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Quantifying the Impact of Sustainable Farming Practices on Environment and Climate

Greenhouse gas emissions, carbon sequestration and nutrient loss data from meta-analysis

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Contents

Abstract

Agriculture plays a pivotal role in meeting global food demands and maintaining socio-economic stability. However, some agricultural practices may negatively contribute to environmental concerns such as greenhouse gas emissions, nutrient losses, and biodiversity decline, impacting climate change, water quality and quantity, and ecosystem functioning. In response, the European Commission (EC) fosters the integration of environmental sustainability within agricultural policy frameworks, including the Common Agricultural Policy (CAP).

In this context, identifying sustainable farming practices is crucial for achieving the EU's sustainability objectives, in particular to assess the environmental and climate performance of the agricultural sector. To support this, we present in this report a comprehensive collection of coefficients quantifying the environmental impacts of farming practices. Focusing on greenhouse gas emissions, carbon sequestration, and nutrient losses, these coefficients are sourced from scientific articles, primarily meta-analyses, which have been reviewed through systematic literature analysis. They offer valuable insights into the effects of different agricultural management options, directly relevant to assess the likely impacts of CAP interventions on the environment and the climate.

The report presents a collection of over 100 tables containing quantitative coefficients that assess the environmental impacts of 35 farming practices (ranging from single farming practices to cropping systems or conservation and restoration actions), offering clear links to specific scholar references, along with detailed contextual information.

This work is part of the iMAP4agri project, commissioned by the EC's Directorate-General for Agriculture and Rural Development (DG AGRI) to the Joint Research Centre (JRC), drawing on experience from previous CAP periods and materials developed for the current policy cycle. By synthesizing existing meta-analyses and applying a systematic review framework, we ensure a robust and transparent approach that can be implemented within the time and data availability constraints of policymaking processes.

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1 Introduction

Agriculture not only fulfils the basic need for food supply but also plays a role in providing other ecosystem services, while ensuring the socio-economic stability of societies worldwide. In Europe, agriculture is intrinsically linked to the environment, rural development, and the well-being of its citizens (Stoate et al. 2009). Recognizing the dual role of agriculture as both a driver and a mitigating factor to environmental change, the European Union (EU) has consistently fostered the integration of environmental sustainability within its agricultural policy framework (Dos Santos & Ahmad 2020).

Agriculture is a major contributor to environmental problems such as greenhouse gas (GHG) emissions, nutrient losses or biodiversity declines, with implications for climate change, water quality and quantity, and ecosystem functioning (Campbell et al. 2017). The EU has recognized these environmental issues and is enhancing the policy to support environmentally sustainable farming practices. Not only through the CAP, but other recent EU initiatives, including the European Green Deal[1], the Farm to Fork Strategy[2], the Biodiversity Strategy for 2030[3] and the proposal of the Nature Restoration Law[4] reflect a strategy towards sustainability by setting targets for reducing the sector's environmental footprint.

Sustainable farming practices are crucial for achieving the EU's objectives. The application of these practices may minimize impacts, such as GHG emissions and nutrient losses, by employing methods that enhance efficiency and promote ecological balance (Pretty 2018), thereby leading to substantial environmental benefits.

The role of scientific knowledge in this context is multifaceted. Identifying sustainable agricultural practices is dependent on strong scientific evidence from global research and observations. Thus, research provides the necessary evidence base for designing policies that effectively address the environmental challenges of agriculture (Aznar-Sánchez et al. 2019). It also supports the implementation of these policies by offering technical guidance and best practices for farmers and land managers. Furthermore, ongoing scientific evaluation is essential for monitoring the outcomes of policy interventions, contributing to their refinement and optimization through the policy cycle (Head 2016).

Nevertheless, all this knowledge, while comprehensive, requires to be effectively communicated to policymakers for evidence-based decision. Systematic reviews and meta-analyses (MAs) are key methods for knowledge synthesis in various scientific fields. They constitute valuable tools for converting complex data into actionable information, here applied for sustainable agricultural policy development (Gurevitch et al. 2018). A systematic review involves the exhaustive compilation, evaluation and synthesis of all relevant studies dealing with a specific question. It is based on a

[1] *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. The European Green Deal COM(2019) 640 Fina*l, 2019

[2] *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. A Farm to Fork Strategy COM(2020) 381 Final*, 2020

[3] *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. EU Biodiversity Strategy for 2030 COM(2020) 380 Final*, 2020

[4] Proposal for a REGULATION OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL on nature restoration COM(2022) 304 Final, 2022

detailed protocol limiting the bias and favouring a transparent and reproducible approach (Chalmers et al. 2002).

Meta-analysis combines systematic review and statistical analysis and aims to provide quantitative information from a set of relevant primary studies (Lau 1997). This quantitative information can be critical to assess the environmental impacts of agricultural practices, often resolving conflicting research findings and considering the potential variability in the effectiveness of these practices under diverse conditions. Moreover, by aggregating data from individual studies, meta-analysis can increase the accuracy and robustness of outcome estimates by enhancing the overall sample size (Makowski et al. 2019; Philibert et al. 2012). The quantitative coefficients obtained from meta-analysis are valuable not only for informing policy decision-making at various levels, such as the design of targeted interventions, or the evaluation of their impacts, but also for filling data gaps. They can complement data from other sources or feed predictive models that aid in these tasks (Guerrero 2023). The use of these coefficients can streamline policy development and ensure that interventions are grounded in comprehensive and robust empirical evidence.

This report presents a collection of coefficients quantifying the environmental impacts of various farming practices, with a focus on their potential to increase carbon sequestration, and to reduce GHG emissions and nutrient losses. These coefficients are sourced from scientific articles, specifically meta-analyses, which have been identified through systematic literature review. They have been selected based on their quality and relevance to provide decision support tools for evaluating the agricultural policies' environmental performance.

Adapted for publication, this project deliverable provides clear links to the specific bibliographic references accompanying each coefficient. Additionally, it includes detailed contextual information to aid in understanding the coefficients and their caveats. This collection offers insights into how different agricultural management options can affect the environment, with relevance to Impact indicators[5] of the Common Agricultural Policy.

1.1 Purpose and scope of the document

The purpose of this document is to systematically compile and synthesize scientific evidence on the environmental impacts of various agricultural practices. By presenting a collection of quantitative coefficients derived from peer-reviewed meta-analyses, this report aims to inform and support the decision-making process for agricultural policy development within the EU. These coefficients quantify the potential of farming practices to increase carbon sequestration and reduce greenhouse gas (GHG) emissions and nutrient losses, thereby providing decision support tools for evaluating the environmental performance of agricultural policies.

The scope of this document is confined to the environmental impacts of a group of selected agricultural practices and management options, specifically focusing on GHG emissions, nutrient losses, and carbon sequestration. This document does not present an exhaustive list of farming practices and does not address other potential environmental and socio-economic impacts that may result from the application of these and other agricultural management options.

[5] More information on Impact indicators: https://agriculture.ec.europa.eu/common-agricultural-policy/cap-overview/cmef_en#towardsthepmef

2 Context and methodological approach

This work is part of the developments carried out under the project Integrated Modelling Platform for Agro-economic and resource Policy analysis (iMAP4agri) [6] , an Administrative Agreement commissioned by the European Commission's Directorate-General for Agriculture and Rural Development (DG AGRI) to the Joint Research Centre (JRC). iMAP4agri provides scientific support and tools necessary for the implementation, monitoring, and evaluation of the Common Agricultural Policy (CAP) Strategic Plan regulation (Regulation (EU) 2021/2115), with a particular focus on achieving the CAP objectives related to environmental sustainability and climate change mitigation.

The project aims to establish a comprehensive scientific understanding of the impacts that various farming practices have on the environment and climate. Drawing on experience from previous CAP periods and materials developed for the current policy cycle, the project has gathered scientific evidence of the positive and negative environmental impacts of selected sustainable farming practices, including their estimation through quantitative information.

2.1 Systematic review of meta-analyses and result synthesis

In this work, building the scientific evidence base is grounded on a large systematic review of synthesis papers (meta-analysis and others). The starting point is the large number of meta-analyses published in agricultural science that allow to explore general trends from large numbers of experimental studies and to identify key moderating factors. Thus, a methodological framework has been developed for assessing the impacts of farming practices on the environment and climate based on a systematic review of published meta-analyses. The framework helps to report the results and quality of meta-analysis in a rigorous and transparent manner. In addition, the framework can be implemented in a time-frame operational and compatible with the time constraints of policymaking.

The proposed framework's primary goal is to collect and synthesize meta-analyses (MAs) on specific farming practices, as defined by the above mentioned policy needs. It consists of four steps: (1) a systematic literature search to compile MAs, (2) screening and selecting relevant MAs based on predefined criteria, (3) extracting data and assessing the quality of selected MAs, and (4) generating reports on the impacts of farming practices.

The search string for Step 1 includes keywords and synonyms pertinent to the farming practice and filters out unrelated papers. It focuses on MAs and systematic reviews that include quantitative results, excluding unsystematic reviews. The use of multiple scientific databases ensures a comprehensive coverage.

In Step 2, the initial screening is based on titles and abstracts, and studies are categorized based on their relevance. Full-text screening follows for those deemed potentially relevant.

Step 3's data extraction covers various aspects of the MAs, such as references, objectives, methods, experimental details, and significant results. These results are rated, and MAs are evaluated using 16 quality criteria covering the following aspects:

[6] https://wikis.ec.europa.eu/display/IMAP/IMAP+Home+page

- Scoping: objective is specified
- Search: search databases are mentioned, search string is reported, and list of studies is reported
- Study selection: selection criteria are reported, number of selected studies at each step is reported
- Data extraction: method for data extraction is described
- Statistical analysis: quantitative results are provided, statistical method is described, individual effect sizes are reported, heterogeneity is analysed, individual studies are weighted, and confidence intervals are presented
- Bias and uncertainty: dataset is available, funding sources are reported, publication bias is analysed

Reporting in Step 4 delivers detailed, summarized, and synthetic information through different report types, offering a progressively synthesized view of the impacts, from detailed individual analyses to a broad overview in a general report.

A synopsis matrix in this general report presents a comprehensive scoring of the farming practice for all impacts, informed by the data compiled in Step 3. From this matrix, the nature of the overall effect of each farming practice on different environmental impacts, such as greenhouse gas emissions or nutrient use efficiency, can be determined. For this report, the determination of the direction of the effect is done using an accepted fast-track method for synthesizing the results of MAs without using the original primary data: relative majority of first-order MAs results (Figure 1). This method consists on the vote counting of MAs results reporting positive, negative, and non-significant effect (Makowski et al. 2023).

The final outputs of this work, as well as the intermediate results of each step, including search strings or exclusion/inclusion critera, can be consulted in the dedicated wiki space and in the published database (EC JRC 2023; Schievano et al. 2023).

It is important to note at this stage that the results presented in all the reports (EC JRC 2023), the dataset (Schievano et al. 2023) and this document, are based on the impacts retrieved from the selected meta-analyses after screening. This means that the search strings used did not include keywords related to specific or expected outcomes, so the impacts collected have been extracted solely from what was reported by the authors. For presentation and policy adequacy, in particular with regard to the Impact Indicators of the Performance and Monitoring Evaluation Framework[7], some of the metrics provided by the authors have been aggregated under the general nomenclatures that title the sections of the report. Some are straightforward, such as the case of greenhouse gas emissions but others require some further conceptualization, as nutrient balance (see Annex 1 for detailed information).

It is also worth emphasizing that the work is based on synthesis literature, which does not prevent having primary studies linking these practices to other impacts or outcomes not reported here. Besides, this document focuses only on results obtained through synthesis literature review concerning the impact of the selected set of farming practices on GHG emissions, carbon sequestration and nutrient losses.

[7] https://agriculture.ec.europa.eu/common-agricultural-policy/cap-overview/cmef_en#towardsthepmef

2.2 Selection of synthesis papers for the extraction of quantitative data

To extract relevant and suitable quantitative data, a method to select the most appropriate synthesis papers has been developed. It is intended to be simple, feasible and time-wise efficient. The method is based on a modified version of the quality score used in Step 3 of the above explained literature review framework (Figure 1). The original quality score evaluates the synthesis papers according to criteria covering three dimensions of the research synthesis procedure: i) literature review and studies selection, ii) quality of statistical analysis, and iii) potential bias (EC JRC 2023).

Given the aim of this selection to gather robust quantitative information on the environmental impacts of selected farming practices, we calculated a relevance score by assigning more weight to the criteria related to the statistical part (7 criteria out of 16 have been assigned 70% of the total weight). This ensures the statistical quality of the synthesis articles, without discarding the information on the quality provided by the other characteristics (30%).

The relevance score is further complemented with the addition of five more components related to:

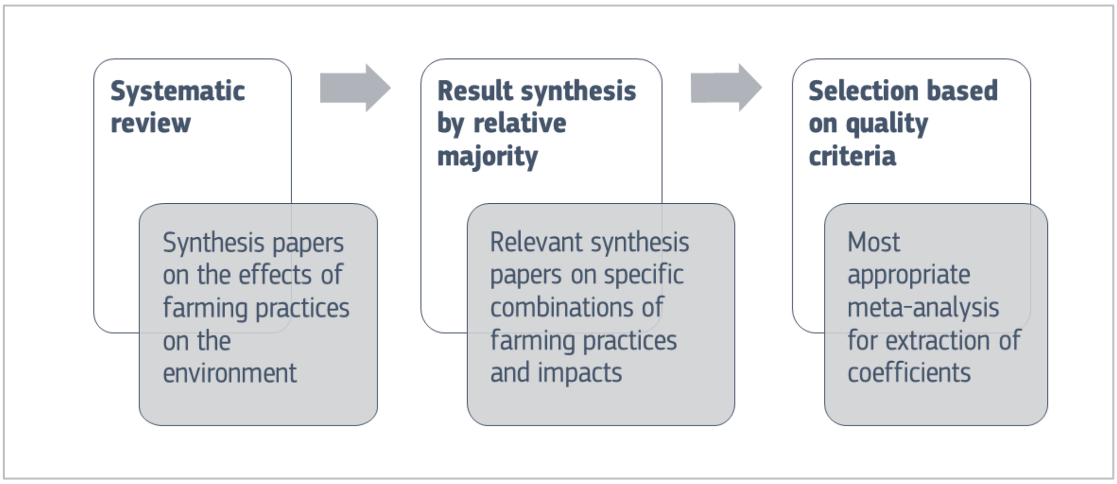
- the number of individual papers analysed in each synthesis paper, including a standardisation that gives higher value to the synthesis paper including more individual studies and weights the others according to this maximum.
- the year of publication, including a standardisation that gives higher value to the most recent paper and weights the others according to this last publication date.
- the inclusion of data from European Member States, giving higher value to those synthesis including European cases.
- the relevance of crops for the European Union agricultural context, giving higher value to those synthesis including two or more EU relevant crops. In some specific cases (i.e. agroforestry), this "bonus" is used to penalize comparators or interventions less pertinent to the policy.
- the pertinence of the impact metric to the policy framework, giving higher value to those output metrics more applicable to the policy context. This is needed because there are many different metrics used in systematic reviews to express effect sizes (indexes measuring the magnitude of the effects) and not all of them allow to derive a percentage or absolute change of the measured effect when applying the practice compared to control.

All these values, once aggregated, result in a final selection score. This evaluation and selection exercise is repeated for each farming practice-impact pair for which a significant positive or negative effect has been found (Figure 1). The studies are ordered according to their selection score and the first ranked is used for the extraction of numerical coefficients. In a few cases when not all the synthesis papers retrieved in the systematic review report unanimously the same qualitative (positive/negative) effect (hence the overall effect is determined by vote counting), it may happen that the first ranked synthesis paper belongs to the minority of those reporting a different qualitative effect, e.g. negative when the majority of the other paper report positive. In this case and to ensure consistency, the highest ranked synthesis paper reporting results in line with the overall effect is selected for the extraction of data.

From the subset of selected synthesis papers, corresponding quantitative information is extracted from texts, tables and figures provided in the publications. Usually, synthesis papers report the effect size i.e. a ratio of a certain metric affected by the practice compared to the same metric measured without applying it (control study). For example, if the mean effect size of the use of controlled released fertilizers with regard to ammonia emission is 0.8, it means that the use of these techniques reduces ammonia emissions by 20% on average, compared to control (i.e. not using it, other things being equal). Effect sizes, whenever possible, are transformed to percentage of change, and confidence intervals and number of pairwise comparisons are provided when applicable and available.

For a single combination of farming practice and its measured impact, a synthesis paper may report more than one effect size based on different factors such as crop type, biogeographic region, or specific techniques used. This further stratified information is extracted when it can be relevant for the policy. In this cases, the stratified results can differ in their direction (positive or negative) or statistical significance from the overall result defined by relative majority of first-order MAs results, as discussed earlier. In some cases, data were extracted from two papers if they provide consistent but complementary information with regard to the examined environmental effects, e.g. by providing more detailed results with respect to specific sub-practices or geographic locations.

Figure 1 Representation of the decision phases for the selection of meta-analyses for quantitative data extraction. Following a systematic review process in synthesis literature, the results are evaluated using the relative majority fast-track method. In cases where statistically significant majority results are determined, the most appropriate meta-analysis is selected for coefficient extraction. See text for further details.



Source: Own elaboration.

3 Results

3.1 Summary table

The following table (Table 1) summarizes the results presented in this document. It displays, for each combination of farming practice and environmental impact, the number of coefficients extracted from selected synthesis papers, once the process described in the previous section is completed (Figure 1). These coefficients can represent overall values (i.e. average result considering all sub-groups) and/or disaggregated results regarding different sub-groups, which may be crop type, soil type, geographical area, etc. As explained earlier in the methodological approach, the selection of farming practices has been guided by the policy preferences and needs that contextualize the project. From all the farming practice-impact combinations found in the systematic literature review, only those determined to be statistically significant by the majority of results, have been considered for quantitative data extraction. Thus, the empty cells in the following summary table may reflect situations where the practice predominantly does not present statistically significant effects or where the literature review did not retrieve results. For policy related preferences, the authors of this collection have decided not to include LCA (Life Cycle Assessment) results, as their adoption in European policy is not widespread (Sala et al. 2021), therefore, these results wer not taken into account at any of the steps of the process.

The focus is on data concerning GHG emissions, carbon sequestration and nutrient losses disaggregated into ammonia emissions, nutrient leaching and run-off and the effect of farming practices on the agricultural nutrient balance (more information on the nutrient balance impact can be found in Annex 1). For the coefficients extracted on soil organic carbon (SOC) additional information on geographical area, time and soil depth, is provided as supporting material available at Bosco et al. 2024.

Table 1 Summary of results from the systematic literature review and study selection process: Number of coefficients extracted for each combination of farming practice and environmental impact, based on statistically significant results and, in brackets, the corresponding section. Empty cells may indicate both situations of not statistically significant effects or of not retrieved results from literature review.

	GHG emissions	Soil Organic Carbon	Ammonia emissions	Nutrient balance	Nutrient leaching and run-off
Agroforestry		6 (3.2.1)			
Organic systems	6 (3.3.1)	26 (3.3.2)			3 (3.3.3)
Low-ammonia mineral fertilisation techniques			31 (3.4.1)	35 (3.4.2)	6 (3.4.3)
Enhanced Efficiency Fertilisers	14 (3.5.1)		17 (3.5.2)	83 (3.5.3)	26 (3.5.4)
Organic fertilisation	1 (3.6.1)	12 (3.6.2)	4 (3.6.3)	9 (3.6.4)	10 (3.6.5)
Green manuring	12 (3.7.1)	7 (3.7.2)		1 (3.7.3)	2 (3.7.4)
Soil amendment with biochar	15 (3.8.1)	18 (3.8.2)	9 (3.8.3)	8 (3.8.4)	5 (3.8.5)
Soil amendment with lime and gypsum		12 (3.9.1)			
Manure land application techniques	8 (3.10.1)		5 (3.10.2)		

	GHG emissions	Soil Organic Carbon	Ammonia emissions	Nutrient balance	Nutrient leaching and run-off
Manure storage techniques	15 (3.11.1)		12 (3.11.2)		
Manure processing techniques	20 (3.12.1)		10 (3.12.2)		
Livestock housing techniques	6 (3.13.1)		22 (3.13.2)		
Livestock feeding techniques	30 (3.14.1)		8 (3.14.2)	34 (3.17.2)	
Landscape Features		9 (3.15.1)			16 (3.15.2)
Fallowing		26 (3.16.1)			
Intercropping		1 (3.17.1)		4 (3.17.2)	
Crop rotation	5 (3.18.1)	13 (3.18.2)			
Cover and catch crops	8 (3.19.1)	7 (3.19.2)		2 (3.19.3)	23 (3.19.4)
Leguminous crops		1 (3.20.1)			
Tillage practices	30 (3.21.1)	15 (3.21.2)			
Crop residue management	10 (3.22.1)	12 (3.22.2)		10 (3.22.3)	23 (3.22.4)
Mulching	30 (3.23.1)		5 (3.23.2)	20 (3.23.3)	5 (3.23.4)
Grassland conservation and restoration	1 (3.24.1)	16 (3.24.2)			1 (3.24.3)
Grassland management	18 (3.25.1)	4 (3.25.2)			8 (3.25.3)
Grazing	16 (3.26.1)	9 (3.26.2)			
Peatland management	5 (3.27.1)				
Peatland conservation	14 (3.28.1)	2 (3.28.2)			
Peatland restoration	9 (3.29.1)	1 (3.29.2)			
Wetland management	3 (3.30.1)				
Wetland conservation	17 (3.31.1)	4 (3.31.2)			
Wetland restoration	1 (3.32.1)				5 (3.32.2)
No irrigation	1 (3.33.1)			19 (3.33.3)	4 (3.33.2)
Water-saving irrigation in flooded lands	3 (3.34.1)	1 (3.34.2)	2 (3.34.3)	3 (3.34.4)	8 (3.34.5)
Water-saving irrigation in non-flooded lands	2 (3.35.1)			4 (3.35.2)	4 (3.35.3)

Source: own elaboration.

3.2 Agroforestry

Agroforestry is a type of land-use system where perennial woody plants (trees, shrubs) are deliberately used in combination with agricultural crops, pastures and/or animals. In Europe, six agroforestry practices are identified: silvoarable agroforestry, forest farming, riparian buffer strips, improved fallow, multipurpose trees and silvopasture. In the literature analysed, agroforestry impacts are compared to: (i) land use without trees, including cropland, pasture and, sometimes, fallow land; and (ii) forests, either natural or planted. The comparator considered in the present deliverable is cropland without trees, so just results related to this comparator are considered here.

3.2.1 Effects on soil organic carbon

The selected synthesis paper for the extraction of quantitative data is the study from Chatterjee et al. 2018. Data were extracted for two relevant agroecological regions: Mediterranean regions (located on the western sides of continents between the latitudes of 30 °N and 45 °N) and temperate regions (located in latitudes between 40 °N and 60 °N). The assessed metric is the percentage change in soil organic carbon stock between agroforestry systems and cropland without trees, measured within different soil depth classes: 0-20, 0-40, 0-60 and 0-100 cm for Mediterranean regions and 0-20 and 0–40 cm for temperate regions, as reported in the study (Table 2).

The conversion from agriculture to agroforestry generally increased SOC stocks in both Mediterranean and temperate regions and at different soil depth classes. This increase was found significant at 0-20 and 0-100 cm for Mediterranean regions, and at 0-40 cm for temperate regions. The highest increase in SOC stock was found in the 0-20 cm soil depth class for Mediterranean regions (+43.51%; 95% confidence interval (CI): 12.13%, 75.73%; number of samples or pairwise comparisons (Nc) = 8), and in the 0-40 cm soil depth class for temperate regions (+18.64%; 95% CI: 12.99%, 25.14%; Nc=15).

Table 2 Effect of agroforestry on carbon sequestration (% change in SOC stock) in Mediterranean and temperate regions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Biogeographic regions	soil depth (cm)	CI_LOW	Mean	CI_HIGH	Nc	Effect
Mediterranean regions	0-20	12.13	43.51	75.73	8	Positive
	0-40	-3.00	10.21	21.92	15	Non-significant
	0-60	-12.50	8.5	27.72	7	Non-significant
	0-100	4.15	5.80	7.77	6	Positive
Temperate regions	0-20	-3.49	1.45	6.69	15	Non-significant
	0-40	12.99	18.64	25.14	15	Positive
	other	Not available	Not available	Not available		

Source: Chatterjee et al. 2018.

3.3 Organic farming systems

Organic production is an overall system of farm management and food production that combines best environmental and climate action practices, a high level of biodiversity, the preservation of natural resources and the application of high animal welfare standards and high production standards in line with the demand of a growing number of consumers for products produced using natural substances and processes[8]. Organic farming systems have been examined both mixed and considering cropping systems and livestock systems separately.

3.3.1 Effects on greenhouse gas emissions

Results of GHG using the Life Cycle Assessment method (LCA) have not been included in the analysis. For this reason, there are no results for organic mixed livestock and crop systems but only for organic cropping systems.

The selected synthesis paper (Skinner et al. 2014) reports effects on GHG emissions from arable land and grasslands soils. Studies were conducted in the Northern hemisphere under temperate climate. All arable soils showed an average methane uptake, which was slightly higher (both area-scaled and yield-scaled) under organic than under non-organic management with a mean difference of 3.2 ± 2.5 kg CO2 eq ha^{-1} yr^{-1} for organic management (8 comparisons; $p = 0.01$). The important exceptions to soils as a methane sink are rice paddies, waterlogged anaerobic systems, which are emitters of large amounts of methane produced by methanogenic *Archaea*. However, only one comparative study on rice paddies had been included in their meta-analysis. The study on paddy rice cropping showed pronounced methane emissions under both systems. Significantly more methane was emitted under organic compared to non-organic management with a mean difference of 950 kg CO2 eq ha^{-1} yr^{-1} (Nc=3; $p < 0.01$). The authors (Skinner et al. 2014) also found that across all annual GHG measurements on arable and grassland use, soils under organic farming emit less N2O than the non-organic counterparts. The impact for arable land was -497 ±162 kg CO2-eq. ha^{-1} yr^{-1} (p 0.001) (Table 3).

