematic assessment of benefits and costs of landscape features (LFs) for agricultural production, yields and farm economy, we summarise below the evidence from the literature we reviewed here.

The literature consistently highlights the importance of incorporating economic aspects into decision-making, yet, such evidence remains scarce. Several reviews stress that many nature-based solutions lack robust economic evaluation, even though such analysis is essential for farmer adoption and developing incentives. For example, the review of England et al. (2020) note that no economic metrics were analysed in studies of planting trees on dairy farms. In the context of temperate European agriculture, Christen and Dalgaard (2013) discuss need for thorough cost–benefit analyses at both farm and landscape scales when using networks of buffer strips for biomass production.

A recurring theme is that many interventions of creating or maintaining LFs offer multifunctional benefits, but also impose costs or trade-offs. Above all, the need to understand opportunity costs when taking land out of production for the establishment or reintroduction of LFs is high where land is scarce or crop revenues are high. For example, vegetated buffer strips and riparian buffers can reduce nutrient losses, but their construction and maintenance costs can be substantial, with seed or sod prices ranging from €2.34 to €5.40 per m² (USD to EUR exchange rate of 2008) (review in Liu et al., 2008).

Wei et al. (2016) suggest that terracing provides one of the clearest examples of both high potential benefits and significant economic constraints. While terraces can dramatically increase yields (in their review, 3–4 times for maize and 6–7 times for wheat compared to slopes), they also involve high construction and maintenance costs, which increase with slope gradient. Slumps in agricultural prices further erode returns in such long-term investments.

In contrast, field-edge flower strips require little land and low upfront investment, while providing several ES and offering potential seed-production income (Lowe et al., 2021). Nevertheless, small landowners may still hesitate to adopt such measures due to perceived economic barriers, although the limited size of these patches may support the adoption as well (Donkersley et al., 2023).

Low-cost ecological engineering measures such as ditches, ponds, and ecological ditches (which the review defines as vegetated water channels in contrast to concrete channels lacking vegetation), emerge as economically attractive alternatives to buffer strips, which require comparatively more space. (Bai et al., 2025). Such ditches are repeatedly described as "low-cost and land-saving" elements (Shen et al., 2021; Stehle et al., 2011) and "effective tools for removing agricultural pollutants" (Bai et al., 2025). Constructed wetlands can also be cost-efficient for reducing eutrophication (Land et al., 2016), although their feasibility depends on land availability and local regulations. Cost efficiency is highly context-specific: for example, in Denmark it is required that new wetlands will have capacity to remove more than 300 kg N $ha^{-1}$ $yr^{-1}$ to qualify for implementation according to the national regulation and implementation strategy (Carstensen et al., 2020).

Finally, several primary studies evaluate economic outcomes of LFs on farmland and discuss incentives. In Italy, water-retention ponds would provide economic and social benefits, thereby justifying incentive schemes to support their implementation (Staccione et al., 2021). In Spain, enhancing landscape aesthetics in olive groves through green cover and stone walls is estimated to raise local economics from recreational use by a factor of 1.8 (Rodríguez-Entrena et al., 2017).

In Greece, Masiero et al. (2024) calculated that investment into the restoration of native riparian forests would be paid back in five years while contributing to multiple benefits. Similarly, Matzek et al. (2020) modelled the value of additional carbon restored in degraded riparian forests, estimating a benefit of approximately USD 15,000 per km of the restored forest over a 20-year period, and proposed carbon accounting as a means to diversify farmers' incomes. Although not including economics in their study, Hallberg et al. (2014) refer to evidence of floodplain remediation outperforming constructed wetlands in cost-efficiency in the United States (Roley et al., 2016).

Together, these studies underscore that economic viability of LFs can be shaped through a combination of targeted incentives, ecosystem service payments, and strategic landscape design.

## 2.5. Evidence synthesis for biodiversity and ecosystem services

In this chapter we synthesise evidence from the literature review in Chapters 2.2-2.4. The review results are discussed in relation to key foci of LAFERIA, namely context dependence across different pedo-climatic regions and (intensive) agricultural systems, as well as scale dependence. Additional literature is referenced on a few occasions to further illustrate specific aspects.

### 2.5.1. Integrated evidence on biodiversity and ecosystem services

There is substantial evidence on the functioning of landscape features (LFs) in agricultural landscapes, as reflected in this synthesis of 40 review papers and additional primary studies. (Table 6).

**Table 6.** Summary of the effects of LFs on ES and biodiversity, and count of LF effects per effect direction. Numbers are cases of the LF-ES combination in reviewed literature.

| | Woody | | | | | Grassy | | Wet | | | | Stony | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Hedgerows | Tree rows | Individual trees | Groups of trees/shrubs | Small forests | Herbaceous strips | Herbaceous parcels | Ditches | Small wetlands | Small ponds | Riparian vegetation | Terraces | Stonewalls | Rock outcrops | Cultural features |
| **Provisioning services** | | | | | | | | | | | | | | | |
| Animal production | | 1 | 1 | | | | | | | | | | | | |
| Grassland production | | 1 (±) | 1 (±) | | | | | | | | | | | | |
| Crop yield | 1 (±) | | | | | 3 | | | | | | 1 (±) | | | |
| Biomass production | *1* | | | | | 2 | | | | | 2 | | | | |
| **Regulating services - biological** | | | | | | | | | | | | | | | |
| Pest and disease control | 1 | | | | | 3 | *1 (+)* | | | | | | | | |
| Pollination | 1 (+) | | | | | 3 (+) | | | | | *1* | | | | |
| **Regulating services - climate** | | | | | | | | | | | | | | | |
| Carbon sequestration | 3 | 1 | 1 | | | 2 | *1* | | | | 1 | | | | |
| **Regulating services - soil** | | | | | | | | | | | | | | | |
| Soil erosion control | 3 | | | 1 | | 6 | | 1 | | | 1 | 4 | | | 1 |
| Nutrient retention | 2 | | | | | 6 | | | | | 5 | | | | |
| Soil nutrient levels | | | | | | | | | | | | 1 | | | |
| Soil water retention | | | | | | | | | | | | 2 | | | |
| **Regulating services - water** | | | | | | | | | | | | | | | |
| Water quality | | | | | | *1* | *1* | 5 | 3 | 1 | 5 | | | | 1 |
| Water regulation | *1* | | | | | | | | | *1* | *1* | | | | |
| **Cultural services** | | | | | | | | | | | | | | | |
| Cultural services | *3* | *2* | | *1* | | *1* | | | | *1* | 1 | | *2* | | 1 |
| **Biodiversity** | | | | | | | | | | | | | | | |
| Biodiversity | 2 (+) | 1 | 1 | 1 | | 9 (±) | 2 | 1 | | *1* | 3 (±) | *1* | *3 (±)* | *1* | 1 |
| **Count of LF effects per effect direction** | | | | | | | | | | | | | | | |
| Positive | 8 / *5* | 3 / *2* | 3 | 2 | | 26 / *1* | 2 | 7 | 3 | 1 / *3* | 17 / *2* | 7 / *1* | *2* | *1* | 2 |
| Negative | | | | | | | | | | | | | | | 2 |
| Neutral | 1 | | | *1* | | 5 / *1* | *2* | | | | | | | | |
| Mixed | 4 (±) | 1 (±) | 1 (±) | | | 3 (±) | *1 (+)* | | | | 1 (±) | 1 (±) | *3 (±)* | | |

Evidence type:
**review paper**
*primary paper*

Direction of effect:
Positive
Negative
Neutral
Mixed

Mixed effect type:
(+) mix of positive and neutral effects
(-) mix of negative and neutral effects
(±) mix of positive and negative effects



The number of reviews differs greatly across the LF types, with woody, grassy, and wet features all supported by substantial evidence, while stony features are represented by fewer studies.

There are also large differences in the amount of available evidence across response categories, with biodiversity being the most studied, followed by the ES of soil erosion, water quality, and nutrient retention. Across broad ES categories, regulating services has been subjected to substantially more reviews compared with provisioning services and cultural services.

The results also show many blank cells, which may suggest substantial knowledge gaps. However, many of these do not represent true gaps, but instead arise from a combination of factors, including illogical LF-ES combinations. Examples include biomass production and carbon sequestration from stony and wet features, as well as cases where the ES effects are localized within a LF itself and are therefore of limited relevance for an agricultural context (e.g., ES "soil nutrient levels" and "soil water retention" delivered from wet and grassy features, and the ES "water quality" which is a process studied within wet features and not in grassy or woody features). Moreover, in cases where neither reviews nor primary studies were identified, this absence may partly be attributed to the methodological decision to include the term "ecosystem servic*" in the search strategy for primary papers, which restricted the results to literature that explicitly engages with the concept of ES. An example could be crop and biomass yields that are not necessarily framed as provisioning ES in literature.

During the screening of primary studies aimed at filling gaps in LF–ES combinations not covered by reviews, most retrieved studies addressed combinations that were already the focus of existing reviews and were therefore excluded. This suggests that some LF–ES combinations are more frequently studied while other combinations remain comparatively underrepresented. The variation in number of available studies for LF-ES combinations may be influenced by their relevance to policy (e.g., water quality in the light of Water Framework Directive) and by the type of research required to provide evidence. For instance, studies on water regulation often require watershed- or landscape-scale analyses and modelling approaches that do not readily align with the PICO framework. In addition, historical research traditions may also to some extent explain why some LF-ES combinations were less studied than others.

A clear knowledge gap exists regarding the effects of small forest patches in agricultural landscapes. In our LF classification, riparian forests were included under "riparian vegetation," so relevant literature is partly captured there; however, no studies were identified for small forest patches in non-riparian contexts. Similarly, no literature was found for streams and floodplains. For streams, this may reflect their widespread conversion into ditches, while for floodplains, our definition (limited to those associated with streams <20 m wide) excluded much of the literature on larger river systems.

Additionally, gaps remain in our understanding of ES provided by woody features other than hedgerows. Further knowledge gaps include the influence of wet features on pollination and on pests and disease control, as well as the effects of woody features on soil nutrient levels and water retention in adjacent agricultural fields.

Across the different LF types, most reported effects on ES and biodiversity are positive, with consistent patterns across several ES (Table 6). For woody features, a total of fourteen positive effects is documented in reviews, with the strongest evidence for carbon sequestration, soil erosion control, and biodiversity. The strongest positive evidence base of all LF types pertain to grassy features, with a total of 28 positive findings in review studies, showing particularly strong support for biodiversity, soil erosion control, and nutrient retention. The evidence for positive effects of wet features is almost as strong, with 26 positive findings across review studies showing particularly strong support for water quality regulation and nutrient retention. The evidence base for stony features is less strong, but reviews mainly showing positive effects, most notably for soil erosion control, soil water retention and cultural services.

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

Across the review-level evidence, only six cases report neutral effects of LFs on ES (Table 6): three for herbaceous strips in relation to crop yield, two for pollination, and one for hedgerows in relation to pest and disease control. These neutral effects likely reflect a mismatch between local biodiversity increases within LFs and ecosystem service delivery at the field scale. While reviews consistently report higher abundance of service-providing organisms (e.g. pollinators and natural pest enemies) within LFs, these effects are spatially limited. Pollination and pest control depend on mobile organisms whose activity declines rapidly with distance into crops; for example, only 20% of pollination was recorded at 20 m from a flower strip compared with the field edge (Albrecht et al., 2020). Consequently, local edge effects are diluted when assessed at the whole-field scale, resulting in neutral outcomes for crop yield, pollination, and pest control. Neutral effects may also arise from high between-study variation, often linked to LF quality. For instance, newly established LFs may show time-lagged effects, with positive outcomes only emerging after several years as habitat structure and populations develop (Blaauw and Isaacs, 2014; Morandin et al., 2016; Venturini et al., 2017b; Albrecht et al., 2020).

Importantly, there is no evidence that LFs impose negative effects on crop yield, pollination, or pest control. Concerns features such as herbaceous strips may divert pollinators from crops or increase pest, disease, or weed pressure on the crop are not supported by the synthesized evidence. Overall, these neutral outcomes primarily reflect scale constraints on the delivery of the three closely linked services of crop yield, pollination, and pest control, while the same LFs simultaneously deliver positive effects on biodiversity and a wide range of regulating ES (e.g. carbon sequestration, soil erosion control, nutrient retention) without compromising crop production.

There is little evidence of LFs in agricultural landscapes having entirely negative effects on ES or dis-services (as they are sometimes framed). The only such case is provided by a review on the cultural feature sunken lanes, which negatively affects soil erosion and water quality, in contrast to their positive effects on cultural services. However, many LF-ES and LF-biodiversity combinations had mixed effects, which includes cases of negative impacts (or dis-services) in some contexts.

Eleven cases of mixed effects were identified in the review-level evidence, highlighting the strong context dependence of LF performance (Table 6). Mixed responses arise because different ecological mechanisms dominate under different conditions, most commonly depending on spatial configuration, species-specific functional traits, or land-use context. For biodiversity responses, the direction and magnitude of effects depend strongly on the taxa considered. This context dependence is evident across both ES and biodiversity responses. For example, hedgerows show mixed effects on crop yield because yields decline close to hedgerows due to competition for resources but increase further into fields due to shelter effects and improved microclimatic conditions (Van Vooren et al., 2017). Whether positive or negative effects dominate at the parcel level, therefore, depends largely on parcel configuration.

Beyond spatial configuration, mixed effects can also emerge from species-specific functional traits. For example, individual trees in grasslands have mixed effects on grassland production because tree species differ in functional traits and resource use: nitrogen-fixing species can facilitate pasture growth, particularly under dry conditions, whereas species with high water demand impose strong competitive effects (Rivest et al., 2013). Comparable patterns are also observed for biodiversity, where responses vary across taxonomic groups and land-use contexts. For example, both grass strips and riparian woody buffers show mixed biodiversity outcomes depending on previous land use. Conversion of intensively managed cropland into a LF typically enhances biodiversity by increasing habitat heterogeneity and resource availability, whereas establishment of these features on existing grasslands can result in neutral or negative effects by replacing already valuable open habitat conditions, especially for grassland-adapted species (Scheper et al., 2013; Ferrarini et al., 2017; Christen & Dalgaard, 2013).

Overall, the mixed effects observed across ES and biodiversity emphasize that the outcomes of LFs are inherently conditional. Rather than indicating inconsistency of these interventions, the review-level evidence reflects predictable ecological trade-offs linked to spatial design, species traits, and land-use context. This highlights the importance of context-specific implementation of LFs to maximize benefits and minimize trade-offs. Multifunctionality of LFs and their combinations

The reviewed literature indicates that the same LFs can enhance multiple ES, providing strong evidence of their multifunctionality. In particular, hedgerows, herbaceous strips, and riparian vegetation are associated with simultaneous positive effects on multiple ES. However, some limitations should be considered, as a number of trade-offs related to LF design and management are reported in the literature. A trade-off in hedgerow systems exists between crop yield and regulating ES (e.g., carbon sequestration, nutrient interception, and erosion control). While wider and taller hedgerows enhance these services, they can reduce adjacent crop yields, with only limited additional benefits compared to smaller hedgerows. Therefore, low and narrow hedgerows are generally preferable. (Van Vooren et al., 2017). A second design consideration is the trade-off in ditches between hydrological function and water quality: partially lined ditches enhance water conveyance and runoff control, while dense vegetation impedes flow but improves nutrient retention (Bai et al., 2025; Shen et al., 2021).

Trade-offs also occur when enhancement by design or management of one ES adversely impacts another ES or biodiversity. For example, maintaining ditch water flow and reducing erosion often requires mowing, biomass removal, and dredging, which can negatively affect biodiversity. These impacts can be mitigated by reducing disturbance (e.g., leaving refuge areas) and timing maintenance carefully (Dollinger et al., 2015). Additional trade-offs arise between nutrient retention and pesticide removal: pesticide removal is enhanced by dead organic matter (Christen & Dalgaard, 2013; Mayer et al., 2007), whereas nutrient retention generally requires biomass removal (Van Vooren et al., 2017; Zheng et al., 2020). However, LF management is not the focus of this report; the examples above nevertheless highlight the importance of acknowledging that LF-specific management can differentially affect biodiversity and various ES.

LF classes also vary in complexity, and while some can be broadly classified as woody, grassy, wet, or stony, in practice they often include multiple elements. For example, ditches typically include vegetated banks, wetlands, ponds and hedges often contain grassy vegetation along their margins, and riparian zones and terraces may comprise a mix of vegetated and other features. As a result, the effects of individual components within these systems cannot be disentangled. Consequently, LFs such as streams and floodplains were excluded from the classification, as they are encompassed within the riparian vegetation category. The only studies directly comparing composite LFs with simple LFs focused on grassy-woody buffer strips, categorized under riparian vegetation. Riparian vegetation is highly variable in terms of shape, size, structure, and species composition. Riparian forests, in particular, are regarded as notable “multiservice hotspots,” providing a range of ES (Affek et al., 2021). Four studies compared the efficiency of woody, grassy, and mixed grassy-woody buffer strips in nutrient and pesticide removal. Although these studies differ somewhat in type and focus metrics, the conclusion is that combining woody and grassy components generally results in higher removal efficiency (Christen et al., 2013; Ferrarini et al., 2017; Mayer et al., 2007; Zhang et al., 2020).

Finally, two LF classes of herbaceous vegetation (that is, linear elements or parcels) are distinct from the other LFs insofar as they include also elements created on arable land as a short-term addition to a more permanent landscape structure. These include such LFs as sown flower strips, bird seed areas, rotational fallows, and bioenergy buffers. We could not separate them in this synthesis because most reviews include multiple types of elements analysed from a functional perspective, considering e.g. biomass production, soil and nutrient retention, or pollination and pest control. These LFs are established around or across fields, commonly with vegetation tailored to a desirable ES or overall biodiversity (e.g., grass-based or special seed mixtures). They are typically re-sown or rotated every few years to maintain a desirable vegetation profile. However, flower strips are also established on “unused field margins” (Lowe et al., 2021). While such LFs deliver rapid and targeted enhancement of resources to functionally beneficial organisms (pollinators, pest predators) and specific ES they provide, they should not be seen as a substitute to more permanent LFs, such as permanent field margins. Field margins are the remnants of existing vegetation from cleared lands, result from natural dispersal of plants or established through sowing without regular re-sowing (Zamorano et al., 2020). They provide broad and long-term biodiversity benefits by sustaining stable habitats and structurally complex habitats that support a wide range of species and ecological functions (Lawrence and Hackett, 2014). They are also likely to be better buffers to other LFs and natural habitats.

The two can be regarded complementary in functionality and values and, therefore, used in combination according to a whole-farm or landscape design. Field margins as well as long-term set-asides can serve as a permanent network contributing to the landscape-scale resilience, especially along sensitive areas buffering other LFs and areas. They provide permanent connectivity, perennial nesting sites and sites, and sustain broad biodiversity allowing natural evolution of communities. Sown strips or fallow parcels can be established as an additional, more flexible and service-tailored layer, targeting flower-visiting arthropods, predators, birds, enhancing local soil biota and adding to landscape amenity. These can be best added in priority areas, such as in vicinity of pollinator-dependent crops or crops with high added revenue from improved pest protection (a plant mix can be tailored to support specific prey predators without benefiting pests), or in simplified landscapes with low-diversity crops (to diversify the existing resource base for flower-visiting organisms, birds or game), or as landscape amenity fields in areas of public access. Since in a structurally complex landscape with existing diverse resources, additional benefits for targeted organisms may be relatively low, the focus could be on quality and connectivity (e.g., linking existing hedgerows and margins, improving management of existing margins).