Table 3 Effect of organic systems on greenhouse gas emissions (change of emissions per hectare[9]). All figures are in kg CO2eq/ha/yr. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

GHG	Systems	CI_LOW	Mean	CI_HIGH	Nc	Effect
CH4	Arable	-5.7	-3.2	-0.7	8	Positive
	Rice paddies	+535	+950	+1365	3	Negative
N2O	Arable + grassland	-652	-492	-332	70	Positive
	Arable	-659	-497	-335	67	Positive
	Grassland	-3622	-1091	+1440	3	Non-significant
	Rice-paddies	-1686	-646	+394	3	Non-significant

Source: Skinner et al. 2014

[8] https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:32018R0848&from=EN

[9] The conversion factors applied are 298 kg CO2eq/kg N2O, 468,198 kg CO2eq/kg N2O-N, 25 kg CO2eq/kg CH4..

3.3.2 Effects on soil organic carbon

Results are typically provided for soil organic carbon concentration, soil organic carbon stocks and soil carbon sequestration rates. SOC concentrations describe the organic carbon concentration on a weight by weight basis, SOC stocks on a weight by area basis, and C sequestration rates, on weight by area and elapsed time since conversion to organic (Schievano et al. 2023).

3.3.2.1 Organic livestock systems

The data provided in the selected paper (Gattinger et al. 2012) mainly cover top-soil and temperate zones, whereas only few data from tropical regions and subsoil horizons exist. Authors argue that, although these data clearly showed that organic management increased SOC, it is often stated that increased SOC stocks originate from massive imports of organic matter. To examine the potential impact of imported organic matter, the authors analyse a subset of studies representing organic farming systems with zero net input separately ("*Zero net input systems*"). These should represent mixed livestock–crop production farms with forage crops in the crop rotation, such that the livestock can be fed entirely from fodder produced on-farm. In such systems, no import of organic matter should occur. On the other hand, these systems could also be stockless farms that import organic matter from elsewhere but to an extent that is supported by their own systems' productivity. For their analysis, the cut-off for a zero net input system was set at an amount of organic fertilisers applied to the trials that corresponds to the manure amount from 1.0 (European) livestock unit (LU) ha−1. This is lower than the maximum stocking rates that they consider to reflect a zero net input system. Adopting this conservative threshold, they may have neglected some studies that actually represent zero net input systems, but they could be sure that all organic systems with net inputs were excluded.

Table 4 Effect of organic systems on carbon sequestration[10] mostly for top soil and temperate zones. CI_LOW: confidence interval bottom. CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Impact and metric	Farm types	CI_LOW	Mean	CI_HIGH	Nc	Effect
SOC concentration (g/kg)	All farms	1.2	1.8	2.4	200	Positive
	Zero net input	0.4	1.3	2.2	60	Positive
SOC stock (Mg C/ha)	All farms	2.4	3.5	4.6	204	Positive
	Zero net input	0.52	2.17	3.81	60	Positive
C sequestration rate (Mg C/ha/yr)	All farms	0.24	0.45	0.66	41	Positive
	Zero net input	-0.10	0.27	0.64	19	Non-significant

Source: Gattinger et al. 2012.

[10] Results are provided for soil organic carbon concentration, for soil carbon stocks and carbon sequestration. SOC concentrations describe the organic carbon concentration on a weight by weight basis, SOC stocks on a weight by area basis, and C sequestration rates, on weight by area and elapsed time since conversion to organic.

The selected synthesis paper (Gattinger et al. 2012) found significant differences and higher values for organically farmed soils of 3.50 ± 1.08 Mg C ha−1 for stocks, and 0.45 ± 0.21 Mg C ha−1 yr−1 for sequestration rates compared with nonorganic management. For the "zero net input" subset (systems with zero net input of organic matter, see previous paragraph), SOC stocks were 2.16 ± 1.65 Mg C ha−1 higher ($p < 0.01$), and C sequestration rates were no longer significantly different from non-organically managed soils (0.27 ± 0.37 Mg C ha−1 yr−1; $P > 0.1$). SOC concentration and stocks increase with time, while C sequestration rates decrease with the number of years. Data are not provided in terms of % change, but considering typical SOC values in soils the absolute changes in SOC stocks are in the order of magnitude of 6-10% (Table 4).

3.3.2.2 Organic cropping systems

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the overall effect is positive. Data for organic cropping systems are extracted from the synthesis paper by Aguilera et al. 2013 (Table 5). Additional results from Smith et al. 2019 have been added as it differentiates SOC concentration from SOC stock and USA from European studies (Table 6)

In the synthesis paper by Aguilera et al. 2013 SOC concentration was preferentially chosen for comparisons because it is considered to be a more direct measure of SOC, which is not influenced by soil volume and bulk density estimations. Shallow sampling was often the case in organic farming studies, where average sampling depth was just 19.2 cm. SOC increment in organic systems was greater under irrigation than under rainfed conditions (25% vs. 13% increase over conventional, respectively). When analysed by crop type, the data shows that the best performing organic group is horticulture, where SOC is increased by 48%. Groups in Irrigation and Crop Type categories differed in the intensification of C input rate. Mean C inputs (additional C input over conventional) were 4.8 and 3.2 Mg C ha−1 yr−1 in irrigated and rainfed systems, respectively, and 6.1, 2.5 and 3.1 Mg C ha−1 yr−1 in horticulture, cereals and woody crops, respectively.

The type of organic input employed in the organic system also influences the differences found between systems. Compost, either applied alone or in combination with cover crops, is the input associated to highest increases in SOC (48% and 26.2% for compost alone and mixtures with cover crops, respectively) and in C sequestration rate (1.32 and 0.97 Mg C ha−1 yr−1 for compost and mixtures, respectively). Manure application obtains poorer results, being the increase in C sequestration rate over conventional non-significant when this amendment is applied alone, although the sample size is lower in this case (9 paired comparisons versus 27 in Compost alone). When manure is combined with cover crops, the increase is higher and significant (35.8% and 0.62 Mg C ha−1 yr−1). We have only two paired data of cover crops used alone. This category, however, was also studied in the general meta-analysis, where the number of studies was larger. Finally we have found some organic treatments where no organic inputs were applied, or at least no more than in their conventional counterpart. These treatments were the only ones with a lower SOC level than conventional, although the differences were not significant. The last variable studied was the type of experimental approach. The increase in SOC concentration and sequestration rate in organic plots was much more pronounced in plots of controlled experiments (51.6% and 1.28 Mg ha−1 yr−1) than in real farms (11.4% and 0.31 Mg ha−1 yr−1, being the latter non-significant).

Table 5 Effect of organic systems on carbon sequestration: change in SOC concentration or C stock and in C sequestration rate. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Impact, metric and source	Systems	CI_LOW	Mean	CI_HIGH	Nc	Effect
SOC concentration (or SOC stock if missing) *Source:* Aguilera et al. 2013	All crops	11.1%	19.2%	29.9%	53	Positive
	Cereals rotation	2.2%	7.8%	17.2%	20	Positive
	Horticulture	26.7%	48.3%	74.5%	25	Positive
	Woody crops	4.6%	15.7%	27.9%	35	Positive
SOC concentration [%] *Source:* Smith et al. 2019	All crops	10.2 %	13.9 %	17.3 %	209	Positive
	Annual	12.7%	16.2%	20.9%	138	Positive
	Perennial	-0.6%	5.1%	11.6%	54	Non-significant
SOC stock [kg C/ha] *Source:* Smith et al. 2019	All crops	9.1 %	11.6 %	15 %	209	Positive
	Annual	11.6%	15%	19.7%	138	Positive
	Perennial	-1.7%	24.6%	8.5%	54	Non-significant
C sequestration rate (Change in Mg C/ha/yr) *Source:* Aguilera et al. 2013	All crops	0.54	0.97	1.48	24	Positive
	Cereals rotation	0.30	0.76	1.44	10	Positive
	Horticulture	0.46	1.05	1.56	24	Positive
	Woody crops	0.54	0.92	1.58	10	Positive

Source: references in the first column.

Interestingly, important differences have been found by Smith et al. 2019 between European and US studies (Table 6). USA studies showed higher effect (double) of organic farming on SOC concentration and SOC stock than the European ones.

Table 6 Effect (% change) of organic systems on carbon sequestration by regions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Impact, metric and source	Region	CI_LOW	Mean	CI_HIGH	Nc	Effect
SOC concentration [%]	All	10.2	13.9	17.3	209	Positive
	EU	4.8	9.4	13.9	108	Positive
	USA	17.3	24.6	32.3	47	Positive
SOC stock [kg C/ha]	All	9.1	11.6	15	209	Positive
	EU	5.3	9.4	13.9	108	Positive
	USA	13.9	20.9	27.1	47	Positive

Source: Smith et al. 2019

3.3.3 Effects on nutrient leaching and run-off

The selected synthesis paper is the one by Mondelaers et al. 2009. Mean response ratios showed that nitrogen leaching per unit of area was 32% lower from organic farming compared to conventional farming (95% CI: -47%, -13%). These values are rather in line with those for arable land (cropping systems) (-35%; 95% CI: -54%, -8%) while for mixed livestock systems the effect on N leaching is not significant (Table 7).

Table 7 Effect of organic systems on nitrogen leaching (% change of kg N per hectare). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

	Systems	CI_LOW	Mean	CI_HIGH	Effect
N leaching	All	-47	-32.1	-13	Positive
	Mixed livestock (without arable)	-55	-30.5	+8	Non-significant
	Arable	-54	-34.8	-8	Positive

Source: Mondelaers et al. 2009.

3.4 Low-ammonia emission techniques for mineral fertilisation

Low-ammonia-emission techniques are fertilisation techniques used to reduce agricultural emissions of ammonia from mineral fertilisers. They include several practices that have been analysed separately, namely:

- Deep placement: the fertiliser is placed into the soil at greater depth than in conventional application.
- Irrigation after fertilisation.
- Split application: the total amount of fertiliser is applied through multiple applications in time (usually 2 or 3).
- Use of specific amendments with sorbents (such as pyrite or zeolites) added to the applied fertiliser, to reduce NH3 volatilisation.
- Use of non-urea based fertilisers, like ammonium nitrate and ammonium sulphate.
- Leaving crop residues on soil.

3.4.1 Effects on ammonia emissions

The selected paper is Ti et al. 2019. However, we also include Pan et al. 2016 which includes all the techniques, comprising also use of sorbents and crop residues left on the soil (Table 8).

Most of the mitigation strategies studied by Ti et al. 2019 were effective in reducing NH3 emission. For organic fertilisers, deep manure placement has the highest mitigation potential relative to other mitigation strategies, with reduction range of 93.8 to 99.7% relative to the control. Regarding mineral fertilisers, the fertiliser type and deep placement application are the most effective for reducing NH3 emission. The application of the following chemical fertilisers: ammonium nitrate, ammonium sulphate, urea phosphate, calcium ammonium nitrate, monoammonium phosphate, and diammonium phosphate could significantly reduce NH3 emission by 88.3, 82.9, 76.2, 67.3, 51.9, and 51.2%, respectively, compared with the use of urea. Deep placement of mineral fertilisers could reduce NH3 emission by 48.0%. Irrigation right after fertilisation significantly reduced NH3 emission by 38.8%. Split application of fertilisers has no significant reduction effect according to Ti et al. 2019. Among crop types, significant reductions through fertiliser management (deep placement, split application and irrigation) were achieved in wheat and maize.

Table 8 Effect (% change) of low-ammonia emission techniques on ammonia emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Specific practice	Subpractice or factor	CI_LOW	Mean	CI_HIGH	Effect
Overall for deep placement, split application & irrigation *Source:* Ti et al. 2019	Overall	-46.8	-39.8	-31.4	Positive
	pastures	-40.2	-13.9	25.3	Non-significant
	rice	-42.0	-19.7	11.4	Non-significant
	maize	-46.8	-28.5	-2.9	Positive
	wheat	-71.3	-61.8	-48.3	Positive
Deep placement *Source:* Ti et al. 2019	Overall	-57.4	-48.3	-36.6	Positive
Deep placement *Source:* Pan et al. 2016	Overall	-66	-54.7	-41	Positive
Irrigation after fertilisation *Source:* Ti et al. 2019	Overall	-49.0	-39	-26	Positive
Irrigation after fertilisation *Source:* Pan et al. 2016	Overall	-50	-34.5	-17.3	Positive
Use of sorbents *Source:* Pan et al. 2016	Overall	-41.6	-31.4	-20.6	Positive
	zeolite	-59.0	-43.5	-26.1	Positive
	pyrite	-31.6	-21.0	-9.8	Positive
	organic acid	-20.8	-15.9	-11.0	Positive
Use of non-urea fertilisers *Source:* Ti et al. 2019	ammonium nitrate	-91.3	-88.3	-83.9	Positive
	ammonium sulphate	-86.8	-82.9	-78.2	Positive
	urea phosphate	-89.6	-76.2	-43.3	Positive
	calcium ammonium nitrate	-81.0	-67.3	-41.6	Positive
	monoammonium phosphate	-76.5	-51.9	-1.0	Positive
	diammonium phosphate	-68.5	-51.2	-23.9	Positive
	urea ammonium nitrate	-58.2	1.2	145.8	Non-significant
	ammonium bicarbonate	-26.8	84.1	364.7	Non-significant
Use of non-urea fertilisers *Source:* Pan et al. 2016	Overall (17 types)	-70.0	-63.5	-55.5	Positive
	non-urea based fertilisers	-80.6	-74.5	-67.1	Positive

Specific practice	Subpractice or factor	CI_LOW	Mean	CI_HIGH	Effect
	urea-containing mixed fertilisers	-42.9	-30.8	-15.8	Positive
	ammonium nitrate	-94.2	-87.9	-76.8	Positive
	ammonium sulphate	-86.5	-78.8	-69.0	Positive
	other 8 types	-	-	-	Positive
	other 4 types	-	-	-	Non-significant
	other 3 types	-	-	-	Negative
Split application *Source:* Ti et al. 2019	Overall	-34.0	-1.6	47.3	Non-significant
Split application *Source:* Pan et al. 2016	Overall	-20.6	7.4	33.5	Non-significant
	4 times	-32.3	11.3	46.1	Non-significant
	3 times	-12.9	2.6	21.9	Non-significant
Crop residues left on soils *Source:* Pan et al. 2016	Overall	6.7	25.5	50	Negative

Source: references in the first column.

Pan et al. 2016 found that, when compared to urea fertilisers, the application of non-urea based fertilisers and urea-containing mixed fertilisers significantly decreased NH3 volatilization by 74.5% and 30.8%, respectively, with an overall reduction of 63.5%. Ammonium nitrate and ammonium sulphate were the two most effective non-urea fertilisers in reducing NH3 volatilization (by 87.9% and 78.8%, respectively) relative to urea application. Deep placement significantly decreased NH3 volatilization through incorporation of fertilisers by 54.7% when compared to surface application. Split applications of N fertiliser did not affect NH3 volatilization, regardless of splitting frequency. Irrigation significantly decreased NH3 volatilization by 34.5% compared to rainfed or supplementary (minimal) irrigation.

Soil cover with crop residues significantly increased NH3 volatilization by 25.5%. Amendments with sorbent materials significantly reduced NH3 volatilization by 31.4%. In particular, zeolite decreased NH3 volatilization by 43.5%, followed by pyrite (20.9%) and organic acid (15.9%) when applied with fertilisers.

3.4.2 Effects on the nutrient balance

The selected synthesis paper (L. Xia et al. 2017) conducted a comprehensive meta-analysis for staple grain (rice, wheat, and maize) production in China. The authors assessed the responses to knowledge-based N management practice of crop productivity (yield; NUE; aboveground N uptake), N2O emission and major Nr losses (NH3 emission, N leaching and runoff), and economic indicators (input cost, yield profit, and NEB). They focused on seven knowledge-based N management practices, including the applications of several enhanced efficiency fertilisers techniques, increasing splitting frequency of fertiliser N application, deep placement of N fertiliser, lower basal N fertiliser (BF) proportion, and optimal N rate based on soil N test.

The aboveground N uptake was significantly increased by 5.1% increasing splitting frequency of fertiliser N application. According to the authors “responses of aboveground N uptake and NUE were in general similar for various crops, N application rates, and soil properties”. In Table 9 it can be observed that values for the different crops are rather similar, but not the values for soil pH: there is a positive effect for acidic and neutral soils (pH<8) but non-significant effect for alkaline soils.

The paper by L. Xia et al. 2017 also provides results for nitrogen use efficiency (NUE) which they define as "grain NUE, calculated by dividing the difference in the grain N uptake between the treatments with and without fertilisation by fertiliser N rate." This definition corresponds to what other authors have defined as fertiliser recovery (NR, see Annex 1).

For deep placement, the selected paper for data extraction (Nkebiwe et al. 2016) provides results for nitrogen, phosphorus and potassium uptake and for all nutrients globally. We also report the results by L. Xia et al. 2017 for comparison with split application and because they provide an overall value for nitrogen fertilisation.

Table 9 Effect (% change) of low-ammonia emission fertiliser techniques on nutrient uptake and nutrient recovery (NR). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Specific practice and source	Factor and metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Split application or increased splitting frequency *Source:* L. Xia et al. 2017	Overall- uptake	3.5	5.1	6.8	52	Positive
	Overall- NUE (NR)	20.5	30	41.5	77	Positive
	Rice - uptake	2.7	5.6	8.2	11	Positive
	Wheat - uptake	1.0	3.9	7.1	19	Positive
	Maize - uptake	3.9	6.3	9.0	22	Positive
	Rice – NUE (NR)	25.4	46.1	75.9	20	Positive
	Wheat – NUE (NR)	-3.9	10.2	27.2	16	Non-significant
	Maize – NUE (NR)	19.8	32.0	46.6	41	Positive
	pH≤6 - uptake	5	7.3	9.7	10	Positive
	6<pH<8 - uptake	1.7	5.5	9.4	10	Positive
	pH≥8 - uptake	-1.9	1	4	11	Non-significant
Nutrient (N, P, K & S) deep placement *Source:* Nkebiwe et al. 2016	Overall above-ground biomass content	10	12	14	245	Positive
	Rapeseed (maximum)	30	36	43	36	Positive
	Maize	9	12	16	112	Positive
	Winter wheat	3	7	11	57	Positive
	Soybean	1	2	4	2	Positive
	Mixed grass species	0	0	0	3	Non-significant
	Winter rye (minimum)	-9	-3	1	16	Non-significant

Specific practice and source	Factor and metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Nitrogen deep placement *Source:* Nkebiwe et al. 2016	Overall - above-ground biomass content	NA	NA	NA	NA	Not available
	Depth > 10 cm	6	11	17	47	Positive
	Depth 5-10 cm	4	4	10	80	Positive
	Depth 0 cm	7	9	12	14	Positive
Nitrogen deep placement *Source:* L. Xia et al. 2017	Overall- uptake	4.8	8.3	12.2	39	Positive
	Overall- NUE (NR)	11.2	28.5	48.6	26	Positive
	Rice - uptake	4.7	13.4	24.9	10	Positive
	Wheat - uptake	2.0	6.7	11.4	14	Positive
	Maize - uptake	2.0	6.4	10.9	15	Positive
	Rice – NUE (NR)	-4.3	5.5	16.8	5	Non-significant
	Wheat – NUE (NR)	8.9	23.0	34.7	6	Positive
	Maize – NUE (NR)	9.4	42.6	82.8	15	Positive
	pH≤6 - uptake	13.6	26.2	39	6	Positive
	6<pH<8 - uptake	3.9	8	12.4	16	Positive
	pH≥8 - uptake	-0.7	2.9	6.6	17	Non-significant
Phosphorus deep placement vs. broadcast *Source:* Nkebiwe et al. 2016	Overall- above-ground biomass content	NA	NA	NA		Not available
	Depth > 10 cm	5	9	12	27	Positive
	Depth 5-10 cm	11	20	31	25	Positive
	Depth 0 cm	4	5	6	5	Positive

Source: references in the first column.

According to Nkebiwe et al. 2016, the overall deep placement effect on crop uptake (above-ground biomass) content of all nutrients (NPK and S) was 11.9% (95% CI: 9.7,14.5; Nc= 245). Results are also split by nutrient type, crop, crop growth stage, fertiliser placement methods, fertiliser type and placement depth. By crop type only, relative placement effect on above-ground biomass nutrient (NPK and S) content for different crop species were in the following order: rapeseed (36.4%; 95% CI: 30.2%, 43.3%; Nc= 36); turnip rape (30.3%; 95% CI: 28.1%, 32.9%; Nc= 2); sorghum(17.7%; 95% CI: 10.8%, 26.4%; Nc= 12); maize (12.2%; 95% CI: 8.7%,16.1%; Nc= 112); cauliflower (12.2%; 95% CI: 9.1%–15.4%; Nc= 2); winter wheat(7.2%; 95% CI: 3.5%–11.1%; Nc= 57); soybean (2.2%; 95% CI: 0.6%–4.1%; Nc= 2); mixed grass species (0.0%; 95% CI: 0.0%–0.0%; Nc= 3); lettuce (−2.0%; 95% CI: −17.4%, 13.2%; Nc= 2); winter rye (−3.1%; 95% CI: −9.0%, 0.8%; Nc= 16).

During vegetative growth, relative placement effect on nutrient (NPK and S) content in above-ground biomass showed the following decreasing trend according to placement depth: >10 cm (24.9%; 95% CI: 16.8%, 33.8%; Nc= 36); 5–10 cm (23.4%; 95% CI: 16.2%,32.5%; Nc= 51); surface (−5.7%; 95%

CI: −15.4%, 6.3%; Nc= 4) as compared with fertiliser broadcast. According to fertilizer type and placement depth, there was also a tendency for the effect on uptake of N, P and K to increase with increasing placement depth (see Table 9).

Nkebiwe et al. 2016 also analysed differences by:

- crop type and development stage vegetative and generative stages)
- eight fertiliser placement methods (e.g. subsurface shallow band, subsurface deep band, subsurface shallow point injection, subsurface deep point injection, etc.)
- fertiliser type (placement of a combination of ammonium or urea with soluble P showed a tendency to lead to stronger effect on N or P uptake than placement of ammonium, urea or soluble P not combined).

3.4.3 Effects on nutrient leaching and run-off

The selected synthesis paper (L. Xia et al. 2017) showed reduced N leaching and run-off for both deep fertiliser placement compared to superficial placement and for split application techniques compared to single application. However, no evidence from other low-ammonia techniques other than from deep placement and split application has been found (Table 10).

L. Xia et al. 2017 conducted a comprehensive meta-analysis for staple grain (rice, wheat, and maize) production in China. Their results indicate that increasing splitting frequency of fertiliser N application significantly reduced N leaching by 24.7% (95% CI: -46%, -3%) and N run-off by 36.5% (95% CI: -46%-26%). Deep placement also significantly decreased N run-off by 15.5% (95% CI: -25%, -9.5%) while no values are available for N leaching.

A stronger reduction in N runoff was shown in rice (45.7%) than wheat (24.5%) for increasing splitting frequency of fertiliser N application. Responses of N leaching and runoff were in general similar for different soil properties.

Table 10 Effect (% change) of low-ammonia emission fertiliser techniques on NO3- leaching. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Specific practice	Metric	CI_LOW	Mean	CI_HIGH	Effect
Split application or increased splitting frequency	N leaching –all crops	-46.2	-24.7	-3.0	Positive
	N runoff – all crops	-45.9	-36.5	-26.1	Positive
	N runoff - rice	-57.3	-45.7	-30.3	Positive
	N runoff - wheat	-28.3	-24.5	-19.2	Positive
	N runoff - maize	-57.2	-37.5	-10.6	Positive
Deep placement	N leaching – all crops	NA	NA	NA	Not available
	N runoff – all crops	-25	-15.5	-9.5	Positive

Source: : L. Xia et al. 2017

3.5 Enhanced Efficiency Fertilisers

This set of practices comprises different specific techniques aimed to increase the nitrogen use efficiency, i.e. the ratio of N uptake by crops over total N input (either mineral or organic). This can be achieved by slowing down the rate of N release from conventional fertilisers or by delaying the chemical and biological N transformation processes occurring in the soil leading to N volatilization in the form of ammonia and/or nitrous dioxide (N2O) and N leaching and runoff. The specific techniques included in this review are:

- controlled-release fertilisers, which use partially permeable coating material or encapsulating material to control N release.
- urease inhibitors, which delay urea hydrolysis thus lowering NH3 emission potential.
- nitrification inhibitors, which reduce the activities of nitrifying bacteria to decrease NO3 leaching and N2O emissions
- double inhibitors, combining nitrification inhibitors and urease inhibitors, they lower both NH3 and NO3/N2O losses.

3.5.1 Effects on greenhouse gas emissions

Enhanced efficiency fertilisers have an effect on N losses, therefore this practice is relevant for N2O emissions. The selected source of data is the synthesis paper by T. Li et al. 2018. The assessed metric is N2O emissions per hectare and year in production systems of major crops. Major crops or crop groups considered are: pastures, rice, and drylands, which include vegetables, maize and wheat. Results are summarised in Table 11.

For controlled-release fertilisers (polymer-coated fertilisers), the mean effect on N2O emissions is close to -74% for grasslands and paddy rice areas, while it is much lower for drylands (-23%; 95% CI: -32%, -13%).

For urease inhibitors, the synthesis paper (T. Li et al. 2018) found a reducing effect of -37% (95% CI: -59%, -7%) for drylands only (wheat, maize, vegetables). Additional research is needed to further clarify or validate the effect of urease inhibitors on N2O emissions. Although not showed in the table, Fan et al. 2018 found a mean value for all soils of -24% (95% CI: -33%, -14%; Nc=65). However, they found significant differences depending on the soil pH, with a positive effect for alkaline soils (-41%; 95% CI: -52%, -27%; Nc=43) but not significant effect for acid soils (-12%; 95% CI: -24%, 0.1%; Nc=22). This can be in relation with the not significant effect on grasslands, which in Europe are often located on acid soils.

For nitrification inhibitors, there is a mean effect of almost -60% effect for grasslands and drylands. The effect is lower for paddies (-33%; 95% CI: -43%, -23%). Interestingly, the reduction of N2O emission by nitrification inhibitors was more important with higher baseline N2O emission (i.e., emissions from the conventional fertiliser treatment). For example, the average N2O reduction was 67% for situations where baseline emissions exceeded 20 kg N/ha, compared to a 50% reduction for baseline emissions of <1 kg N/ha. Differences were also observed between different types of nitrification inhibitors used: from a mean decrease of -22.4% for Neem to a mean of -62.6% for 3,4-Dimethylpyrazole phosphate (DMPP).

For double inhibitors, a clear positive effect of approximately -50% reduction (95% CI: -55%,-41 and -44%) is observed for grassland and drylands.