### 2.5.2. Context dependence of delivery of ecosystem services

Though LAFERIA focuses specifically on intensive **agricultural systems**, it was not possible to extract information specifically for these systems from the reviewed literature. Beyond the literature included in our review, several review studies have demonstrated that research on biodiversity and ES is highly more prevalent in Europe (and Northwest European countries of the EU) and the US (Rau et al., 2020; Sutcliffe et al., 2014; Haddaway et al., 2018; Zolyomi et al., 2023). This is where most intensive agricultural systems tend to occur. Therefore, the results synthesised here are likely to be relevant to intensive systems and regions.

Most review studies describe agricultural systems only in general terms, and differences between systems are rarely incorporated into analyses, with a few exceptions where cropland, permanent crops, and grasslands are reported separately. Where such distinctions are made, a pattern emerges across multiple ES. For carbon sequestration, vegetative LFs established on grassland generally show no effect or smaller increases compared to cropland and may even have an initial negative effect when grassland is replaced by woody vegetation (Christen & Dalgaard, 2013; Drexler et al., 2021; Ferrarini et al., 2017). A similar pattern is observed for biodiversity, with LF effectiveness tending to be greater in croplands than in grasslands, and some evidence even suggesting negative effects in grasslands (Scheper et al., 2013; Staggenborg et al., 2022; Ferrarini et al., 2017).

Taken together, our findings suggest that while LFs can be effective in both types of systems, positive effects are generally larger and more consistently reported in croplands than in grasslands. A pattern emerges in which higher ES effectiveness of LFs is achieved in systems where ES delivery is lower and pressures on the natural environment are higher. For example, buffers and terraces tend to have larger impacts on soil erosion control, nutrient retention, and soil moisture retention in croplands than in land with permanent vegetation (Wei et al., 2016; Zhan et al., 2010). Eurostat data shows that arable farmland typically contains fewer LFs than permanent crops or grasslands (European Commission, Eurostat (n.d.), suggesting a greater potential effectiveness for such features in these systems, particularly in relation to challenges such as habitat provision, erosion, and nutrient runoff.

At the same time, although LFs may need to be implemented where their effectiveness and the pressures are the highest, the role of wet features and buffer strips in nutrient and water quality management must be considered carefully. There is a risk for pollution of the LFs themselves, particularly when water pollution loads exceed their purification capacity, negatively affecting their ecological quality. Although wet features are generally more effective at retaining nutrients than buffers, it may be undesirable for them to bear the full pollution load. Instead, the literature recommends intercepting nutrients as close to the source as possible, for example through buffer strips, thereby reducing pressure on downstream features such as wetlands and maintaining their ecological quality (Bei et al., 2025; Stehle et al., 2011).

Beyond agricultural system type, biodiversity responses are also strongly influenced by the degree of habitat contrast created by LF implementation. One review focusing on flower strips shows that their effectiveness increases when they are placed in structurally simple agricultural landscapes (Pérez-Sánchez et al., 2023). This could be expected based on earlier conceptual work, according to which adding high-quality flower strips as the LF interventions into agricultural landscapes dominated by intensively managed fields will maximise the ecological contrast for the benefit of biodiversity (Kleijn et al., 2011). For similar reasons, implementing LFs in agricultural landscapes that have a low share of non-crop habitats, can effectively increase the marginal biodiversity benefit (Kleijn et al., 2011). However, it is important to note that implementing flower strips in landscapes without any LFs may not effectively benefit flower-visiting organisms, because these typically also rely on non-arable land e.g. to establish nest sites (Bishop et al., 2025; Kleijn et al., 2011).

Similarly, although the evidence base analysed here were often drawn from a range of **pedo-climatic regions** (the exception being Conservation Evidence synopsis for the Mediterranean climate (Shackelford et al., 2017)), the results were generally not presented separately for these contexts. It therefore remains difficult to draw clear conclusions from literature about how the effectiveness of LFs varies across different agricultural and environmental settings. Nevertheless, some effects of the influence of climate on LF effectivity emerged from the literature. As an example, colder temperatures generally lower vegetated buffer retention capacity, suggesting that recommended buffer widths may differ between warm and cold climates (Lind et al., 2019) (also see Chapter 3 on climate change).

Furthermore, information on study scale and the **scale dependence** of effects was collected to assess how the scale of implementation influences effectiveness, including whether LFs function differently at larger scales and whether a minimum scale is required to achieve relevant effects. As most studies followed the PICO framework, most primary studies included in the reviews were conducted at the field scale, typically comparing the presence of a landscape feature with a control situation without that feature. While scale was occasionally discussed, such as the need for watershed scale studies on wet features, scale factors were generally not formally analysed with a few exceptions. Albrecht et al. (2020) showed that pollination benefits of floral plantings (e.g. flower strips and hedgerows) are highly localized and decline rapidly with distance into the crop. As planting area had little effect, a dense network of small plantings appears more effective than a few large ones. This suggests that floral plantings can enhance ES, but only when their design and placement are tailored to the appropriate spatial scale. A second possible reason for the lack of studies explicitly focusing on scale effects may be the emphasis on individual LFs in our review. While many studies assess the combined influence of all LFs in agricultural landscapes on landscape metrics, it becomes increasingly difficult above the field scale to disentangle the effects of different LF types.

Based on the limited evidence on the implementation and effectiveness of LFs across contexts, some inferences can be made about which LF types are most suitable for implementations in different contexts. In intensive agricultural landscapes, LF selection for reintegration can be guided by the ES that are most needed and valuable within a given system. Table 7 presents classes of agricultural systems and their key challenges and needs, and corresponding LF types to address them.

**Table 7.** Overview of agricultural system types and their most pressing challenges and needs and the LF that can mitigate those challenges according to the literature review results. Classifications are broadly based on Table 5 of milestone report M1. Categories that are newly introduced in this table are indicated by an asterisk (*) in the first column.

| Intensive agricultural system | Key challenges and needs | LF types with mitigation capacity |
|---|---|---|
| **Arable systems** | | |
| All* | Low nutrient retention polluting surface and ground waters | Woody and grassy buffer strips (no drainage, with biomass removal, in cold climates broader strips with cold-tolerant species), Ditches, Small wetlands (for P removal, not placed on farmland) (Land et al., 2016) |
| Drained arable land* | Low nutrient retention polluting surface and ground waters | Small wetlands or ditches instead of buffer strips (Christen & Dalgaard, 2013). |
| Irrigated arable land | Water availability and storage in production landscapes | Ditches, Ponds |
| Non-irrigated arable land | Soil water retention, Preventing evapotranspiration | Terraces, Woody features (for windbreaking) |
| | Excess water → Efficient drainage | Ditches |
| Sloped arable land* | Runoff water → Soil erosion | Grass strips, Hedgerows, Riparian buffers, Terraces (Wei et al., 2016) |
| | Runoff water → Nutrient losses | Ponds, Ditches |

| | | |
|---|---|---|
| Rice fields | None identified | |
| **Permanent crops** | | |
| Fruit trees and berry plantations | Insect pollination | Hedgerows, Herbaceous strips (pollination increases close to the LFs, thus a dense network of small plantings can be effective; Albrecht et al., 2020) |
| Vineyards | Soil erosion | Grass strips, Terraces |
| Olive groves | Droughts | Ditches, Ponds, Terraces |
| | Soil erosion | Grass strips, Terraces |
| **Grassland systems** | | |
| Permanent grasslands | Heat stress in livestock | Tree rows, Individual trees (shade reduces heat stress and improves stock productivity; England et al., 2020) |
| | Pasture yield | Individual trees (nitrogen fixing species increase pasture production; Rivest et al., 2013) |
| | Pasture quality | Tree rows (shade can increase the crude protein content of forage; England et al., 2020) |
| **Mixed crop systems** | | |
| Complex agricultural landscapes | None identified | |
| Croplands with significant natural vegetation | | |
| **All intensive agricultural systems*** | | |

| | | |
|---|---|---|
| | High land costs, high production demand | Hedgerows (in right configuration) (Van Vooren et al., 2020), Wet features for water quality management (cost-efficient) (Shen et al., 2021; Stehle et al., 2011) |
| All* | Biodiversity loss | Woody LFs: Hedgerows (pollinators, Donkersley et al., 2023; predatory insects, Van Vooren et al., 2017), Tree rows (invertebrates and habitat function, England et al., 2020), Individual trees (invertebrates; England et al., 2020), Groups of trees (vertebrates and invertebrates; England et al., 2020) |
| | | Grassy LFs: Herbaceous strips (pollinators, Donkersley et al., 2023; Lowe et al., 2021; Pérez-Sánchez et al., 2023; Scheper et al., 2013; Zamorano et al., 2020; natural enemies, Jachowicz et al., 2025; insects, Haaland et al., 2011), Herbaceous parcels (birds, Staggenborg et al., 2022; Van Buskirk et al., 2004; insects, spiders and harvestmen, and plants, Van Buskirk et al., 2004) |
| | | Wet LFs: Ditches (plants, invertebrates, insects, amphibians, birds, and mammals, Dollinger et al., 2015), Riparian vegetation (stream macroinvertebrates, England et al., 2020; soil and aquatic biodiversity, Christen & Dalgaard 2013) |
| | | Stony LFs: Cultural features (plants and animals, Zgłobicki et al., 2021) |
| All* | Loss of cultural values | LFs regarded as having cultural and traditional value in the region |
| High human population density* | Low aesthetic and recreation value | A diversity of LFs, especially woody features |
| Low woody cover* | Low carbon sequestration | Woody features (line plantings) (England et al., 2020) |

# 3. The role of landscape features in a changing climate

## 3.1. Background

Climate change is increasing the frequency and intensity of extreme weather events, such as droughts, heatwaves, and heavy precipitation, posing growing challenges to agricultural systems. These impacts affect both agricultural production and the functioning of landscape features (LFs), for example through heat and water stress (Mittler et al.,2025). Strengthening the capacity of farming systems to cope with and adapt to these pressures is therefore increasingly important. In this context, LFs can play a key role in enhancing the resilience of agricultural landscapes. This chapter synthesises evidence on what role LFs have adaptation and resilience of agricultural systems to climate change, as well as the impacts of climate change on the ecosystem service delivery and the structural integrity of LFs.

## 3.2. Methodology

A semi-structured literature search was conducted in Web of Science using the search strings used for the JRC-FP database (Schievano et al., 2026) for LFs, and agriculture. These were combined with terms for climate change and for climate adaptation and resilience (Annex 1). The search results were screened by screening titles and abstracts of the first result pages for relevant studies.

Due to the large number of studies (often several hundreds to thousands) with low relevance, the search original search terms were simplified to subsets of the original search strings. This refinement yielded a smaller set of more relevant publications. Due to the limited number of relevant studies identified, supplementary searches were conducted in Google Scholar and Google using the same search strings. In total, these searches identified seven relevant papers.

In addition, papers included in the review-of-reviews and in the rapid review of primary studies were searched for the keyword *climate*, and relevant information was extracted. In some cases, papers initially excluded from these reviews were revisited to assess their relevance to climate change resilience. From the reviews, seven papers were identified as relevant for this chapter. Additionally, a small number of relevant studies were identified through prior knowledge of the literature of the contributing authors.

## 3.3. Results

To date, few studies have explicitly examined the role of LFs in climate change adaptation and resilience, either in general terms or with respect to specific LFs, such as hedgerows and grass strips. Likewise, LFs rarely feature prominently in the most cited literature on climate-smart agriculture, or climate adaptation and resilience in agriculture (for example; Anderson et al., 2020; Anwar et al., 2013; Chandra et al., 2018; Howden et al., 2007; Scherr et al., 2012; Steenwerth et al., 2014; Taylor, 2018).

Despite the apparent disconnect between research on LFs and studies focusing on climate adaptation and resilience in agriculture, several publications do provide insights that are relevant to climate resilience. By synthesising results from these studies, we present a first overview of the contribution of LFs to the climate resilience of agricultural systems (Table 8). While outside the scope of this report, relevant insights may also be drawn from research focussing on landscape complexity, including studies on grassy and woody vegetation in general, as well as specific systems such as agroforestry.

### 3.3.1. Landscape features in general

Kuntzman & Brom (2025) state that the role of landscape structure in buffering climatic extremes and stabilizing local climate is becoming increasingly important, yet the understanding of how landscape structures shape local climatic conditions remains limited. To address this gap, they analysed two contrasting neighbouring landscapes using one year of satellite data to derive spectral and microclimatic indicators. Their results show that a simplified landscape structure intensifies climatic extremes. They hypothesize that the widespread use of large, homogeneous fields with low edge density under uniform agricultural management, as is common in intensive agricultural areas, may amplify these extremes and their negative consequences. Such fields are generally more susceptible to wind and water erosion (Devátý et al., 2019). The relatively lower amount of edges surrounding larger fields also increases vulnerability to heavy precipitation by increasing runoff volume and speed, thereby reducing water infiltration, causing soil erosion and nutrient losses (Kuntzman & Brom, 2025).

By contrast, a more complex landscape was associated with fewer climatic extremes. This buffering effect is likely due to the presence of a greater diversity of crops, a higher vegetation cover, a more heterogeneous spatial configuration, and a more balanced temporal distribution of agricultural activities (Kuntzman & Brom, 2025).

The study by Kuntzman & Brom (2025) did not explicitly assess the role of individual LFs, but instead focussed on related metrics such as field size and edge density. Nevertheless, LFs such as hedgerows, grass strips, and ditches are -depending on regional context- commonly present between individual fields in more complex landscapes. As such, these widely occurring LFs are likely to contribute at least partially to the climate-buffering effects observed in complex landscapes. However, further research is needed to determine to what extent these features can be optimised to enhance climate resilience while simultaneously delivering multifunctional benefits under global change.

Indirect climate adaptation effects provided by increases in (semi-)natural habitats, and their associated benefits for wildlife, are also highlighted in a study on climate change adaptation of agriculture: “As climate change intensifies, ecological connectivity of wildlife habitats and hydrological resources will become increasingly important as an adaptation strategy” (Scherr et al., 2012). The authors also note the importance of field margins, riparian buffers, and forest edges as part of a climate-resilient strategy against pests, as these features support natural enemies and other beneficial insects.

**Table 8**. An overview of climate stressors and the mechanisms by which landscape features can mitigate their effects in agricultural systems. (-) indicates that the capacity for service delivery will possibly be negatively influenced by stressor.

| **Climate change stressor** | | **Heat stress crops and livestock** | **Drought stress** | **Waterlogging/ flooding from heavy precipitation** | **Runoff water from heavy precipitation** | **New pests and diseases or outbreaks** | **Stronger winds** | **Biodiversity loss overall and species turnover** |
|---|---|---|---|---|---|---|---|---|
| Potentially negatively affected ES by climate change | | Animal production<br>Grassland production<br>Crop yield | Crop yield<br>Grassland production<br>Soil moisture retention<br>Water storage | Soil erosion control<br>Nutrient retention<br>Water storage | Soil erosion control<br>Nutrient retention | Pest control<br>Crop yield | Soil erosion control<br>Crop yield | All |
| Mitigating mechanism | Woody | Shading<br>Cooling by evapotranspiration | Shading<br>Cooling by evapotranspi-ration<br>Deep soil water retention<br>Reduced evapotranspi-ration by wind breaking | Water removal by evapotranspi-ration<br>Soil water infiltration<br>Deep layer interception and storage | (-) Surface interception and infiltration<br>(-) Slowing down runoff, reducing erosivity | Habitat for natural enemies<br>Increased structural and functional diversity | Windbreak<br>Reduced evapotranspi-ration<br>Soil retention | Functional redundancy, habitat connectivity and ecosystem condition |
| | Grassy | Cooling by evapotranspiration | Cooling by evapotranspi-ration | Soil water infiltration and storage | | | | |
| | Stony | | Soil water infiltration and retention | | | Increased structural and functional diversity | | |
| | Wet | Thermal buffering | (-) Water source for irrigation | (-) Flood water storage | | | | |

### 3.3.2. Grassy features

Vegetation, whether in the form of crops or edge elements such as grassy strips and woody borders, plays a central role in several different aspects of climate adaptation. A key regulating service is the cooling effect provided by vegetation through evapotranspiration, which helps moderate local climate conditions. Furthermore, vegetation increases surface roughness and soil porosity, thereby enhancing water infiltration and water retention. These properties are beneficial in case of heavy precipitation events as well as droughts (Kuntzman & Brom, 2025).

However, during periods of heavy precipitation, the flow of runoff water may exceed the infiltration and interception capacity of vegetative strips, leading to less effective filtering and increased losses of nutrient- and sediment-rich water to lower-lying areas and watercourses. Climate change may therefore increase the need for LFs to manage these impacts (Dorioz et al., 2006; Lyu et al., 2021). On the other hand, nutrient uptake by vegetated buffers increases under warmer conditions (Lind et al., 2019) and may therefore improve under a warming climate.

### 3.3.3. Woody features

Woody LFs are of particular interest due to their high capacity for evapotranspiration. A modelling study by Zimmerman et al. (2024) shows that tree cover in a landscape is a key explanatory factor for land surface temperature. The authors note that, because evapotranspiration is an area-based process, the extent of land covered by woody plants is crucial for generating a cooling effect. This highlights the potentially important role of a high cover of woody LFs in buffering local climate conditions. In addition, due to their height and structure, woody elements can reduce wind speed. This can lower evapotranspiration rates in adjacent crop fields, increase soil moisture retention, and buffer soil temperatures, benefits that may be particularly valuable during heatwaves and droughts (Campi et al., 2009; England et al., 2020; Speakman, 2018; Karki and Goodman, 2015). In livestock systems, they can provide shade (and fodder), with potential positive effects on production and animal health (England et al., 2020).

Evapotranspiration is also important for removing excess water from soils during periods of waterlogging or flooding. In general, the evapotranspiration capacity increases with increasing plant biomass. However, there are also concerns that woody LFs may evaporate large amounts of water, which could be undesirable during drought conditions (Jobbágy et al., 2004; Li et al., 2026). A meta-analysis on alley cropping (agroforestry) shows variable effects on crop yield depending on climatic conditions. In particular, the effect of trees on crop yield is less detrimental or even positive under low rainfall and high evapotranspiration conditions, such as those typical of Mediterranean climates (Panozzo et al., 2026). There may be a trade-off between removing excess water during wetter periods and losing excessive amounts through evapotranspiration during dry periods.

Furthermore, woody vegetation in riparian zones along watercourses can provide multiple benefits. Shading helps maintain lower air and water temperatures, while vegetation also protects water quality by reducing erosion and nutrient losses. As such, riparian woody vegetation is expected to play a critical role in climate change adaptation (Matzek et al., 2020).

Agroforestry may have similar effects as woody LFs. The literature mentions potential benefits of agroforestry for climate adaptation and resilience, including the reduction of extreme weather effects through crop and forage shading to improve production and quality, reduction of livestock heat stress, and general microclimate buffering. Indirect benefits of more diversified systems are also emphasized, such as enhanced structural and functional diversity with a stronger ecological connectivity that supports biodiversity and beneficial organisms, including pollinators and other insects, ultimately increasing the overall resilience of the system (Hernández-Morcillo et al., 2018; Schoeneberger et al., 2012).