Table 11 Effect (% change) of controlled-release fertilisers, urease inhibitors, nitrification inhibitors, and double inhibitors on N2O emissions for major crop groups. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Specific practice		CI_LOW	Mean	CI_HIGH	Effect
Controlled-release fertiliser	Overall	-46.6	-37.5	-27.1	Positive
	Grasslands	-86.2	-73.5	-8.6	Positive
	Drylands (Wheat, maize, vegetables)	-40.2	-31	-20.8	Positive
	Paddies	-86.6	-74.5	-51.0	Positive
Urease inhibitors	Overall	-39.9	-20.6	+1.1	Non-significant
	Grasslands	-11.4	+3.9	+33.6	Non-significant
	Drylands (Wheat, maize, vegetables)	-59.3	-37.2	-7.1	Positive
Nitrification inhibitors	Overall	-60	-56.6	-52.8	Positive
	Grasslands	-61.7	-58.2	-54.5	Positive
	Drylands (Wheat, maize, vegetables)	-64	-57.7	-48.4	Positive
	Paddies	-42.9	-33	-22.9	Positive
Double Inhibitors	Overall	-53.5	-48.8	-43.7	Positive
	Grasslands	-55.3	-50.1	-43.8	Positive
	Drylands (Wheat, maize, vegetables)	-55.5	-48.8	-41.2	Positive

Source: T. Li et al. 2018.

3.5.2 Effects on ammonia emissions

The source of data is the synthesis paper by Ti et al. 2019 for the first three techniques and T. Li et al. 2018 for double inhibitors. The assessed metric is ammonia (NH3) emission per hectare and year in production systems of major crops. Major crops or crop groups considered are: vegetables, pastures, rice, maize, wheat, and the aggregate of these ones for the first three techniques; and three main land uses (grasslands, drylands and paddy) for double inhibitors. Extracted data are presented in Table 12.

Table 12 Effect (% change) of controlled-release fertilisers, urease inhibitors, nitrification inhibitors, and double inhibitors on NH3 emissions for major crop groups. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Specific practice		CI_LOW	Mean	CI_HIGH	Effect
Controlled-release fertilisers *Source:* Ti et al. 2019	Overall	-58.0	-53.1	-47.8	Positive
	vegetables	-60.6	-43.9	-19.5	Positive
	pastures	-65.8	-53.3	-34.9	Positive
	rice	-76.0	-63.2	-43.5	Positive
	maize	-60.6	-43.1	-16.5	Positive
	wheat	-80.3	-33.2	124.3	Non-significant

Specific practice		CI_LOW	Mean	CI_HIGH	Effect
Urease Inhibitors *Source:* Ti et al. 2019	Overall	-63.3	-56.80	-48.8	Positive
	vegetables	-99.3	-51.9	65.1	Non-significant
	pastures	-86.5	-75.8	-56.1	Positive
	rice	-79.9	-72.7	-60.6	Positive
	maize	-55.7	-37.7	-11.8	Positive
	wheat	-58.8	-33.2	9.0	Non-significant
Nitrification inhibitors *Source:* Ti et al. 2019	Overall	16.5	41.7	77.4	Negative
	Individual crop types/groups	-	-	-	Non-significant
Double Inhibitors *Source:* T. Li et al. 2018	Overall	-64.1	-52.7	-40.4	Positive
	Grasslands	-48.1	-36.8	-24.2	Positive
	Drylands (wheat, maize, vegetables)	-87.2	-69.9	-36.7	Positive
	Paddies	-85.3	-75.5	-58.8	Positive

Source: Ti et al. 2019 *and* T. Li et al. 2018.

For controlled-release fertilisers, the mean effect on NH3 emissions is a reduction of -53% (95% CI: -58.0%, -47.8%), when examining all crop groups together. This overall positive effect holds true for all examined crop groups when examined separately, with the exception of wheat for which a non-significant effect was found.

For urease inhibitors, the mean effect of NH3 emission is -56.8% (95% CI: -63.3%, -48.8%) when examining all crop groups together. This overall positive effect holds true for all crop groups when examined separately, except vegetables and wheat, for which a non-significant effect was found. Another synthesis paper (T. Li et al. 2018) found that the higher the baseline NH3 emission, the greater the percentage reduction by urease inhibitors. Additionally, areas with annual rainfall <800 mm benefitted more from urease inhibitors, with a 70% reduction in NH3 emission, compared to a 50% reduction in areas with higher annual rainfall. This is consistent with the aggregated results showing a stronger NH3 loss reduction effect of urease inhibitors on in rainfed than in irrigated systems (T. Li et al. 2018).

For nitrification inhibitors, results are significantly different from zero only when the overall effect is considered (i.e., when examining all crops together). In this case, the effect is negative, as results indicate an average increase of ammonia emission of 41.7% (95% CI: 16.5% – 77.4%). Results per crop groups are based on a limited number of studies so they are not shown.

For double inhibitors, a clear positive effect for all examined land use groups is reported. In particular, in drylands the mean effect size is -70% (95% CI: -87.3%, -36.7%).

In Europe, nitrogen losses as NH3 from fertilisers application are estimated to be on average 17.8 kg of N/ha (median), corresponding to circa 13% of the total applied N (Pan et al. 2016). Applying this baseline value, *indicative* absolute values of potential reduction/increase of NH3 emissions from fertiliser application due to the use of each technique is:

- Controlled-release fertilisers: -9.5 Kg N/ha/yr (95% CI: -10.3 – 8.5)

— Urease inhibitors: -10.1 Kg N/ha/yr (95% CI: -11.3 – 8.7)

— Nitrification inhibitors: +7.4 Kg N/ha/yr (95% CI: 2.9 – 13.8)

— Double inhibitors (in drylands): -12.5 Kg N/ha/yr (95% CI: -15.2 – -6.5)

3.5.3 Effects on the nutrient balance

Selected papers for data extraction presented in Table 13 and Table 14 are:

— L. Xia et al. 2017 for controlled-release fertilisers and urease inhibitors;

— Yang et al. 2016 and Sha et al. 2020 for nitrification inhibitors but as Yang et al. 2016 provide results for N uptake only for Dicyandiamide (DCD) and DMPP, we selected the overall value by Qiao et al. 2015.

— For double inhibitors no values on N uptake are available. The best paper among the two providing results on N fertiliser recovery (NR, see Annex 1) is Sha et al. 2020.

Table 13 Effect (% change) of controlled-release fertilisers, urease inhibitors, nitrification inhibitors and double inhibitors on N use efficiency (crop N fertiliser recovery) or N plant uptake, for major crop groups. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Technique and source	Crop or subtechnique	Metric	CI_LOW	Mean	CI_HIGH	Effect
Controlled-release fertilisers *Source:* L. Xia et al. 2017	Overall	Uptake	9.0	10.9	12.8	Positive
	Overall	NUE (NR)	29.6	34.3	39.5	Positive
	Rice	Uptake	7.3	11.4	15.6	Positive
	Wheat	Uptake	7.4	10.6	13.8	Positive
	Maize	Uptake	7.9	10.7	14	Positive
	Rice	NUE (NR)	29.5	34.3	39.4	Positive
	Wheat	NUE (NR)	28.1	36.3	44.8	Positive
	Maize	NUE (NR)	22.6	29.9	37.5	Positive
Controlled-release fertilisers (polymer coated fertilisers) *Source:* T. Li et al. 2018	Overall	NUE (NR)	3.7	8.7	13.9	Positive
	Maize & wheat	NUE (NR)	-10.5	-2.1	6.4	Non-significant
	Vegetables	NUE (NR)	3.7	11.2	18.9	Positive
	Grasslands	NUE (NR)	-23.3	-4.8	16.3	Non-significant
	Paddy	NUE (NR)	15.6	26.4	38.7	Positive
Urease inhibitors *Source:* L. Xia et al. 2017	Overall- NUE (NR)	NUE (NR)	22	31.3	42.0	Positive
	Rice	Uptake	3.2	5.8	8.5	Positive
	Wheat	Uptake	0.3	6.6	12.8	Positive
	Rice	NUE (NR)	-1.9	7.1	16.4	Non-significant
	Wheat	NUE (NR)	15.9	25.3	39.5	Positive
	Maize	NUE (NR)	24.7	39.6	56.6	Positive

Technique and source	Crop or subtechnique	Metric	CI_LOW	Mean	CI_HIGH	Effect
Urease inhibitors *Source:* Sha et al. 2020	Overall	NUE (NR)	18.7	24.1	29.8	Positive
	Dryland	NUE (NR)	10.7	16.5	21.7	Positive
	Paddy	NUE (NR)	24.5	34.9	45.2	Positive
Urease inhibitors *Source:* T. Li et al. 2018 – very much dependent on inhibitor type	Overall	NUE (NR)	13.1	20.1	28.1	Positive
	maize & wheat	NUE (NR)	4.9	14.3	28.4	Positive
	Vegetables	NUE (NR)	NA	NA	NA	Not available
	Grasslands	NUE (NR)	9.5	17.9	26.7	Positive
	Paddy	NUE (NR)	9.3	28.7	53.6	Positive
Nitrification inhibitors *Source:* Qiao et al. 2015	Overall	Uptake	11	15	20	Positive
	Overall	NUE (NR)	34	58	93	Positive
	Cropland	Uptake	12	18	25	Positive
	Cropland	NUE (NR)	22	33	46	Positive
	Pasture	Uptake	6	12	19	Positive
	Pasture	NUE (NR)	65	297	923	Positive
Nitrification inhibitors *Source:* Sha et al. 2020	Overall	NUE (NR)	4	10.5	18	Positive
	Dryland	NUE (NR)	-3.6	2.7	10.2	Non-significant
	Paddy	NUE (NR)	18.8	33.2	52.1	Positive
Nitrification inhibitors *Source:* T. Li et al. 2018 – depends a lot on inhibitor type	Overall	NUE (NR)	15.4	31.7	53.9	Positive
	Maize & wheat	NUE (NR)	-3.9	12.7	41.9	Non-significant
	Vegetables	NUE (NR)	-11.6	-2.1	8.3	Non-significant
	Grasslands	NUE (NR)	22.5	48.4	86.1	Positive
	Paddy	NUE (NR)	4.3	11.1	19.5	Positive
Double Inhibitors *Source:* Sha et al. 2020	Overall	NUE (NR)	29	47.6	75.4	Positive
	Dryland	NUE (NR)	NA	NA	NA	Not available
	Paddy	NUE (NR)	NA	NA	NA	Not available
Double Inhibitors *Source:* T. Li et al. 2018	Overall	NUE (NR)	15.3	22.1	29.6	Positive
	Maize & wheat	NUE (NR)	-4	2.8	10.2	Non-significant
	Vegetables	NUE (NR)	NA	NA	NA	Not available
	Grasslands	NUE (NR)	26.7	32.7	39.9	Positive
	Paddies	NUE (NR)	18.9	43.3	76.3	Positive

Source: references in the first column.

Controlled-release fertilisers increase nitrogen uptake by 11% (95% CI: 9%, 13%); N-(n-butyl) thiophosphoric triamide (NBPT) urease inhibitor by 6% (95% CI: 3.5%, 8.5%); nitrification inhibitors by 15% (95% CI: 11%, 20%); there were no results for double inhibitors.

Given that some results depend greatly on the type of urease inhibitor used, we also provide results by inhibitor type (Table 14). For urease inhibitors, we only have a mean value for uptake from L. Xia et al. 2017 of +6% for (NBTP) but values for NR can vary between +30% of NBTP and +14% of hydroquinone (HQ) according to Sha et al. 2020.

For nitrification inhibitors, the mean value has been found between +11 and +18% for DCD, while it is very low and non-significant for DMPP (T. Li et al. 2018; Qiao et al. 2015; Yang et al. 2016.) For double inhibitors there was not information on N uptake, but T. Li et al. 2018 repot high differences in NUE(NR), from 19% (95%CI: 12%, 26%) for DCD+NBPT to 62% (95%CI: 32%, 97%) for other products.

Table 14 Effect (% change) of controlled-release fertilisers, urease inhibitors, nitrification inhibitors and double inhibitors on N use efficiency or N plant uptake for different inhibitor types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Technique / source	Subtechnique	Metric	CI_LOW	Mean	CI_HIGH	Effect
Controlled-release fertilisers *Source:* L. Xia et al. 2017	Overall	Uptake	9.0	10.9	12.8	Positive
	Overall	NUE (NR)	29.6	34.3	39.5	Positive
Controlled-release fertilisers *Source:* T. Li et al. 2018	Overall (polymer coated fertiliser)	NUE (NR)	3.7	8.7	13.9	Positive
Urease inhibitors *Source:* L. Xia et al. 2017	NBPT	Uptake	3.4	5.9	8.5	Positive
	Overall	NUE (NR)	22	31.3	42.0	Positive
	NBPT	NUE (NR)	29.3	41.6	55.4	Positive
	HQ	NUE (NR)	5.7	15.7	29.7	Positive
Urease inhibitors *Source:* Sha et al. 2020	Overall	NUE (NR)	18.7	24.1	29.8	Positive
	Phenyl-phosphoryl-diamine (PPD)	NUE (NR)	11.6	20.5	30.0	Positive
	NBTP	NUE (NR)	22.2	30.3	39.5	Positive
	HQ	NUE (NR)	5.3	14.2	26.8	Positive
Urease inhibitors *Source:* T. Li et al. 2018 – depends a lot on inhibitor type	Overall	NUE (NR)	13.1	20.1	28.1	Positive
	NBPT	NUE (NR)	10.3	17.2	24.7	Positive
	PPD	NUE (NR)	-16.7	0.1	20.4	Non-significant
	Limus	NUE (NR)	6.2	20.9	40.5	Positive
Nitrification inhibitors *Source:* Yang et al. 2016	DCD	Uptake	8.2	18.1	19.1	Positive
	DMPP	Uptake	-1.9	1.9	6.5	Non-significant
Nitrification inhibitors *Source:* Qiao et al. 2015	Overall	Uptake	11	15	20	Positive
	Overall	NUE (NR)	34	58	93	Positive
	DCD	Uptake	9	16	23	Positive
	DCD	NUE (NR)	25	90	226	Positive
	DMPP	Uptake	-5	6	19	Non-significant
	DMPP	NUE (NR)	1949	2611	3487	Outlier values, (only 2 obs.)
	Nitrapyrin	Uptake	NA	NA	NA	Not available
	Nitrapyrin	NUE (NR)	-10	7	32	Non-significant

Technique / source	Subtechnique	Metric	CI_LOW	Mean	CI_HIGH	Effect
	Organic NI	Uptake	16	24	32	Positive
	Organic NI	NUE (NR)	33	50	70	Positive
Nitrification inhibitors *Source:* Sha et al. 2020	Overall	NUE (NR)	4	10.5	18	Positive
	DCD	NUE (NR)	-1.4	11.4	26.6	Non-significant
	DMPP	NUE (NR)	-4.1	2.8	9.6	Non-significant
	Nitrapyrin	NUE (NR)	4.0	13.8	28.4	Positive
Nitrification inhibitors *Source:* T. Li et al. 2018	Overall	NUE (NR)	15.4	31.7	53.9	Positive
	DCD	NUE (NR)	17.4	34.4	59	Positive
	DMPP	NUE (NR)	-6.0	31.4	111	Non-significant
	Else	NUE (NR)	4.6	12.2	20.7	Positive
Double Inhibitors *Source:* Sha et al. 2020	Overall	NUE (NR)	29	47.6	75.4	Positive
Double Inhibitors *Source:* T. Li et al. 2018	Overall	NUE (NR)	15.3	22.1	29.6	Positive
	DCD+NBPT	NUE (NR)	12.3	19	26	Positive
	Else	NUE (NR)	32.3	61.9	97.4	Positive

Source: references in the first column.

3.5.4 Effects on nutrient leaching and run-off

The selected synthesis paper for data extraction is T. Li et al. 2018 for all techniques. In the case of nitrification inhibitors, two synthesis papers are selected: Yang et al. 2016 and Qiao et al. 2015.

Non-significant effect on nitrate leaching was found for urease inhibitors and for double inhibitor overall, even though for double inhibitors a positive effect was found for grasslands and a negative one for cropland (dryland).

Only two practices have proven effective to decrease NO3- leaching: controlled-release fertilisers by -66% (95 CI: -81%, -42%) and nitrification inhibitors by -57% (95% CI: -65%, -38% for DCD and -63%, -52% for DMPP) (Table 15).

Table 15 Effect (% change) of controlled-release fertilisers, urease inhibitors, nitrification inhibitors and double inhibitors on NO3- leaching for major crop groups. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Intervention and source	Factor	CI_LOW	Mean	CI_HIGH	Effect
Controlled-release fertilisers *Source:* T. Li et al. 2018	Overall	-81.3	-65.9	-41.8	Positive
	dryland	-66.9	-42.3	-20.2	Positive
	paddy	-96.8	-91.9	-83.6	Positive
Urease inhibitors *Source:* T. Li et al. 2018	Overall	-83.4	-38.9	45.1	Non-significant
	dryland	-61.6	-13.3	19.3	Non-significant
	grasslands	NA	-94.8	NA	Non-significant
	paddy	NA	NA	NA	Not available

Intervention and source	Factor	CI_LOW	Mean	CI_HIGH	Effect
Nitrification inhibitors *Source:* T. Li et al. 2018	Overall – NO3-	-50.4	-44.5	-38.1	Positive
	Dryland – NO3-	-51.5	-35.3	-12.6	Positive
	Grasslands – NO3-	-51.8	-45.2	-38.3	Positive
	Paddy – NO3-	-58.3	-52.3	-44.9	Positive
Nitrification inhibitors *Source:* Yang et al. 2016	DCD – NO3-	-64.8	-56.9	-38.1	Positive
	DMPP - NO3-	-63.2	-57.6	-51.7	Positive
	DCD - DIN	-47.7	-38.0	-23.2	Positive
	DMPP-DIN	-53.0	-46.9	-39.3	Positive
Nitrification inhibitors *Source:* Qiao et al. 2015	Overall – NO3-	-59	-47	-32	Positive
	Overall - DIN	-56	-48	-38	Positive
	Cropland – NO3-	-64	-37	+9	Non-significant
	Pasture – NO3-	-63	-52	-37	Positive
	Cropland - DIN	-47	-38	-29	Positive
	Pasture - DIN	-63	-53	-42	Positive
	DCD – NO3-	-61	-45	-24	Positive
	DMPP - NO3-	-59	-54	-48	Positive
	DCD - DIN	-63	-53	-41	Positive
	DMPP-DIN	-45	-38	-29	Positive
Double inhibitors *Source:* T. Li et al. 2018	Overall	-53.9	-29.3	4.2	Non-significant
	dryland	17.9	58	111.2	Negative
	grasslands	-68.7	-50.5	-31.7	Positive

Source: references in the first column.

3.6 Organic fertilisation

Organic fertilisation is the application to soils of plant or animal-derived materials containing organic forms of nutrients that soil microorganisms decompose, making them available for use by crops (FAO 2009). This section covers the application of organic fertilisers, alone or combined with mineral fertilisation, from different animal (cattle, pig, sheep, poultry, and earthworms), plant and mixed (municipal and agro-industrial waste) sources used both as composted and non-composted manures. The fertilisation with green manure or green manuring (organic fertilisation from crops grown on the same field as cover-crops) is addressed as a separate practice in section 3.7.

To assess the quantitative effect of the practice on the environment, the used comparator is the application of mineral fertilisers alone.

3.6.1 Effects on greenhouse gas emissions

The selected paper for data extraction is Linquist et al. 2012, that reported a negative effect of organic fertilisation as compared to mineral fertilisation in rice systems. They found that farmyard manure forming all or part of the total N rate increased CH4 emissions by 26% (95% CI: 12%, 47%) when compared to a treatment receiving only urea N at the same total N rate. Wei et al. 2020 found a non-significant effect on CH4 emissions in maize systems (Table 16).

Table 16 Effect (% change) of organic fertilisation as compared to mineral fertilisation on GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

GHG	Source	Population	CI_LOW	Mean	CI_HIGH	Nc	Effect
CH4	Linquist et al. 2012	Rice	12	26	47	14	Negative
	Wei et al. 2020	Maize	-	-	-	-	Non-significant

Source: references in the second column.

3.6.2 Effects on soil organic carbon

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the overall effect of organic fertilisation compared to mineral fertilisation is positive. The selected synthesis paper for data extraction in Table 17 is Y. Chen et al. 2018, a paper that covers a wide geographic spectrum (but includes in their analysis also green manure).

The authors (Y. Chen et al. 2018) compare the effect of organic amendments (OA) and organic amendments with inorganic fertiliser (OA+IF) for different levels of inorganic N application rates, with the standard control of inorganic fertiliser. They also compare with a nil control of no fertilisation, but these results are not reported here. Overall, there was a significant mean relative increase in final SOC content, with 29% (95% CI: 27%, 31%) with the use of OA and OA+IF as compared to the standard control. The variables that explained the greatest variation in organic carbon when compared with either nil or standard controls were (1) initial SOC, (2) climate, (3) initial total N, (4) soil texture, (5) soil order, and (6) initial soil pH. The type, rate, and duration of organic amendment application, as well as the inorganic N application rate added along with the organic amendment also explained a considerable amount of SOC variation. The types of organic amendments that were compared with the standard control are: straw (crop straw, straw husk, straw compost), manure (farmyard manure, livestock manure, manure-based materials such as composts) and green manure (green manure crops such as rape crop, lantana and some leguminous plants). Therefore, the results would be a bit higher if green manure were not included, the effect for animal manure being +39% (95% CI: 35%, 43%).

It shall be noted that > 90% of the straw-treated soils received additional application of inorganic fertiliser compared with >71% of those treated with manure applied soils. The addition of organic amendments to soils located in warm regions (Mediterranean/subarid/arid, tropical and semi-tropical) had the greatest relative increases in final SOC (more than 30% when compared with the standard control), whereas relative increases were below these threshold values in soils under humid-temperate and continental-type climates.

Table 17 Effect (% change) of organic fertilisation on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

		CI_LOW	Mean	CI_HIGH	Nc	Effect
Overall		27	29	31	150	Positive
Tropical		36.7	45.0	54.3	18	Positive
Mediterranean, arid and semi-arid		23.0	31.3	40.4	16	Positive
Subtropical		27.6	31.6	35.5	68	Positive
Continental		17.4	22.8	28.3	31	Positive
Humid-temperate		7.0	13.9	21.4	17	Positive
Years of application	20-10	34	38	42	82	Positive
	10 - 20	19	23	27	70	Positive
Manure		35	39	43	78	Positive
Straw		8	13	19	33	Positive
Green manure*		15	23	31	21	Positive

Source: Y. Chen et al. 2018. **Please note that green manure is addressed in a specific independent section of the document. This value is shown here just to highlight that the overall value also includes green manure.*

3.6.3 Effects on ammonia emissions

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the overall effect of organic fertilisation compared to mineral fertilisation alone is positive, with a significant reduction of ammonia (NH3) emissions, although Ti et al. 2019 found that the significant reduction only occurs if the mineral fertiliser substitution rate is 100%.

The selected paper according to the established criteria is Ti et al. 2019 (Table 18). When comparing with 50, 75 and 100% substitution of synthetic N fertiliser by livestock manure (with an equivalent total N rate), only the 100% substitution significantly reduced NH3 emissions (by -67%; 95% CI: -79%, -47%). The retention of crop residues, such as wheat straw and leaf vegetable, on the soil surface could increase NH3 emission by 22.3% on average, but it is not statistically significant as compared with the control.

Table 18 Effect (% change) of organic fertilisation on ammonia emissions as compared to urea. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Specific practice		CI_LOW	Mean	CI_HIGH	Effect
Organic fertilisation	Residue retention (straw)	-5.7	22.5	59.4	Non-significant
	Manure 100 %	-78.9	-67.2	-46.9	Positive
	Manure 75 %	-69.7	-16.2	133.2	Non-significant
	Manure 50 %	-44.5	-16.8	25.6	Non-significant

Source: Ti et al. 2019.

3.6.4 Effects on the nutrient balance

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the overall effect of organic fertilisation on N use efficiency is mostly negative but dependent on N fertilisation rates (Table 19).

Wei et al. 2020 found that the substitution of mineral fertiliser with organic fertiliser had non-significant effect on nitrogen use efficiency in maize systems. However, the effect of substitution of mineral fertilisation with organic fertilisers on nitrogen use efficiency tended to increase with increasing N fertilisation rate[11]. The low fertilisation rate had a negative impact on nitrogen use efficiency (−17.3%). Given that in EU the Nitrates Directive does not allow putting more than 170 kg N/ha, we can consider that for all regions/countries where this threshold is respected the effect would be negative. The optimal fertilisation rate (180-250 kg N/ha in EU) compromised nitrogen use efficiency to some extent. The high fertilisation rate significantly increased nitrogen use efficiency (8.77%). The authors did not take the effects of organic treatment (compost or digestate) into account due to the limited amount of available data on maize production. The studies included in their analysis included the following types of organic fertilisers: animal manure (47%), compost (37%), commercial organic fertiliser (e.g., industrially processed, standardized poultry or livestock manure; 9%), digestate (5%), slurry (2%);

Huygens et al. 2020 cover all arable crops. They found that the response ratios of nitrogen utilization efficiency[12] for organic fertilizers were lower than those of mineral fertilizers, but results depended on the type of organic fertilizer. Highest values corresponded to manure-derived N fertilizers that were more mineral-like or were dominated by urea, an easily degradable mineral N precursor. The processed manure materials (with mineral-N/total-N ratio < 90% or TOC:TN > 3) showed a response ratio of nitrogen utilization efficiency value below 75% (significantly lower than mineral fertilizers), with the lowest values observed for materials of high organic matter content, such as compost and solid digestate fractions. Manure and processed manure fertilizers with high organic content are: Digestate liquid fraction (n=11); Raw manure (n=78); Acidified manure (n=1); Manure solid fraction (n=17); Digestate slurry (n=65); Manure liquid fraction (n=17); Digestate solid fraction (n=6); Compost (n=5).

Table 19 Effect (% change) of organic fertilisation on nitrogen use efficiency. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Population	Metric and Source	Factors	CI_LOW	Mean	CI_HIGH	Nc	Effect
Arable crops	NUE (NR) *Source:* Huygens et al. 2020	Overall	NA	NA	NA	NA	Negative
		Organic fertiliser type					
		Raw manure	-34.4	-28.7	-22.7	78	Negative

[11] The N fertilisation rate for maize production was categorized as low, optimal, or high, where these categories were region-specific: the optimal N rates for Africa, Asia, and other regions were 50–80, 150–210, and 180–250 kg N ha−1, respectively.