### 3.3.4. Wet features

Wet LFs play a different role in climate adaptation and resilience than vegetative ones. They contribute to climate adaptation primarily by regulating water availability and storage and may therefore gain importance under climate change scenarios characterized by increased droughts and heavy precipitation. Wet features can capture excess precipitation and help prevent flooding, while also storing water that can later be used for irrigation during drought periods (Staccione et al., 2021).

Zimmerman et al. (2024) discuss water management in the context of climate change, noting that subsurface drainage systems in crop fields have long been installed to handle high rainfall through drainage networks and ditch systems. However, with the increasing frequency of droughts, there may be a growing need to enhance water storage within the landscape so that water can be retained and used during dry periods. Water availability in agricultural landscapes is also critical for supporting the cooling evapotranspiration function of vegetation (Zimmerman et al., 2024). Additionally, during periods of heavy precipitation, infrastructure is required to capture and convey excess water to prevent flooding. Small ponds, wetlands, streams, natural floodplains and ditches may therefore play an increasingly important role in the water management of agricultural areas. Two-levelled ditches that are re-profiled to allow for such flooding and can be especially valuable to protect arable land (Hallberg et al., 2024).

Rising temperatures are expected to improve the nutrient removal performance of vegetated wet features due to increased plant growth and nutrient uptake (Carstensen et al., 2020). However, the sensitivity of water bodies to phosphorus influx is also expected to increase with rising temperatures (Tsai et al., 2022), while increases in pest and disease outbreaks may lead to higher pesticide use (Stehle et al., 2011). Together, these developments could increase the demand for nutrient and pesticide purification services in agricultural landscapes.

At the same time, heavy precipitation events may exceed the hydraulic and water purification capacity of wet LFs, thereby reducing their effectiveness (Carstensen et al., 2020). It has therefore been argued that a well-functioning network of vegetated ditches may be one of the few ways to meet this purification demand when large volumes of water must be captured and treated within short time periods, as such loads would not be effectively mitigated even by wide grassy or woody buffer strips (Stehle et al., 2011).

### 3.3.5. Stony features

Although specific studies on stony features are limited, based on similar mechanisms observed for grassy, woody, and wet features, they can be expected to contribute to climate adaptation by intercepting and slowing runoff during heavy precipitation events, and are therefore likely to play an increasingly important role in the future

## 3.4. Knowledge gaps

- There is a general lack of studies targeting the potential benefits of grassy, woody, wet, and stony LFs on climate adaptation and resilience in agricultural landscapes.
- No literature was found on what the functional resilience is of the LFs themselves to a changing climate. It is likely that the functional resilience of grassy and woody features will differ, based on the species' responses to changing climatic conditions, including changing hydrological regimes. Especially present-day investments into tree planting may be risky due to increased incidents of drought. Wet features may be threatened due to increased irrigation demands and heatwaves, as well as aridisation. Some of the wet features may be inundated in the events of rising sea levels. Stony features are not expected to be influenced by a changed climate if they are sturdy enough to survive intense precipitation events.
- Some trade-offs are not sufficiently understood, such as those related to planted trees evaporating considerable amounts of water leading to water stresses in surrounding production land.
- The resilience of biodiversity associated with LFs in the context of the forecasted climate change is poorly understood. Because of their linear nature and small sizes, as well as exposure to agricultural disturbances, communities may be prone to considerable species turnover.
- There is a limited understanding of how biodiversity-driven functional redundancy contributes to ecosystem resilience, and how factors such as habitat connectivity and ecosystem condition interact to sustain ecosystem functioning under disturbance.

# 4. Natural Capital Accounting: from empirical evidence to monetary values

## 4.1. Background

This report synthesises evidence on how landscape features (LFs) affect biodiversity and ecosystem services (ES). Chapter 2 shows that while empirical findings are extensive, they remain fragmented and highly context dependent, underscoring the complexity of ecological functioning. Scaling this knowledge for applications such as **natural capital accounting** requires broader statistical and modelling approaches to generate better insights into ES and their variability to assess their value. Incorporating LF impacts into accounting frameworks may also require **monetising ecosystem services**.

This chapter outlines the key concepts behind ES monetisation, **natural capital modelling** (NCM), and natural capital accounting (NCA). It introduces their purposes and economic foundations, then explains how modelling approaches can be applied - particularly in the context of LFs.

## 4.2. Methodology

The conceptual basis of this chapter draws on a snowball-type literature review. Initial sources included outputs from EU projects focused on ecosystem accounting and natural capital (e.g. SELINA, MAIA and NCAVES), followed by academic and grey literature on monetising ecosystem services and policy documents such as SEEA-EA.

A semi-structured literature review was then conducted to **identify economic valuations of ES associated with LFs**. This semi-structured literature review was conducted in Scopus, Web of Science, and Google Scholar using the search terms "ecosystem services of landscape features" and "economic valuation" and "landscape features". In addition, backward citation tracking and targeted searches for feature-specific stated-preference (SP) studies were conducted. Only peer-reviewed empirical studies explicitly analysing ES associated with identifiable landscape features or landscape components (e.g., hedgerows, tree rows, orchards, woodland patches, structural elements, or spatial landscape characteristics) were considered. While the focus was on economic (monetary) valuation, selected socio-cultural valuation studies were added when they explicitly assessed discrete LFs and discussed their ES contributions; these are clearly distinguished and used to contextualise economic results.

A second semi-structured search identified examples of **natural capital accounting** applied to LFs. Searches in Web of Science, Google Scholar, and Google used broad combinations of terms related to NCA (e.g. "economic valuation" and specific LFs (e.g., "hedge"). Results were screened for representative cases demonstrating how NCA approaches have been used to assess LF impacts on different ES.

## 4.3. Monetisation of ecosystem services

### 4.3.1. Purposes of monetising ecosystem services

As described by Immerzeel et al. (2025), the purpose of monetising ES is to provide a common metric for policymakers and private actors to assess and compare the benefits provided by different ES with each other and with other goods and services relevant for human well-being. That need for comparability has driven the rapid development of valuation approaches over recent decades (see e.g. Dasgupta, 2021; Kumar et al., 2010).

Economic valuation serves multiple functions - from illustrating the importance of intact ecosystems for human well-being to informing compensation after environmental damage (Costanza et al., 2014). A central role of monetisation, however, is to generate **shadow prices** (i.e. estimated monetary value assigned to an intangible asset) for non-market components of **natural capital**, enabling their integration into **economic accounting** frameworks (Dasgupta, 2021; Arrow et al., 2004).

### 4.3.2. Conceptual basis of monetising ecosystem services

In the context of NCA, the monetisation of ES refers to the integration of nature's contributions into a national accounting framework consistent with the System of National Accounts[1] (SNA). As described by Ruijs et al. (2019), NCA extends traditional accounts – which primarily record market-based economic flows – by incorporating environmental assets (natural capital stocks) and the flows between the environment and the economy. Within this structure, natural capital is recorded as an asset (e.g. forests, water bodies, soils or LFs), while ES represent the measurable flows generated by these assets contributing to economic and societal benefits. Following Costanza (2020), ES can be understood as the relative contribution of natural capital to the production of human well-being, in interaction with built, human and social capital. Monetisation in NCA therefore primarily concerns the valuation of ES flows, which can subsequently inform the monetary value of (changes in) natural capital stocks.

From an economic perspective, ES valuation is commonly organised through the **Total Economic Value** (TEV) framework (Figure 4), which distinguishes use, option, and non-use values. **Use values** relate to direct or indirect uses of ecosystems, here largely corresponding to ES such as recreational benefits or crop-related services linked to LFs. **Non-use values** reflect intergenerational or interspecies altruism (e.g., valuing traditional LFs for preservation). **Option values** capture the importance of maintaining ecosystems for potential future benefits (e.g., LFs that may gain relevance under climate change). Not all value components apply equally to every LF: use values matter for productive features (e.g., fruit tree rows), non-use values for traditional or biodiversity-rich features, and option values where future benefits are uncertain.

[1] SNA is the internationally agreed standard set of recommendations on how to compile measures of economic activity, adopted by the UN statistical commission.

These values are typically estimated for (marginal) changes in ES provision using three main methodological families: **market and cost-based methods**, **revealed preference** (RP), and **stated preference** (SP) methods.  Market-, cost-based, and RP methods primarily capture use values, while SP methods can elicit both use and non-use values. Together, they form the methodological foundation for monetary ES assessment within NCA (Pascual et al., 2012).

Within the SEEA-EA statistical framework, only '**exchange values**' are included, i.e. the amounts that would be paid in actual or hypothetical market transactions. These are typically estimated using direct market valuation and cost-based methods. As a result, monetary ecosystem accounts do not record the full welfare value of ES (NCAVES & MAIA, 2022).

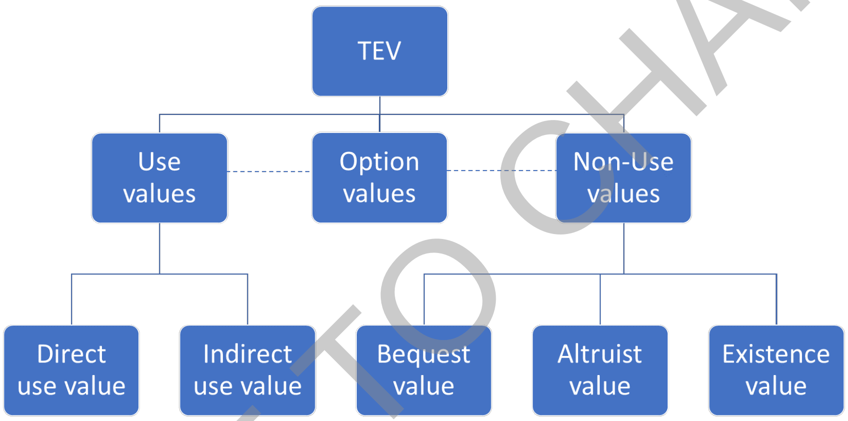



**Figure 4.** Conceptual structure of Total Economic Value (TEV), distinguishing between use, option, and non-use values (existence and bequest). Adapted from Pascual et al. (2012) and Plottu & Plottu (2007).

## 4.4. Literature review: economic valuation of ecosystem services by landscape features

A semi-structured literature review was performed to analyse how ES delivered by LFs have been economically valued in existing literature. The results show some recurring patterns concerning the LF types, valuation methods used, and ES considered, as described below. More detailed findings per study are found in ▯▯.

### 4.4.1. Types of economically valued landscape features and ecosystem services in literature

Woody elements such as trees, hedgerows, orchards, woodlots and forests emerge as the most frequently examined LFs. These elements appear across nearly all valuation approaches, including discrete choice experiments (DCE), contingent valuation and socio-cultural preference assessments. Wet features such as rivers, lakes, small pools and coastal elements are also regularly represented, though less systematically than woody structures. In these studies, agricultural landscape components, including cultivated fields, olive groves, green cover, crop diversity and agricultural management practices are consistently evaluated in relation to landscape quality, but these are not the focus of the LAFERIA project. Many studies additionally value structural landscape characteristics - such as diversity, heterogeneity, openness, patchiness and spatial depth- which often serve as proxies for perceived landscape quality. These structural attributes are especially prominent in stated-preference methods.

The cultural ES, such as aesthetic and recreational values, provided by these LFs are mostly studied. Provisioning and regulating ES are less frequently assessed at the level of individual LFs and are rarely monetised.

Overall, the evidence base is strongly concentrated on visually perceivable and spatially structured landscape components, while smaller-scale structural elements such as LFs and non-cultural ES remain comparatively underrepresented.

### 4.4.2. Patterns in the valuation of ES provided by LFs

The literature review reveals several recurring patterns in the valuation of ES provided by LFs. Depending on whether studies aim to capture (i) the full range of ES or only specific categories, and (ii) microscale *versus* macroscale features, different valuation approaches are used. For example, Rambonilaza & Dachary-Bernard (2007) provide an approach for valuing cultural ES of microscale features such as hedgerows, while Häfner et al. (2018) demonstrate how preferences for linear or point features (e.g., tree rows, individual trees, kettle holes) can be elicited without relying on methods based on 'willingness to pay'. However, comprehensive valuations covering the full ES bundle provided by specific LFs are largely absent. Studies attempting broader valuation (e.g., Włodarczyk-Marciniak et al., 2020) make it difficult to isolate values for individual ES, and no study was found that fully monetises the complete ES portfolio of small-scale landscape features.

The semi-structured review highlights three main conceptual and methodological patterns:

1. **Concentration on woody and mosaic agricultural features**
   Trees, hedgerows, orchards, and agricultural mosaics are the most consistently analysed LFs. Their prominence reflects both ecological relevance and strong visual salience within cultural landscapes. These features are easily represented in stated-preference methods and mapping approaches, and they are closely tied to landscape identity and aesthetic appreciation.
2. **Broad operationalisation of "landscape features"**
   Many studies extend the concept of LFs beyond discrete physical elements to include broader, structural landscape characteristics such as heterogeneity, land-cover diversity, openness, and spatial depth. This broad use enhances analytical flexibility but blurs the boundary between different types of LFs and larger-scale land-use categories. The distinction between discrete structural elements (e.g., stone walls, ponds, tree rows) and macroscopic spatial patterns becomes a key definitional issue, especially in contexts like NCA where classification consistency matters.
3. **Dominance of cultural ES**

Valuation studies overwhelmingly focus on cultural ES, particularly aesthetic and recreational benefits. These services are well-suited to stated-preference approaches, which rely on visual perception and subjective judgments (Häfner et al., 2018; Van Ruymbeke et al., 2026). Provisioning and regulating services are far less frequently monetised at the feature level. Even when LFs underpin regulating functions (e.g., biodiversity support, water regulation), these contributions are rarely valued explicitly. This imbalance partly reflects methodological practice: provisioning and regulating services are often assessed at highly disaggregated units (e.g., biomass per tree, water retention per m² of hedgerow), and therefore such studies may not explicitly refer to "landscape features" and thus fall outside LF-focused search terms.

The described patterns have direct implications for integrating LF valuation into natural capital accounting (NCA). The dominance of cultural ES may appear imbalanced, but it reduces the risk of double counting ES flows. LFs typically generate multiple ES simultaneously; monetising several services independently can lead to overlapping valuations when intermediate and final services are not clearly distinguished. For example, if fruit production is valued and pollination services from adjacent hedgerows are also monetised, the same ecological contribution may be counted twice. Focusing on a single ES category - most often cultural services - implicitly limits this risk. However, this comes at the cost of underrepresenting the full economic contribution of LFs outside the NCA framework. In this sense, the literature reveals a trade-off between accounting robustness and comprehensiveness of value capture, a central tension when integrating LF valuation into NCA frameworks.

### 4.4.3. The role of scale

As also discussed in section 2.5, scale emerges as an implicit but important dimension. Some studies assess micro-scale elements (e.g., hedgerows, stone walls, kettle holes), while others evaluate broader landscape components (e.g., orchards, forests, urban structures) or even abstract structural indicators. This variability reflects methodological differences but also points to the absence of a consistent operational definition of "landscape features" within the ES literature. Clarifying the scale and definitional boundaries of LFs is therefore essential for advancing conceptual coherence and improving comparability across studies.

### 4.4.4. Conclusions from literature review

The reviewed valuation studies suggest that cultural ES provided by LFs can generate positive 'willingness to pay' (WTP), but the reported monetary values vary widely and are not directly comparable. More specifically, the literature does not provide a single coherent price range for LFs, but rather context-specific estimates that are dominated by cultural-services valuations. Estimates range from small monthly premiums for particular landscape attributes to daily or one-off payments for preferred landscape configurations or preservation scenarios. This variation reflects differences in payment format, scale, valuation context and study design. Negative values reported for some attributes likely reflect heterogeneous preferences and trade-offs rather than a general absence of value.

Overall, the findings demonstrate that existing research provides substantial evidence for the cultural value of visually salient LFs, but reveals clear gaps regarding regulating and provisioning services and the comprehensive economic valuation of the full range of ES provided by individual small-scale features.

## 4.5. Modelling of Natural Capital for biophysical and monetary valuation

### 4.5.1. Concept of Natural Capital Modelling

Natural Capital Modelling (NCM) refers to the approach of quantifying ES through modelling exercises. This can be done on national, landscape or regional scales. NCMs aim to estimate the supply, flow and sometimes the demand of ES across landscapes considering current situations, future scenarios, or both (e.g. UN, 2024). NCM is used to explore how changes in land use, management, or climate affect the capacity of landscapes to provide benefits such as pollination, water purification, carbon sequestration, or flood mitigation (e.g. De Knegt et al., 2022; Natural Capital Alliance, 2026).

An important aspect of NCM is its reliance on spatial input data, including landcover maps, soil and climate data, topography, and hydrological networks, combined with simplified ecological relationships to model the supply, flow and demand of ES (e.g. De Knegt et al., 2022). The main purpose of NCM is to support scenario analysis, land-use planning, and policy evaluation, rather than to capture the mechanistic ecological effects of individual landscape features.

There are several tools available, such as *InVEST*, *ARIES for SEEA*, and *Natural Capital Model* (see Table 9). These tools usually operate based on several sub-models, each representing a specific ecosystem service.

**Table 9.** Examples of three models used to assess ecosystem services (ES) and their monetary values (Natural Capital Alliance, 2026; ARIES User Guide 2026; De Knegt et al., 2022).

| Name | Developed by | Sub-models (ES) | Specifications |
|---|---|---|---|
| InVest | Stanford University & Natural Capital Alliance | Annual Water Yield<br>Carbon Storage & sequestration<br>Coastal Blue Carbon<br>Coastal Vulnerability Model<br>Crop Production<br>Forest Carbon Edge Effect<br>Habitat Quality<br>Habitat Risk Assessment<br>Nutrient Delivery Ratio<br>Offshore Wind Energy Production<br>Crop Pollination<br>Sediment delivery ratio<br>Seasonal water yield<br>Scenic quality<br>Urban cooling | Mostly focused on biophysical modelling, less on valuation.<br>Run in different sub-models, which makes it flexible to exclude some.<br>GIS-based.<br>Open-source.<br>Useful for scenario analyses. |

| | | | |
|---|---|---|---|
| | | Urban flood risk mitigation<br>Urban nature access<br>Urban stormwater retention<br>Visitation: Recreation & tourism<br>Wave energy production | |
| ARIES for SEEA | Basque Centre for Climate Change, UNDESA and UNEP | Crop provisioning<br>Timber provisioning<br>Livestock grazing<br>Biomass provisioning<br>Fish provisioning<br>Water supply (surface & groundwater)<br>Carbon sequestration & storage<br>Flood regulation<br>Water purification<br>Soil erosion control<br>Pollination<br>Coastal protection<br>Air filtration<br>Local climate regulation<br>Nature-based recreation<br>Aesthetic/amenity services<br>Nursery habitat for fisheries<br>Habitat connectivity | Developed specifically as a tool to provide measurements based on the SEEA-EA statistical framework.<br>Focus on monetary valuation and national accounting. |
| Natural Capital Model | Wageningen University & Research, Netherlands Environmental Assessment Agency (PBL) | Wood production<br>Drinking water production<br>Biomass for energy (forest)<br>Carbon sequestration (forest)<br>Carbon sequestration (peat soils)<br>Pollination<br>Water purification<br>Air quality regulation<br>Prevention of heat islands<br>Pest control<br>Water retention<br>Soil fertility (hydrology)<br>Natural heritage<br>Outdoor recreation<br>Food and fodder production | Developed for the Netherlands context.<br>Detailed input maps.<br>Focus on biophysical modelling. |



### 4.5.2. Modelling of different ecosystem services and landscape features

Natural Capital Modelling primarily addresses regulating and provisioning ES, which are closely linked to biophysical processes and therefore more readily represented in economic-ecological models than cultural or supporting services (e.g. De Knegt et al., 2022). Services that are mostly influenced by broad land-cover patterns (e.g. carbon sequestration and biomass production) are relatively easy to model, because they can be directly associated with specific ecosystem types (UN, 2024). For example, forest carbon sequestration can be estimated using land-cover maps that distinguish forest types with different growth rates, resulting in relatively robust estimates (e.g. De Knegt et al., 2022).

In contrast, ecosystem services that depend on finer-scale ecological processes and are strongly mediated by LFs - such as biodiversity, pollination, or pest control - are only partially captured by NCM approaches (see e.g. Alexandridis et al., 2021). These services are more dynamic and context-dependent, and model resolutions often do not accommodate small-scale features such as hedgerows or field margins. In addition, landscape quality and management are rarely included. For example, in the 'Natural Capital Model' (De Knegt et al., 2022) pollination is evaluated based on ecosystem types such as 'tree lines', 'meadows' and 'public green space'. For each type, a suitability score is set to assess the potential for nesting and floral suitability to support pollination. However, how well such a type of ecosystem is designed or managed will have a strong effect on the ES supply capacity, and this type of detail is typically lost in the modelling approach. An Italian study by Valeri et al. (2024) has demonstrated the value of including LF quality into assessments, as they say a lot more about the actual biodiversity services delivered. They came up with a first proposal of how to assess this for economic valuation, but that is not directly applicable to the prevailing NCM approaches.

To illustrate these limitations, Table 10 summarizes how the individual ES discussed in Chapter 2 are generally represented in the models of InVest, ARIES for SEEA and Natural Capital Model.



**Table 10.** Overview of how the different ecosystem services (ES) are modelled in Natural Capital Modelling (NCM) approaches of InVest (Natural Capital Alliance, 2026), ARIES for SEEA (NCAVE & MAIA, 2022) and Natural Capital Model (De Knegt et al., 2022), including some analysed strengths and limitations. (prod. = production).

| Grouped ES | Inputs for modelling | Strengths | Limitations regarding LFs |
|---|---|---|---|
| Animal prod.<br>Grassland prod.<br>Crop yield<br>Biomass prod. | Production functions based on landcover classes, productivity indices, soil fertility, climate data, and simplified yield data or models. | Good for broad productivity mapping and scenario analysis. | Does not capture field edge effects, species composition, management intensity, or microclimate created by hedgerows/trees. |
| Soil erosion control, Nutrient retention, Soil nutrient levels, Soil water retention, Water quality, Water storage | Models water flows based on erosion models, vegetation cover, hydrological models, soil texture/depth data, infiltration/runoff models, and wetland/groundwater proxies. | Effective at catchment or landscape scale; identifies hotspots and broad patterns. | Small features (buffer strips, ponds, hedgerows) often below resolution; fine scale retention or soil processes not represented. |

| | | | |
|---|---|---|---|
| Pollination, Pest control | Models potential supply based on habitat, nesting and floral availability of different land uses. Coupled with crop-specific demand by species-specific foraging ranges. | Among the better developed NCM services; captures general landscape level ecological patterns. | Simplifies species communities; cannot model specific flower strips, hedgerow structure, management and landscape quality. |
| Biodiversity | Modelled using habitat suitability models, species distribution models, and connectivity metrics. | Useful for broad assessments of habitat availability. | Does not capture fine-scale habitat quality, or species-specific responses, which are relevant for LFs. |
| Cultural services | Proxies such as accessibility, landscape diversity, presence of water to model recreational potential. | Useful for broad assessments of recreational potential. | Does not capture experiential values or heritage related to smaller LFs. |



### 4.5.3. From Natural Capital Modelling to Natural Capital Accounting

As described in sections 4.3 and 4.4, several approaches exist to economically value ecosystem assets and services. A common approach is based on modelling of ES, as outlined above, which are subsequently translated into a monetary value. For national accounts, this valuation is typically carried out within the UN SEEA-EA framework, which restricts valuation to exchange values (i.e. market-equivalent prices). Different methods are used to convert ES flows into exchange values. A few examples of such valuation methods in the SEEA-EA framework are presented in Table 11, based on the SEEA-EA report (NCAVES & MAIA, 2022).

**Table 11.** Different methods that can be used to valuate ecosystem services (ES) according to the SEEA-EA framework.

| Valuation method from SEEA-EA framework | What it measures | How it works | Examples |
|---|---|---|---|
| Market price | Value of ecosystem goods sold in markets | Market price is used, but adjusted for taxes, subsidies and processing costs to isolate the ecosystem's contribution | Timber from forests at stumpage prices |

| | | | |
|---|---|---|---|
| Resource rent | Ecosystem's share of production value | Subtract all other inputs (labour, capital) from total value to get to contribution of ecosystem. | Price of sold crops, minus costs for labour, machinery, inputs and capital costs |
| Production function | Ecosystem's marginal contribution to output | Using a biophysical model to link ecosystem processes to production | Model how wild pollinators increase crop yields |
| Avoided damage | Costs avoided due to ES | Identifying real damages and model how ecosystems reduce those damages | Floodplains reducing flood damage to infrastructure |
| Replacement cost | Costs of replacing ES with built infrastructure | Identify a real substitute and estimating the costs of providing a similar service | Wetlands replaced by water treatment facilities<br>Urban trees replaced by cooling systems |

As all approaches described (Table 11) rely on available data, the most appropriate method can differ depending on purpose and regional context. Within the SEEA-EA framework, the resource rent is identified as preferred approach for many ecosystem services. A study from the MAIA project (Schenau et al., 2022) examined the applicability of this approach in the Netherlands. They found that preferred valuation methods differ depending on the type of ES: rent-based approaches are most suitable for provisioning services; replacement cost or avoided damage cost methods are preferred for regulating services; and consumer expenditure and hedonic pricing are most appropriate for cultural services. It should be noted that these methods are assessed as best applicable for national accounting and do not necessarily represent the best methods for all different purposes.

## 4.6. Examples from studies combining LFs and NCA or NCM

Three studies (Powell et al., 2020; le Clech et al., 2024; Kruit et al., 2025) were identified that explicitly combine LF with NCA or NCM approaches, all focusing on specific regions. The studies clearly demonstrate the potential of LFs (mainly linear and woody in these studies) to support (functional) biodiversity, to contribute to climate mitigation and regulation and to cultural heritage. However, these values should be treated as indicative because of the variation and context-dependency of processes underlying modelled ES. Next to that, it also shows that approaches are quite different, and the use of harmonised methods would be beneficial for comparisons.

**Heritage, natural capital and ecosystem services – boundaries and linear landscape features in the Lower Severn Vale (Powell et al., 2020)**
The study developed an ES accounting model for linear LFs, tested across three case study areas in Gloucestershire, UK. Linear features comprise field boundaries, routeways, and water-management and flood-protection

structures associated with the River Severn (e.g. dykes, embankments, ditches, and channels). The study assessed a broad range of provisioning, supporting, regulating and cultural ES, based on developed indicators (legibility, time-depth, inter-relationship). Across all three case studies, overall benefit flows were comparable, with boundary features contributing most strongly. Cultural ES made accounted for a substantial share of total value, particularly in the study areas famous for its cultural heritage landscape.

**Effects of linear landscape elements on multiple ecosystem services in contrasting agricultural landscapes (le Clech et al., 2024)**
This study assessed the potential effect of linear landscape features (woody, grassy, flower and mix) on six different ES (food and feed provision, pollination, pest control, climate regulation, aesthetics and habitat maintenance), combining scenarios and spatially explicit modelling approaches (NCM). This was done for three regions in Denmark, the Netherlands and Hungary, respectively. The results showed positive effects on all regulating and cultural ES, with higher delivery of ES linked to higher abundance of LFs. Greatest potential increase in ES was found in places with a low initial supply.

**Exploration of use of the Natural Capital Model and Accounting to study effects of increased agroforestry and landscape features on ecosystems services (Kruit et al., 2025)**
Kruit et al. (2025) explored how agroforestry and woody LF can potentially contribute to different ES using the Natural Capital Model in the Netherlands. The studied ES included pollination, pest control, carbon sequestration, water infiltration and recreation, as these were the only ES sub-models that gave an effect of adding tree lines or forest on agricultural land. To enable modelling, land use categories have been added or adapted to the land cover maps for woody LFs, alley cropping and food forests, each with specific habitat qualities to assess the effect on the mentioned ES. Results show that large scale introduction of woody LF - around 67,000–69,000 ha in the future scenarios – would substantially increase pest control (up to 30% more natural enemies), pollination (5% higher production in pollination dependent crops), and carbon sequestration (10% increase, equal to 0.22 Mton $CO_2$ per year). Effects on water infiltration and recreation were found limited within the applied modelling framework.



## 4.7. Discussion

This discussion synthesises converging and conflicting findings across the reviewed studies, focusing first on modelling approaches and subsequently on monetisation.

### 4.7.1. Modelling of ecosystem services by LFs

Sections 4.5 and 4.6 show that modelling can be a useful tool to assess the ES delivered by LFs, which can in some cases be successfully translated into monetary values. However, current models have clear limitations. They often lack the capacity to account for more complex ecological interactions, small-scale LFs, and variation in LF quality. A key constraint is the typical course spatial resolution of input data, which limits the representation of fine-scale landscape elements such as hedgerows or field margins. In addition, ES with a high context-dependency and temporal variability -such as pollination- are inherently more difficult to capture through modelling.

Based on the reviewed models, two different approaches can be distinguished. First, models based on biophysical modelling (e.g., InVest & Natural Capital Model), which are well-suited for scenario analyses and working on regional or landscape levels. Second, models based on broader statistics (e.g., ARIES for SEEA), which are clearly meant for national accounting purposes and are therefore more applicable at national scales. Both types have value but serve different purposes. To analyse specific regional or landscape context, biophysical models such as InVest & Natural Capital Model may be more applicable. Their outputs should be interpreted as indicative estimates of the potential of LFs to support and enhance ecosystem services, and as a complementary tool for upscaling empirical evidence to broader landscape-scale scenarios.

### 4.7.2. Monetisation of ecosystem services by LFs

**Methods for monetization**

There are different methods and approaches to economically value ES. The presented SEEA EA framework focusses purely on exchange values (market-based prices), while the literature review revealed that there are also many successful examples using SP (stated preference) methods (e.g. willingness to pay).

The compatibility of valuation methods with accounting principles remains debated, particularly concerning the use of SP approaches, which dominate the studies identified in the literature review of section 4.4. It is sometimes argued that the monetary values estimated by SP methods are not compatible with the exchange-value-based accounting approaches, such as in SEEA EA. While this has led to some scepticism towards SP methods in accounting contexts, it has been argued that SP methods can provide consistent proxies for exchange values (Droste & Bartkowski, 2018; Obst et al., 2016; Dasgupta, 2021).

Through this literature review, we provide an exemplary overview about available monetary values for LF-relevant ES drawing on the whole range of valuation methods, based on the assumption that they are all relevant for the purposes of NCA. Nonetheless, it should be considered that monetisation cannot capture all values that ecosystems may have, and that it does not capture all values within Total Economic Value (TEV) equally well (Pascual et al., 2023).

Looking specifically at monetisation of LFs, sections 4.4 and 4.5 highlight that there is no perfect valuation method available to monetise the contribution of all LFs to all ES. While biophysical modelling combined with market prices can be a good tool to monetise carbon sequestration by LFs, this will not work as well for assessing cultural services, which can best be assessed by SP methods. Given the current state of knowledge, the valuation of ES associated with LFs would therefore ideally rely on a combination of valuation approaches, selected according to the type of ES and the nature of the benefits involved. However, such pluralistic valuation also introduces complexity. The underlying theoretical assumptions of different methods vary, and combining monetary values derived from different approaches in a consistent and coherent way remains challenging. Moreover, combining valuation methods increases the risk of the selective use of evidence: choosing ES or valuation approaches based on convenience, data availability or biased expectations of outcomes rather than conceptual consistency.

**Conceptual limitations of monetisation**

Overall, the monetisation of ES by LFs remains subject to important conceptual and practical limitations. Human well-being depends on the changes across multiple forms of capital, including natural capital (Dasgupta, 2021). To avoid natural capital being neglected - as is currently the case in the way economic metrics such as gross domestic product - it needs to be systematically incorporated in national accounts, a gap that the SEEA-EA framework seeks to address. However, the use of economic valuation alone does not automatically translate into policy influence. As shown by Zolyomi et al. (2023), meaningful policy impact also depends on factors such as stakeholder engagement and institutional context. At the same time, limitations of monetisation need to be acknowledged to avoid misinterpretation and misuse. The monetisation of ES raises some conceptual and normative concerns, as it reduces complex ecological and ethical values to monetary metrics, and may exclude alternative value considerations from decision-making (Pascual et al., 2023). First, it has been widely questioned whether all ES – particularly those linked to biodiversity, which underpins the provision of many ES, and some cultural ES with strong plural value components – can or should be meaningfully expressed in monetary terms. Critics argue that monetisation may oversimplify ecological complexity, obscure incommensurable values, and impose an instrumental utilitarian framework that fails to capture alternative ethical perspectives on nature (Scholtes, 2015). In particular, biodiversity is often considered to hold value beyond human preferences, making its reduction to WTP measures conceptually problematic. Second, aggregating diverse ecosystem functions into a single monetary metric risk neglecting ecological thresholds, irreversibility and distributional concerns. These critiques highlight that, while monetary valuation can support certain policy contexts, it should be applied with caution and complemented by pluralistic valuation approaches that recognise the existence of multiple, incommensurable values and integrate diverse value frameworks (De Valck et al., 2023; Spash, 2008). Third, the functioning of LFs can be strongly context-dependent, as also discussed in section 2.5.3, and this context dependency extends to their economic valuation. It remains unclear to what extent monetary values can be transferred across contexts, for example when applying values derived for other natural capital assets (e.g. soil functioning) to LFs. In theory, some ES values are relatively homogenous – such as carbon sequestration, implying that the economic value of one tonne of $CO_2$ should be independent of whether it is sequestered by a grassland soil or a hedge. In practice, however, there are some tentative indications that economic values are more context-specific than theory suggests, limiting their transferability across contexts (Johnston et al., 2018). For services such as pollination or pest control, transferability is particularly constrained, as benefits only materialise where local demand coincides with local supply.

# 5. Evidence from study areas

Chapter 5 summarises evidence on the benefits of landscape features within the LAFERIA case-study areas across Europe: Bulgaria (Plovdiv), Finland (Uusimaa), Belgium (Brabantse Wouden), the Netherlands (South Holland), Sweden (Kalmar County) and Germany (Saxony). Evidence was provided by the case study partners. By combining location-specific scientific studies with regional reports and grey literature, this chapter provides an integrated overview of the ecological and socio-economic impacts of landscape features on biodiversity and ecosystem services. It further assesses the extent to which this empirical evidence confirms, refines or adds new insights to the overarching conclusions of the LAFERIA synthesis report.

While many relevant studies should already have been captured through the literature review in earlier chapters, this section focuses more specifically on evidence generated in, or directly relevant to, the case-study regions. In some cases, this includes studies conducted within LAFERIA case-study areas themselves, whereas in other regions such location-specific evidence remains limited or absent. Though some studies included here could have been included also in the review, this section contextualises and complements the broader evidence base from the review by highlighting how ecosystem-service benefits of LF manifest within the specific case-study settings.

Six primary papers have been identified in the review for **Bulgarian case study** region: Ahmed et al., 2025; Aneva et al., 2020; Concepción et al., 2020; Dyulgerova & Biserkov, 2024; Dyulgerova et al., 2015; Popgeorgiev et al., 2024. Across the reviewed studies, the added evidence confirms the findings on ES and biodiversity benefits presented in this report (Sections 2.2 and 2.3) while providing additional empirical detail. Ahmed et al. (2025) found that tree–shrub belts function as refuges and corridors for mammals and birds in agricultural landscapes, which is directly in line with the report's synthesis that woody linear features enhance connectivity and biodiversity. Similarly, Aneva et al. (2020) showed that plant diversity is highest in semi-natural systems such as grazing land with trees, wetlands, and field margins, which supports the report's conclusion that grassy and woody LFs (Sections 2.2.5–2.2.6) are key biodiversity hotspots in farmland. Popgeorgiev et al. (2024) demonstrated that many amphibians depend on agricultural landscapes where wet features such as ponds and ditches are present, which aligns with the report's findings on the importance of wet LFs (Sections 2.2.7–2.2.10). In terms of landscape structure, Dyulgerova and Biserkov (2024) found that edge density increases overall bird diversity, confirming the report's emphasis on connectivity and heterogeneity (Section 2.5), while Dyulgerova et al. (2015) showed that woody encroachment following land abandonment reduces farmland bird diversity, reinforcing the report's discussion of trade-offs between LF types. Finally, Concepción et al. (2020) demonstrated that most LFs support biodiversity but in a context-dependent and sometimes non-linear way, which is fully consistent with the report's overarching conclusion on context dependence (Section 2.5.2).

Taken together, these studies strongly support the report's synthesis by confirming that biodiversity benefits arise from a combination of woody, grassy and wet LFs in heterogeneous landscapes, while also reinforcing the need to balance these features to avoid negative impacts on specialist species (Ahmed et al., 2025; Aneva et al., 2020; Concepción et al., 2020; Dyulgerova & Biserkov, 2024; Dyulgerova et al., 2015; Popgeorgiev et al., 2024).

Across the studies developed in the **Finnish case study area**, the evidence confirms and nuances the report by highlighting the strong dependence of LF performance on hydrology, seasonality and design. The focus is on wet structures, such as constructed wetlands and two-stage channels. A two-stage channel is a modified agricultural drainage channel with a narrow main channel for low flows and adjacent floodplain benches that are inundated during high flows, designed to improve drainage while enhancing water quality and habitat functions (REF).

Constructed wetlands consistently reduce nutrient loads, but with moderate and variable efficiency. Valkama et al. (2017) found annual retention of ~13% for phosphorus and 14% for nitrate, largely controlled by retention time and flow conditions. Similarly, Koskiaho et al. (2003) reported highly variable retention (6–67% for phosphorus), strongly dependent on hydraulic residence time and input loads. A key insight is the strong seasonal mismatch between load and retention. Koskiaho et al. (2003) showed that most nutrient transport occurs during snowmelt and autumn peaks, when retention is often limited, while Valkama et al. (2017) found that retention is highest during the growing season, especially for nitrogen under warm, low-oxygen conditions. This supports the report's emphasis on context-dependent LF functioning.

Mechanistically, studies confirm that phosphorus is mainly retained via sedimentation, whereas nitrogen removal is biologically driven. Koskiaho et al. (2003) showed that most phosphorus is particle-bound, while nitrogen is removed mainly via denitrification, in line with the report's differentiation between physical and biogeochemical processes. The importance of LF design and vegetation is also clear. Koskiaho and Puustinen (2019) found much higher retention in larger, well-dimensioned wetlands (up to ~58% TP), while vegetation enhances retention by slowing flow and trapping particles (Valkama et al., 2017).