[12] Plant N use efficiency was calculated as the difference in N uptake between fertilised and unfertilised plants, expressed relative to the fertiliser application rate. As the NUE was corrected based on the N uptake of a blank without fertiliser, it was referred to as blank-corrected NUE (NUE(bc)).

		Manure liquid fraction	-48.6	-41.3	-33.0	17	Negative
		Manure solid fraction	-38.8	-31.4	-23.4	17	Negative
		Compost	-71.7	-63.1	-51.5	5	Negative
		Digestate slurry	-38.5	-31.6	-24.0	65	Negative
Maize systems	NUE (NR) *Source:* Wei et al. 2020	Overall	-5.0	0.6	6.2	185	Non-significant
		Fertilisation rate					
		Low (< 180 kg N/ha in EU)	-27.0	-17.3	-7.6	30	Negative
		Optimal (180-250 kg N/ha in EU)	-14.8	-4.1	6.6	48	Non-significant
		High (> 250 kg N/ha in EU)	0.9	8.8	16.6	107	Positive

Source: references in the second column.

3.6.5 Effects on nutrient leaching and run-off

Results in the literature quantify the decrease in leaching of nitrates ($NO3^-$) and sometimes also in leaching of ammonium (NH_4^+), total dissolved inorganic nitrogen (DIN) and N runoff. We show results, where available, for $NO3^-$ leaching and for DIN leaching (Table 20).

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the overall effect is positive.

One synthesis paper (Wei et al. 2020) analyses the link between the substitution of mineral fertiliser with organic fertilisers and nitrogen losses by leaching and run off with positive results (i.e. decreased losses). Wei et al. 2020 performed a meta-analysis of 133 maize studies, conducted worldwide, to assess maize yield and environmental performance with substitution of mineral fertiliser with organic fertiliser (Table 20). This substitution is performed at different substitution rates (Rs), defined as organic nitrogen (N) input/total N applied (%)). At an equivalent nitrogen (N) rate, substituting mineral fertiliser with organic fertiliser reduced N leaching and runoff by 26.9%. The maximum decrease in N runoff and leaching (by 45.5%) was achieved at the optimal fertilisation rate. Results also changed with the substitution rate and the treatment duration. Partial substitution (0 < Rs < 100) of organic with mineral fertiliser did not affect N runoff and leaching except for substitution at a rate of 40–60%, at which N runoff and leaching significantly decreased by 41.9%. Full substitution of organic with mineral fertiliser significantly decreased runoff and leaching by 50.0%.

Long-term organic fertiliser substitution (≥ 3 years) significantly decreased runoff and leaching were by 47%; this was significantly greater than the 11.2% reduction under short-term treatments (<3 years).

Table 20 Effect (% change) of organic fertilisation on N leaching and runoff. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons. Rs: mineral fertiliser substitution rate.

		CI_LOW	Mean	CI_HIGH	Nc	Effect
N leaching & run-off	N leaching & runoff overall	-39.6	-26.9	-14.1	66	Positive
	Low fertilisation rate	-62.9	-18.7	26.1	10	Non-significant
	Optimal fertilisation rate	-64.2	-45.5	-26.5	13	Positive
	High fertilisation rate	-39.3	-22.2	-5.1	43	Positive
	0 < Rs ≤ 40	-20.6	-1.7	16.6	28	Non-significant
	40 < Rs ≤ 60	-64.8	-41.9	-18.6	11	Positive
	60 < Rs < 100	-39.5	-3.2	33.0	7	Non-significant
	Rs = 100	-69.7	-50.0	-29.5	20	Positive
	Long term ≥ 3 years	-66.1	-47	-27.5	25	Positive
	Short term < 3 years	-25.2	-11.2	2.9	41	Non-significant

Source: Wei et al. 2020.

3.7 Green manuring

Fertilisation with green manure, or green manuring, refers to the use of herbaceous plants, often but not necessarily planted as cover crops, as source of nutrients for the subsequent cash crops. Plants are killed (either mechanically or chemically) and then usually incorporated into the soil, or in some cases left on it. Leguminous plants are often used to this purpose due to their ability of fixing atmospheric nitrogen which is subsequently made available as nutrient for the following crops. From an environmental point of view, the purpose of the practice is to replace, partially or totally, mineral-nitrogen (N) fertilisers. Therefore, examined synthesis papers included only studies where cover crops were explicitly used as green manure, assessing their effects by comparing their use to either mineral-N fertilisation or to absence of mineral-N fertilisation.

3.7.1 Effects on greenhouse gas emissions

Results are provided for N2O and CH4 emissions in Table 21. The first selected synthesis paper for N2O emissions is Basche et al. 2014. Forty percent of the studies assessed in this analysis showed that cover crops not harvested decreased N2O emissions and 60% of the studies showed that cover crops not harvested increased N2O emissions. There are both environmental and management factors that modified the impact of cover crops not harvested on N2O emissions, including fertiliser N rate, soil incorporation, and the period of measurement and rainfall. At low N rates, legume cover crops not harvested had higher relative N2O emissions. N2O emissions of non-legume cover crops not harvested were negative at low N rates but increased as N rate increased. In general, it seems that cover crops not harvested have a greater potential to reduce N2O emissions when non-legume species are utilized and cover crop residue is not incorporated into the soil.

With the same level of fertilisation both in treatment and control, Muhammad et al. 2019 found an overall decrease of N2O emissions, but also that the effect varies depending on several factors. In particular, N2O emission increased when leguminous crops were used and decreased when non-leguminous crops were used. Incorporation of crop residues increased N2O emissions while leaving them on the soil decreased them.

Results are also provided for paddy rice fields on the effect of legume incorporated to the soil both on N2O and CH4 emissions (Linquist et al. 2012).

Table 21 Effect (% change) of green manuring on GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

GHG	Technique type or crop type	CI_LOW	Mean	CI_HIGH	Effect
N2O *Source:* Basche et al. 2014	Overall	NA	NA	NA	Not available
	Legume	469	490	511	Negative
	Non-legume	1.4	7.3	13	Negative
	Biculture (mixed legume and non-legume)	-23.5	8.8	54.7	Non-significant
	Incorporation	482	494	507	Negative
	Surface	-48	-17	33.5	Non-significant
N2O *Source:* Muhammad et al. 2019	Overall	-26.8	-16.3	-5.6	Positive
	Leguminous cover-crop	29.8	60.8	96.0	Negative
	Non-leguminous cover-crop	-46.2	-36.5	-25.8	Positive
	Incorporated	109.0	133.3	166.0	Negative
	Surface	-62.3	-54.9	-45.9	Positive
N2O Rice systems only *Source:* Linquist et al. 2012	Paddy rice with *Sesbania* (legume) incorporated to the soil	-41	-12	12	Non-significant
CH4 from paddy rice *Source:* Linquist et al. 2012	Paddy rice with *Sesbania* (legume) incorporated to the soil	71	192	396	Negative

Source: references in the first column.

3.7.2 Effects on soil organic carbon

Results are typically provided for soil organic carbon concentration, soil organic carbon stocks and soil carbon sequestration rates. SOC concentrations describe the organic carbon concentration on a weight by weight basis, SOC stocks on a weight by area basis, and C sequestration rates, on weight by area on an annual basis. Results on SOC stock change have been found for green manure.

Two synthesis papers have been selected: Muhammad et al. 2019 and Shackelford et al. 2019. The first one includes data from 41 experimental sites from various regions of the world with different soil and climatic conditions where cover crops were rotated with various cash crops. The second one focused on arable farmland in Mediterranean climate regions, in California and the Mediterranean, based on 326 experiments reported in 57 publications. They all refer to arable land, where the practice of cover crop is compared to the effect of bare soil. All or part of the N fertiliser rate was added as green manure (Table 22).

Table 22 Effect (% change) of green manuring on soil organic carbon (stock per hectare). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Biogeographic regions	Technique or crop type	CI_LOW	Mean	CI_HIGH	Effect
All regions *Source:* Muhammad et al. 2019	Overall (all types of green manure, all treatments)	9.5	15.2	22.4	Positive
	Removed from the soil (only aboveground biomass)	-1.4	1.2	3.7	Non-significant
	Incorporated into the soil	14.1	22.1	31.5	Positive
	Legume	2.4	10.7	22.4	Positive
	Non-legume	0.4	12.9	28.9	Positive
	Mixed legume & not	13.5	22.1	31.9	Positive
Mediterranean climate regions *Source:* Shackelford et al. 2019	All types of green manure, all treatments	4	9	15	Positive

Source: references in the first column.

The results of non-harvested cover crops compared to no cover crop were positive for SOC. Overall, non-harvested cover crops increased SOC by 15% (95% CI: 9.5%, 22.4%) compared to no cover crop (Muhammad et al. 2019). The mixed cover crop resulted in a higher SOC than legume or non-legume cover crop alone. Incorporation of cover crop residue into the soil increased SOC compared to residue removal.

3.7.3 Effects on the nutrient balance

No synthesis paper explicitly reported results in terms of gross nutrient balance or nutrient surplus. Nevertheless, impacts have been found on other metrics related to the gross nutrient balance (Annex 1): Partial Factor Productivity of Nitrogen (only for rice in China), that is a specific measure of the nitrogen use efficiency (yield divided by total N fertilisation including nitrogen from green manure)

Ding et al. 2018 found that the fertilisation of rice fields in China using herbaceous cover crops as green manure (no distinction between legume and non-legume as experiments included Chinese milk vetch, oilseed rape and rye-grass), compared to harvested cover crops, increases Partial Factor Productivity of Nitrogen by 4.9% (95% CI: 3.0%, 6.8%). Experimental and control treatments in each study received the same amount of inorganic NPK fertiliser (Table 23).

Table 23 Effect (% change) of green manuring on N use efficiency. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Population	Cover crop / technique type	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Rice in China	Leguminous and non-leguminous, non-harvested compared to harvested cover crop	PFPN	3.0	4.9	6.8	84	Positive

Source: Ding et al. 2018

3.7.4 Effects on nutrient leaching and run-off

Results in the literature quantify the decrease of leaching of nitrates (NO3-), of ammonium (NH4+), of total dissolved inorganic nitrogen (DIN) and N runoff. We show results, where available, for NO3- leaching and for DIN leaching in Table 24.

A synthesis paper (Tonitto et al. 2006) collected evidence on the effect of green manuring on nitrogen leaching. Tonitto et al. 2006 concluded that this practice has a statistically significant potential in reducing nitrogen leaching relative to conventional fertiliser-based systems due to increased organic nitrogen retention. Their comprehensive review of the literature (including several sites in North-America and one in Brazil) revealed few studies where leaching or soil inorganic N was simultaneously measured in conventional fertiliser-driven and legume-based systems. For this reason, the synthesis paper reported comparisons of Haber-Bosch[13] N fertilized systems in which the cash crop was followed by either a non-legume cover crop or a bare fallow. The meta-analysis compares nitrate leaching from non-legume cover crop followed by an inorganic N fertilized cash crop and nitrate leaching from legume cover crop followed by an unfertilized cash crop with the control, which was a winter bare fallow followed by a cash crop fertilized at recommended fertiliser rates . Nitrate leaching was clearly reduced when a cover crop was present. The meta-analysis showed a 70% overall reduction in leaching under the non-legume cover crop relative to bare fallow systems, with a narrow 95% CI (-77%, -62%; Nc=69). Though a limited sample size (Nc=8), the comparison between legume-fertilized systems and conventional systems showed a significant 40% reduction in nitrate leaching (95% CI: -55%, -15%).

Table 24 Effect (% change) of green manuring on NO3- leaching for major crop groups. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

		CI_LOW	Mean	CI_HIGH	Effect
North-America and north-western Europe	Overall	NA	NA	NA	Not available
	Legume	-54.7	-37.9	-15.2	Positive
	Non-legume	-77.5	-70.3	-62.5	Positive

Source: Tonitto et al. 2006

[13] The Haber-Bosch process is a process that fixes nitrogen with hydrogen to produce ammonia.

3.8 Soil amendment with Biochar

Biochar is charcoal that is produced by pyrolysis[14] of biomass; different biomass can be used to produce it, including manure, crop residues, sewage sludge, municipal wastes and compost. The use of Biochar has been proposed as a way to augment the long-term Carbon sink potential of soils and to ameliorate soil properties through the increase of soil organic matter.

3.8.1 Effects on greenhouse gas emissions

The effects of soil amendment with biochar on the emissions of different greenhouse gases from soils can vary. Contrasting results were found for CH_4 and CO_2 (soil respiration), with a prevalence of non-significant effects in both cases, while results for N_2O are more consistently positive. When considering the overall effect in terms of aggregated greenhouse gas potential, the resulting effect is also considered positive (Schievano et al. 2023).

The selected paper for overall GHG emissions is Q. Zhang et al. 2020. They found that the weighted response ratio of soil GHG emissions exhibited a difference among management strategies, biochar characteristics, and soil properties. In general, global warming potential[15] and the intensity of greenhouse gas emissions[16] significantly decreased, and crop yield greatly increased, with an average change of -23%, -41%, and 21%, respectively. The impact of biochar application on GWP was mainly on soil N_2O emission (Table 25).

To assess biochar characteristics, biochar was divided into five categories according to the raw material or feedstock for the meta-analysis: (1) shell residue (nutshell, oat hull, walnut shell, peanut hull, and bagasse); (2) straw waste (peanut straw, maize stalks, wheat straw, sorghum stalks, and rape stalks);(3) wood waste (bark, wood chips, pruning, trunk, and branches); (4) livestock manure (pig, cow and sheep manure); (5) municipal solid waste (household waste and excess sludge). Biochar derived from straw waste had a highly remarkable effect: it lowered GWP by 26% and increased crop yield by 21%, 3% more than biochar derived from crop residues. Biochar from straw waste and crop residue increased the crop yield by 18% and 22% and mitigated GHGI by 35% and 48%, respectively.

Table 25 Effect (% change) of biochar GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

GHG	Technique type or crop type	CI_LOW	Mean	CI_HIGH	Effect
All GHG emissions Global Warming Potential *Source:* Q. Zhang et al. 2020.	Overall	-29	-23	-16	Positive
	Straw waste	-33	-26	-18	Positive
	Crop residues	-18	-16	-13	Positive

[14] the thermal decomposition of materials at elevated temperatures in absence of oxygen

[15] GWP in CO_2-C equivalents (kg ha-1) was estimated for the emission of different greenhouse gases using the relative radiation effect of forcing factors by following equation $GWP = 25 \times R_{CH4} + 298 \times R_{N2O} + R_{CO2}$ where R_{CH4}, R_{N2O}, and R_{CO2} are the soil CH_4, N_2O, and CO_2 emissions (kg ha-1), respectively.

[16] Greenhouse gas emission intensity (GHGI) is the ratio of GWP to crop yield, which can be used to relate agricultural production to soil GHG emissions. GHGI was calculated as GHGI = GWP/crop yield. A smaller value of GHGI indicates that a lower GWP is produced to obtain the same crop yield.

GHG	Technique type or crop type		CI_LOW	Mean	CI_HIGH	Effect
All GHG emissions	Overall		-50	-41	-31	Positive
GHG emissions intensity	Straw waste		-46	-35	-24	Positive
Source: Q. Zhang et al. 2020.	Crop residues		-59	-48	-37	Positive
N2O emissions *Source:* Z. Wu et al. 2019	Overall		-23.4	-18.7	-13.9	Positive
	Rice		-23.8	-16.2	-8.3	Positive
	Wheat		-32.9	-20.6	-8.3	Positive
	Maize		-32.1	-21.8	-11.1	Positive
	Others		-34.9	-19.0	-2.8	Positive
	Experimental duration (years)	≤ 0.5	-21.0	-15.0	-9.1	Positive
		0.5 - 1	-36.8	-25.0	-13.1	Positive
		1 - 2	-51.5	-32.5	-13.1	Positive
		> 2	-42.0	-21.4	-0.8	Positive

Source: references in the first column.

For N2O emissions the selected paper is Z. Wu et al. 2019 (Table 25). According to the meta-analysis, biochar application significantly decreased N2O fluxes by 18.7% (95% CI: -23.4%, -13.9%). The soil and biochar characteristics and the experimental conditions were significantly influencing the response ratios of the soil GHG fluxes and the yield across all the studies. Soil texture, soil organic C and soil C/N were shown to be key factors influencing N2O emissions, while biochar with a higher pyrolysis temperature exerted a greater mitigation of N2O emissions in higher C/N soil with higher N application. Moreover, the long-term effects of biochar amendment on the GHG fluxes were confirmed by the meta-analysis. The decrease in N2O emissions in biochar-amended soil persisted over the studied years after a single amendment in field experiments.

3.8.2 Effects on soil organic carbon

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the effect of soil amendment with biochar on carbon storage in soil is positive, although some variables can lead to locally different results, including soil type and depth, climatic conditions, presence of crop rotation, N-fertilisation rate and biochar characteristics.

The selected paper (Albert et al. 2021) is a meta-analysis showing that biochar significantly increased the soil total organic carbon by 54.3% (95% CI: 47.6%, 61.3%) compared to the control (Table 26). The analysis included different crops: vegetables, grass, legume, maize, wheat, rice, and bamboo. However, when they identified the effect of biochar application on soil property changes, they did not divide the data into subgroups.

Due to the fact that results are very sensitive to a series of variables and that the selected synthesis paper (Albert et al. 2021) provide just a single result, we find it can be useful to provide some additional results to illustrate the variability associated to soil characteristics and climatic conditions. We include data from a second selected synthesis paper (Bai et al. 2019), which found that depending on soil type, SOC increase due to biochar use can be between 32% (clay loam and clay soils) and 63% (silty clay and silty clay loam soils) (Table 26). Depending on soil pH, biochar use increased SOC by 28%, 35% and 65% in acid, neutral and alkaline soils, respectively. Biochar markedly stimulated

SOC increases in irrigated croplands (49%), three times higher than those under rainfed condition (16%). Biochar increased 12% (38% vs. 26%) more SOC in arid areas, respectively, compared to humid areas. In warm areas, biochar applications only increased SOC by half of the enhancement observed in cool areas (17% and 32% respectively). Biochar amendments enhanced SOC by 52% in rotational cropping systems, much higher than that in the continuous cropping system (31%).

Table 26 Effect (% change) of biochar on soil organic carbon (stock per hectare). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Biogeographic regions	Technique type or crop type	CI_LOW	Mean	CI_HIGH	Nc	Effect
All regions *Source:* Albert et al. 2021	Overall	47.6	54.3	61.3	166	Positive
All regions *Source:* Bai et al. 2019	Overall	33	39	45	222	Positive
	In the field	23	28	32	130	Positive
	In incubation and pot experiments	44	57	71	92	Positive
	Irrigated fields	37	49	60	119	Positive
	Rainfed fields	1	16	31	4	Positive
	Cool regions	25	32	39	99	Positive
	Warm regions	8	17	26	24	Positive
	Arid regions	31	38	45	23	Positive
	Humid regions	21	26	31	87	Positive
	Acidic soils	22	28	34	119	Positive
	Neutral soils	21	35	48	23	Positive
	Alkaline soils	50	65	80	67	Positive
	0-10 cm depth	27	40.7	55	42	Positive
	0-20 cm depth	NA	69.2	NA	NA	Positive
	0-30 cm depth	NA	14.2	NA	NA	Positive
	Short term (0-5 years)	36.7	45.2	53.6	146	Positive
	Medium term (6-20 years)	18.8	35.6	52.4	5	Positive

Source: references in the first column.

3.8.3 Effects on ammonia emissions

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the link with ammonia emissions is contrasting. Authors found differences by type of biochar (biochar feedstock), soil pH, soil texture, fertilisation rate, etc.

As shown on Table 27 , the literature analysed by Sha et al. 2019 showed that, overall, biochar application had non-significant effect on ammonia volatilization (RR = 0.92%, 95% CI: −12.45–13.38%). However, they also found differences by type of biochar. Wood-based biochar application significantly reduced ammonia volatilization (RR = −34.6%, 95% CI: −51.1 to −15.9%). However, they did not find a significant impact for other feedstocks such as lignocellulosic waste (including Macadamia nut shell, walnut shells, peanut shell and maize cobs) or herbaceous waste (including crop straw and green waste).

Sha et al. 2019 report that biochar applied to acidic soils could increase ammonia volatilisation by 38.4%. Increases in ammonia volatilisation were also caused by amendment with high pH biochar (30.8%), or combining biochar with ammonium-based fertilisers (67.9%). Reductions in ammonia volatilisation were observed when biochar was applied to fine soils (high SOC (−42.8%) and clay content (−58.4%)) at appropriate rates (5–15 t ha−1; −33.8%), was combined with urea or organic fertilisers (−18.6% and −28.7%), was used with appropriate N-fertiliser rates (<200 kg N ha−1; −33.5%), and was applied as acidified biochar (−42.2%).

Q. Liu et al. 2018 report that biochar significantly increases soil NH3 volatilization by 17% (P= 0.034) on average across different studies (Table 27). However, results depend on the type of biochar and soil pH. Manure biochar and straw biochar stimulate soil NH3 volatilization by an average of 43% and 27%, respectively, whereas wood biochar tends to decrease soil NH3 volatilization by an average of 30%. Biochar stimulates soil NH3 volatilization to a larger extent from acidic soils (pH ≤ 5) than from moderately acidic soils (5 < pH ≤ 6.5), while it shows little effect on neutral or alkaline soils. Soil NH3 volatilization from clay textured soils are more prone to be increased by biochar than that from other types of soil. Biochar addition to soils with less than 10 g SOC kg−1 induces a significant increase in soil NH3 volatilization, while a non-significant response to biochar is observed in soils with SOC> 10 g kg−1. In general, biochar characterized by pH > 9, or being applied at the rate > 40 t ha−1 induces a significant increase in soil NH3 volatilization, however, biochar with pH lower than 9, or being applied at less than 40 t ha−1 shows non-significant effect.

Table 27 Effect (% change) of biochar on ammonia volatilisation by biochar feedstock. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Biochar feedstock	Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
Overall	Sha et al. 2019	-12.4	0.9	13.4	141	Non-significant
	Q. Liu et al. 2018	0	17	38	99	Negative
Wood	Sha et al. 2019	-51.1	-34.6	-15.9	26	Positive
	Q. Liu et al. 2018	-48	-31	1	13	Positive
Lignocellulosic waste	Sha et al. 2019	-48	-20	24	13	Non-significant
Herbaceous waste	Sha et al. 2019	-3.7	11.4	29.7	92	Non-significant
Straw	Q. Liu et al. 2018	12	26	44	75	Negative
Manure	Sha et al. 2019	-55	40	370	10	Non-significant
	Q. Liu et al. 2018	3	42	96	11	Negative

Source: references in second column.

3.8.4 Effects on the nutrient balance

The selected paper for data extraction is Q. Liu et al. 2018. They found that, on average, biochar leads to an increase of 11% (P < 0.001) in plant N uptake, which is derived from an increase of 12% in plant biomass (P<0.001) and a minor decrease of −2% (P = 0.014) in plant tissue N concentration (Table 28).

There are, however, different factors that can modify this impact: soil pH, soil texture, soil cation exchange capacity, biochar type and biochar application rate. Biochar generally increases N uptake in acidic soils (defined by the authors as soil with pH ≤ 6.5), but it shows little effect in neutral or alkaline soils. The increasing impacts of biochar on N uptake are usually maximized in soils with poor structure (rich in either sand or clay) and in soils with low cation exchange capacity. Manure biochar induces a higher N uptake (27%) than wood biochar (10%) or straw biochar (9%). The relationship of biochar application rate with the response of N uptake follows a convex curve, and over application of biochar (> 80 t ha^{-1}) will significantly inhibit N uptake.

Table 28 Effect (% change) of biochar on N plant uptake. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

		CI_LOW	Mean	CI_HIGH	Effect
Biochar	Overall- uptake	8	11	15	Positive
	Soil pH 3-5	26	35	44	Positive
	Soil pH 5- 6.5	-6	7	19	Non-significant
	Soil pH 6.5- 7.5	0	3	6	Non-significant
	Soil pH > 7.5	-10	3	18	Non-significant
	Wood biochar	5	10	15	Positive
	Straw biochar	4	9	14	Positive
	Manure biochar	18	27	36	Positive

Source: Q. Liu et al. 2018

3.8.5 Effects of nutrient leaching and run-off

Results in the literature quantify the decrease leaching of nitrates (NO_3^-), of ammonium (NH_4^+), of total dissolved inorganic nitrogen and sometimes also N runoff.

The selected synthesis paper is Borchard et al. 2019. Overall, nitrates (NO_3^-) leaching was significantly reduced by 13% with biochar. Results were not significant for experiments of less than 30 days, but for studies with an experimental time of more than 30 days (30-60, 60-120, >120), biochar reduced NO_3^- leaching by 26 to 32% (Table 29). Nitrates leaching was not affected when biochar was applied to grassland but was reduced by 36% for a majority of agricultural crops (except horticulture).

Results depended on the feedstock and manufacturing of biochars. Biochars produced from lignocellulosic biomass and biochars produced at temperatures of >500 °C reduced NO_3^- leaching. Low biochar application rates of <10 Mg ha^{-1} increased NO_3^- concentration in soils. Larger biochar application rates (e.g., >10–20 Mg ha^{-1}) tended to reduce NO_3^- leaching.

Regarding soil types, there was also a large variability. Only coarse textured soils (i.e. sand) showed a reduced leaching of NO_3^-. Leaching was exclusively reduced in Cambisols (i.e. soils of limited age), and semi-arid soils (e.g. Calcisol, Solonetz).

The authors also looked at the effects of biochar with fertilisation: leaching was reduced by biochar in unfertilized soil and when the fertiliser application rate was below 150 kg N ha^{-1}. However, nitrates leaching progressively increased in response to increased N application rates (i.e. -26% for <150 kg N ha^{-1}, −7% for 150–300 kg N ha^{-1}, +46% for >300 kg N ha^{-1}). Last, nitrates leaching remained unaffected when biochar was applied in combination with organic fertilisers. Hence, over all N fertiliser application rates, the nitrates leaching reduction was not significant (i.e. +3% with mineral fertiliser, −7% with organic fertiliser, and −35% with organo-mineral fertiliser).