In terms of biodiversity, Finnish studies show clear co-benefits. Wahlroos et al. (2015) demonstrated rapid colonisation of constructed wetlands by diverse plant and animal communities, while Huttunen et al. (2024) found that two-stage channels increase riparian diversity and overall gamma diversity in agricultural landscapes. At the same time, trade-offs are evident. Koskiaho et al. (2003) showed that poorly designed systems can act as nutrient sources, and Huttunen et al. (2024) found that in-stream biodiversity does not always improve despite gains in riparian zones.

Overall, the Finnish studies strongly support the report's synthesis: hydrological LFs enhance water quality and biodiversity, but their effectiveness is highly context-dependent, design-sensitive and often involves trade-offs (Koskiaho et al., 2003; Valkama et al., 2017; Wahlroos et al., 2015; Huttunen et al., 2024).

Across the **Flemish case study** literature, the evidence consistently reinforces and refines the LAFERIA report's conclusions, particularly regarding the multifunctionality and trade-offs of landscape features (LFs). Studies show that woody LFs such as hedgerows are key contributors to climate regulation. Van Den

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

Berge et al. (2021a; 2021b) found that hedgerow-grown trees exhibit sustained biomass growth and carbon sequestration, while soil carbon stocks are substantially higher in hedgerows than in adjacent land uses and are largely lost following removal, which is in line with the report's characterisation of woody linear features as important long-term carbon sinks (Van Den Berge et al., 2021a; 2021b).

In terms of biodiversity and ecosystem functioning, Eeraerts et al. (2021) demonstrated that fruit orchards and woody semi-natural habitats provide complementary nesting and floral resources for pollinators over time, which directly supports the report's emphasis on landscape mosaics and temporal resource continuity. Similarly, Hulsmans et al. (2023) showed that pollinator abundance and diversity increase with landscape diversity and the presence of semi-natural elements, especially at orchard edges, confirming the report's conclusions on the importance of connectivity and landscape heterogeneity (Section 2.5). At a smaller scale, Van Vooren et al. (2018) found that hedgerows and grass strips enhance multiple ecosystem services, including soil carbon storage, pest control potential and water quality regulation, while simultaneously reducing crop yield close to the feature, illustrating the trade-offs highlighted in the report between provisioning and regulating services.

At the landscape scale, Rosier et al. (2024) demonstrated that vegetated landscape elements such as hedgerows can strongly reduce runoff and flood risk depending on their spatial configuration, which aligns with the report's treatment of LFs as key regulating elements. In parallel, Marcantonio et al. (2024) showed that expanding small woody features can significantly improve habitat connectivity in intensive agricultural landscapes across the EU, further supporting the report's conclusions on the importance of spatial arrangement and connectivity (Sections 2.2 and 2.5).

Other studies highlight the role of woody patches and structural habitat features beyond linear elements. Proesmans et al. (2019) found that small forest patches in agricultural landscapes support diverse pollinator communities, with species richness driven by both local habitat quality and surrounding landscape structure in Northwestern European rural landscapes, including Belgian. This is consistent with broader European evidence that such patches act as biodiversity reservoirs and contribute to multiple ecosystem services (Decocq et al., 2016), reinforcing the report's classification of woody patches as high-value LFs (Sections 2.2.3–2.2.4). Additionally, Kratschmer et al. (2024) showed that hedgerow structural complexity (e.g. layering, density) is a key driver of biodiversity and associated ecosystem services, directly supporting the report's emphasis on LF quality and design.

Finally, agroforestry studies provide clear evidence of production–ecology trade-offs. Pardon et al. (2025) found that crop yield decreases with proximity to trees in alley cropping systems, although crop quality remains largely unaffected, which aligns with the report's discussion of competition effects in tree-based systems. Similarly, Panozzo et al. (2026) showed that intercrop yields in temperate alley cropping systems are on average lower than in treeless conditions, particularly near tree rows, confirming that integrating woody LFs can reduce local productivity while delivering broader ecosystem services.

Taken together, these studies also support the report's synthesis by showing that woody, grassy and semi-natural LFs enhance biodiversity, pollination, carbon storage and regulating services in agricultural landscapes. At the same time, they provide robust empirical evidence for the report's core conclusion that outcomes are context-dependent and involve trade-offs: benefits are highest in heterogeneous, well-connected landscapes with structurally diverse LFs, but these gains may be accompanied by local reductions in crop yield or shifts in species composition depending on LF type, configuration and management (Decocq et al., 2016; Eeraerts et al., 2021; Hulsmans et al., 2023; Kratschmer et al., 2024; Marcantonio et al., 2024; Panozzo et al., 2026; Pardon et al., 2025; Proesmans et al., 2019; Rosier et al., 2024; Van Den Berge et al., 2021a; 2021b; Van Vooren et al., 2018).

Evidence of the **Dutch case study area** strongly confirms the synthesis, showing that field margins, flower strips and wet features function as key multifunctional LFs, but with effects that are highly local, design-dependent and often limited in terms of population-level outcomes.

A consistent finding is that flower strips and field margins increase the abundance and species richness of pollinators and other insects. For example, Bukovinszky et al. (2016) found that increasing flower availability in agricultural landscapes led to more abundant and species-rich bumblebee communities, with clear spill-over into surrounding habitats after two years. Similarly, Stip et al. (2021) reported significantly higher abundance and richness of butterflies, bees, and hoverflies in flower-rich field margins compared to reference edges. This confirms the report's conclusion that LFs enhance regulating and supporting services through increased resource availability.

However, Dutch studies also clearly show that responses differ among taxa and depend on limiting factors beyond food availability. Bukovinszky et al. (2016) found no short-term increase in solitary bee populations despite increased floral resources, likely due to limited nesting habitat. Likewise, Stip (2019) found no consistent effect of bird fields on bee abundance, even though other insect groups (butterflies, hoverflies) increased significantly. This aligns with the report's emphasis that multiple resources (food, nesting, habitat continuity) jointly determine LF effectiveness. A key recurring mechanism is the role of plant diversity as a driver of ES. Stip et al. (2021) showed that both insect abundance and species richness increase with the number of flowering plant species, while broader synthesis work (e.g. Bos et al., 2014) demonstrates that field margins consistently enhance pollinators and natural enemies through increased floral resources. This directly supports the report's conclusion that LF quality (vegetation composition and diversity) is more important than mere presence.

The Dutch literature provides particularly strong evidence for biological pest control as a key ES. Stip et al. (2021) found higher abundances of aphid-eating hoverflies in field margins, indicating enhanced natural pest regulation, while the broader review by Bos et al. (2014) shows that field margins consistently increase natural enemies, although effects on actual pest reduction are less often quantified. This again reflects the gap between intermediate and final ecosystem outcomes highlighted in the report.

Longer-term and larger-scale experiments confirm strong biodiversity gains but mainly for generalist species. In the PARTRIDGE project, Maas and Van der Arend (2023) found that beetle banks and flower blocks increased insect abundance up to fivefold and species richness substantially compared to conventional fields. Similarly, Stip (2019) reported strong increases in butterflies and other insects on perennial bird fields. However, both studies emphasise that most responding species are widespread generalists, with limited evidence for recovery of specialists - confirming the report's findings on biodiversity quality versus quantity. Dutch studies also highlight clear spatial limitations of LF effects. Effects are strongest within or directly adjacent to the feature, with limited evidence for landscape-scale population increases. Stip et al. (2021) explicitly caution that observed benefits are local and not necessarily reflected at landscape scale. Similarly, Bukovinszky et al. (2016) show delayed spill-over effects, indicating that landscape-level impacts require time and sufficient spatial coverage. This strongly supports the conclusion that landscape configuration and density of LFs are critical.

For wet and riparian features, Dutch monitoring shows consistent but moderate biodiversity gains. Verhofstad et al. (2021) found that nature-friendly banks increase plant species richness, habitat width and macrofauna diversity compared to conventional banks, while long-term monitoring by Waterschap Aa en Maas (2011) shows increases in plant and dragonfly diversity following ecological restoration. However, effects are often constrained by water quality and management, again highlighting context dependency.

Finally, the reports consistently point to trade-offs and management constraints. Narrow or annually disturbed margins provide limited habitat value, favour generalists, and may lack nesting opportunities (Stip et al., 2021). High nutrient levels, mowing regimes and surrounding land use further influence outcomes across both dry and wet LFs. Overall, Dutch case studies confirm the LAFERIA synthesis: field margins, flower strips and wet features enhance biodiversity and ecosystem services, but effects remain context-dependent, local and design-driven (Bukovinszky et al., 2016; Bos et al., 2014; Stip et al., 2021; Stip, 2019; Maas & Van der Arend, 2023; Verhofstad et al., 2021).

Grey literature from Kalmar County, the **Swedish case study area**, supports the LAFERIA synthesis, showing that landscape features — particularly wetlands, farmland with individual oaks, riparian zones and small non-cropped biotopes — enhance biodiversity, ecosystem services and landscape resilience, but with strong dependence on landscape structure and connectivity (Länsstyrelserna, 2019). Multiple regulating ES, including nutrient retention, water flow regulation, groundwater recharge and flood buffering are highlighted, alongside habitat provision for birds, amphibians and aquatic species. Similarly, riparian zones and buffer strips contribute to water quality and erosion control while functioning as ecological corridors, reinforcing their role as key hydrological LFs (Skogsstyrelsen, 2018; Rambol et al., 2019; Länsstyrelsen Kalmar län.,n.d).

A central contribution of the Swedish evidence is the emphasis on green infrastructure and landscape connectivity. Kalmar planning frameworks explicitly treat LFs as part of functional ecological networks (e.g. core areas, corridors and stepping stones) rather than isolated features. This aligns with the report's conclusion that spatial configuration is critical for LF effectiveness. Presence of oaks on farmland also provides habitat continuity for specialist taxa (e.g. insects, fungi and birds) and contributes to long-term ecological resilience (Länsstyrelsen Kalmar län.,n.d.).

A distinctive aspect of the Swedish case is the inclusion of coastal and shallow marine habitats as LFs, where restoration and grazing management enhance nursery habitats, ecological productivity and fisheries-related ecosystem services. Though this goes beyond LAFERIA scope, the approach highlights cross-system connectivity (Länsstyrelsen Kalmar län.,2017; Länsstyrelserna, 2019).

Finally, a **German case study** paper of Brandenburg (Schaan et al., 2025), closely related to the Saxony case study in the LAFERIA project, aligns with the synthesis, but mainly adds a landscape-planning and targeting perspective rather than direct ecological effect measurements. Schaan et al. (2025) show that agricultural landscape features (LFs) provide multiple ecosystem services -such as pollination, pest control, erosion reduction and water regulation- confirming their multifunctional role highlighted in this report. However, the study demonstrates that their potential is currently strongly constrained by very low availability: average LF cover is only about 3.6%, and 94.4% of landscapes fall below the 10% EU target, indicating a major structural deficit in agricultural landscapes. Schaan et al. (2025) show that simple, large-scale arable landscapes, with large field sizes and low edge density, have the lowest LF cover and diversity, while more complex mosaic landscapes contain higher LF shares. This directly supports the conclusion that LF effectiveness depends not just on presence but on overall landscape configuration. The authors recommend prioritizing simple, intensively managed landscapes for restoration, where increases in LF cover can generate the greatest ecological gains. This is consistent with our emphasis on landscape-scale optimisation, rather than uniform implementation.

At the same time, the paper explicitly integrates socioeconomic feasibility, an aspect less developed in many ecological studies. By identifying areas with low productivity or high erosion risk as priority zones, Schaan et al. (2025) show that LF implementation can minimise opportunity costs and improve farmer uptake. This reinforces the idea that LF deployment depends not only on ecological suitability but also on practical and economic constraints.

In the **Portuguese case study**, no peer-reviewed studies focusing on LF presence as compared to their absence on intensive agriculture were identified additionally to the reviewed ones. A master thesis modelled ES in one site within the case study region (25 000 ha of mixed landscape, including irrigated olive) concluding that riparian vegetation provides the highest ES capacity overall (Lemos 2017).

# 6. Cross-cutting insights and patterns

## 6.1. Key insights on biodiversity and ecosystem services

Our results show that considerable scientific evidence demonstrate the benefits of all types of landscape features (LFs) typical of European agricultural landscapes to biodiversity on farmland. Reviews consistently report benefits of herbaceous strips or parcels and ditches for the studied biodiversity taxa. Hedgerows and riparian vegetation (often including tree lines or forest patches) display mixed benefits depending on the taxonomic groups, and on respective preferences towards open landscapes. Prioritisation and careful planning of establishing such LFs may be needed in such cases to minimise adverse impacts on vulnerable biodiversity values. Many taxa supported by LFs are providers of important ES on farmland, such as pollination and pest and disease control. However, reviews demonstrated that more diverse and abundant local assemblages of beneficial organisms do not necessarily translate into the respective ES and benefits in crop production (Figure 1; section 1.2). However, in cases when improved biodiversity of beneficial organisms leads to reduced chemical applications on crops, it provides public benefits that are seldom assessed in empirical work on LFs.

Evidence on LFs providing essential ecological structures (Figure 1) that are necessary for regulating ES is ample and diverse. Based on existing evidence, most multifunctionally beneficial features are hedgerows, herbaceous strips, and riparian vegetation. There is little evidence that LFs in agricultural landscapes would have strongly negative effects on ES by mainly contributing to dis-services. Ecological structures contribute also to some direct production benefits, particularly demonstrated regarding woody elements on pastures. Importantly, a predominantly neutral impact on yield in adjacent crops demonstrates that adding or retaining LFs can deliver multiple ES on agricultural land and beyond it, without negative impacts on crop yields.

Natural capital modelling (NCM) is a powerful approach that can translate empirical evidence, as collated here, into broader, spatially explicit landscape-scale scenarios. These methods could estimate how landscapes currently supply ES and how this may change given established or restored LFs or climate scenarios, supporting land-use planning and policy evaluation across scales. Because of a high level of context-dependence, the NCM outputs are indicative for the potential of LFs to sustain and enhance ES, rather than quantifying the amount of ES delivered at scales of individual field parcels or land ownership units.

The most apparent trade-off of establishing or reintroducing LFs on agricultural land, and especially in intensive production, is the crop area removed from production of agricultural commodities. Most multifunctional and complex LFs – hedgerows, riparian vegetation – require relatively large parcels of land in comparison to margins or groups of trees. Thus, the opportunity cost to farmers is likely to be considerable in comparison to measurable private benefits (in yield quantity and/or quality). This signifies the critical importance of applying various methods of identifying and measuring ES flows also to other beneficiaries than farmers, and to identify and develop incentives for farmers to reintroduce LFs. Valuating these ES in monetary terms is a step towards incorporating values of LFs into a cost-benefit analysis (both on farm and societal levels) and natural capital accounting (NCA). A comprehensive valuation may provide much needed leverages to sustain and increase incentives for restoration of LFs as providers of measurable benefits (Figure 5).

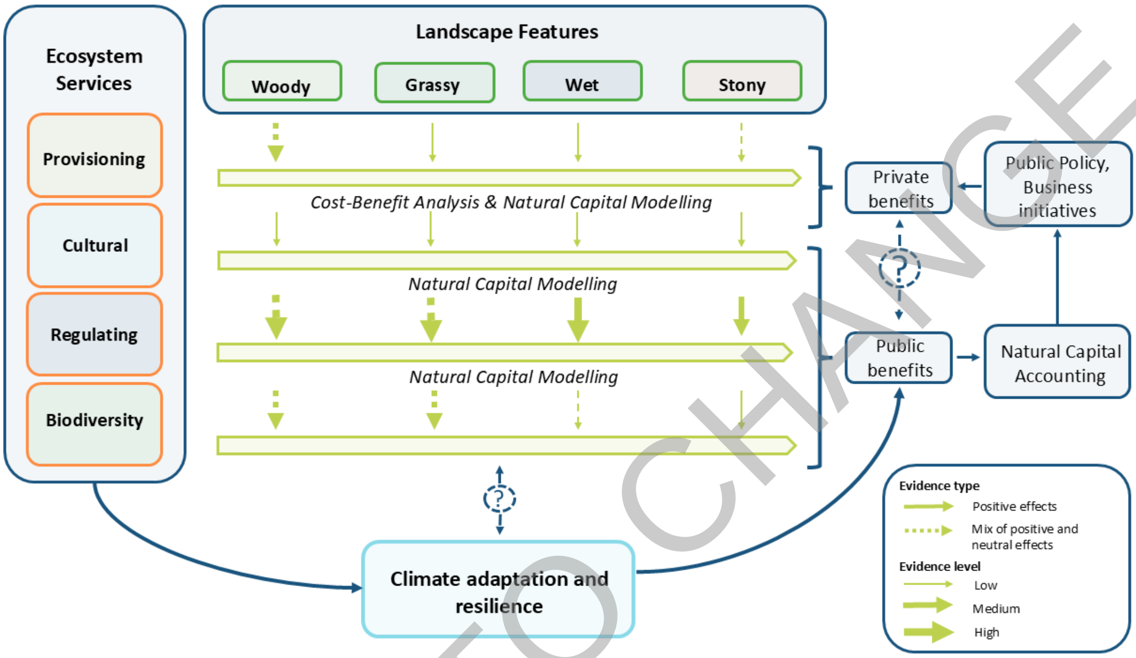



**Figure 5.** A graphical summary of the state of evidence reviewed for this synthesis including: the flows of the ecosystem service classes arising from the major types of landscape features on the agricultural land, confirmed associated private and public benefits, as we all as identified relevant valuation approaches. The relationships where the evidence has most knowledge gaps have an added "?".

## 6.2. Climate change impacts

Though the current LFs contribution to natural green and blue infrastructure may have considerable values for increasing resilience of landscapes under climate change, the evidence for the roles of LFs is highly fragmented and incomplete, particularly in the context of intensive agricultural landscapes. Current research provides almost no insight into how resilient these features themselves are under future climate conditions. It is plausible that linear and small-scale grassy and woody elements may be particularly vulnerable if plant species are poorly matched to new climatic regimes, and newly planted trees may face high failure rates. Wet features could be compromised by rising irrigation demands, heatwaves, and droughts, particularly in the Mediterranean region. In contrast, stony features are likely to be more robust. A potential trade-off in tree planting increasing local water stress through high evapotranspiration is still poorly understood across European biophysical zones, though existing examples can be used in risk assessments.

The biodiversity supported by LFs within intensive agricultural landscapes may also be at risk, as their small parcel size, linear shape, and exposure to agricultural disturbance potentially make them prone to rapid species turnover under climate change. At the same time, maintaining high biodiversity remains essential because it provides functional redundancy, ensuring that ES continue even if some species decline. Sufficient areal cover to sustain functioning at landscape or watershed levels, robust connectivity and overall ecosystem condition maintained through adaptive management emerge as critical factors for sustaining resilience in the long term.

## 6.3. Key insights on economic valuation of ecosystem services

Economic valuation can help demonstrate the societal benefits of ES provided by LFs, and such valuation is well established in environmental policy appraisal. Although economic valuation is increasingly incorporated into NCA, its application to farmland LFs remains uneven. Existing monetary estimates, particularly for cultural ES, vary widely depending on context, valuation method, and spatial scale, which limits their comparability and transferability across regions and LF types. To address this, uniform frameworks such as the UN's SEEA Ecosystem Accounting have been developed to support integration into national statistical systems. While these frameworks improve consistency, this report suggest that they may lose detail and complexity and oversimplify important socio-ecological complexities. Methodological simplification might serve national-level accounting purposes, but these approaches may be less suitable for assessments on a regional or landscape level, where such details matter.

Because no single method can fully capture all ES provided by LFs, a pragmatic approach is to combine biophysical data with existing valuation databases to derive aggregated or disaggregated values at the level of feature types. Monetary values can be powerful in policy and business settings by clarifying the benefits of LFs for decision-making, but they must be interpreted with caution and alongside with ecological and biogeochemical evidence. Monetisation reduces values of LFs to a highly condensed, one-dimensional form that hides multidimensionality and uncertainty. Looking ahead, progress in ES valuation and NCA for LFs will depend on better accounting for context dependence, non-market values, and the diversity of societal perspectives on nature's benefits.

## 6.4. Strategic recommendations

1. Prioritise climate resilient landscape features and choices of species and designs in (re)introduction.

Because the resilience of grassy, woody, and wet features under climate change is poorly understood, investments should prioritise species and designs adapted to projected local climate conditions. This includes selecting tree and shrub species adapted to projected climatic conditions, designing wet features that can cope with fluctuating water levels, and ensuring that new plantings are not placed in areas where water scarcity is expected to intensify. Choosing sown seed mixes and planted trees with redundancy in functioning instead of focusing on minimal species numbers may also be necessary.

2. Enhance landscape features quality and persistence as a core climate resilience strategy.

High biodiversity provides functional redundancy, making ecosystem services more stable under stress. Policies should focus not only on the introduction of new LFs but also improving habitat quality of existing LFs, increasing their structural diversity, and reducing disturbances around LFs. Planting short-term LFs (e.g., flower strips) should be seen as an additional strategy and not replacement of the permanent LFs in agricultural landscapes. Improving landscape connectivity through LFs is also important.

3. Address trade-offs between ecosystem services, ecosystem services and production, and between species groups explicitly in planning and policy.

Some mixed effects of LFs on different biodiversity groups and ES emphasize that the outcomes of establishing LFs on agricultural land are inherently conditional and are linked to spatial design, ecological needs of species, and land-use context. This highlights the importance of context-specific implementation of LFs to maximize benefits and minimize trade-offs. Guidance should be provided to farmers on choosing LFs that enhance biodiversity, most relevant ES to their agricultural systems (e.g., sufficient pollination for crops known to respond well to the addition of LFs) and multiple ES with minimal impact on production. NCM could be a powerful tool to identify and visualise trade-offs and provide a solution space optimised for multiple ES.

Because tree planting can increase local water stress through high evapotranspiration and negatively affect conservation priority species dependent on open landscapes, strategic planning should assess water availability, crop needs, local hydrology and vulnerable species before promoting large-scale tree planting in intensive agricultural areas.

4. Consider multifunctionality of landscape features and their combinations.

Though many LFs have a specific primary reason of being established, optimising delivery of multiple ES at farm and societal benefits is a cost-effective strategy, especially in intensive production landscapes with limited land resources and high opportunity costs.

5. Use natural capital modelling and valuation cautiously and at the right spatial scale.

Monetary valuation and Natural Capital Modelling can support high-level planning and make benefits visible to decision-makers. Their use is most appropriate at national or regional scales. At landscape and farm scale, more detailed ecological evidence is needed to avoid oversimplification. At the landscape feature level, combining biophysical data with existing valuation databases may be needed. Such approaches need to be transparent on the assumptions, simplifications and uncertainty.

6. Integrate value pluralism and non-market benefits into policy frameworks.

There are no easy approaches to monetise many benefits of landscape features, such as cultural identity, aesthetics, resilience and ecological integrity. Policies should therefore incorporate qualitative and participatory approaches to ensure these values are recognised alongside economic metrics.

7. Use of voluntary markets.

Explore possibilities of using novel mechanisms of value-transfer, such as carbon markets and nature markets, to mobilise private finance into emissions reductions and removals and nature restoration. This requires the adoption of the highest integrity approaches to substantiate respective claims.

8. Support farmers with guidance and incentives tailored to local conditions and specific to a region and agricultural system.

Given the context-dependence of both benefits and risks, advisory services and incentive schemes should be region-specific. Farmers need clear, practical guidance on which LFs are most effective under specific local condition in climate, soil, and hydrology, as well production systems. It is important not to overstate private benefits when most of them accrue on a societal level.

9. Integrate landscape features establishment and maintenance into CAP.

The new CAP instruments need to be designed in a way that they effectively support establishment and maintenance of landscape features, given their evidence-based efficiency. Preservation of the existing LF needs to be part of the baseline conditions of the Statutory Management Requirements. Payments under Agri-environmental and climate actions need to focus on climate-resilient designs and species according to projected local climate and water conditions. Payment schemes need to reward not only the presence of features, but also on their ecological quality, connectivity, and long-term maintenance. Non-productive investments can be used more widely than currently to establish new permanent LFs such as hedges or trees. Robust indicators need to be in place track to the progress on establishment and reintroduction of LFs in agricultural landscapes across the EU.

10. Strengthen the alignment with the EU Bird- and Habitat Directives.

In the light of the EU Bird- and Habitat Directives, landscape features should be matched to the existing landscapes and target habitats and species of conservation concern. Although most LFs have positive effects on general biodiversity, woody LFs can also have negative effects on target species, such as grassland and farmland birds.

11. Incentivize the implementation of landscape features on landscape level.

Coordination at the landscape and watershed levels is needed to improve strategic placement of LFs where they are most effective for multiple ES biodiversity and are cost-efficient, instead of targeting measures on the farm level. Coordination and area-based planning may be especially critical for LFs with considerable opportunity cost of land, investment costs or maintenance by specialised machinery, as well as LFs that buffer watercourses, affect hydrology and connectivity with natural and restored sites to enhance regulating ES and biodiversity at the landscape and regional scales. This could be achieved with a special bonus for collaborative implementation.

## 6.5. Knowledge gaps and research needs

This report lists specific knowledge gaps in respective chapters above. More generic knowledge gaps for future research include the following:

- A better understanding of trade-offs between different ES and research focusing on multiple ES.
- An increase of long-term studies accounting for the longevity and age of LFs.
- More focused research on pollination outcomes. Most of the research on pollinators and pollination deals with pollinator conservation rather than pollination services or yield, leaving knowledge gaps in explicit functional outcomes. Similar gaps exist regarding pest control studies, which rarely measure crop damage or yield directly, limiting assessments of effectiveness.
- Estimates of the economic values of ES of relevance to landowners that have LFs.
- A better understanding of context-dependency (including type of agricultural system and scale of implementation) in biodiversity and ES delivery, which is important for tailored design recommendations.
- An improved understanding of values of LFs for climate change adaptation and resilience, and structural resilience of LFs to climate change stressors.
- A methodological development of valuation at the scales of relevance to LFs.
- An improved reporting of metadata in scientific research to strengthen the consistency and sufficiency of existing scientific knowledge in relation to stakeholder-relevant, evidence-based syntheses.

## 7. Acknowledgements

The authors acknowledge valuable advice of the project Advisory Board members Tatiana Nemcova and Maria Luisa Paracchini, administrative support of Marta Neto, team of Pensoft for illustrations, all project partners for inspiration and fruitful ideas. Our work builds on all the work from thousands of researchers before us, which is greatly acknowledged.

## 8. References


Affek, A.N., Regulska, E., Kołaczkowska, E., et al. (2021). Pollination potential of riparian hardwood forests: A multifaceted field-based assessment in the Vistula Valley, Poland. Forests, 12(7), 907. https://doi.org/10.3390/f12070907

Albaladejo-García, J.A., Martínez-García, V., Martínez-Paz, J.M., et al. (2025). Gaining insight into best management practices for climate change impact abatement on agroecosystem services and disservices. Journal of Environmental Management, 384, 125629. https://doi.org/10.1016/j.jenvman.2025.125629

Albrecht, M., Kleijn, D., Williams, N.M., et al. (2020). The effectiveness of flower strips and hedgerows on pest control, pollination services and crop yield: A quantitative synthesis. Ecology Letters, 23(10), 1488–1498. https://doi.org/10.1111/ele.13576

Alexandridis, N., Marion, G., Chaplin-Kramer, R., Dainese, M., Ekroos, J., Grab, H., ... & Clough, Y. (2021). Models of natural pest control: Towards predictions across agricultural landscapes. Biological control, 163, 104761.

Anderson, R., Bayer, P.E., Edwards, D. (2020). Climate change and the need for agricultural adaptation. Current Opinion in Plant Biology, 56, 197–202. https://doi.org/10.1016/j.pbi.2019.12.006

Anwar, M.R., Liu, D.L., Macadam, I., et al. (2013). Adapting agriculture to climate change: A review. Theoretical and Applied Climatology, 113(1), 225–245. https://doi.org/10.1007/s00704-012-0780-1

ARIES User Guide (2026). Available at: https://aries.integratedmodelling.org/aries-for-seea-user-guide/

Assandri, G., Bogliani, G., Pedrini, P., et al. (2017). Assessing common birds' ecological requirements to address nature conservation in permanent crops: Lessons from Italian vineyards. Journal of Environmental Management, 191, 145–154. https://doi.org/10.1016/j.jenvman.2016.12.071

Assandri, G., Bogliani, G., Pedrini, P., et al. (2018). Beautiful agricultural landscapes promote cultural ecosystem services and biodiversity conservation. Agriculture, Ecosystems & Environment, 256, 200–210. https://doi.org/10.1016/j.agee.2018.01.012

Bai, J., Qin, Y., Zhao, J., et al. (2025). Investigation of agricultural nutrient removal by ecological ditches using meta-analysis. Agriculture, Ecosystems & Environment, 380, 109401. https://doi.org/10.1016/j.agee.2024.109401

Bartkowski, B. (2017). Economic valuation of biodiversity: An interdisciplinary conceptual perspective. Routledge.

Booth, A., Sutton, A., Papaioannou, D. (2016). Systematic approaches to a successful literature review (2nd ed.). SAGE Publications.

Bullock, J.M., McCracken, M.E., Bowes, M.J., et al. (2021). Does agri-environmental management enhance biodiversity and multiple ecosystem services? A farm-scale experiment. Agriculture, Ecosystems & Environment, 320, 107582. https://doi.org/10.1016/j.agee.2021.107582

Campi, P., Palumbo, A.D., Mastrorilli, M. (2009). Effects of tree windbreak on microclimate and wheat productivity in a Mediterranean environment. European Journal of Agronomy, 30(3), 220–227. https://doi.org/10.1016/j.eja.2008.10.004

Carstensen, M.V., Hashemi, F., Hoffmann, C.C., et al. (2020). Efficiency of mitigation measures targeting nutrient losses from agricultural drainage systems: A review. Ambio, 49(11), 1820–1837. https://doi.org/10.1007/s13280-020-01345-5

Chandra, A., McNamara, K.E., Dargusch, P. (2018). Climate-smart agriculture: Perspectives and framings. Climate Policy, 18(4), 526–541. https://doi.org/10.1080/14693062.2017.1316968

Chen, D., Wei, W., Chen, L. (2017). Effects of terracing practices on water erosion control in China: A meta-analysis. Earth-Science Reviews, 173, 109–121. https://doi.org/10.1016/j.earscirev.2017.08.007

Chen, D., Wei, W., Chen, L. (2020). How can terracing impact on soil moisture variation in China? Agricultural Water Management, 227, 105849. https://doi.org/10.1016/j.agwat.2019.105849

Christen, B., Dalgaard, T. (2013). Buffers for biomass production in temperate European agriculture: A review and synthesis. Biomass and Bioenergy, 55, 53–67. https://doi.org/10.1016/j.biombioe.2012.09.053

Colbach, N., Cordeau, S., Garrido, A., et al. (2018). Land sharing vs land sparing: Reconciling crop production and biodiversity. Agriculture, Ecosystems & Environment, 251, 203–217. https://doi.org/10.1016/j.agee.2017.09.005

Costanza, R. (2020). Valuing natural capital and ecosystem services toward efficiency, fairness, and sustainability. Ecosystem Services, 43, 101096. https://doi.org/10.1016/j.ecoser.2020.101096

Crowther, L.I., Wilson, K., Wilby, A. (2023). The impact of field margins on biological pest control: A meta-analysis. BioControl, 68(4), 387–396. https://doi.or1007/s10526-023-10205-6

Czúcz, B., Baruth, B., Terres, J., et al. (2022). Classification and quantification of landscape features in agricultural land across the EU. Publications Office of the European Union. https://doi.org/10.2760/59418

Czúcz, B., Baruth, B., Angileri, V., et al. (2022). Landscape features in the EU Member States: A review of existing data and approaches. European Commission, Joint Research Centre.

de Knegt, B., Biersteker, L., van Eupen, M., et al. (2022). Natural capital model. WOT Natuur & Milieu. https://doi.org/10.18174/583796

De Valck, J., Jarvis, D., Coggan, A., et al. (2023). Valuing ecosystem services in complex coastal settings: An extended ecosystem accounting framework for improved decision-making. Marine Policy, 155, 105761. https://doi.org/10.1016/j.marpol.2023.105761

De Valck, J., Landuyt, D., Broekx, S., et al. (2017). Outdoor recreation in various landscapes: Which site characteristics really matter? Land Use Policy, 65, 186–197. https://doi.org/10.1016/j.landusepol.2017.04.009

Devátý, J., Dostál, T., Hösl, R., et al. (2019). Effects of historical land use and land pattern changes on soil erosion: Case studies from Lower Austria and Central Bohemia. Land Use Policy, 82, 674–685.

Dicks, L.V., Ashpole, J.E., Dänhardt, J., et al. (2020). Farmland conservation. In: Sutherland, W.J., Dicks, L.V., Petrovan, S.O., et al. (eds.), What works in conservation 2020. Open Book Publishers.

Dollinger, J., Dagès, C., Bailly, J.S., et al. (2015). Managing ditches for agroecological engineering of landscape: A review. Agronomy for Sustainable Development, 35(3), 999–1020. https://doi.org/10.1007/s13593-015-0301-6 [1](https://sci-hub.st/10.1007/s13593-015-0301-6)

Donkersley, P., Witchalls, S., Bloom, E.H. (2023). A little does a lot: Can small-scale planting for pollinators make a difference? Agriculture, Ecosystems & Environment, 343, 108254. https://doi.org/10.1016/j.agee.2022.108254

Dorioz, J.M., Wang, D., Poulenard, J., et al. (2006). The effect of grass buffer strips on phosphorus dynamics. Agriculture, Ecosystems & Environment, 117(1), 4–21. https://doi.org/10.1016/j.agee.2006.03.029

Drexler, S., Gensior, A., Don, A. (2021). Carbon sequestration in hedgerow biomass and soil. Regional Environmental Change, 21(3), 74. https://doi.org/10.1007/s10113-021-01798-8

Droste, N., Bartkowski, B. (2018). Ecosystem service valuation for national accounting. Environmental and Resource Economics, 71(1), 205–215. https://doi.org/10.1007/s10640-017-0146-3

Elek, Z., Růžičková, J., Ádám, R., et al. (2020). Mixed effects of ecological intensification on pest control providers. PeerJ, 8, e8746. https://doi.org/10.7717/peerj.8746

England, J.R., O'Grady, A.P., Fleming, A., et al. (2020). Trees on farms to support natural capital. Science of the Total Environment, 704, 135345. https://doi.org/10.1016/j.scitotenv.2019.135345

Englund, O., Börjesson, P., Mola-Yudego, B., et al. (2021). Strategic deployment of riparian buffers and windbreaks in Europe. Communications Earth & Environment, 2, 176. https://doi.org/10.1038/s43247-021-00247-y

European Commission, Eurostat (n.d.). Landscape features in agricultural land. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Landscape_features_in_agricultural_land

Faccioli, M., McVittie, A., Glenk, K., et al. (2018). Natural capital accounts: Review of available data and accounting options. The James Hutton Institute.

Ferrarini, A., Serra, P., Almagro, M., et al. (2017). Multiple ecosystem services provision and biomass logistics management in bioenergy buffers. Renewable and Sustainable Energy Reviews, 73, 277–290. https://doi.org/10.1016/j.rser.2017.01.052

Ghofrani, Z., Sposito, V., Faggian, R. (2017). A comprehensive review of blue-green infrastructure concepts. International Journal of Environment and Sustainability, 6(1), 15–36.

Gómez-Baggethun, E., Barton, D.N., Berry, P., et al. (2016). Concepts and methods in ecosystem services valuation. In: Potschin, M., Haines-Young, R., Fish, R., et al. (eds.), Routledge handbook of ecosystem services. Routledge. https://doi.org/10.4324/9781315775302-9

Haaland, C., Naisbit, R.E., Bersier, L.F. (2011). Sown wildflower strips for insect conservation. Insect Conservation and Diversity, 4(1), 60–80. https://doi.org/10.1111/j.1752-4598.2010.00098.x

Haddaway, N.R., Brown, C., Eales, J., et al. (2018). The multifunctional roles of vegetated strips. Environmental Evidence, 7, 14. https://doi.org/10.1186/s13750-018-0126-2

Häfner, K., Zasada, I., van Zanten, B.T., et al. (2018). Assessing landscape preferences in agricultural regions. Landscape Research, 43(6), 846–861. https://doi.org/10.1080/01426397.2017.1386289

Haines-Young, R., Potschin, M. (2023). Common International Classification of Ecosystem Services (CICES) V5.2.

Hallberg, L., Hallin, S., Djodjic, F., et al. (2024). Trade-offs between nitrogen and phosphorus removal. Water Research, 258, 121770. https://doi.org/10.1016/j.watres.2024.121770

Hatan, S., Fleischer, A., Tchetchik, A. (2021). Economic valuation of cultural ecosystem services. Ecological Economics, 184, 107005. https://doi.org/10.1016/j.ecolecon.2021.107005

Hein, L., Bagstad, K.J., Obst, C., et al. (2020). Progress in natural capital accounting for ecosystems. Science, 367(6477), 514–515. https://doi.org/10.1126/science.aaz8901 [3](https://research.wur.nl/en/publications/progress-in-natural-capital-accounting-for-ecosystems/)

Hernández-Morcillo, M., Burgess, P., Mirck, J., et al. (2018). Agroforestry-based solutions for climate change mitigation. Environmental Science & Policy, 80, 44–52. https://doi.org/10.1016/j.envsci.2017.11.016

Howden, S.M., Soussana, J.F., Tubiello, F.N., et al. (2007). Adapting agriculture to climate change. Proceedings of the National Academy of Sciences, 104(50), 19691–19696. https://doi.org/10.1073/pnas.0701890104

Immerzeel, B., Brander, L., Grammatikopoulou, I., et al. (2025). SELINA D5.3.

IPCC (2014). Summary for policymakers. In: Climate change 2014: Impacts, adaptation, and vulnerability. Cambridge University Press.

IPCC (2021). Annex VII: Glossary. In: Climate change 2021: The physical science basis. Cambridge University Press.