Table 29 Effect (% change) of biochar on NO3- leaching. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

		CI_LOW	Mean	CI_HIGH	Effect
Biochar	Overall	-19	-13	-7	Positive
	Cereals, other crops and perennials	-42	-36	-31	Positive
	Horticulture	-68	-53	-29	Positive
	Grassland	-17	-6	6	Non-significant
	Forest	-46	-6	+66	Non-significant

Source: Borchard et al. 2019

3.9 Soil amendments with lime and gypsum

Soil amendments are practices used to improve the soil quality in terms of its structure and biochemical function. Most amendments use calcium-containing minerals, like lime and gypsum. Lime refers to a material that can come in different forms, especially calcium carbonate ($CaCO_3$) and magnesium carbonate ($MgCO_3$), and it is used to reduce soil acidity and to add calcium or magnesium to the soil. Gypsum is also a commonly used calcium containing mineral (calcium sulphate dihydrate, $CaSO_4 \cdot 2H_2O$). It is used to improve soil calcium and sulphur contents, and to alleviate a range of subsoil problems due to its solubility.

3.9.1 Effects on soil organic carbon

From the systematic review of synthesis papers (Schievano et al. 2023), soil liming resulted in a marginal, non-significant increase in soil carbon stocks as found by a paper of global geographical extent (Eze et al. 2018), and one synthesis paper (Y. Wang et al. 2021) reported significant effects of the application of gypsum amendment on SOC content (Table 30). They analysed the impacts of gypsum on saline-sodic soils in China. Saline-sodic soils cover around 10% of the global land surface and deliver various ecosystem services to human society in the arid/semiarid regions. Flue gas desulfurization gypsum (FGDG), a byproduct from coal-fired power plants, is widely used to ameliorate saline-sodic soils. The FGDG application significantly reduced soil bulk density (by 7.3% ± 3.6%) and increased SOC content by 24.5% ± 15.3%. As shown on Table 30 FGDG application significantly increased SOC only in saline soil (not in different types of sodic soils), in the application season of spring, and in two soil depths (0-20 cm and 20-40 cm). The logarithmic response ratio values of SOC were positively correlated with the initial values of soil exchangeable sodium percentage and negatively with the initial values of soil electrical conductivity, nitrate, and available potassium, and the logarithmic response ratio values of soil exchangeable sodium percentage.

Table 30 Effect (% change) of flue gas desulfurization gypsum on soil organic carbon in China. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

All techniques types		CI_LOW	Mean	CI_HIGH	Nc	Effect
Flue gas desulfurization gypsum on different types of saline-sodic soils in China	Overall	9	24.5	40	76	Positive
	All	8	25	43	76	Positive
	Heavy sodic soil	-7	15	43	38	Non-significant
	Strong sodic soil	-27	9	63	9	Non-significant
	Moderate sodic soil	-20	11	54	21	Non-significant
	Saline soil	47	94	158	8	Positive
	Incorporation 0-20 cm	6	24	43	66	Positive
	Incorporation >20 cm	-14	35	113	10	Non-significant
	Application autumn	-6	16	42	55	Non-significant
	Application spring	10	33	61	21	Positive
	Soil depth 0-20 cm	3	18	34	48	Positive
	Soil depth 20-40 cm	21	39	60	28	Positive

Source: Y. Wang et al. 2021

3.10 Manure management: manure land application techniques

Improved manure land application techniques are mainly aimed at limiting nutrients losses and emissions of ammonia and/or greenhouse gases during manure application (either solid or liquid fractions) to the soil. These are compared to the conventional broadcast spread application. Results have been found in literature on the effects of the following techniques:

- Injection of liquid manure (deep placement) or immediate incorporation of solid manure into the soil;
- Band application of manure by trailing hoses or other equivalent systems, by which the manure is applied beneath the crop canopy, but not incorporated into the soil;
- Irrigation coupled to manure application;
- Application of processed manure fractions (e.g. digestate, composted, solid fraction, liquid fraction, etc.);
- Separate application of solid or liquid fractions;
- Use of additives to manure, like nitrification inhibitors, lava meal, biochar, superphosphate or sawdust as emission mitigation options.

3.10.1 Effects on greenhouse gas emissions

Manure land application techniques are found to have differing effects on GHG gases, depending on the technique and the gas. Several papers have been selected for extraction as not a single one provided information for all cases (Table 31).

Different results are reported for deep placement and incorporation techniques. For CH4, two results are available from one synthesis study (Emmerling et al. 2020): a positive one for deep placement or injection (-23%; 95% CI: -34%, -13%) and non-significant effect for incorporation compared to surface application. For N2O, non-significant effect was found for this practice.

A negative effect on GHGs is reported for band application. A synthesis paper (Emmerling et al. 2020) found increased emissions of CH4 (+153%; 95% CI: 108%, 197%), while a non-significant change for N2O.

Application of composted/digested manure showed a positive effect on N2O emissions (F. Xia et al. 2020) while the effects of the application of manure digestates were found non-significant according to Y. Wang et al. 2017 and Hou et al. 2015 as compared to untreated manure. Hou et al. 2015 examined the application of separated solid and liquid fractions on N2O emissions, on manure from dairy cows and swine. The overall effect of liquid fractions on N2O emissions did not differ from that of raw slurry. Field-applied solid fractions showed on average 46% ($p < 0.01$) lower N2O emissions than field-applied untreated manure.

Similarly, results on the use of additives are provided by two papers with regard to N2O only. One synthesis paper (Y. Wang et al. 2017) reports a decrease in N2O emissions of -33 (±10.3)% with the use of nitrification inhibitors on swine manure, and the other (Y. Wang et al. 2018a) reports non-significant effect of the application of additives (biochar and nitrification inhibitors) on cattle manure.

Table 31 Effect (% change) of manure land application techniques on greenhouse gas emissions for major GHG types. SE: Standard Error, CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

GHG	Technique and source	CI_LOW	Mean	CI_HIGH	Nc	Effect
CH4	Banding vs. surface application *Source:* Emmerling et al. 2020	108	153	197	2	Negative
	Deep placement or injection vs. surface incorporation *Source:* Emmerling et al. 2020	-34	-23	-13	5	Positive
	Immediate incorporation vs. surface application *Source:* Emmerling et al. 2020	- 167	-37	+117	12	Non-significant
N2O	Banding vs. surface application *Source:* Emmerling et al. 2020	-2	25	57	6	Non-significant
	Deep placement and immediate incorporation *Source:* Emmerling et al. 2020	-	-	-	-	Non-significant
	Composted/digested manure (vs non composted or vs. raw slurry) — Anaerobic digestate *Source:* Hou et al. 2015	-46	-25	6	19	Non-significant

	Separated solid or liquid fractions	Solid fraction *Source:* Hou et al. 2015	-66	-46	-15	10	Positive
		Liquid fraction *Source:* Hou et al. 2015	-25	-2	25	40	Non-significant
	Additives	Nitrification inhibitors – swine *Source:* Y. Wang et al. 2017	(-SE): -43.6 Min: -83	Mean: -33.3 Median: -28	(+SE): -20.0 Max: 34	12	Positive
		Biochar & nitrification inhibitors – cattle *Source:* Y. Wang et al. 2018a	-	-	-	-	Non-significant

Source: references in second column.

3.10.2 Effects on ammonia emissions

Results on ammonia emissions differ depending on the technique employed, the majority being positive for deep placement or injection, immediate incorporation and irrigation, and banding and non-significant for the rest (Schievano et al. 2023).

On the effect of deep placement/incorporation techniques compared to conventional broadcast spread application, the selected paper is Hou et al. 2015. Authors report that emissions of ammonia from manures following incorporation and injection were 70% (95% CI: 50%, 82%) and 80% (95% CI: 72%, 86%) lower than that from surface broadcasted manures, respectively. These differences were statistically significant ($p < 0.01$).

For band application, Hou et al. 2015 found that emissions of ammonia from manures following band spreading were 55% (95% CI: 37%, 67%) lower than that from surface broadcasted manures. These differences were statistically significant ($p < 0.01$).

For irrigation coupled to manure application, 1 synthesis paper with a global scale coverage including studies from Europe is available (Ti et al. 2019), reporting a significant reduction of ammonia emissions (-73.6%) compared with the control.

This same paper (Ti et al. 2019) provides a value for all of mitigation measures analysed together (deep placement, incorporation, injection, irrigation, separation, digestion, crude protein content reduction and band application). Overall, the average reduction of land application strategies was 60.7% (95% CI: 53.5%, 66.6%).

Table 32 Effect (% change) of the use of manure land application techniques on ammonia emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Technique		CI_LOW	Mean	CI_HIGH	Effect
Overall *Source:* Ti et al. 2019		-66.6	-60.7	-53.5	Positive
Land application with deep placement/incorporation *Source:* Hou et al. 2015	Injection	-86	-80	-72	Positive
	Incorporation	-82	-70	-50	Positive

Land application with banding or band application *Source:* Hou et al. 2015	-67	-55	-37	Positive
Irrigation coupled to manure application *Source:* Ti et al. 2019	-92	-73.6	-25	Positive

Source: references in the first column.

3.11 Manure management: manure storage techniques

This set of practices aims at limiting the loss of nutrient from manure in the form of ammonia and N2O during the storage and composting phase. Different specific techniques can be used, among which the following ones have been examined:

- Use of physical, chemical or microbial additives that alters the bio physical processes responsible for the emissions of nutrients.
- Use of covers of either solid or liquid manure storage facilities. This includes plastic membranes, floating biomass or inert materials, and natural crusts.
- Manure acidification during storage. This decreases manure's pH, limiting the activity of urease-producing bacteria for the purpose of reducing NH3 emissions.
- Compaction of solid manure heap, to increase its density and decrease the air fraction within it.
- Storage with biofilters that intercept and treat air emissions from storage facilities.
- Manure cooling during storage.

3.11.1 Effects on greenhouse gas emissions

The effects of improved manure storage techniques on greenhouse gases are diverse across specific techniques and gases.

Results on the effect of the use of additives are available for the main greenhouse gas separately and for the total global warming potential. Specific effects may vary depending on the type of additives used, the type of manure and the considered greenhouse gas. Only one synthesis paper (Cao et al. 2019) provides results on the aggregate global warming potential, reporting positive results. This meta-analysis covers various types of additives, as well as different types of manure – livestock, food/green waste and sewage sludge – with a global coverage including Europe. On average, during composting, additives reduced total GHG emissions expressed as global warming potential (GWP) by 54.2% (95% CI: 65%, 42%). Physical additives[17] (e.g. biochar and zeolite) were more

[17] Physical additives are additives involved in adsorbing or changing the physical factors of the compost: Biochar; Zeolite; Bentonite; Medicine stone; Expandable perlite; Expandable vermiculite; pumice; Sand; Clay; soil; Raw attapulgite; Tannin; Activated charcoal; Biomass fly ash;

effective at reducing the total GHG emissions (67.2%) compared to chemical additives[18] (29%). Regarding storage with microbial[19] inocula (compared to no inoculation) the selected synthesis paper (Z. Zhang et al. 2021) reports no significant effects on CH4 emissions and positive ones on N2O emissions that decrease on -48% (95% CI: -72%, -23%; Nc=5).

Results on the use of storage covers are available for both CH4 and N2O. The selected synthesis paper for data extraction (Hou et al. 2015) reports an overall no significant effect on emissions of these gases. Nevertheless, the authors report that emissions of N2O were enhanced by a factor of 8.6, when stored slurry was covered by chopped straw ($P < 0.01$), and that slurry covered with artificial film decreased N2O emissions by 98% ($p < 0.01$).

For acidification, a positive effect can be assumed both for CH4 and N2O.The results of the selected synthesis paper (Emmerling et al. 2020) showed that slurry acidification was effective in reducing CH4 (mean −86%; 95% CI: -92%, -80%; Nc=33) and N2O emissions (-21%; 95% CI: -27%, -15%; Nc=15) including individual studies from Europe. Slurry acidification was mainly achieved using sulphuric acid, with a pH-value between 5.1 and 6.5.

The effect of manure compaction on greenhouse gases was examined by one synthesis paper with global coverage including Europe (Pardo et al. 2015), finding non-significant effects for N2O and CH4.

One synthesis paper based on studies from different world locations including Europe (Y. Wang et al. 2017) investigated the effect of cooling on CH4, finding a statistically significant reduction of -76.3±5.5% (Nc=6).

Table 33 Effect (% change) of the use of additives and acidification on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

GHG	Technique		Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
GWP	Additives	Overall	Cao et al. 2019	-65	-54.2	-42	NA	Positive
		Physical	Cao et al. 2019	-78	-67.2	-52	NA	
		Chemical	Cao et al. 2019	-44	-29	-12	NA	
CH4	Additives	Overall	Cao et al. 2019	-77	-68	-58	68	Positive
		Physical	Cao et al. 2019	-83	-72	-57	40	Positive

[18] Chemical additives are additives containing chemicals promote chemical reactions with the compost substrate relating to N turnover processes: Chemical additives PO4^{3-} and Mg^{2+} salts; Superphosphate; Gypsum; KH2PO4; H3PO4; CaSO4;Al2(SO4)3; CaCl2;H2SO4; FeCl3; MgCl2; Modified red mud; MgSO4; NaH2PO4 H2O and Na2HPO4 2H2O; S; KHSO4 and H2SO4; Ferrisulfas, sodium humate and superphosphate; Vinegar; NaAc; Apple pomace; Citric acid; Calcium magnesium phosphate.

[19] Microbial additives are additives containing microorganism that affect N turnover processes: NOB; NTB (ammonifiers, nitrobacteria, azotobacter) agent; TAT105 (thermophilic, ammonium-tolerant bacterium); Ammonia- oxidizing bacteria; commercial microbiological agent; T. thioparus 1904 inoculum.

GHG	Technique		Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
		Chemical	Cao et al. 2019	-74	-62	-46	28	Positive
		Microbial	Z. Zhang et al. 2021	-	-	-	3	Non-significant
	Covers – all types		Hou et al. 2015	-	-	-	-	Non-significant
	Acidification		Emmerling et al. 2020	-92	-86	-80	33	Positive
	Compaction		Pardo et al. 2015	-	-	-	-	Non-significant
	Cooling		Y. Wang et al. 2017	-44.8 (-SE) Min -61	-39.3 (Median -38)	-33.8 (+SE) Max -23	6	Positive
N2O	Additives	Overall	Cao et al. 2019	-56	-44	-31	99	Positive
	Additives	Physical	Cao et al. 2019	-76	-65	-51	51	Positive
	Additives	Chemical	Cao et al. 2019	-21	-3	14	44	Non-significant
	Additives	Microbial	Z. Zhang et al. 2021	-72	-48	-23	5	Positive
	Covers	Overall	Hou et al. 2015	-	-	-		Non-significant
	Covers	Artificial (plastic) film	Hou et al. 2015	-100	-98	-69	4	Positive
	Covers	Straw cover	Hou et al. 2015	43	861	6335	11	Negative
	Acidification		Emmerling et al. 2020	-27	-21	-15	15	Positive
	Compaction		Pardo et al. 2015	-	-	-	-	Non-significant

Source: references in the third column.

3.11.2 Effects on ammonia emissions

While the measured effects vary depending on the specific additive used, overall a significant reduction is reported across a variety of different types of manure and additives, and from different geographic contexts, including Europe (Schievano et al. 2023).

Ti et al. 2019 found that, overall, additives used by manure storage showed a significant reduction potential of 29.3% (95% CI: 17%, 40%). Specifically regarding microbial inocula, Cao et al. 2019, report a reduction of 43% (95% CI: -29%, -57%) in ammonia emissions thanks to this technique.

The use of covers has an overall positive effect. Hou et al. 2015 do not provide an average value for all covers but specific values by type of techniques (crust, straw, granules, plastic films, oil), which values range between -78% and -99%. Another synthesis paper (Ti et al. 2019) reports an average decrease in ammonia emissions from manure cover of -76% (95% CI: 72%, 78%).

Regarding the effect of manure acidification versus non-acidified manure, the selected paper for data extraction (Hou et al. 2015) reports a reduction of 83% in ammonia emission (95% CI: -59%, -93%).

The use of biofilters for emissions reduction during manure storage was investigated in two synthesis studies, both reporting a significant reduction of ammonia emissions for chicken and cattle manure in different geographic context, including Europe. The selected papers are Y. Wang et al. 2019a for chicken manure and Y. Wang et al. 2018a for cattle manure. For broilers and layers, the ammonia reduction reported by Y. Wang et al. 2019a is 73% (95% CI: ±11.7%), while for beef cattle, the average decrease is -91% (Y. Wang et al. 2018a).

Finally, only one synthesis paper (Pardo et al. 2015) investigated the effect of compaction of solid manure heaps, finding non-significant effects.

Table 34 Effect (% change) of the use of manure storage techniques on ammonia emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top.

Technique		CI_LOW	Mean	CI_HIGH	Effect
Additives *Source:* Ti et al. 2019		-40	-29.3	-17	Positive
Additive: microbial inocula *Source:* Cao et al. 2019		-57	-43	-29	Positive
Covers	All *Source:* Ti et al. 2019	-78	-76	-72	Positive
Covers *Source:* Hou et al. 2015	Crusting	-89	-65	14	Non-significant
	Straw cover	-90	-78	-53	Positive
	Granules cover	-93	-85	-69	Positive
	Artificial film	-100	-98	-93	Positive
	Peat/kitchen oil	-100	-99	-97	Positive
Acidification *Source:* Hou et al. 2015		-93	-83	-59	Positive
Compost biofilters	Broiler and layers *Source:* Y. Wang et al. 2019a	-85	-73	-62	Positive
	Beef cattle *Source:* Y. Wang et al. 2018a	-100	-91	-80	Positive
Compaction *Source:* Pardo et al. 2015		-83	-54	+6	Non-significant

Source: references in the first column.

3.12 Manure management: manure processing techniques

Manure processing techniques can be used to change manure chemical/physical properties and composition and thus increasing manure management efficiency and/or limit manure emissions compared to raw manure. Under the literature review of synthesis papers, manure processing techniques included the following techniques (note that this is not an exhaustive list of manure processing techniques, but of those found in the literature that meet the requirements to be included in the review):

- Composting of solid manure using improved techniques (e.g. turning, forced-air, bulking agents, C/N adjustment).
- Anaerobic digestion of slurries. Here, the environmental impacts of anaerobic digestion of manure alone (i.e. mono-digestion) or with other substrates (i.e. co-digestion) are reviewed and are considered either as result of the overall processing chain (pre-storage, digesters, post-storage, land distribution, energy generation using biogas) or from single steps.
- Liquid manure storage in anaerobic lagoons or in aerobic lagoons.
- Solid manure storage in piles. Improved techniques for manure storage in static stockpiles (e.g. physical, chemical or microbial additives, etc.) are excluded here, and included in a separate set of fiches regarding 'Improved manure storage techniques'.
- Solid-liquid separation (e.g. decanter centrifuges, screw press, roller presses, decantation). The effects are considered by comparing either handling, storage or land application of the separated fractions, as compared to the raw manure.
- Drying of solid fractions
- Recovery of nutrients through physical or chemical treatments (e.g. struvite precipitation, ammonia stripping)
- Manure pasteurization

3.12.1 Effects on greenhouse gas emissions

The effects of manure processing techniques on greenhouse gases are diverse across specific techniques and gases according to the results retrieved from the synthesis literature review (Schievano et al. 2023).

Regarding composting techniques, for both CH4 and N2O, results were overall positive for all composting techniques[20] for pig manure according to Z. Zhang et al. 2021. They report a mean decrease of -18 % in methane (95% CI: -28%, -9%) and -24% in N2O (95% CI: -32%, -15%). Nevertheless, the significance of the results vary depending on specific techniques. To illustrate this, in Table 35 we provide quantitative coefficients extracted from various selected synthesis papers (Ba et al. 2020; Pardo et al. 2015; Y. Wang et al. 2017; S. Zhao et al. 2020).

Regarding anaerobic digestion, we provide results concerning the processing/storage GHG emissions as compared to raw slurry emissions and the saving of emissions from avoided fossil fuels for energy production. The selected synthesis paper (Miranda et al. 2015) reports a reduction of emissions from stored digested slurry of -82% (median, Wilcoxon's P=0.01), and a reduction from field application of digested slurry of -36% (median, Wilcoxon's P=0.0026). Emissions from avoided fossil-derived energy

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[20] The techniques included in the analysis are: optimal C/N ratios, optimal moisture, turning once weekly, intermittent ventilation or optimized aeration rates, covering and additives.

are -115 % (median, P<0.01). The other selected synthesis paper (Emmerling et al. 2020) report a decrease in methane emissions of -46 % (95% CI: -65%,-27%) and a non-significant effect on N2O emissions.

Solid-liquid separation, compared to no manure processing, showed a positive effect on CH4 reduction from 1 synthesis paper (Emmerling et al. 2020). They report a reduction of -27% (95% CI: -46%, -8%). Regarding N2O emissions, the same paper reports non-significant effect.

Table 35 Effect (% change) of the use of manure processing techniques on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique		GHG	Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
Composting	All	CH4[20]	Z. Zhang et al. 2021	-28	-18.1	-9	77	Positive
		N2O[20]	Z. Zhang et al. 2021	-32	-23.8	-15	107	
	C/N adjust-ment	CH4		NA	NA	NA		Not available
		N2O	S. Zhao et al. 2020	-93	-30	+30	15	Non-significant
	Bulking agents	CH4	Pardo et al. 2015	-91	-71	-31	12	Positive
		N2O	Pardo et al. 2015	-72	-53	-21	12	Positive
	Vermi-composting	CH4	Ba et al. 2020	NA	Median -0.78 (P=0.32)	NA	1	Non-significant
		N2O	Ba et al. 2020	NA	Median -0.53 (P=0.32)	NA	1	Non-significant
	turning	CH4	Pardo et al. 2015	-90	-71	-18	7	Positive
		N2O	S. Zhao et al. 2020 & Pardo et al. 2015	NA -75	-55 -50	NA +2	NA 10	Positive
	aeration	CH4	Pardo et al. 2015	-95	-63	+24	11	Non-significant
		N2O	Pardo et al. 2015	-35	+45	+161	9	Non-significant
	chemical, physical or microbial additives	CH4	Y. Wang et al. 2017	Min -0.63	Mean -0.16 p=0.465	Max +0.15	4	Non-significant
		N2O	Y. Wang et al. 2017	Min -0.94	Mean -0.45 (p < 0.01)	Max -0.09	11	Positive
Anaerobic digestion	All AD types	GHG storage	Miranda et al. 2015		Median -82 Wilcoxon's P = 0.01		24	Positive
		GHG field application	Miranda et al. 2015		Median		35	Positive

Technique		GHG	Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
					−36 Wilcoxon's P = 0.0026			
		GHG avoided fossil fuels	Miranda et al. 2015		Median -11 ($p < 0.01$)		18	Positive
	All AD types	CH_4	Emmerling et al. 2020	-65	-46	-27	NA	Positive
		N_2O	Emmerling et al. 2020	-10	+10	+30	NA	Non-significant
Solid-liquid reparation		CH_4	Emmerling et al. 2020	-46	-27	-8	NA	Positive
		N_2O		-	-	-	-	Non-significant

Source: references in the third column.

3.12.2 Effects on ammonia emissions

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), the effect of manure composting techniques can be significant on reducing ammonia emissions, but depends on the analysed technique.

Regarding composting techniques, the overall effect is positive, but varies depending on the considered composting technique (e.g. C/N adjustment, vermicomposting, addition of bulking agents, periodical turning, forced aeration, and/or the use of either chemical or physical or microbial additives to the composting piles). Z. Zhang et al. 2021, whose meta-analysis was based on 68 studies related to pig manure composting, report an average positive of -33% (95% CI: -38%, -28%) although they do not include all techniques. This overall value does not include vermicomposting nor bulking agents, but includes other physical, chemical and microbial additives, such as biochar and superphosphate. Separately, Z. Zhang et al. 2021 report an average positive effect of additives of -36% (95% CI: -42%, -30%) with 128 observations. Similarly, Ti et al. 2019, report a positive effect with a reduction of -29% (95% CI:-385, -17%) with 49 observations and a -92.5% reduction for acidifiers alone (95% CI: -94%, -86%; Nc=17) . In Table 36, we provide data on additional techniques such as adjusted moisture, vermicomposting, C/N adjustment, turning frequency and bulking agents with diverse effects on NH_3 emissions.

Table 36 Effect (% change) of the use of composting manure processing techniques on ammonia emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique/Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
Overall *Source:* Z. Zhang et al. 2021	-38	-32.7	-28	154	Positive
C/N adjustment *Source:* S. Zhao et al. 2020	-27	-9	+7	22	Non-significant
Adjusted moisture *Source:* Z. Zhang et al. 2021	-33	-19	-10	6	Positive

Technique/Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
Vermicomposting *Source:* Ba et al. 2020		median - 33.5 (p = 0.002)		6	Positive
Additives *Source:* Z. Zhang et al. 2021	-42	-36	-30	128	Positive
Additives *Source:* Ti et al. 2019	-38	-29.3	-17	49	Positive
Acidifiers *Source:* Ti et al. 2019	-94	-92.5	-86	17	Positive
Turning *Source:* Pardo et al. 2015	+26	+54	+88	13	Negative
Turning *Source:* Z. Zhang et al. 2021	+6	+9	+12	2	Negative
Bulking agents *Source:* Pardo et al. 2015	+0.5	+34.5	+88.5	14	Negative

Source: references in the first column.

3.13 Livestock housing techniques

Livestock housing techniques include several improved strategies mainly used to reduce emissions to the environment and to improve animal welfare. Some of these techniques are floor type, in-house litter amendment with conditioners, additives or inhibitors, frequency of litter/manure removal, shading, space allowance, light management and ventilation. However, we are not including here manure management outside the house or on livestock feeding, nor the impact of livestock housing techniques on animal welfare, as these will be assessed separately.

3.13.1 Effects on greenhouse gas emissions

The effect of livestock housing techniques on greenhouse gas emissions depend on the technique applied and gas type (Table 37).

Biofilters for exhaust air, compared to livestock houses without biofilters, have different effects on GHG emissions. Y. Wang et al. 2017 found that biofilters decrease CH_4 emissions by -24% ($p < 0.01$) but have weak non-significant negative effect on N_2O emissions. However, they also found that some studies suggest that biofilters may increase N_2O emissions because the absorbed NH_3 from the exhaust air may be nitrified and denitrified, generating N_2O. The overall effect of biofilters on GHG emissions might be positive but cannot be determined without additional information.

High frequency manure removal, compared to low frequency manure removal, has different effects on GHG emissions. The only synthesis paper including a statistical analysis (Hou et al. 2015) has reported a positive effect on CH_4 emissions (-55%; 95% CI: -70%, -33%) and not significant weak positive effect on N_2O emissions. However, it is based in a very low number of observations (4 from one single study) therefore it cannot be considered a strong result.

Slurries underneath slatted floor showed higher CH_4 emissions (+50%; 95% CI: 22%, 85%), but lower N_2O emissions (-89%; 95% CI: -96%, -19%) than buildings with deep- litter, according to the only available synthesis paper (Hou et al. 2015).

Table 37 Effect (% change) of livestock housing techniques on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	GHG	CI_LOW	Mean	CI_HIGH	Nc	Effect
Biofilters for exhaust air	GHG	NA	NA	NA	NA	Not available
	CH4 *Source:* Y. Wang et al. 2017	Min -83	-24 $p < 0.01$	Max +10	14	Positive
	N2O *Source:* Y. Wang et al. 2017	Min -22	+0.13 P=0.195	Max +81	13	Non-significant
High frequency manure removal	GHG	NA	NA	NA	NA	Not available
	CH4 *Source:* Hou et al. 2015	-70	-55	-33	4	Positive
	N2O *Source:* Hou et al. 2015	-100	-42	>+900	4	Non-significant
Slurries underneath slatted floor as compared to deep-litter	GHG	NA	NA	NA	NA	Not available, mixed effect
	CH4 *Source:* Hou et al. 2015	+22	+50	+85	8	Negative
	N2O *Source:* Hou et al. 2015	-96	-89	-19	10	Positive

Source: references in second column.

3.13.2 Effects on ammonia emissions

Results on ammonia emissions differ depending on the technique applied. Several synthesis papers have been selected to cover the different techniques.

According to Y. Wang et al. 2017 the use of biofilters on swine systems is one of the most effective mitigation measures for limiting NH_3 emissions from animal houses (median -72%, $p < 0.001$). For comparison we also provide results by Y. Wang et al. 2019b, that reported a decrease of -85% in ammonia from the use of biotrickling filters on broilers systems, but it was not significant due to the low number of observations (Nc=3).

Non-significant effects due to the low number of observations were reported for acid scrubbers (Y. Wang et al. 2019b).

Mechanical management of exhaust air and manure, compared to livestock houses without mechanical management, was addressed only by one synthesis paper (Ti et al. 2019). The use of mechanical management of air and manure was the most effective NH3 mitigation measure among those analysed by these authors, decreasing NH3 emissions by -58% (95% CI: -73.7%, -31.7%). This value is reflected in the results figure of the synthesis paper, while in the text the value is -73.6%, which in the figure exactly corresponds with the lower 95% CI.

Results on the use of different floor types have been found in the literature review. A synthesis paper (Ti et al. 2019) found that several floor types and management reduced NH3 emissions by 48% (95% CI: -61.2, -29.6%; Nc=25) mainly in pigs but also cattle barns. These practices include: for cattle, extra-straw addition compared to straw-bedded housing; for pigs: slatted floors compared with straw-flow system, compared with concrete floors, compared with straw-based deep litter or sawdust-based deep litter; etc. The second best paper that can give more detailed information on specific floor types is Hou et al. 2015. They used data derived from 11 farm-scale studies to test the effects of animal houses with distinct floor constructions on emissions. Animal houses with alternative floors tended to have lower NH3 emissions compared to the reference floor construction. The difference between slatted-floor/deep litter stables compared with solid floor stables (-57%; 95% CI: -77.5%, -16.9%; Nc=4) was statistically significant ($p < 0.01$) while the difference between deep litter and slatted floor is not significant. Deep litter with extra straw addition has non-significant effect compared to deep litter without extra straw addition (Hou et al. 2015).

High frequency litter/manure removal, compared to low frequency manure removal, has non-significant effect on NH3 emissions according to 2 synthesis papers (Hou et al. 2015; Y. Wang et al. 2018a).

In-house litter amendment with conditioners, additives or inhibitors, compared to untreated litter, has several results. Urease inhibitors showed a decrease on ammonia emissions of around 50% according to two synthesis papers (Ti et al. 2019; Y. Wang et al. 2018a). The selected synthesis paper is Ti et al. 2019 with value -51.5%. A synthesis paper (de Toledo et al. 2020) finds an overall positive effect for litter treatment methods, but then they find a positive effect for acidifiers and gypsum, while non-significant effects for alkalinizers, adsorbents and superphosphate. However, we cannot use their results as they provide them as the standardized mean difference, which is calculated as the raw mean difference between the treatment and control groups divided by their pooled standard deviations. Y. Wang et al. 2019b found that acidifiers and zeolite (adsorbent) significantly reduced by 44% ammonia emissions on poultry. Y. Wang et al. 2018a reported acidifiers decrease NH3 emissions (82%; 95% CI: -107%, -58%; Nc=4) for beef cattle, but only four observations were available, resulting in an insignificant outcome ($p = 0.068$). Ti et al. 2019 reported positive effects for a type of adsorbents (XF & GY) in China different from the ones analysed by other authors (sepiolite, zeolite, bentonite, lignite and other types of coal, sawdust). Non-significant effects were reported for the rest of amendments, often due to the low number of observations.

Table 38 Effect (% change) of livestock housing techniques on ammonia emissions. SMD: Standardized mean difference CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique		Source	Population	CI_LOW	Mean	CI_HIGH	Nc	Effect
Biofilters for exhaust air	Biofilters	Y. Wang et al. 2017	Swine	Min: -86 Mean-SE: -71	-63 (median -72)	Max: 0.01 Mean + SE: -56	14	Positive
	Biotrickling filters	Y. Wang et al. 2019b	Broilers	-	-85	-	3	Non-significant
Acid scrubbers		Y. Wang et al. 2019b	Poultry	-	-92.5	-	2	Non-significant
High frequency manure removal		Hou et al. 2015 / Y. Wang et al. 2018a	Livestock / Beef cattle	-	-	-	-	Non-significant
Slatted floor compared with several floors and extra straw compared with standard straw		Ti et al. 2019	Pigs and cattle	-61.2	-48	-29.6	25	Positive
Deep litter compared with solid floors and slatted floor compared to solid floors		Hou et al. 2015	Livestock	-77.5	-56.7	-16.9	4	Positive
Deep litter compared to slatted floor		Hou et al. 2015	Livestock	-55.5	-26.9	19.9	13	Non-significant
Deep litter with extra straw compared to standard deep litter		Hou et al. 2015	Livestock	-70.2	-16.2	138	13	Non-significant
Mechanical management exhaust air and manure		Ti et al. 2019	Cattle, pig, poultry	-73.7	-57.9	-31.7	16	Positive
In-house litter amendments	Urease inhibitors	Ti et al. 2019	Cattle, pig, poultry	-68.7	-51.5	-23.9	9	Positive
		Y. Wang et al. 2018a	Beef cattle	-63.4	-43.3 (Median -59.5)	-23.2	20	Positive
	Overall: acidifiers, gypsum, alkalinizing, adsorbents, superphosphate)	de Toledo et al. 2020	Poultry (Broilers)	NA (SMD - 1.722)	NA (SMD - 1.014)	NA (SMD -0.306)	206	Positive
	Gypsum	de Toledo et al. 2020	Poultry (Broilers)	NA (SMD - 11.354)	NA (SMD - 6.375)	NA (SMD -1.396)	32	Positive

Technique		Source	Population	CI_LOW	Mean	CI_HIGH	Nc	Effect
	Alkalizers[21]	de Toledo et al. 2020	Poultry (Broilers)	NA (SMD -0.803)	NA (SMD +0.184)	NA (SMD +1.170)	24	Non-significant
	Acidifiers[22]	de Toledo et al. 2020	Poultry (Broilers)	NA (SMD -1.898)	NA (SMD -1.075)	NA (SMD -0.251)	122	Positive
	Acidifiers[23]	Y. Wang et al. 2018a	Beef cattle	-107 (Max -98)	-82.3 (Median -85)	-58 (Min -61)	4	Non-significant
	Litter additives (acidifiers + adsorbent)[24]	Y. Wang et al. 2019b	Poultry	-75.1	-44.2	-13.3	17	Positive
	Adsorbents (XF & GY)[25]	Ti et al. 2019	Cattle in China	-50.9	-30.5	0	26	Positive
	Adsorbents[26]	de Toledo et al. 2020	Poultry (Broilers)	NA (SMD -3.410)	NA (SMD -0.231)	NA (SMD +2.949)	20	Non-significant
	Adsorbent (Lignite)	Y. Wang et al. 2018a	Beef cattle	Max -66	-41.3	Min -28	3	Non-significant
	Adsorbent (Sawdust)	Y. Wang et al. 2018a	Beef cattle	Max -45	-20	Min - 6	3	Non-significant
	Adsorbents: Yallourn brown coal and lignite	Ti et al. 2019	Cattle	-91	-37.1	333.9	3	Non-significant
	Superphosphate	de Toledo et al. 2020	Poultry (Broilers)	NA (SMD -5.474)	NA (SMD -2.099)	NA (SMD +1.276)	8	Non-significant

Source: references in second column.

[21] The alkalizing sub-group was formed by hydrated lime, quicklime, calcitic limestone and dolomitic limestone.

[22] Acidifiers include aluminum sulfate, sodium bisulfate, potassium permanganate, aluminum chloride, ferrous sul- fate, acidified clay, alum, hydrochloric-citric phosphoric acid and SoftAcid™.

[23] Chemical amendments such as alum ($Al_2(SO_4)_3$) and calcium chloride ($CaCl_2$). They reduceNH3 emissions by decreasing the pH and increasing the function of H+ through cation exchange

[24] Litter additives were usually used in broiler litter house, including alum, sodium bisulfate (PLT), and zeolite, with the median NH3 mitigation for all the litter additives being 39.9% ($p < 0.01$). Alum and PLT were acidulants, and they could reduce NH3 emissions by decreasing the pH and increasing the function of H+ through cation exchange. Zeolite was an effective, non-corrosive and non-hazardous litter additive, which could reduce NH3 emissions by adsorption

[25] XF-4, GY1, GY2, GY3 and GY4 for cattle.

[26] Adsorbents include sepiolite, zeolite, bentonite and coal.

3.14 Livestock feeding techniques

Livestock feeding techniques include two main groups of interventions used to improve animal feeding management and to reduce emissions to the environment:

- Diet formulation. It includes these interventions: dietary legume; forage with higher digestibility; high concentrate level in diet; low crude protein diet; tannin rich forage; fermented feed; lipids or oils.
- Feed additives. They include these interventions: Coccidiostats and histomonostats (substances intended to kill or inhibit protozoa); enzymes (carbohydrase, phytase, not specified); nutritional additives (vitamins, amino acids, urea, etc.); sensory additives (any feed additive that improves or changes the organoleptic properties of the feed or the visual characteristics of the food derived from animals, e.g. colorants, flavourings); technological additives (preservatives; antioxidants, emulsifiers, stabilisers, thickeners, acidity regulators, silage additives); zootechnical additives (digestibility enhancers, gut flora stabilisers, etc.); and non-specified feed additives.

This is not an exhaustive list of livestock feeding techniques but of those found in the literature that meet the requirements to be included in the meta-analysis review. The impact of livestock feeding techniques on animal welfare is not included here. Given the high number of substance under each category, we will only report the quantitative data when the impact is positive or negative.

3.14.1 Effects on greenhouse gas emissions

The effect on greenhouse gas emissions varies among livestock feeding techniques.

Regarding diet formulation techniques, the analysis of synthesis papers found significant effects only for lipids.

- Lipids, compared to no lipid, show a positive or non-significant effect on CH_4 emissions, depending mainly on the lipid type (e.g. medium chain fatty acids, long chain fatty acids; specific oil type). Given that lipids do not have N in their composition, a direct impact of lipids on N_2O emissions is not expected. The selected paper for lipids is Y. Wang et al. 2018a. Their results for beef cattle showed that the median mitigation efficiency on CH_4 emissions for dietary lipid additives was −14.9% ($p < 0.05$) and the mean -16.8% (95% CI: -28.6%, -5.0%; Nc=15).

Table 39. Effect (% change) of the use of diet formulation techniques on GHG emissions by GHG type. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	Source	GHG	CI_LOW	Mean	CI_HIGH	Nc	Effect
Lipids overall	Y. Wang et al. 2018b	CH_4	-28.6	-16.8 (Median -14.9)	-5.0	15	Positive

Source: references in second column.

The significant impacts of the different feed additives are synthesised in Table 40.

- Coccidiostats and histomonostats: results are not reported given that ionophores are not traditionally allowed for use in ruminants by the EU legislation (Y. Wang et al. 2018a)[27].

- Sensory additives, compared to no feed additive, have mixed results depending on the additive used. There are no results regarding N2O emissions. Observed effects on CH4 emissions are either non-significant or positive. As it depends on the type of additive, we report results from several papers, ordered by the ranking performed based on the quality of the paper:

  - The highest rank paper (Torres et al. 2021) compares essential oils (sensory additives) to monensin (ionophore), finding no significant differences on CH4 emissions of beef cattle.

  - The second rank paper is Belanche et al. 2020. They found that long term (≥ 28 days) supplementation with Agolin® Ruminant essential oils blend decreased the CH4 production per day (-8.8%), per dry matter intake (-12.9%) and per fat and protein corrected milk yield (-9.9%). The meta-analysis addressing short-term exposure found non-significant effects.

  - Darabighane et al. 2021 found that using saponin-rich sources on sheep reduced CH4 production by 1.246 g CH4/day (but it is not significant) and CH4by 0.849 g CH4/kg dry matter intake (DMI) (significant).

  - Y. Wang et al. 2018b found non-significant effect for sensory additives (saponins and essential oils) on beef cattle with only 3 observations.

  - A paper that provides a high number of results for specific additives is Lewis et al. 2015, but they have not been statistically tested. Nayak et al. 2015 found that tannins and (tea) saponins have proven effective to decrease CH4 emissions with a mean reduction of 15% ± 4%.

  - Yanza et al. 2020 analysed the effects of lauric and mystiric acids but only found positive impacts for in vitro experiments, not for in vivo ones.

  - van Gastelen et al. 2019 did not find a significant effect for garlic oil and tannins supplementation on ruminants.

  - Dai & Faciola 2019 looked at the effect of different sensory additives on ruminants CH4 emissions. The global impact of phytochemicals overall was -20%, mainly led by tannins (-20%) and saponins (-15%) while for essential oils they did not find a significant effect.

  - Harahap et al. 2020 found a positive effect in in-vitro experiments for Chitosan, but this product seems not to be among the approved feed additives in EU28.

- Zootechnical additives, have a majority of positive results on CH4. As the effect on CH4 emissions depends on the type of additive, we collect results for the different types, starting from the highest ranking synthesis paper to the lowest:

[27] See https://food.ec.europa.eu/safety/animal-feed/feed-additives/eu-register_en for an up to date version of the European Union Register of Feed Additives

[28] https://food.ec.europa.eu/safety/animal-feed/feed-additives/eu-register_en

- X. Y. Feng et al. 2020 looked at the effects of nitrates supplementation on dairy and beef cattle. The overall effect size when slow release urea was included was -13.9±0.95% for CH_4 production (g/day) and -11.4±1.25% for yield (g/kg of DMI). If results with slow release urea were removed from the sample, the effect was a bit higher: -15±1.12% for CH_4 production and -13.2 ±1.46% for CH_4 yield. Results depend on different factors, mainly dry matter intake, nitrate dose (g/kg of DMI) and type of cattle (beef or dairy). Nitrate effect sizes were significantly different between the 2 types of cattle for CH_4 production (dairy: −20.4 ± 1.89%; beef: −10.1 ± 1.52%) and CH_4 yield (dairy: −15.5 ± 1.15%; beef: −8.95 ± 1.764%). When data from slow-release nitrate sources were removed from the analysis, there was no significant difference in type of cattle anymore for CH_4 yield while values for CH_4 production were -21.1±1.87% for dairy cattle and -8.54±2.21% for beef cattle. Nitrate dose enhanced the mitigating effect of nitrate on CH_4 production and yield by 0.911 ± 0.1407% and 0.728 ± 0.2034%, respectively, for every 1 g/kg of DM increase from its mean dietary inclusion (16.7 g/kg of DM). An increase of 1 kg of DM/d in DM intake from its mean dietary intake (11.1 kg of DM/d) decreased the effect of nitrate on CH_4 production by 0.691 ± 0.2944%.

- Dijkstra et al. 2018 look at results of 3-nitrooxipropanol (3NOP) on CH_4 emissions of dairy and beef cattle. Using RVE random-effects models, an average dose of 123 mg of 3NOP/kg of dry matter in dairy and beef cattle reduced CH_4 production ($p < 0.001$) by 32.5 ± 5.74% and CH_4 yield ($p < 0.001$) by 29.3 ± 5.63%. When adjusted for 3NOP dose and dietary neutral detergent fibre (NDF) content, the CH_4-mitigating effect of 3NOP was less in beef cattle (−22.2 ± 3.33%) than in dairy cattle (−39.0 ± 5.40%). An increase of 10 mg/kg of DM in 3NOP dose from its mean (123 mg/kg of DM) enhanced the 3NOP effect on CH_4 production decline by 2.56 ± 0.550%. However, a greater dietary NDF content impaired the 3NOP effect on CH_4 production by 1.64 ± 0.330% per 10 g/kg DM increase in NDF content from its mean (331 g of NDF/kg of DM). The factors included in the final mixed-effect model for CH_4 yield were −17.1 ± 4.23% (beef cattle) and −38.8 ± 5.49% (dairy cattle), −2.48 ± 0.734% per 10 mg/kg DM increase in 3NOP dose from its mean, and 1.52 ± 0.406% per 10 g/kg DM increase in NDF content from its mean.

- Lewis et al. 2015 analysed fumaric and linoleic acids but the results have not been statistically tested.

- Nayak et al. 2015 found positive effects for nitrates and chemical inhibitors, but the main chemical inhibitors analysed were ionophores and bromochloro-methane, banned in the EU. They also looked at methods for altering the microbial community: probiotic supplementation of yeast or bacteria, and defaunation (i.e., removal of ciliate protozoa from the rumen ecosystem), but they did not prove successful in reducing CH_4 emissions (-1% ± 4% and 0% ± 9% mean reduction respectively).

- Instead, Z. Li et al. 2018 found a positive result for the elimination of rumen protozoa (defaunation) from a high number of studies. However, the effect size used (standardized mean difference, SMD) does not allow us to extract useful quantitative coefficients.

- Ungerfeld et al. 2007 found a weak positive effect for fumarate. They found a linear relationship between CH_4 emissions decrease and fumarate concentration, with a decrease in CH_4 emissions of 0.037 µmol/µmol of added fumarate ($P= 0.03$). The observed decrease was more than 6-fold lower than the theoretical stoichiometry of -0.25 mol of CH_4/mol of added fumarate. Fumarate added to continuous culture appears in general to be more effective in decreasing CH_4 production than in batch cultures, which suggests that adaptation of ruminal microbiota to metabolize fumarate occurs.

Table 40. Effect (% change unless otherwise specified) of the use of feed additives on GHG emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	GHG	Source	Population	Factors	CI_LOW	Mean	CI_HIGH	Nc	Effect
Sensory additives – plant extracts – phytochemicals overall									
Tea saponins and tannins	CH_4	Nayak et al. 2015	Cattle & sheep		-20	-15	-11	29	Positive
Plant bioactive compounds (saponin and essential oil)	CH_4	Y. Wang et al. 2018b	Beef cattle		-	-	-	3	Non-significant
Several	CH_4	Lewis et al. 2015	Ruminants		--	-	-	-	Not tested
Tannins, saponins and essential oils	CH_4 g/Kg DMI[29]	Dai & Faciola 2019	Ruminants		NA	-20	NA	NA	Positive
Sensory additives – plant extracts - Tannins									
Tannins supplementation	CH_4	van Gastelen et al. 2019	Ruminants		-	-	-	-	Non-significant
Tannins supplementation	CH_4 g/kg DMI	Dai & Faciola 2019	Ruminants		NA	-20	NA	NA	Positive
Sensory additives – plant extracts - Saponins									
Saponins supplementation	CH_4 g/kg DMI	Darabighane et al. 2021	Sheep		NA	0.849 g/kg	NA	15	Positive
	CH_4 g/day				-	-	-	12	Non-significant
Saponins supplementation	CH_4 g/kg DMI	Dai & Faciola 2019	Ruminants		NA	-15	NA	NA	Positive
Sensory additives – Essential oils									
Agolin® Ruminant essential oils blend supplementation	CH_4 production (g/day)	Belanche et al. 2020	Dairy cattle	long term (>= 28 days)	-13.2	-8.8	-4.2	7	Positive
				overall (short + long term)	-7.9	-4.6	-1.3	8	Positive

[29] DMI: Dry matter intake

Technique	GHG	Source	Population	Factors	CI_LOW	Mean	CI_HIGH	Nc	Effect
	CH4 yield (g/kg DMI)	Belanche et al. 2020		long term (>= 28 days)	-19.8	-12.9	-5.5	7	Positive
				overall (short + long term)	-8.2	-1.8	5	8	Non-significant
	CH4 intensity (g/kg FPCM[30])	Belanche et al. 2020		long term (>= 28 days)	-19.3	-9.9	0	5	Positive
				overall (short + long term)	-13.6	-7.5	-1.1	8	Positive
Essential oils compared to monensin	CH4	Torres et al. 2021	Beef cattle		-	-	-	-	Non-significant compared to monensin
Garlic oil	CH4	van Gastelen et al. 2019	Ruminants		-	-	-	-	Non-significant
Essential oils	CH4 g/kg DMI	Dai & Faciola 2019	Ruminants		-	-	-	-	Non-significant
Sensory additives – fatty acids									
Lauric acid	CH4	Yanza et al. 2020	Ruminants	In vivo	-	-	-	-	Non-significant
				In vitro	NA	-41%	NA	102	Positive
Mystiric acid	CH4	Yanza et al. 2020	Ruminants	In vivo	-	-	-	-	Non-significant
				In vitro	-	-	-	-	Non-significant
Mixed lauric and mystiric acids	CH4	Yanza et al. 2020	Ruminants	In vivo	-	-	-	-	Non-significant
				In vitro	NA	-51%	NA	102	Mixed depending on the metric
Sensory additives- Miscellaneous chemical and natural substances									
Chitosan (not	CH4	Harahap et al. 2020		In vitro only	-	-	-	-	Positive

[30] FPCM: Fat and protein corrected milk yield.

Technique	GHG	Source	Population	Factors	CI_LOW	Mean	CI_HIGH	Nc	Effect
approved in EU)									
Zootechnical additives									
Zootech-nical additives	N2O	Y. Wang et al. 2018b			-	-	-	-	Not tested
Nitrates supplemen-tation including slow release urea	CH4 production (g/day)	X. Y. Feng et al. 2020	Overall		-14.9 (SE)	-13.9	-12.9 (SE)	NA	Positive
			Dairy cattle		-22.3 (SE)	-20.4	-18.5 (SE)	NA	Positive
			Beef cattle		-11.6 (SE)	-10.1	-8.6 (SE)	NA	Positive
	CH4 yield (g/kg dry matter intake)	X. Y. Feng et al. 2020	Overall		-12.6 (SE)	-11.4	-10.2 (SE)	NA	Positive
			Dairy cattle		-16.6 (SE)	-15.5	-14.4 (SE)	NA	Positive
			Beef cattle		-10.7 (SE)	-9.0	-7.2 (SE)	NA	Positive
Nitrates supplemen-tation excluding slow release urea	CH4 production (g/day)	X. Y. Feng et al. 2020	Overall		-16.1 (SE)	-15.0	-13.9 (SE)	NA	Positive
			Dairy cattle		-23.0 (SE)	-21.1	-19.2 (SE)	NA	Positive
			Beef cattle		-10.7 (SE)	-8.5	-6.3 (SE)	NA	Positive
	CH4 yield (g/kg dry matter intake)	X. Y. Feng et al. 2020	Overall		-14.7 (SE)	-13.2	-11.7 (SE)	NA	Positive
			Dairy cattle		NA	NA	NA	NA	Positive
			Beef cattle		NA	NA	NA	NA	Positive
3-nitrooxpropanol (3NOP)	CH4 production (g/day)	Dijkstra et al. 2018	Overall		-38.2 (SE)	-32.5	-26.8 (SE)	NA	Positive
			Dairy cattle		-40.4 (SE)	-39.0	-33.6 (SE)	NA	Positive
			Beef cattle		-25.5 (SE)	-22.2	-18.9 (SE)	NA	Positive
	CH4 yield (g/kg dry matter intake)	Dijkstra et al. 2018	Overall		-34.9 (SE)	-29.3	-23.7 (SE)	NA	Positive
			Dairy cattle		-44.3 (SE)	-38.8	-33.3 (SE)	NA	Positive
			Beef cattle		-21.3 (SE)	-17.1	-12.9 (SE)	NA	Positive
Fumaric acid	CH4	Lewis et al. 2015	Cattle & buffalo		-	-	-	-	Not tested
Linoleic acid	CH4	Lewis et al. 2015	Cattle, buffalo,		-	-	-	-	Not tested

Technique	GHG	Source	Population	Factors	CI_LOW	Mean	CI_HIGH	Nc	Effect
			sheep & goat						
Chemical inhibitors (bromochloro –CH4 –not allowed)	CH4	Nayak et al. 2015			-	-	-	-	Positive but not allowed
Elimination of rumen protozoa	CH4	Nayak et al. 2015			-	-	-	-	Non-significant
	CH4	Z. Li et al. 2018	Sheep and cattle	Metric: SMD	NA	NA	NA	49	Positive
Fumarate	CH4 production	Ungerfeld et al. 2007	Ruminal batch cultures		NA	0.037 µmol/µmol of added fumarate	NA	7	Positive

Source: references in second column.

3.14.2 Effects on ammonia emissions

The effect on ammonia emissions varies among livestock feeding techniques. However, a synthesis paper (Ti et al. 2019) provides an average value for several aggregated feeding techniques, including crude protein content reduction, dietary yucca extracts, dietary bacteria, dietary acidifiers and other dietary additives. NH_3 emissions decrease on average by -39.6% (95% CI: -44.5, -34.3%; Nc= 276).