Jachowicz, N., Sigsgaard, L. (2025). Highly diverse flower strips promote natural enemies more in annual field crops: A review and meta-analysis. Agriculture, Ecosystems & Environment, 381, 109412. https://doi.org/10.1016/j.agee.2024.109412

Jia, L., Zhao, W., Fu, B., et al. (2019). Effects of minimum soil disturbance practices on controlling water erosion in China's slope farmland: A meta-analysis. Land Degradation & Development, 30(6), 706–716. https://doi.org/10.1002/ldr.3258

Jobbágy, E.G., Jackson, R.B. (2004). Groundwater use and salinization with grassland afforestation. Global Change Biology, 10(8), 1299–1312. https://doi.org/10.1111/j.1365-2486.2004.00806.x

Joint Research Centre, Schievano, A., Bosco, S., et al. (2025). The JRC farming practices evidence library. Publications Office of the European Union. https://doi.org/10.2760/9473570

Karki, U., Goodman, M.S. (2015). Microclimatic differences between mature loblolly pine silvopasture and open pasture. Agroforestry Systems, 89(2), 319–325. https://doi.org/10.1007/s10457-014-9773-5

Koch, T., Deumlich, D., Chifflard, P., et al. (2023). Using model simulation to evaluate soil loss potential in diversified agricultural landscapes. European Journal of Soil Science, 74(1), e13332. https://doi.org/10.1111/ejss.13332

Kruit, J., Lof, M., Roelofsen, H., et al. (2025). De bijdrage van agroforestry en landschapselementen aan water-, klimaat- en natuurdoelen. Wageningen Environmental Research. https://doi.org/10.18174/679713

Kuntzman, J., Brom, J. (2024). From fields to microclimate: Assessing the influence of agricultural landscape structure. Remote Sensing, 17(1), 6. https://doi.org/10.3390/rs17010006

Keränen, F., Selin Norén, I., Ekroos, J., Lomba, A., Moreira, F., Herzon, I. (2026). Literature review data from: LAFERIA D2.1 Potential of landscape features to benefit biodiversity and ecosystem services [Dataset]. Zenodo. https://doi.org/10.5281/zenodo.20813701

Land, M., Granéli, W., Grimvall, A., et al. (2016). How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? Environmental Evidence, 5, 9. https://doi.org/10.1186/s13750-016-0060-0

Länsstyrelsen Kalmar län. (2017). *Naturvärdesbedömning av kustnära miljöer i Kalmar*

Länsstyrelserna. (2019). *Ekosystemtjänster som skapats av projektet LIFE Coast Benefit* [Rapport]. Länsstyrelsen Östergötland.  Ramboll et al. (2019). *Ecosystem services linked to wetlands and floodplains*

Länsstyrelsen Kalmar län. (n.d.). *Wetlands in Kalmar County* le Clech, S., van Bussel, L.G.J., Lof, M.E., et al. (2024). Effects of linear landscape elements on ecosystem services. Ecosystem Services, 67, 101616. https://doi.org/10.1016/j.ecoser.2024.101616

Lemos, A.B. (2017) Evaluating Ecosystem Services trade-offs due to land use changes: Transition to an irrigated agriculture landscape. MSc Thesis. University of Lisbon. 63.

Li, J., Yu, M., Wang, W., et al. (2026). Divergent responses of evapotranspiration and water yield. Journal of Forestry Research, 37(1), 71. https://doi.org/10.1007/s11676-026-02011-4

Lind, L., Hasselquist, E.M., Laudon, H. (2019). Towards ecologically functional riparian zones. Journal of Environmental Management, 249, 109391. https://doi.org/10.1016/j.jenvman.2019.109391

Liu, X., Zhang, X., Zhang, M. (2008). Factors influencing vegetated buffers. Journal of Environmental Quality, 37(5), 1667–1674. https://doi.org/10.2134/jeq2007.0437

Lowe, E.B., Groves, R., Gratton, C. (2021). Field-edge flower plantings and pollinator conservation. Agriculture, Ecosystems & Environment, 310, 107290. https://doi.org/10.1016/j.agee.2020.107290

Lyu, C., Li, X., Yuan, P., et al. (2021). Nitrogen retention in riparian zones. Journal of Cleaner Production, 315, 128143. https://doi.org/10.1016/j.jclepro.2021.128143

Maetens, W., Poesen, J., Vanmaercke, M. (2012). Soil conservation techniques in Europe. Earth-Science Reviews, 115(1–2), 21–36. https://doi.org/10.1016/j.earscirev.2012.08.003

Masiero, M., Bottaro, G., Righetti, C., et al. (2024). Riparian forests as nature-based solutions. Forests, 15(5), 760. https://doi.org/10.3390/f15050760

Matzek, V., Lewis, D., O'Geen, A., et al. (2020). Carbon stocks from riparian restoration. Carbon Balance and Management, 15(1), 16. https://doi.org/10.1186/s13021-020-00147-9

Mayer, P.M., Reynolds, S.K., McCutchen, M.D., et al. (2007). Nitrogen removal in riparian buffers. Journal of Environmental Quality, 36(4), 1172–1180. https://doi.org/10.2134/jeq2006.0462

Millennium Ecosystem Assessment (2005). Ecosystems and human well-being: Synthesis. Island Press.

Mittler, R., Karlova, R., Bassham, D.C., et al. (2025). Crops under stress. Philosophical Transactions of the Royal Society B, 380(1927). https://doi.org/10.1098/rstb.2024.0228

Müller-Bloch, C., Kranz, J. (2015). Identifying research gaps in qualitative reviews. ICIS Conference Proceedings.

Natural Capital Alliance (2026). InVEST 3.18.0. https://doi.org/10.60793/natcap-invest-3.18.0

NCAVES, MAIA (2022). Monetary valuation of ecosystem services and assets. United Nations. https://seea.un.org/sites/seea.un.org/files/techreportvaluationv15_final_21072022.pdf

Obst, C., Hein, L., Edens, B. (2016). Valuation of ecosystem assets. Environmental and Resource Economics, 64(1), 1–23. https://doi.org/10.1007/s10640-015-9921-1

Oteros-Rozas, E., Martín-López, B., Fagerholm, N., et al. (2018). Social media and ecosystem services. Ecological Indicators, 94, 74–86. https://doi.org/10.1016/j.ecolind.2017.02.009

Panozzo, A., Quataert, P., De Swaef, T., et al. (2026). Trees and intercrop yield. Agricultural Systems, 232, 104578. https://doi.org/10.1016/j.agsy.2025.104578

Pascual, U., Muradian, R., Brander, L., et al. (2012). Valuing ecosystem services and biodiversity. In: The economics of ecosystems and biodiversity. Routledge.

Pérez-Sánchez, A.J., Schröder, B., Dauber, J., et al. (2023). Flower strip effectiveness for pollinating insects in agricultural landscapes depends on habitat quality. Ecological Solutions and Evidence, 4(3), e12261. https://doi.org/10.1002/2688-8319.12261

Plottu, E., Plottu, B. (2007). The concept of total economic value of environment. Ecological Economics, 61(1), 52–61. https://doi.org/10.1016/j.ecolecon.2006.09.027

Potschin, M.B., Haines-Young, R.H. (2011). Ecosystem services: Exploring a geographical perspective. Progress in Physical Geography, 35(5), 575–594. https://doi.org/10.1177/0309133311423172

Powell, J.R., Lake, J., Berry, R., et al. (2020). Heritage, natural capital and ecosystem services: Boundaries and linear landscape features. University of Gloucestershire.

Rambonilaza, M., Dachary-Bernard, J. (2007). Land-use planning and public preferences. Landscape and Urban Planning, 83(4), 318–326. https://doi.org/10.1016/j.landurbplan.2007.05.013

Rau, A.L., Burkhardt, V., Dorninger, C., et al. (2020). Temporal patterns in ecosystem services research. Ambio, 49(8), 1377–1393. https://doi.org/10.1007/s13280-019-01292-w

Rivest, D., Paquette, A., Moreno, G., et al. (2013). Influence of scattered trees on pasture yield. Agriculture, Ecosystems & Environment, 165, 74–79. https://doi.org/10.1016/j.agee.2012.12.010

Rodríguez-Entrena, M., Colombo, S., Arriaza, M. (2017). Olive grove landscapes and the rural economy. Land Use Policy, 65, 164–175. https://doi.org/10.1016/j.landusepol.2017.03.017

Ruggiero, A., Céréghino, R., Figuerola, J., et al. (2008). Farm ponds and aquatic insect biodiversity. Comptes Rendus Biologies, 331(4), 298–308. https://doi.org/10.1016/j.crvi.2008.01.009

Ruijs, A., Vardon, M., Bass, S., et al. (2019). Natural capital accounting for better policy. Ambio, 48(7), 714–725. https://doi.org/10.1007/s13280-018-1107-y

Santos, M., Garcês, C., Ferreira, A., et al. (2023). Side effects of European eco schemes on endangered species conservation. Ecological Indicators, 148, 110155. https://doi.org/10.1016/j.ecolind.2023.110155

Sayadi, S., González-Roa, M.C., Calatrava-Requena, J. (2009). Public preferences for landscape features. Land Use Policy, 26(2), 334–344. https://doi.org/10.1016/j.landusepol.2008.04.003

Schaan, L.N., Finch, E.A., Wartenberg, A.C., et al. (2025). Mapping and prioritising landscape feature restoration. Land Use Policy, 154, 107531. https://doi.org/10.1016/j.landusepol.2024.107531

Schaak, H., Musshoff, O. (2020). Public preferences for pasture landscapes. Land Use Policy, 91, 104371. https://doi.org/10.1016/j.landusepol.2019.104371

Scheper, J., Holzschuh, A., Kuussaari, M., et al. (2013). Environmental factors driving effectiveness of agri-environmental measures. Ecology Letters, 16(7), 912–920. https://doi.org/10.1111/ele.12128

Scherr, S.J., Shames, S., Friedman, R. (2012). Climate-smart landscapes. Agriculture & Food Security, 1, 12. https://doi.org/10.1186/2048-7010-1-12

Schievano, A., Perez-Soba, M., Bosco, S., et al. (2026). JRC farming practices dataset. European Commission. https://doi.org/10.2905/JRC.MH3RZJG

Schmidt, K., Martín-López, B., Phillips, P.M., et al. (2019). Landscape features in ecosystem service provision. Land Use Policy, 82, 353–366. https://doi.org/10.1016/j.landusepol.2018.12.022

Schoeneberger, M., Bentrup, G., De Gooijer, H., et al. (2012). Agroforestry for climate change mitigation. Journal of Soil and Water Conservation, 67(5), 128A–136A. https://doi.org/10.2489/jswc.67.5.128A

Scholtes, F. (2015). Umweltherrschaft und Freiheit: Naturbewertung im Anschluss an Amartya K. Sen. transcript Verlag.

Shackelford, G.E., Kelsey, R., Robertson, R.J., et al. (2017). Sustainable agriculture in Mediterranean climates. Conservation Evidence, University of Cambridge. Available from https://about.conservationevidence.com/

Shen, W., Li, S., Mi, M., et al. (2021). Ditches and ponds in nitrogen control. Agriculture, Ecosystems & Environment, 314, 107409. https://doi.org/10.1016/j.agee.2021.107409

Shen, Y.T., Hou, S.N., Hu, S.L., et al. (2024). Ecological ditch water purification. Ecological Engineering, 204, 107280. https://doi.org/10.1016/j.ecoleng.2024.107280

Skogsstyrelsen. (2019). *Ekosystemtjänster knutna till våtmarker i skogslandskapet*

Spash, C.L. (2008). Valuation of ecosystems. Environmental Values, 17(2), 259–284. https://doi.org/10.3197/096327108X303882

Spash, C.L. (2013). Ecological economics debate. Ecological Economics, 93, 351–362. https://doi.org/10.1016/j.ecolecon.2013.05.016

Speakman, D. (2018). Adaptation in marginal lands. Weather, Climate, and Society, 10(1), 121–136. https://doi.org/10.1175/WCAS-D-17-0064.1

Staccione, A., Broccoli, D., Mazzoli, P., et al. (2021). Natural water retention ponds. Journal of Environmental Management, 292, 112849. https://doi.org/10.1016/j.jenvman.2021.112849

Staggenborg, J., Anthes, N. (2022). Fallows and farmland birds. Conservation Letters, 15(4), e12904. https://doi.org/10.1111/conl.12904

Steenwerth, K.L., Hodson, A.K., Bloom, A.J., et al. (2014). Climate-smart agriculture research agenda. Agriculture & Food Security, 3, 11. https://doi.org/10.1186/2048-7010-3-11

Stehle, S., Elsaesser, D., Gregoire, C., et al. (2011). Pesticide mitigation by vegetated systems. Journal of Environmental Quality, 40(4), 1068–1080. https://doi.org/10.2134/jeq2010.0510

Sutcliffe, L.M.E., Batáry, P., Kormann, U., et al. (2014). Biodiversity in European farmland. Diversity and Distributions, 20(12), 1380–1390. https://doi.org/10.1111/ddi.12218

Tagliafierro, C., Boeri, M., Longo, A., et al. (2016). Landscape valuation methods. Ecological Economics, 127, 11–22. https://doi.org/10.1016/j.ecolecon.2016.03.022

Tagliafierro, C., Longo, A., Van Eetvelde, V., et al. (2013). Landscape economic valuation. Environmental Science & Policy, 32, 26–36. https://doi.org/10.1016/j.envsci.2012.12.001

Taylor, M. (2018). Climate-smart agriculture critique. Journal of Peasant Studies, 45(1), 89–107. https://doi.org/10.1080/03066150.2017.1312353

TEEB (2010). The economics of ecosystems and biodiversity. Earthscan.

Tsai, Y., Zabronsky, H.M., Zia, A., et al. (2022). Riparian buffers and phosphorus removal. Frontiers in Water, 4, 882560. https://doi.org/10.3389/frwa.2022.882560

UN (2024). System of environmental economic accounting. United Nations. https://seea.un.org/sites/seea.un.org/files/documents/EA/seea_ea_f124_web_12dec24.pdf

Valeri, S., Schmitz, M.F., Acosta-Gallo, B., et al. (2024). Landscape elements and habitat quality. Agriculture, Ecosystems & Environment, 375, 109199. https://doi.org/10.1016/j.agee.2024.109199

Valkama, E., Usva, K., Saarinen, M., et al. (2019). Nitrogen retention by buffer zones. Journal of Environmental Quality, 48(2), 270–279. https://doi.org/10.2134/jeq2018.03.0120

Van Berkel, D.B., Verburg, P.H. (2014). Cultural ecosystem services valuation. Ecological Indicators, 37, 163–174. https://doi.org/10.1016/j.ecolind.2012.06.025

Van Berkel, J., Bogaart, P., Driessen, C., et al. (2021). Natural capital accounting in the Netherlands. Wageningen University.

Van Buskirk, J., Willi, Y. (2004). Farmland biodiversity enhancement. Conservation Biology, 18(4), 987–994. https://doi.org/10.1111/j.1523-1739.2004.00359.x

Van Ruymbeke, K., Zanni, A.M., Bailey, A., et al. (2026). Valuing agricultural landscapes. Journal of Rural Studies, 123, 104037. https://doi.org/10.1016/j.jrurstud.2026.104037

Van Vooren, L., Reubens, B., Broekx, S., et al. (2017). Ecosystem services of agri-environmental measures. Agriculture, Ecosystems & Environment, 244, 116–127. https://doi.org/10.1016/j.agee.2017.04.015

Van Zanten, B.T., Zasada, I., Koetse, M.J., et al. (2016). Landscape aesthetics and ecosystem services. Ecosystem Services, 17, 87–98. https://doi.org/10.1016/j.ecoser.2015.11.011

Wang, J., Qi, Z., Wang, C. (2023). Phosphorus loss and crop yields. Agriculture, Ecosystems & Environment, 357, 108683. https://doi.org/10.1016/j.agee.2023.108683

Wei, W., Chen, D., Wang, L., et al. (2016). Global terracing synthesis. Earth-Science Reviews, 159, 388–403. https://doi.org/10.1016/j.earscirev.2016.06.010

Wersebeckmann, V., Burstedde, K., Leyer, I. (2024). Vineyard terracing and biodiversity. Applied Vegetation Science, 27(2), e12777. https://doi.org/10.1111/avsc.12777

Win, K.Z., Arriaza, M., Villanueva, A.J. (2026). Mediterranean landscapes valuation. Spanish Journal of Agricultural Research, 23(2), 21502. https://doi.org/10.5424/sjar/2025232-21502

Włodarczyk-Marciniak, R., Frankiewicz, P., Krauze, K. (2020). Socio-cultural valuation of landscapes. Journal of Rural Studies, 74, 190–200. https://doi.org/10.1016/j.jrurstud.2020.01.017

Xiong, M., Sun, R., Chen, L. (2018). Soil conservation effects. Science of the Total Environment, 645, 753–760. https://doi.org/10.1016/j.scitotenv.2018.07.124

Zamorano, J., Bartomeus, I., Grez, A.A., et al. (2020). Floral enhancements and pollinators. Insect Conservation and Diversity, 13(6), 519–531. https://doi.org/10.1111/icad.12454

Zgłobicki, W., Poesen, J., De Geeter, S., et al. (2021). Sunken lanes. Earth-Science Reviews, 221, 103757. https://doi.org/10.1016/j.earscirev.2021.103757

Zhang, X., Liu, X., Zhang, M., et al. (2010). Vegetated buffers and pollution. Journal of Environmental Quality, 39(1), 76–84. https://doi.org/10.2134/jeq2008.0496

Zheng, Y., Wang, H., Qin, Q., et al. (2020). Hedgerows and agricultural pollution. Environmental Science and Pollution Research, 27(20), 24831–24847. https://doi.org/10.1007/s11356-020-08988-7

Zimmermann, B., Kruber, S., Nendel, C., et al. (2024). Cooling potential of adaptation measures. Journal of Environmental Management, 366, 121595. https://doi.org/10.1016/j.jenvman.2024.121595

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

Zolyomi, A., Franklin, A., Smith, B., et al. (2023). Ecosystem services in policy. Earth System Governance, 16, 100178. https://doi.org/10.1016/j.esg.2023.100178.