Table 41. Overall effect (% change) of analysed feeding techniques on ammonia emissions (effect varies with the specific technique used, see following tables). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	Source	Population	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Feeding techniques overall	Ti et al. 2019	Cattle, pigs and poultry	NH_3 emissions	-44.5	-39.6	-34.3	276	Positive

Source: references in second column.

The effects on diet formulation techniques are collected in Table 42.

— The systematic review of synthesis papers only found a significant effect on ammonia emissions for low crude protein diet. The selected synthesis paper is Ti et al. 2019. According to this paper, crude protein content reduction, compared to no reduction, abates NH_3 emissions by 42.8% (95% CI: -51%, -33%; Nc= 56).

Table 42. Effect (% change) of the use of diet formulation techniques on ammonia emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	Source	Population	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Low crude protein diet[31]	Ti et al. 2019	Cattle, pigs and hens	NH_3 emissions	-51.1	-42.8	-33.1	56	Positive

Source: references in second column.

The rest of interventions for which significant effects were found are feed additives. Results are shown in Table 43.

- Coccidiostats and histomonostats: results are not reported here as they apply to ruminants and these substances are allowed for use on some types of poultry but not on ruminants by the EU legislation (European Commission 2022).
- Sensory additives, compared to no feed additive, have different effects on air pollutant emissions. Ti et al. 2019 showed a significant reduction of 33% of NH_3 emissions with the supplementation of Yucca extracts[32] on broilers. Harahap et al. 2020 showed a non-significant effect for Chitosan on ruminants. Last, Lewis et al. 2015 looked at a large number of additives (*Allium arenarium* oil; *Anethum graveolens* oil; *Armoracia rusticana* extract; *Capsicum oleoresin*; Cinnamaldehyde; *Cinnamomum verum*; Condensed tannins; Eucalyptus oil; Fenugreek; *Juniperus communis* berry oil; *Mentha piperita* oil; *Origanum vulgare*; Psidium guajava; *Quillaja saponaria* extract; *Syzygium aromaticum*; Tannic acid; Thymol; *Thymus vulgaris*) but this paper does not include the required statistical testing.
- Technological additives, compared to no feed additive, have mixed effects on ammonia emissions, depending on the additive. The literature found non-significant effect on ammonia emissions and electron receptors additives (Y. Wang et al. 2018b) and for acidifiers supplementation (Ti et al. 2019) but a positive effect -37.8 % (95% CI: -46.6%, -27.5%) for bacteria supplementation of pigs and poultry (Ti et al. 2019).
- Non-specified feed additives. The selected paper for livestock overall is Ti et al. 2019. They found that dietary additives can reduce NH_3 emissions in pigs, poultry and cattle systems by -45.5% (95% CI: -54.2%, -35.2%), from 100 pairwise comparisons. For the types of additives included in this group see footnote of Table 43. Higher results are reported for poultry, as Y. Wang et al. 2019c found that the additives in feed reduced the in-house NH_3 emissions by -64% (mean, -36.3% median) ($p < 0.001$). Instead, for pigs, H. Wang et al. 2020 report that mean emissions of NH_3 decreased with the usage of other additives by -21.5% ($p < 0.01$).

[31] Low protein diet, all the low protein diets are supplemented with the first four crystalline amino acids (L-lysine, DL-methionine, L-threonine and L-trypto-phan) to balance for ideal protein ratio;

[32] Under the "Dietary yucca extracts" category the authors have included: 150 mg/kg Yucca extract; Yucca; Cinnamon 0.625g/L; Cinnamon 1.25g/L; Cinnamon 2.5g/L; Cinnamon 5g/L; Cinnamon 10g/L; Garlic 0.0625g/L; Garlic 0.125g/L; Garlic 0.25g/L; Garlic 0.5g/L; Garlic 1g/L; Ydeca 0.025g/L; Ydeca 0.05g/L; Ydeca 0.1g/L; Ydeca 0.2g/L; Ydeca 0.4g/L.

Table 43. Effect (% change) of the use of feed additives on ammonia emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	Source	Population	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Sensory additives								
Yucca extracts[33]	Ti et al. 2019	Broiler	NH3 emissions	-53.7	-33.0	-3.1	19	Positive
Chitosan	Harahap et al. 2020	Ruminants	NH3 emissions	-	-	-	-	Non-significant
Several sensory additives	Lewis et al. 2015	Cattle, buffalo, sheep and goat	NH3 emissions	-	-	-	-	Not tested
Technological additives								
Electron receptors additives	Y. Wang et al. 2018b	Beef cattle	NH3 emissions	-	-	-	-	Non-significant
Acidifiers	Ti et al. 2019	Poultry and pigs	NH3 emissions	-37.4	-8.6	33.5	12	Non-significant
Bacteria	Ti et al. 2019	Pigs and poultry	NH3 emissions	-46.6	-37.8	-27.5	89	Positive
Non specified additives								
Dietary additives[34]	Ti et al. 2019	Cattle, pigs and poultry	NH3 emissions	-54.2	-45.5	-35.2	100	Positive
Other additives[35]	H. Wang et al. 2020	Swine	NH3 emissions	-26.4	-21.5	-11.0	108	Positive
Diet additive[36]	Y. Wang et al. 2019c	Poultry	NH3 emissions	-77.7	-64.0 (-36.3 median)	-50.3	30	Positive

Source: references in second column

[33] Under the "Dietary yucca extracts" category the authors have included: 150 mg/kg Yucca extract; Yucca; Cinnamon 0.625g/L; Cinnamon 1.25g/L; Cinnamon 2.5g/L; Cinnamon 5g/L; Cinnamon 10g/L; Garlic 0.0625g/L; Garlic 0.125g/L; Garlic 0.25g/L; Garlic 0.5g/L; Garlic 1g/L; Ydeca 0.025g/L; Ydeca 0.05g/L; Ydeca 0.1g/L; Ydeca 0.2g/L; Ydeca 0.4g/L.

[34] Dietary additives included here: Cinnamon extract; Yucca schidigera extract; SS extract; OFL extract; Copper silicate nanoparticale; Aromex ME Plus at 100 mg/kg; Fresta F Plus at 150 mg/kg; Liquid Al+Clear1; Liquid Al+Clear2; Liquid Al+Clear4; Granular Al+Clear 0.5; Granular Al+Clear 1.0; Granular Al+Clear 1.5; GranularFerix-3 0.5; GranularFerix-3 1.0; GranularFerix-3 1.5; PLT0.5; PLT1.0; PLT1.5; Zn1000; Zn2000; Zn3000; 0.45 % Tannin; 1.80 % Tannin; Dried distillers grains with solubles.

[35] Other additives: e.g. fermentable carbohydratesm acidifying agent/salts, probiotics and/or prebiotics.

[36] The feed additives used during chicken rearing include zeolite, probiotics, Yucca, etc.

3.15 Landscape Features

Landscape features in general, comprise small areas of non-productive semi-natural vegetation embedded in farmlands, as well as anthropogenic structures. They are usually defined as a group/list of elements ("features"), such as hedges, ponds, ditches, trees in line, in group or isolated, field margins, terraces, dry-stone or earth walls, vegetated areas, individual monumental trees, water streams, springs or historic canal networks presented in agricultural land, either as historical remnants or newly established.

Nevertheless, there is no standard definition and typology of landscape features, existing different interpretations in the various sectors and disciplines. The literature review feeding this document applies an ad hoc "typology", synthesized from the feature types addressed in the synthesis scientific literature (i.e., it is not an exhaustive list but comprises only the features found in the papers that meet review's requirements). Thus, the quantitative information here provided is related to the feature definition proposed by the authors of the selected papers.

The impacts of landscape features include spatial and temporal comparisons between agricultural land (cropland or grassland) with and without landscape features embedded within the farm or with and without landscape features within the surrounding agricultural landscape.

3.15.1 Effects on soil organic carbon

Quantitative estimations on the effects of field margins/grass strips on SOC stocks are extracted from the MA by Van Vooren et al. 2017. The influence of this landscape feature was calculated considering field plots adjacent to a grass strips as intervention and without it as control. The width and age of the grass strip were considered too in the analysis as covariables. The overall effect is a significant increase in carbon stock (total carbon accumulated in the soil profile): +24.6% on average (95% CI: 15.0%, 30.0%) within the grass strip itself. Authors warn that this result is reliable for the upper soil layer (up to 62 cm in this case), because the majority of observations (101 out of 108) were situated in the upper 30 cm. When only the 0-30 cm depth range is considered, the effect is even more marked, the average increase in SOC being +37% (CI not provided).

Table 44 Overall effect (%) of field margins on SOC stock. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
Field margins (grass strips) adjacent to field plot	Full soil profile	15.0	24.6	35	108	Positive
	Upper layer (0-30 cm)	NA	37	NA	NA	Positive

Source: Van Vooren et al. 2017

Drexler et al. 2021 provide a meta-analysis on the effect of hedgerows on SOC and carbon sequestration. The review is based on 10 studies comprising 108 observations in total. Data from European countries were included in this global study. Two metrics calculated by the authors are reported here: change in SOC stock and average additional Carbon sequestration rate (compared to cropland alone) over a period of 20 years.

Results show that the establishment of hedgerows on cropland leads to a significant increase in SOC stocks. The SOC stock under hedgerows was on average 32 % higher than in the adjacent cropland (95 % CI: 15 - 51 %). This translates into an absolute SOC stock increase of 17 ± 12 Mg C per hectare. In contrast, the difference between SOC under hedgerows and adjacent grasslands was close to zero and not significant (9 %; 95% CI: −30%, 19%). As for Carbon sequestration rate, hedgerows compared to cropland can sequester on average 0.9 (95% CI: 0.7, 1.7] more Mg C/ha/yr, calculated as the mean over a period of 20 years.

Table 45: Effect of hedgerows on soil organic carbon (SOC) and carbon stock sequestration (C seq.) compared to cropland and grassland. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Impact	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Hedgerows adjacent to cropland vs. cropland	SOC stock	%	15	32	51	38	Positive
		Mg C/ha	5	17	39	38	Positive
	C sequestration rate (mean over 20 year period)	Mg C/ha/yr	0.7	0.9	1.7		Positive
Hedgerows adjacent to grassland vs. grassland	SOC stock	%	-30.1	-8.6	19.4	45	Non-significant

Source: Drexler et al. 2021

Abera et al. 2020 found that terraces and bunds have a positive effect on soil carbon sequestration compared to cropland and grassland without terraces, but the magnitude and significance of the effect depends on the type of terrace being considered (contour bund or stone terraces, vegetated or not). However, the study applies only to Ethiopia, and provides results for *fanya juu*[37] terraces.

Table 46: Effect (% change) of terraces and contour bunds on soil organic carbon (SOC). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Practice	Practice as named in the paper	Region	CI_LOW	Mean	CI_HIGH	Nc	Effect
Terraces	Fanya juu	Ethiopia	≥8	11	≤15	>10[38]	Positive
Contour bunds (stone or soil walls on slopes) with NO permanent green cover	(Stone or soil) bunds	Ethiopia	-6	4.9	16	>10	Non-significant
Contour bunds (stone or soil walls on slopes) WITH permanent green cover	Bunds + biological[39]	Ethiopia	89	139	164	NA	Positive

Source: Abera et al. 2020

[37] Fanya juu is a special kind of bund constructed by digging trenches along the contour of the slope and heaping the soil on the uphill side.

[38] According to the authors, in Figure 13 "The statistics of effect size is calculated for agroecology and intervention combinations with 10 or more case- studies."

[39] Biological refers to options including agroforestry and tree/forage planting as part of restoration, intensification, and/or diversification options.

3.15.2 Effects on nutrient leaching and runoff

Quantitative estimations on the effects of buffer strips on nutrient leaching and runoff are extracted from Valkama et al. 2019, based on 46 primary studies carried out in different areas of the world. This synthesis paper estimates the retention of nitrogen in the form of nitrates (NO3) and total N in surface runoff (concentration and loads pooled together) and of nitrates (concentration only) in groundwater. Results are summarized in Table 47.

The overall mean effect size is a reduction of 33% of NO3 in surface runoff (95% CI: -48%, -17%) and -57% (95% CI: -68%, -43%) when total N is considered. For N in groundwater, the mean effect size is -70% (95% CI: -78%, -62%). In all cases, results do not significantly change whether N is measured as concentration or load, and for surface runoff, results do not depend on the type of runoff (natural vs artificial).

More specific results are available for different types of pollution sources, i.e. whether buffer strips are located next to cereal fields, grasslands or feedlots. For N in groundwater, the above positive effects are not statistically affected by the source of pollution, while significant differences are found for surface runoff. A more marked positive effect was estimated for buffer strips next to cereal fields (mean effect size -53.3%; 95% CI: -62.9%, -42.2%) and for those next to feedlots (-51.2; 95% CI: -72.8%, -20.7%), i.e. where the initial concentration and loads of N are higher. Conversely, non-significant effect was estimated for buffer strips next to plots used for grass production, which is likely due to their initially low levels of pollution. Finally, the MA also considered effects in different continents and climatic zones, finding not significant differences based on studies from Europe only compared to the general results shown above.

Table 47: Effects (%) of buffer strips on N retention for surface runoff (concentration and loads) and N groundwater (concentration). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Practice	Impact	Pollution source	CI_LOW	Mean	CI_HIGH	Nc	Effect
Buffer strips	NO3-N in surface runoff	Overall	-48	-33	-17	25	Positive
	Total N in surface runoff	Overall	-68	-57	-43	16	Positive
	NO3-N and total N in surface runoff	Close to cereals	-63	-53.3	-42.2	24	Positive
	NO3-N and total N in surface runoff	Close to grasslands	-42.8	-14.3	9.5	10	Non-significant
	NO3-N and total N in surface runoff	Close to feedlots	-72.8	-51.2	-20.5	7	Positive
	NO3-N in groundwater	Overall	-78	-70	-62	38	Positive
	NO3-N in groundwater	Close to cereals	-78.7	-70.3	-60.2	28	Positive
	NO3-N in groundwater	Close to grasslands	-83.7	-72.5	-52.3	9	Positive

Source: Valkama et al. 2019

The effects of hedgerows and field margins (grass strips) on nutrient leaching and runoff are extracted from Van Vooren et al. 2017. They calculated the ratio of Nitrogen and Phosphorous inflow into the hedgerows or grass strips to the Nitrogen and Phosphorous outflow out of them. For Nitrogen, separate results on interception rates for surface and subsurface flows are provided, while for Phosphorous only the effect on surface flows was calculated. In Table 48 below, we provide results expressed in terms of % of reduction of the Nitrogen and Phosphorous load. Experiment results expressed in total mass and concentration are pooled together.

In all cases, results are significant and positive. Hedgerows can intercept on average the surface flow of N by 68.7% (95% CI: 60.5%, 77.7%) and the subsurface flow by 34.3% (95% CI: 9.5%, 52.8%). For P, the average reduction in the surface flow is 67.4% (95% CI: 46.7%, 80.2%).

Similarly, grass strips on average reduce N surface flow by 75.8% (95% CI: 58.5%, 86.1%), N subsurface flow by 31.6% (95% CI: 6.8%, 50.3%) and P surface flow by 72.7% (95% CI: 53.2%, 84.1%).

Table 48: Nutrient interception by hedgerows and grass strips: reduction of outflow compared to inflow (%). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Practice	Impact	CI_LOW	Mean	CI_HIGH	Nc	Effect
Hedgerows	N surface flow	60.5	68.7	77.7	49	Positive
	N subsurface flow	9.5	34.3	52.8	71	Positive
	P surface flow	46.7	67.4	80.2	36	Positive
Grass strips	N surface flow	58.5	75.8	86.1	90	Positive
	N subsurface flow	6.8	31.6	50.3	22	Positive
	P surface flow	53.2	72.7	84.1	116	Positive

Source: Van Vooren et al. 2017

The effects of small wetlands on N and P leaching and runoff are extracted from Carstensen et al. 2020. Results refer to Free Water Surface constructed wetlands (FWS) in agricultural landscapes, and are expressed as raw nutrient removal efficiency. This removal is a percentage calculated as the loading to the system minus the loading from the system divided by the loading to the system. The calculation of the effect size was based 109 replicates from 33 different study sites including data from Europe.

Table 49 below provides the extracted results. For both N and P results are significant and positive. Small wetlands have a N removal efficiency from system outflow of 41% (95% CI: 29%-51%) and a P removal efficiency of 33% (95% CI: 19%-47%).

Table 49: Nutrient removal efficiency (%) by small wetlands (Free Water Surface constructed wetlands). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Practice	Impact	CI_LOW	Mean	CI_HIGH	Nc	Effect
Small wetlands (free water surface constructed wetlands)	Nitrate loading reduction	29	41	51	42	Positive
	Total P removal	19	33	47	41	Positive

Source: Carstensen et al. 2020.

3.16 Fallowing

Fallow land is defined by Eurostat[40] as "all arable land either included in the crop rotation system or maintained in good agricultural and environmental condition (GAEC), whether worked or not, but which will not be harvested for the duration of a crop year. The essential characteristic of fallow land is that it is left to recover, normally for the whole of a crop year. On land lying fallow there shall be no agricultural production. Land lying fallow for more than 5 years for the purpose of fulfilling the ecological focus area shall remain arable land. In terms of land cover, fallow land may be: a) bare land with no crops at all; b) land with spontaneous natural growth which may be used as feed or ploughed in; c) land sown exclusively for the production of green manure (green fallow). The Eurostat definition includes arable land lying fallow for less than 5 years and arable land lying fallow for 5 years or more if for the purpose of fulfilling the requirements of ecological focus area."

In the literature review that feeds this document, fallows have been consider as comprising: bare land bearing no crops at all, land with spontaneous natural growth that may be used as feed or ploughed, recently abandoned and set-aside lands for less than 5 years. The latter have been called "natural fallows". The review also includes "green fallows", i.e. fallow land used for the production of green manure (often with leguminous), but results on this particular practice are provided under the "green manure" section of this document (section 3.7). The review does not include short and seasonal fallowing periods of annual crops.

3.16.1 Effects on soil organic carbon

Only one synthesis paper was retrieved addressing the effect of fallowing on soil organic carbon (SOC) (Kämpf et al. 2016a). This synthesis paper analyses the effect of the abandonment of cultivation of arable land on SOC. Abandonment of arable land can be assimilated to "fallow land" when the abandonment time is 0-4 years, and to "fallow land *for the purpose of fulfilling the ecological focus area (EFA)"* for any time period. Extracted from this meta-analysis, in Table 50 we provide data in %, and in Table 51, we provide data on absolute values of soil Carbon change in t/ ha.

On average, independently of the abandonment time, carbon stocks were 18% higher in soils of ex-arable land (fallow land for EFA according to Eurostat definition) than in arable land. However, carbon sequestration was significantly affected by abandonment time, climate and the initial SOC stock.

Time since abandonment was available for 54 estimates; the mean annual sequestration rate was 0.72 t ha−1 yr−1. For fallowing periods up to 4 years, the lower limit of the 95% confidence interval barely touches 0, therefore it can be considered a marginally no significant positive effect (Table 50). Nevertheless, the absolute soil carbon change effect is significantly positive (see Table 51). Considering that for the >4-8 years period the effects on SOC stock is significantly positive, and that for absolute SOC change in fallows of less than 5 years the effect is positive and statistically significant (2.8 ± 2.3 ton ha^{-1}), a positive effect on fallow land with regard to SOC could be accepted.

[40] https://ec.europa.eu/eurostat/statistics-explained/index.php/Glossary:Fallow_land

The mean effect size on SOC stock increases as duration of fallowing increases. However, after 16 years, SOC increase stagnated at high level (+31%).

The effects of climate and the initial SOC stock are not differentiated by abandonment time. The proportional gain of carbon was positively correlated with the aridity index of the study site. It increased from 12.5% in humid, to 31.7% in semiarid climate. Absolute values, however, did not differ significantly. The effect of the initial SOC stock on SOC change was negative for both proportional change as well as absolute values. SOC stocks were significantly higher in ex-arable land compared to arable land with initial SOC stocks of 0–50 t ha−1. In contrast, land-use abandonment had non-significant effect on SOC when the initial SOC stock was already >50 t ha−1. Although lowest sequestration rates were found in soils with high initial SOC stock, the mean annual sequestration rate did not correlate with the initial SOC stock (linear regression; Nc= 54, p = 0.075, adjusted r2 =0.042).

Table 50 Effect (% of change) of land abandonment on soil organic carbon sequestration. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Practice			Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Fallow as compared to no crops	Overall (EFA fallow)		%	12	18	23	69	Positive
	Abandon ment time	0-4 years	%	-0.2	7.4	15.2	11	Non-significant
		>4-8 years	%	11.0	16.1	21.6	15	Positive
		>8-16 years	%	20.1	26.9	34.7	16	Positive
		>16 years	%	18.2	31.1	44.2	10	Positive
	Global aridity index	humid	%	5.1	12.5	20.5	19	Positive
		moderatel y humid	%	11	19.5	28.0	18	Positive
		sub-humid	%	18.2	31.1	43.6	6	Positive
		semi-arid	%	17.1	31.7	46.4	14	Positive
	Initial SOC stock	> 75 t/ha	%	-1.2	3.8	9.0	21	Non-significant
		> 50-75 t/ha	%	-4.0	0.3	4.6	6	Non-significant
		> 25-50 t/ha	%	10.4	20.1	30.1	20	Positive
		0-25 t/ha	%	23.6	32.9	42.4	22	Positive

Source: Kämpf et al. 2016a.

Table 51 Effect (absolute soil carbon change) of land abandonment on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Practice			Metric	CI_LOW	Mean	CI_HIGH	N	Effect
Fallow as compared to no crops	Overall (EFA fallow)		t/ ha	3.7	5.4	7.1	69	Positive
	Abandon ment time	0-4 years	t/ha	0.5	2.8	5.1	10	Positive
		>4-8 years	t/ha	2.5	5.0	7.5	16	Positive
		>8-16 years	t/ha	7.0	9.2	11.4	15	Positive
		>16 years	t/ha	7.4	11.4	15.4	11	Positive
	Global aridity index	humid	t/ha	2.3	4.6	6.9	19	Positive
		moderately humid	t/ha	4.7	7.5	10.3	18	Positive
		sub-humid	t/ha	11.8	15.6	19.4	6	Positive
		semi-arid	t/ha	4.6	7	9.4	14	Positive
	Initial SOC stock	> 75 t/ha	t/ha	-1.4	2.2	5.8	21	Non-significant
		> 50-75 t/ha	t/ha	-1.7	1.0	3.7	6	Non-significant
		> 25-50 t/ha	t/ha	5.0	8.3	11.6	20	Positive
		0-25 t/ha	t/ha	5.1	6.9	8.7	22	Positive

Sources: Kämpf et al. 2016a.

3.17 Intercropping

Intercropping is a farming method that involves cultivating two or more crop species (i.e., crop mixture cropping) or genotypes (i.e., cultivar mixture cropping) in the same area and coexisting for a time so that they interact agronomically (Brooker et al. 2015; Vandermeer 1989). The results of the literature review here presented include different types of intercropping[41], namely:

Mixed intercropping: sowing multiple crop species or cultivars in the same field at the same time, in mixture with a given seeding ratio but random spatial arrangement.

Row intercropping: sowing multiple crop species in the same field at the same time in alternate rows.

Strip intercropping: sowing two (or more) crop species in the same field at the same time in multi-row strips wide enough to allow independent cultivation.

Relay (strip) intercropping: intercropping of two crop species in which the second species is under-sown in the first at a later point in the growing season.

However, most synthesis papers refer only to a mixed cropping, differentiating in some cases between crop mixture and cultivar mixture.

[41] Two practices are not included here: alley cropping, i.e., the cultivation of food, forage or specialty crops between rows of trees, that is included under Agroforestry; and dual-purpose cropping, i.e. the cultivation of two or more crops such as cereals (e.g. wheat, barley, oats and triticale) and brassicas (mainly canola) with the intended purpose of grazing during the vegetative stage and harvesting grain after the crop matures.

3.17.1 Effects on soil organic carbon

One synthesis paper (Daryanto et al. 2020) reports quantitative results on the effect of intercropping cereals and grain legumes on soil organic carbon content.

The synthesis paper is based on individual studies in several African countries. It is nevertheless considered relevant for the purpose of the present deliverable as the cereals considered includes crops largely cultivated in Europe – maize, wheat, sorghum, barley. The same is true for several intercropped grain leguminous crops covered by individual studies, such as fava bean, soybean and common bean. It is to be noted, however, that the synthesis paper includes also cereals and grain legumes not typical of the tropical and dry climates as millet, cowbean and pigeon pea.

The mean effect size expressed as increase of soil organic carbon content is 14.9%, C.I. = [6.9%, 23.9%] (Table 52)

Table 52 Effects of cereals and grain legume intercropping on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Cereals and grain legumes intercropping	CI_LOW	Mean	CI_HIGH	Nc	Effect
Concentration of Soil organic carbon (% change)	6.9	14.9	23.9	NA	Positive

Source: Daryanto et al., 2020

3.17.2 Effects on the nutrient balance

Most impacts of this practice relate to nutrients excretion that could be linked to the nutrient surplus (assuming there is not manure trade) and to other indicators (e.g. nitrates leaching and runoff, ammonia emissions, etc.).

First, we review diet formulation techniques (see

Table 53):

- The only diet formulation technique with significant results is low crude protein diet, compared to no reduction of dietary crude protein. It has a positive effect on nitrogen excretion of swine (H. Wang et al. 2020). On average, total N excretion significantly decreased by -28.5% with a lowering of dietary crude protein content. The effect of lowering dietary crude protein content on reducing urinary N excretion (39.6%) was stronger than that on reducing faecal N excretion (10.4%). The ratios of total N excretion to average daily gain were reduced by lowering dietary crude protein content by 22%. Reduction of total N excretion varied with the level of lowering dietary crude protein content (see
- Table 53). The largest decrease was derived at a crude protein reduction of > 4%. Faecal N excretion and urinary N excretion were also decreased with lowering crude protein content (not shown in
- Table 53).

Table 53. Effect (% change) of the use of diet formulation techniques on nutrient excretion. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	Population and source	Metric	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
Low crude protein diet[42]	Swine H. Wang et al. 2020	Total N excretion	Overall	-27.6	-24.7	-22.0	175	Positive
		Urinary N	Overall	-36.6	-32.8	-28.8	113	Positive
		Faecal N	Overall	-15.3	-10.0	-4.1	115	Positive
		Total N excretion to average daily gain	Overall	-23.6	-19.7	-15.6	111	Positive
		Total N excretion	>6%	-52	-45	-39	10	Positive
			>6% + AA[43]	-49	-38	-25	3	Positive
			4-6%	-42	-39	-35	33	Positive
			4-6%+AA	-40	-30	-17	5	Positive
			2-4%	-21	-18	-14	71	Positive
			2-4% + AA	-24	-15	-5	16	Positive
			>2%	-18	-12	-5	35	Positive

Source: references in second column.