# 9. Annexes

### Annex 1. Search approach to literature review

**Table A1:** Search terms used for literature review. See sections 2.1.2-2.1.4 for how these search terms were combined and used.

| Category | Type | Search terms |
|---|---|---|
| LFs | Hedgerow | "hedgerow*" OR "shelterbelt*" OR "living fence*" OR "woody border*" |
| LFs | Tree row | "tree row*" OR "tree line*" OR "row of trees" OR "tree strip*" |
| LFs | Individual trees | "individual tree*" OR "solitary tree*" OR "isolated tree*" OR "single tree*" OR "scattered tree*" |
| LFs | Groups of trees/shrub | "tree group*" OR "group* of trees" OR "clump of trees" OR "stand of trees" OR "woodlot*" OR "shrub group*" OR "group* of shrubs" OR "shrub patch*" OR "bush patch*" OR "tree cluster*" OR "shrub cluster*" OR "tree islet" OR "tree patch" OR "woody patch" OR "woody islet" |
| LFs | Small forests | "small forest*" OR "forest patch*" OR "forest islet*" OR "forest island*" OR "forest fragment*" OR "forest clump*" OR "mini forest*" OR "wooded patch*" OR "wooded islet*" OR "wooded fragment*" |
| LFs | Field margins | "field margin*" OR "field boundary*" OR "flower strip*" OR "wildflower strip*" OR "buffer strip*" OR "vegetation strip*" OR "field edge*" OR "conservation margin*" |
| LFs | Grass strips | "grass* strip*" OR "grass buffer*" OR "filter strip*" OR "vegetative strip*" OR "conservation strip*" OR "grass margin*" |
| LFs | Herbaceous parcels | "fallow*" OR "herbaceous parcel" OR "set-aside*" OR "set aside*" OR "idle land*" OR "uncultivated land*" OR "uncultivated field*" OR "resting field*" OR "out-of-production" OR "out of production" OR "non-productive land*" OR "abandoned farmland*" OR "retired farmland" OR "former cropland" OR "agricultural land abandonment" |
| LFs | Ditches | "ditch*" OR "field drain*" OR "drainage channel*" OR "small canal*" |
| LFs | Small wetlands | "small W/4 wetland*" OR "farm wetland*" OR "agricultural wetland*" OR "constructed wetland*" OR "man-made W/4 wetland*" OR "wetland patch*" |
| LFs | Small ponds | "small W/4 pond* OR "farm pond*" OR "agricultural pond*" OR "constructed pond*" OR "man-made W/4 pond*" OR "artificial pond*" OR "pond patch*" OR "waterhole*" OR "stock pond*" OR "ephemeral pond*" |

| | | |
|---|---|---|
| LFs | Riparian vegetation | "riparian W/4 vegetation* OR "riparian forest*" OR "riparian buffer*" OR "riparian zone*" OR "streamside vegetation*" OR "riverine vegetation*" OR "riparian woodland*" OR "riparian shrub*" OR "riparian strip*" OR "riparian corridor*" |
| LFs | Streams | "stream*" OR "brook*" OR "rivulet*" OR "headwater*" OR "tributar*" OR "creek*" OR "drainage channel*" OR "waterway*" OR "watercourse*" |
| LFs | Floodplains | "floodplain*" OR "flood plain*" OR "alluvial plain*" OR "river plain*" OR "fluvial plain*" OR "inundation zone*" OR "inundation area*" OR "overflow zone*" OR "overflow area*" OR "flood-prone zone*" OR "flood-prone area*" |
| LFs | Terraces | "terrace*" OR "agricultural terrace*" OR "terraced field*" OR "terraced slope*" OR "bench terrace*" OR "contour terrace*" OR "step terrace*" OR "terracing*" |
| LFs | Stonewalls | "dry-stone wall*" OR "dry stone wall*" OR "stone wall*" OR "earth wall*" OR "dry wall*" OR "dry-wall*" OR "rubble wall*" OR "stonewall*" OR "rock wall*" |
| LFs | Rock outcrops | "cairn*" OR "boulder*" OR "rock outcrop*" OR "rock formation*" OR "rocky outcrop*" OR "granite outcrop*" OR "stone outcrop*" OR "rocky hill*" OR "secular stone*" |
| LFs | Cultural features | "cultural feature*" OR "archaeological feature*" OR "heritage feature*" OR "sunken lane*" OR "hollow way*" OR "grave mound*" OR "burial mound*" OR "tumulus" OR "tumuli" OR "barrow*" OR "kurgan*" OR "dolmen*" OR "megalith*" |
| LFs | LFs in general | "landscape feature*" OR "landscape characteristic*" OR "green infrastructure*" OR "landscape connectivity" OR "landscape diversity" OR "landscape element*" OR "landscape fragment*" OR "landscape mosaic*" OR "landscape structure*" OR "landscape complexity" OR "landscape heterogeneity" OR "landscape composition" OR "landscape configuration" OR "nature-based feature*" OR "linear feature*" |
| ES | Animal production | "animal product*" OR "livestock product*" OR "meat product*" OR "dairy product*" OR "cattle" OR "sheep" OR "goat*" OR "poultry" |
| ES | Grassland production | "grassland product*" OR "grassland yield" OR "pasture product*" OR "pasture yield" OR "forage product*" OR "forage yield" OR "rangeland product*" OR "rangeland yield" OR "fodder product*" OR "fodder yield" OR "feed product*" OR "feed yield" OR "grazing product*" OR "grazing yield" |
| ES | Crop yield | "yield" OR "crop yield*" OR "agricultural yield*" OR "crop product*" |

| | | |
|---|---|---|
| ES | Biomass production | "biomass production" OR "biomass yield*" OR "biomass" OR "tree biomass" OR "woody biomass" OR "fuelwood" OR "fuel wood" OR "firewood" OR "bioenergy" OR "wood production" OR "wood yield*" OR "timber production" |
| ES | Pest and disease control | "pest*" OR "insect pest*" OR "disease*" OR "crop disease*" OR "plant pathogen*" OR "pest control" OR "biological control" |
| ES | Pollination | "pollinat*" OR "pollinator service*" |
| ES | Soil erosion control | "soil erosion" OR "land degradation" OR "soil loss" OR "erosion control" OR "sediment retention" OR "sediment removal" OR "sediment" |
| ES | Nutrient retention | "nutrient leaching" OR "nutrient runoff" OR "nutrient retention" OR "nutrient loss" OR "fertilizer loss" OR "nitrate leaching" OR "nitrate runoff" OR "nitrate loss" OR "nitrogen leaching" OR "nitrogen runoff" OR "nitrogen loss" OR "phosphorus leaching" OR "phosphorus runoff" OR "phosphorus loss" OR "N leaching" OR "N runoff" OR "N loss" OR "P leaching" OR "P runoff" OR "P loss" OR "nutrient pollution" |
| ES | Soil nutrient levels | "soil nutrient*" OR "soil fertility" OR "soil organic matter" OR "soil nitrogen" OR "soil phosphorus" OR "soil potassium" OR "Soil N" OR "Soil P" OR "Soil K" OR "total nitrogen" OR "total phosphorus" OR "TN" OR "TP" OR "available N" OR "available P" |
| ES | Soil water retention | "soil water retention" OR "soil moisture" OR "water infiltration" OR "water holding capacity" OR "WHC" OR "soil hydraulic retention" OR "soil water storage" OR "soil field capacity" OR "volumetric water content" OR "soil water content" OR "soil porosity" OR "soil infiltration rate" OR "soil hydraulic conductivity" OR "soil water potential" |
| ES | Water quality | "water quality" OR "water pollution" OR "nutrient pollution" OR "eutrophication" OR "freshwater quality" |
| ES | Water regulation | "water storage" OR "flood regulation" OR "flood mitigation" OR "water regulation" OR "water stock*" |
| ES | Carbon sequestration | "carbon sequestration" OR "carbon storage" OR "soil carbon" OR "carbon sink" OR "greenhouse gas mitigation" OR "soil C" OR "SOC" OR "TOC" OR "TC" OR "total carbon" OR "soil organic carbon" OR "total organic carbon" |
| ES | Biodiversity | "biodiversity" OR "wildlife" OR "species richness" OR "species diversity" OR "habitat" |
| ES | Cultural services | "cultural service*" OR "cultural ecosystem service*" OR "cultural benefit*" OR "cultural value*" OR "amenity service*" OR "place-based value*" OR "relational value*" OR "sense of place" OR |

| | |
|---|---|
| | "aesthetic value*" OR "aesthetic quality*" OR "spiritual value*" OR "recreational value*" OR "recreational quality*" OR "educational value*" OR "heritage value*" |
| Climate | "climate change" OR "global warming" OR "climatic change" OR "climate variability" OR "resilien*" OR "climate adaptation" OR "climate change adaptation" OR "adapt* strateg*" OR "climate mitigation" OR "climate change mitigation" |
| Geographical scope | "Europe*" OR "Mediterranean*" OR "temperate*" OR "boreal*" OR "subarctic" OR "Austria*" OR "Belgium*" OR "Belarus*" OR "Bosnia*" OR "Bulgaria*" OR "Croatia*" OR "Cyprus*" OR "Czech*" OR "Denmark*" OR "Estonia*" OR "Finland*" OR "France*" OR "Germany*" OR "Greece*" OR "Hungary*" OR "Iceland*" OR "Ireland*" OR "Italy*" OR "Kosovo*" OR "Latvia*" OR "Lithuania*" OR "Luxembourg*" OR "Malta*" OR "Monaco*" OR "Montenegro*" OR "Netherlands*" OR "North Macedonia*" OR "Norway*" OR "Poland*" OR "Portugal*" OR "Romania*" OR "San Marino*" OR "Serbia*" OR "Slovakia*" OR "Slovenia*" OR "Spain*" OR "Sweden*" OR "Switzerland*" OR "UK" OR "United Kingdom*" OR "England*" OR "Scotland*" OR "Wales*" |
| Agricultural context | "agric*" OR "cultiv*" OR "crop*" OR "farm*" |

## Annex 2. Annotated bibliography of grey literature

This annex lists and briefly summarises reports projects and there that are of relevance for LFs in agricultural systems and use of Natural Capital Accounting.

**NCAVES:**

Mainstreaming biodiversity and ecosystems into policy through natural capital accounts in five countries. UN (SEEA) project, partly funded by the European Union. The documentation from SEEA-EA provides very elaborate information on ecosystem accounting and monetary valuation of ES (UN, 2024, NCAVES & MAIA, 2022).

Website: seea.un.org/home/Natural-Capital-Accounting-Project

**SELINA:**

EU project with the aim to set news standards for international cooperation to promote ES and biodiversity conservation and enhance ecosystem conditions, related to European Biodiversity Strategy 2030 and the Green Deal.

Website: project-selina.eu

**MAIA:**

Mapping and assessment for integrated ecosystem accounting. Aims to mainstream natural capital and ecosystem accounting (NCA) in EU member states.

Website: maiaportal.eu

**INCA:**

To support the development of the SEEA EA, the European Commission set up the INCA project, to connect the EU policy objectives to the establishment of an international NCA initiative. Much of the information and publications are found on the INCA platform.

Website: ecosystem-accounts.jrc.ec.europa.eu

**PARTRIDGE**

This transnational partnership (researchers, farm advisors, conservationists, hunters, civil servants and farmers) is showcasing best practice models at 10 farmland demonstration sites in four North Sea Region nations, to show how new management solutions can improve biodiversity and ecosystem services by a minimum 30% by 2023, and how these can be transferred across all regions of the region and the EU.

Website:

**SHOWCASE:**

Seeks to showcase synergies between agriculture, biodiversity, and ecosystem services, helping farmers capitalize on native biodiversity. The project produced several relevant reports and publications on managing biodiversity in agricultural land, including through LF.
Website: https://showcase-project.eu

**Focus Group 'Enhancing the biodiversity on farmland through high-diversity landscape features'.** Landscape Features and Biodiversity | EU CAP Network

The report presents the findings of the EU CAP Network Focus Group 'Enhancing the biodiversity on farmland through high-diversity landscape features'. The Focus Group experts discussed how farmers can create and maintain (HDLF) that positively impact farmland biodiversity.

**Lawrence, A. & Hackett, M. (2014) Multifunctional Role of Field Margins in Arable Farming.** Report for the European Crop Protection Association (ECPA).

The report highlights that field margins offer substantial biodiversity and environmental benefits by creating semi-natural habitats and functioning as ecological corridors. They also reduce soil erosion, filter agricultural pollutants and improve water quality. It describes distinct benefits of different margin types (natural regeneration margins, sown grass mixes, wildflower strips, pollen-and-nectar mixes, and conservation headlands).

Available at: https://agriculture.basf.com/dam/jcr:6d6f8fe4-7ca7-3ed3-be64-2583d09bc49d/basf/agriculture/global/assets/en/Crop%20Protection/News/Publications/Field_Margins_Arable_Farming_V02.pdf

**Natural Water Retention Measures (NWRM)** .

The NWRM initiative, launched by the DG Environment, European Commission, has launched aims to develop sound and comprehensive European (web-based) knowledge on NWRM. The platform currently covers technical, environmental, socio-economic, governance and implementation aspects of NWRM, mobilizing existing practical experiences, studies and stakeholders' knowledge. It is publicly available and provides a structured catalogue of NWRM, a set of 120 case studies illustrating the implementation of NWRM in a wide set of context, 11 synthesis documents, a serious game to make the NWRM community and the key actors better understand constraints and opportunities of implementing NWRM, and communication material. The NWRM practical guide developed can support the design and implementation of NWRM in Europe. It will become open to new cases.

It also hosts a summary table of benefits of actions, including hedges and buffers, for relevant ES. https://www.nwrm.eu/sites/default/files/a-vs-es.png

Available at: https://www.nwrm.eu/.

**EASAC 2025. Opportunities in Nature Restoration.** ttps://doi.org/10.1553/EASAC_Commentaries_NRR_2025.

The report of the scientific expert body has multiple references to the role of LF. These concern, for example:

Prevention of flooding through Natural Water Retention Measures (NWRM) such as floodplain reconnection, wetlands, riparian buffers, farm ponds. These measures contribute to flatten flood peaks, reduce downstream damages and insurance losses, and recharge aquifers.

Prevention of landslides and erosion through re-establishing contour hedgerows, and forest restoration in critical zones (presumably also small forests).

## D2.1 Potential of landscape features to benefit biodiversity and ecosystem services

Mitigation of drought and heat impacts through, for example, increasing shaded riparian corridors.

**Annex 3. Detailed findings of the semi-structured literature review on economic valuation of ES in relation to LFs**

Table A2. Literature overview of valuation methods of ecosystem services provided by landscape features

| Study | Landscape Feature Type* | Ecosystem Service Category | Valuation Method | Key Findings |
|---|---|---|---|---|
| Oteros-Rozas et al., 2018 | Trees; Shrubs; Grasslands; Lawn; Wood-pasture; Water body; River/lake/pond; Sea; Coastal/beach/dunes; Mountain; Snow/Ice; Rocks; Livestock; Wild; Infrastructures | Cultural: Recreational, social, cultural heritage, spiritual | Content analysis of photographs from social media platforms; no monetary valuation | "Cultural heritage, social and spiritual values were particularly attached to landscapes with wood pastures and grasslands, as well as urban features and infrastructures, i.e. to more anthropogenic landscapes. A positive though weak relationship was found between landscape diversity and cultural ecosystem services diversity. Particularly wood-pastures and shrubs were more frequently portrayed in all study sites in comparison with their actual land cover." |
| Schmidt et al., 2019 | Water bodies (reservoirs, surface water); Bog / Moorland; Grassland; Arable land; Woodland; Heather; Cultural heritage features; Accessibility & recreational infrastructure (paths, car parks); Landscape structure (land cover diversity); Wildness (as biodiversity-related characteristic) | Cultural; Provisioning; Regulating | Participatory mapping (PPGIS), Kernel density analysis, Redundancy analysis (RDA); no monetary valuation | Water and bogs strongly underpin regulating services; Cultural services strongly linked to accessibility and landscape diversity; Mismatch between ES-providing landscape features and management priorities |
| Tagliafierro et al., 2013 | Natural systems; Woods; Urban; Lemon groves; Olive groves; Fruit orchards | Cultural ES (aesthetic, heritage, landscape appreciation) | Discrete Choice Experiment | Positive WTP for heterogeneity & openness; negative for urbanization & disturbance; overall WTP for preservation of landscapes and their characteristics |
| Van Berkel & Verburg, 2014 | Cultural buildings; Tree lines and hedgerows; Lakes, streams and | Cultural ES (recreation, aesthetic, heritage, | Contingent valuation (landscape | Positive WTP for maintaining traditional agricultural landscape structure |

| | | | | |
|---|---|---|---|---|
| | brooks; Wildlife Animals; Villages; Agricultural lands; Stone quarry; | spiritual, inspirational) | preservation scenarios) + Travel cost; Monetary valuation refers to overall landscape character rather than individual features | |
| Rodríguez-Entrena et al., 2017 | Green cover between olive trees; Stone walls; Woodland islets | Cultural ES (aesthetic and recreational) | Choice Experiment | Green cover most valued; Stone walls positively valued; Woodland islets not significant; WTP up to ~6€ per visit (feature-dependent); Landscape improvements trigger multiplier effects in rural economy |
| Häfner et al., 2018 | Linear elements (hedgerows, tree rows); Point elements (trees, kettle holes); Crop diversity; | Cultural ES (aesthetic value) | Choice Experiment; only preferences, no monetary valuation | Strong preference for point elements and high landscape complexity; ~30% preferred low-complexity landscapes; |
| Van Ruymbeke et al., 2026 | Landscape diversity (levels: low/medium/high); Crop dividers (wild/clear); | Cultural ES (aesthetic value) | Discrete Choice Experiment | Higher WTP for increased landscape diversity and visible stewardship; Heterogeneity in preferences; Renewable energy elements partly accepted |
| Win et al., 2026 | Tree alignment; Sloping terrain; Spatial depth / visibility of distant elements; Mature trees; Tree density; Other tree species / tree diversity; Bushes; Rocks; Animals; Inert ground cover (straw, mulch, residues); Weak ground–tree color contrast (vegetation cover/soft contrast); Negative human-made elements (e.g., powerlines, roads, warehouses) | Cultural ES (aesthetic value) | Best-worst scaling choice experiments; Contingent valuation | “Preferences regarding the visual quality of agricultural landscapes are significantly influenced by: (i) farm management... (ii) biotic elements, such as lush green vegetation, dense trees... and diverse tree species... (iii) the alignment of trees... (iv) ephemeral elements... and (v) environmental attitudes and use values...” |

| | | | | |
|---|---|---|---|---|
| Włodarczyk-Marciniak et al., 2020 | Dirt roads; Balks; Small pools; Solitary trees; Groups of trees; Mid-field woodlots; | Provisioning; regulating; supporting; cultural | Socio-cultural valuation using 3-point importance scale (“important / neutral / unimportant”);<br>No monetary valuation | Production-related elements (fields, forests, meadows) valued highest; mosaic elements (woodlots, balks, pools) valued lower; valuation primarily utilitarian |
| Rambonilaza & Dachary-Bernard, 2007 | Hedgerows; Scrubland | Not explicitly specified but social, cultural and aesthetic values are mentioned, thus indicating cultural ES; Use and non-use values | Choice Experiment | Divergence between residents and tourists; positive WTP for landscape attributes; rural landscape has economic value |
| Tagliafierro et al., 2016 | Elements of heritage; Disturbing elements; Openness | Cultural (aesthetic / landscape preservation; not explicitly ES-framed) | Hybrid stated-preference survey (contingent valuation + attribute-based method) | Positive WTP for openness and natural vegetation; negative WTP for fragmentation, heterogeneity, lemon orchards and farmer stewardship; heterogeneous preferences across respondents |
| Schaak & Musshoff, 2020 | Point elements (trees, individual bushes); Linear elements (hedges, groups of bushes);<br>Landscape structure (parcelling) | Cultural ES (aesthetic landscape preferences) | Discrete choice experiment with cost attribute → WTP estimation | Strong preference for structured landscapes;<br>“point elements were more important for the landscape preference than linear elements” |
| Van Zanten et al., 2016 | Linear elements (hedgerows, tree lines); Point elements (trees, forest patches); Agricultural land-use diversity; P | Cultural ES (aesthetic, recreational) | Discrete choice experiment without cost attribute; no monetary valuation | Hedgerows and tree lines highly valued in the Netherlands;<br>point elements and land-use diversity more important in Germany;<br>strong context dependency of preferences;<br>landscape homogenisation reduces aesthetic and recreational value |
| De Valck et al., 2017 | Water elements (ponds, lakes, rivers, canals); Trails; Recreational facilities; | Cultural ES (outdoor recreation) | Distance-based discrete choice experiment without | Strong preferences for water elements, tranquillity and landscape diversity; |

| | | | | |
|---|---|---|---|---|
| | Landscape structure (openness, naturalness, diversity) | | cost attribute (implicit valuation using travel distance; compensating variation); no monetary valuation | heterogeneous preferences across user groups; production landscapes can substitute natural landscapes for recreation; spatial mapping reveals scarcity of high-quality sites |