Results on feed additives are addressed next and shown on Table 54.

- Enzymes supplementation compared to no enzyme, have different effects on nutrient loss, with one synthesis paper (H. Wang et al. 2020) reporting positive effects and non-significant effects depending on the enzyme type (carbohydrase, phytase, enzymes and compound enzymes) and the metric (N or P excretion and the ratio of total N or P excretion to average daily gain). “On average, total N excretion significantly decreased by 8.9% with supplementation of exogenous enzymes”. “The ratios of total N excretion to average daily gain were reduced by enzymes supplementation by 13.3%”. Effects on P excretion are only provided by enzyme type (see Table 54). Supplementation of phytase and compound enzymes led to significant reductions while carbohydrase enzymes supplementation was not effective. There is also a paper (Lewis et al. 2015) with results for N and P excretion but lacking a proper statistical analysis.

[42] Low protein diet, all the low protein diets are supplemented with the first four crystalline amino acids (L-lysine, DL-methionine, L-threonine and L-trypto-phan) to balance for ideal protein ratio.

[43] AA = supplementation of amino acids except for the first four crystalline amino acids (L-lysine, DL-methionine, L-threonine and L-trypto-phan).

— **Nutritional additives**, compared to no feed additive, have 1 paper reporting positive effect on dairy cattle N excretion (Salami et al. 2021) and 1 paper does not have a proper statistical analysis (Lewis et al. 2015). Feeding slow-release urea decreased nitrogen excretion by 2.7% to 3.1% (-12 to -13 g/cow/d). Salami et al. 2021 report that slow-release urea supplementation reduced the amount of nitrogen excreted per dairy cow by 2.7% to 3.1% (-12 to -13 g/cow/day) and per unit of milk by 3.6% to 4.0% (-0.50 to -0.53 g N/kg milk). In these results, N excretion has been calculated by Salami et al. 2021 from result variables obtained from their meta-analysis: dry matter intake (kg/cow); nitrogen intake (g/day); milk yield (kg/day) and depend on the equation used for the calculation.

— **Sensory additives**, compared to no feed additive, have different effects on nutrient loss, with 1 paper (Herremans et al. 2020) reporting positive effect on urinary N excretion (-11%) and negative effect on faecal N excretions (+10%) for tannins supplementation of dairy cows. The meta-analysis shows that tannins have no impact on nitrogen use efficiency (p>0.05) but there is a shift from urinary to faecal N that may be beneficial for environment preservation, as urinary N induces more harmful emissions than faecal N. Another paper (Lewis et al. 2015) that analyses the effects of condensed tannins and *Quillaja saponaria* extract on N excretion lacks a proper statistical analysis.

— **Non specified feed additives**, compared to no feed additive, have different effects on nutrient loss. H. Wang et al. 2020 report for "other additives", including fermentable carbohydrates, acidifying agent/salts, prebiotics etc., a positive effect for swine urinary N excretion, a negative effect for faecal N excretion and non-significant effect for total N excretion and for the ratio of total N excretion to average daily gain (%).

Table 54. Effect (% change) of the use of feed additives on nutrient excretion. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	Population and source	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Enzymes supplementation							
Enzymes overall	Swine *Source:* H. Wang et al. 2020	Total N excretion	-12.9	-8.6	-4.4	174	Positive
		Urinary N	-18.4	-7.8	3.9	28	Non-significant
		Faecal N	-13.1	-1.9	10.5	31	Non-significant
		Ratio total N excretion to av. daily gain	-16.5	-12.4	-8.163	160	Positive
Carbohydrase[44]	Swine	Total N excretion	-14.1	-7.2	0.3	72	Non-significant

[44] Carbohydrase, xylanase, β-glucanase, α-amylase, mannanase, α-galactosidase, cellulose.

Technique	Population and source	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
	Source: H. Wang et al. 2020	Total P excretion	-14.8	-4.0	7.9	31	Non-significant
		Ratio total P excretion to av. daily gain	-13.0	-5.0	4.3	63	Non-significant
Phytase	Swine *Source:* H. Wang et al. 2020	Total N excretion	-13.7	-5.3	3.8	35	Non-significant
		Total P excretion	-32.5	-26.9	-21.0	40	Positive
		Ratio total P excretion to av. daily gain	-24.5	-20.2	-15.5	29	Positive
Protease	Swine *Source:* H. Wang et al. 2020	Total N excretion	-25.6	-16.7	-6.7	24	Positive
		Total P excretion	NA	NA	NA	NA	NA
Compound enzyme[45]	Swine *Source:* H. Wang et al. 2020	Total N excretion	-17.3	-8.1	2.0	43	Non-significant
		Total P excretion	-28.2	-20.6	-12.2	39	Positive
		Ratio total P excretion to av. daily gain	-21.1	-14.6	-6.9	42	Positive
Nutritional additives							
Slow-release urea	Dairy cows *Source:* Salami et al. 2021	N excretion / cow/day	NA	-2.7 to -3.1%	NA	44	Positive
		N excretion / kg milk	NA	-3.6% to -4%	NA	44	Positive
Sensory additives							
Sensory additives: Tannins	Dairy cows *Source:* Herremans et al. 2020	N use efficiency	-2.3	0.7	3.7	26	Non-significant
		Urinary N	-15.6	-10.9	-6.0	34	Positive
		Faecal N	6.7	10.3	14.1	42	Negative
Non specified feed additives							

[45] Compound enzymes, containing at least one kind of phytase, protease and carbohydrase enzymes.

Technique	Population and source	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Other additives	Swine *Source:* H. Wang et al. 2020	Total N excretion	-3.2	1.7	7.1	123	Non-significant
		Urinary N	-14.5	-8.3	-2.1	105	Positive
		Faecal N	11.7	18.9	26.7	105	Negative
		Ratio total N excretion to av. daily gain	-12.2	-5.3	1.9	50	Non-significant

Source: references in second column.

3.17.3 Effects on the nutrient balance

The literature provides the impacts on macronutrients fertilisation, which is a component of the nutrient balance (see Annex 1).

According to the relative majority of results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), compared to monoculture, intercropping decreases fertilisation requirements as nutrient uptake is increased. The selected synthesis paper is C. Li et al. 2020.

The meta-analysis by C. Li et al. 2020 includes results from relay strip cropping, row intercropping and mixed intercropping. They found maximum yield gains, compared with monocultures, for two types of intercropping: (i) mixtures of maize with short-grain cereals or legumes that had substantial temporal niche differentiation from maize, when grown with high nutrient inputs, and using multirow strips of each species (highly fertilised relay strip intercropping); and (ii) growing mixtures of short-stature crop species, often as full mixtures, with the same growing period and with low to moderate nutrient inputs. However, there were significant differences when maize was included.

The N fertiliser equivalent ratio of intercrops (defined as the amount of N fertiliser used in sole crops to produce the same yields as obtained in intercropping) with and without maize were 1.33 ± 0.04 and 1.19 ± 0.05, respectively. So, to achieve the same yield as intercrops, the sole crops used 19–33% more N fertiliser than the intercrops (or intercrops used -16%- -24.8% less fertilisers than monoculture), indicating increased N use efficiency in intercropping if nutrient use efficiency is expressed as fertiliser used per unit yield produced. The N fertilizer equivalent ratio of intercrops with maize was higher (P=0.01) than that of intercrops without maize, indicating that intercrops with maize save more N fertiliser compared with sole crops than intercrops without maize.

The P fertiliser equivalent ratio of intercrops with maize (1.36 ± 0.03) was larger than the P fertilizer equivalent ratio of intercrops without maize (1.19 ± 0.04) (P<0.001), indicating that, while both types of intercrops save P fertiliser compared with sole crops, the savings are greater in intercrops with maize (26.5%) than in intercrops without maize (16%).

Table 55 Effects (% change) of intercropping on nutrients fertilisation. SE: Standard Error of the mean. Nc: number of pairwise comparisons.

Metric	Population	Mean-SE	Mean	Mean+ SE	Nc	Effect
N fertiliser	Crops include maize	-27.0	-24.8	-22.5	407	Positive
	Crops do not include maize	-19.4	-16.0	-12.3	231	Positive
P fertiliser	Crops include maize	-28.1	-26.5	-24.8	325	Positive
	Crops do not include maize	-18.7	-16.0	-13.0	119	Positive

Source: C. Li et al. 2020

3.18 Crop rotation

According to Eurostat's definition[46], crop rotation on arable land is the practice of alternating crops grown on a specific field in a planned pattern or sequence in successive crop years, so that crops of the same species are not grown without interruption on the same field. In a rotation, the crops are normally changed annually, but they can also be multi-annual. If the same crop is grown continuously, the term monoculture can be used to describe the phenomenon.

Results reported here concern rotation of 2 crops compared to monoculture, rotations of 3 or more crops compared to monocultures, and rotations of 3 or more crops compared to 2 crops rotations (also referred to as simplified rotation). Results do not include the effects of cover crops, break crops or fallowing, which are reported in specific sections.

3.18.1 Effects on greenhouse gas emissions

According to the results retrieved from the systematic literature review of synthesis papers (Schievano et al. 2023), significant results are found on the impact of crop rotation on Global Warming Potential (GWP), which includes CO2 equivalent emissions from aggregated N2O and CH4 emissions with, if available, equivalents from soil C sequestration rate (ΔSOC), farm operations, and N fertilisation.

The selected paper, Sainju 2016 found differing effects on the GWP depending on the crop population: negative effects on GWP were found for large grain crops like maize and soybean; conversely, a positive effect was found for small grain crops like barley and pea.

Crop rotation increased GWP[47] by 46% and Greenhouse gas intensity (GHGI)[48] by 41% compared with monocropping (full dataset). This was especially true for large grain crops, such as maize (*Zea mays L.*) and soybean (*Glycine max L.*), where GWP and GHGI were 215 and 325%, respectively, greater under maize-soybean than continuous maize. In contrast, for small grain crops, such as barley (*Hordeum vulgare L.*) and pea (*Pisum sativum L.*), GWP was 22% lower under barley-pea than continuous barley.

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[46] https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Glossary:Crop_rotation

[47] GWP and GHGI account for CO2 equivalents from N2O and CH4 emissions with or without equivalents from soil C sequestration rate (ΔSOC), farm operations, and N fertilisation.

[48] GHGI is expressed as net GWP per unit crop yield

This is explained by the authors. "*Increased ΔSOC due to greater crop residue returned to the soil reduced GWP and GHGI under continuous maize than maize-soybean rotation, although N fertilisation rate to produce sustainable yield was higher in continuous maize. In contrast, greater N_2O emissions following soybean increased GWP and GHGI in maize-soybean rotation. Under small grain crops, however, several researchers found that including legumes, such as pea and lentil (Lens culinaris L.), in rotation with non-legumes, such as wheat (Triticum aestivum L.) and barley, reduced GWP and GHGI compared with continuous non-legumes. They observed this because (1) no N fertiliser was applied to legumes compared with non-legumes which required large amount of N fertilisers to sustain yields, as N fertiliser stimulates N_2O emissions and (2) legumes supplied greater amount of N to succeeding crops due to higher N concentration when above- and belowground residues were returned to the soil and reduced N fertilisation rate than non-legumes.*" They also found that "*legume-non-legume rotation increased ΔSOC because of increased turnover rate of plant C to soil C compared with continuous non-legume. Greater number of experiments and magnitude of changes, however, resulted in higher GWP and GHGI with monocropping than crop rotation under large than small grain crops when values were averaged across experiments during data analysis.*"

Table 56. Effect (% change) of crop rotation on greenhouse gas emissions (together with soil organic carbon where indicated). SE: Standard Error of the mean. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Source	Impact	Crops	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Rotation 2 crops	Sainju 2016	N_2O+ CH_4+ SOC + other GHG	Full dataset	GWP	33.7 (SE)	46	58.3 (SE)	11	Negative
				GHGI	25.3 (SE)	41	56.7 (SE)	11	Negative
			Large grains (maize & soybeans vs. maize)	GWP	166.4 (SE)	215	263.6 (SE)	4	Negative
				GHGI	0 (SE)	325	650 (SE)	4	Negative
			Small grains (cereals like barley or wheat & legumes like pea or lentils vs. cereals)	GWP	-23 (SE)	-22	-21 (SE)	3	Positive
				GHGI	NA	NA	NA	3	NA
Rotation mostly 2 crops	Han et al. 2017	N_2O	Maize–soybean or dry bean vs maize; lupin–wheat vs wheat		-	-	-	31	Non-significant
Rotation 2 crops	Decock 2014	N_2O	Maize-soybean vs. maize		-	-	-	17	Non-significant

Source: references in the second column.

3.18.2 Effects on soil organic carbon

The review of synthesis papers showed that more positive/significant impacts of crop rotation on SOC were found for larger and more diverse crop rotations (i.e. with 3 or more crops and/or different crop families), while simplified successions of 2 crops like maize-soybean tend to show less significant or non-significant effects (Schievano et al. 2023).

The selected synthesis paper is McDaniel et al. 2014. They found that crop rotation overall, as compared to monoculture, increased total carbon by 3.6% (95% CI: 2%, 5%; Nc=367). However, this impact is affected by several factors. The number of crops included in the rotation had a significant effect: increasing the number of crops in rotation from two to three increased total C from 1.9% to 7.5%, but adding more than three appeared to have diminishing returns on total C (3.7% for four crops, and 7.7% for five or more crops). The presence of cover crops in the rotation had a significant impact: rotations without cover crops did not significantly influence SOC. The type of monoculture crop under comparison to rotation also significantly influenced the rotation effect. Soybeans showed the greatest response to rotation, with an 11% increase, whereas introducing a rotation into maize monocultures did not increase total C. Last, mean annual temperature and precipitation correlated positively with rotation effects on total carbon.

Table 57. Effect (% change) of crop rotation on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Technique	Number of crops	CI_LOW	Mean	CI_HIGH	Nc	Effect
Crop rotation overall	All	2.0	3.6	5.1	367	Positive
	2	0.1	1.9	3.7	239	Positive
	3	2.8	7.5	11.4	78	Positive
	4	-0.8	3.7	8.7	25	Non-significant
	5 or more	3.8	7.7	11.8	26	Positive
Crop rotation with no cover crops	All	-0.4	1.5	3.5	251	Non-significant
	2	-0.9	1.0	2.9	215	Non-significant
	3	-2.2	5.8	12.6	44	Non-significant
Crop rotation with cover crops	All	6.8	8.5	10.5	81	Positive
	2	4.2	7.7	12.2	17	Positive
	3	7.9	10.8	13.8	26	Positive
	4	-0.3	3.5	6.7	17	Non-significant
	5	7.5	10.6	14.8	21	Positive

Source: McDaniel et al. 2014.

3.19 Cover and catch crops

The terminology "cover and catch crops" includes various crop types such as those that are sown on purpose during the fallow season, under-sown to winter/spring main crops, or crops that are sown between rows of orchards or vineyards to function as living cover to prevent erosion and thereby minimise the risk of surface runoff by improving infiltration. Cover crops may also function as catch crops, which scavenge the remaining nitrogen in the soil after the main crop is harvested and effectively reduce nutrient losses.

Cover crops are temporary crops and as such may either be cut and removed, or incorporated into the soil. The latter practice provides nutrients to the soil and is referred to as green manuring: in this document is analysed as a type of organic fertilisation in section 3.7. The terminology used in the selected meta-analyses also includes spontaneous vegetation that is left to grow to fulfil the same purpose as cover crops, which is thus considered as a form of cover crops in this exercise. In this work we distinguish between three farming practice interventions: cover crops in general, legume cover crops and non-legume cover crops.

3.19.1 Effects on greenhouse gas emissions

CH4 emissions were significantly increased by cover crops according to Jian et al. 2020, by 97.4% (95% CI: 61.1%, 141.3%) compared to bare soil. This result is obtained from data from North America, Europe, Africa, and Asia, specifically eastern China; cash crop types included are: maize, soybean, wheat, vegetable, maize-soybean rotation, maize-soybean-wheat rotation, and other. Cover crops include legume, grass and multi-species mixture. This effect is explained by the higher amounts of decomposable organic matter.

Differing results are reported for N2O emissions and several factors influence these results, such as the type cover crop (legume versus non-legume), the cover crop residue management methods (removed, left on surface, or incorporated to the soil) and the mineral nitrogen fertilisation rates.

We selected the paper from Muhammad et al. 2019 that has a complete range of results. According to them, if the cover crops are legumes, N2O emissions significantly increased by 60.6% (95% CI: 29.8%, 95.7%) as compared to bare soil on arable cropland. Instead, non-legume cover crops decreased N2O emissions by -36.1% (95% CI: -45.8%, -25.5%).

Effects are also influenced by the management of covers crops (e.g. left on the soil or incorporated into it). Muhammad et al. 2019 report that N2O emissions increased by 133.3% (95% CI: 108.6%, 165.8%) if the residue was incorporated to the soil, but decreased by -54.7% (95% CI: -62%, -45.8%) if it was left on the surface. If cover crop residue was removed from the soil, there were not significant effects. Other factors affecting N2O emissions are cover crop biomass and the C/N ratio of the cover crop residue.

Table 58. Effect (% change) of cover and catch crops on greenhouse gas emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Impact	Intervention	Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
CH_4	Cover crops overall	Jian et al. 2020	61.1	97.4	141.3	25	Negative
N_2O	Cover crops overall	Jian et al. 2020	12.7	40.2	75.1	30	Contrasting effects
		Muhammad et al. 2019	-26.6	-16.1	-5.4	480	
	Cover crop type						
	Legume cover crops	Muhammad et al. 2019	29.8	60.6	95.7	152	Negative
	Non-legume cover crops	Muhammad et al. 2019	-45.8	-36.1	-25.5	303	Positive
	Cover crop residue treatment						
	Left on surface	Muhammad et al. 2019	-62.0	-54.7	-45.8	244	Positive
	Incorporated to the soil	Muhammad et al. 2019	108.6	133.3	165.8	201	Negative
	Removed	Muhammad et al. 2019	-30.1	-2.9	26.5	35	Non-significant

Source: references in the third column.

3.19.2 Effects on soil organic carbon

The majority of results retrieved from the systematic review of synthesis papers showed that cover and catch crops have an overall positive effect on carbon sequestration (Schievano et al. 2023).

The selected synthesis paper is McClelland et al. 2021. It looks at the effects of cover crops on both annual and perennial cropping systems. Results show a strong, positive effect on SOC stock in the upper soil layer from 0 to 30 cm. The relative increase is 12% (95% CI: 7%, 16%), equal to 1.11 Mg C/ha (95% CI: 1.07, 1.16). Growing window (cover crop planting and termination date) and cover crop biomass production were strong predictors of SOC response to cover cropping. Continuous cover and autumn planted and terminated cover crops significantly affect SOC response (36% and 27%, respectively) relative to overwintering (8%) and summer cover crops (7%). However, there were relatively few observations for all growing window categories except for overwinter, thus results should be interpreted with caution. Other factors—C:N ratio, functional type, diversity, and termination method (incorporated or not incorporated)—did not affect SOC response.. SOC response under high cover crop biomass production (>7 $Mg \cdot ha^{-1} \cdot yr^{-1}$) was 30%, which was almost 20% greater than low (12%) or moderate (11%) cover crop biomass production. Grasses and mixtures had strong positive effects on SOC when grown as a continuous cover rather than as an overwinter or summer cover crop, respectively. The effect of legume cover crops on SOC response was greater under fall, summer, and continuous cover growing windows relative to overwinter. "Time since introduction" of a cover crop was a poor predictor of SOC response even after controlling for outliers. Thus, authors of the MA were unable to make a robust estimate of the average annual SOC stock change from 0 to 30 cm under cover crops. A simple calculation dividing the overall mean (1.11 Mg C/ha) by the average time since introduction across the observations in the data set (5.2 yr) indicates an average annual SOC stock change of 0.21 Mg C·ha−1·yr−1 (95% CI: 0.20, 0.22). While not robust, this estimate provides a general indication of the expected annual SOC stock change with cover crops.

Table 59 Effects of cover crops on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Population	Source	Factor and metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
0-30 cm soil on arable and permanent cropland	McClelland et al. 2021.	Overall (%)	7	12	16	181	Positive
		Overall (Mg C/ha)	1.07	1.11	1.16	181	Positive
		Overall (Mg C/ (ha yr))	0.20	0.21	0.22	181	Positive
		Growing window (period) (%)					
		Overwinter	4	8.8	13.7	120	Positive
		Summer	0.9	6.9	13.3	25	Positive
		Autumn	10.6	27.6	47.2	11	Positive
		Continuous	18.8	36.6	57.1	14	Positive

Source: references in the second column.

3.19.3 Effects on the nutrient balance

Only one synthesis paper (Quemada et al. 2013a) was found looking at the effects of cover crops on Nitrogen Use Efficiency (NUE) of the subsequent cash crop calculated as the yield per unit of N applied as fertiliser (Partial Factor Productivity of Fertiliser or PFPN). Their study focuses on irrigated cropping systems. The effect of cover/catch crops, as compared to bare soil, is positive for leguminous species (6%; 95% CI: -13%, 2%; Nc=4), while non-significant effect resulted for non-legume species. However, these results were supported by a limited number of primary studies.

Table 60 Effects (% change) of cover crops on nitrogen use efficiency. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Population	Metric	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
Irrigated cropping systems	PFPN = Yield/N fertiliser	Legume	1.2	13.4	29.6	4	Positive
		Non-legume	-12.7	-6.1	1.6	9	Non-significant

Source: Quemada et al. 2013a

3.19.4 Effects on nutrient leaching and runoff

The majority of results retrieved from the systematic review of synthesis papers showed that the effect on nutrient loss of cover and catch crops overall, as compared to bare soil, is overall positive (Schievano et al. 2023).

The selected papers for extraction of quantitative information are: R. Liu et al. 2021 that studies N and P losses with runoff in permanent cash crops (orchards) and Thapa et al. 2018 that analyses nitrate leaching in arable land.

R. Liu et al. 2021 report that losses of inorganic N forms (NO3−-N and NH4+-N) were significantly reduced by cover crops. Due to limited data, only nutrient losses with runoff were compiled. On average, the presence of ground cover (cover crops and mulch) reduced the losses of total N, NO3−-N and NH4+-N by 60%, 48% and 53%, respectively. Non-legume cover crops reduced total N, NO3−-N and NH4+-N losses more than legume cover crops. Cover crop significantly reduced total phosphorous and dissolved phosphorous loss by 62% and 61%, respectively. Nutrient losses were found to depend on several factors: higher mean annual precipitation, higher mean annual temperature and steeper slopes tended to increase total N loss reduction, and dissolved P reduction, but not total P reduction.

Table 61 Effects of cover crops on nutrients run-off. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Population	Source	Metric	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
Permanent crops (orchards)	R. Liu et al. 2021	Total N runoff	Cover crops overall	-52.5	-60.4	-69.1	57	Positive
			Legume	-32.0	-47.7	-63.7	15	Positive
			Non-legume	-55.6	-65.9	-76.5	38	Positive
		NO3- runoff	Cover crops overall	-39.9	-48.4	-57.4	33	Positive
			Legume	-23.8	-46.1	-69	8	Positive
			Non-legume	-39.3	-49.0	-59.5	25	Positive
		NH4+ runoff	Cover crops overall	-44.3	-52.9	61.8	31	Positive
			Legume	-29.5	-49.6	-70.4	9	Positive
			Non-legume	-42.8	-53.2	-64.3	21	Positive
		Total P	Cover crops overall	-54.8	-62.4	-70.4	56	Positive
			Legume	-43.8	-58.8	-74.0	12	Positive
			Non-legume	-54.0	-63.3	-72.9	43	Positive
		Dissolved P	Cover crops overall	-50.0	-60.8	-72.2	26	Positive
			Legume	-15.3	-35.4	-55.7	5	Positive
			Non-legume	-51.4	-64.8	-78.7	17	Positive

Source: references in the second column.

Most of the studies included in Thapa et al. 2018 evaluated the effectiveness of non-leguminous cover crops in reducing NO3− leaching (216 observations from 27 studies), whereas leguminous and non-legume–legume cover crop mixtures were evaluated in only three studies with 9 and 13 observations, respectively. Compared with no cover crop controls, non-leguminous cover crops significantly reduced NO3− leaching by 56% (95% CI = −66%, −43%). Legumes alone or in combination with non-legumes had non-significant effect on NO3− leaching.

Results by Thapa et al. 2018 depend on several factors: cover crop species; planting dates; cover crop biomass production; mean annual precipitation; and soil texture. Compared with no cover crop controls, grasses and broadleaf species reduced NO3− leaching by 50% (99% CI = −61 to −37%) and 67% (99% CI = −77 to −54%), respectively. Early-planted non-leguminous cover crops significantly reduced NO3− leaching compared with no cover crop controls (mean reduction of 64, 60, and 49% for August-, September-, and October-planted non-legume cover crops, respectively). When planting non-leguminous cover crops after November, there was no advantage of having a cover crop on NO3− leaching (mean = −28%, 99% CI = −50 to 3%).

Table 62 Effects of cover crops on nitrates leaching. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Population	Source	Metric	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
Arable crops	Thapa et al. 2018	NO3− leaching	Overall	NA	NA	NA	238	Positive
			Non-leguminous	-66	-56	-43	216	Positive
			Leguminous	-30	-1	40	9	Non-significant
			Mixtures	-68	-25	78	13	Non-significant
			Grasses	-61 (99% CI)	-50	-37 (99% CI)	145	Positive
			Broadleaf	-77 (99% CI)	-67	-54 (99% CI)	50	Positive
			Planted August October	several	≥49	several	193	Positive
			Planted after November	-50 (99% CI)	-28	3 (99% CI)	14	Non-significant

Source: references in the second column.

3.20 Leguminous (nitrogen-fixing) crops

Leguminous crops —Fabaceae— develop symbiosis with bacteria within nodules in their root systems (rhizobia), producing reactive nitrogen compounds from atmospheric N2, which help the plant to grow. When the plant dies, the fixed nitrogen is released, making it available to other plants; this helps to fertilise the soil. The legume family include crops such as soybean, alfalfa, clover, lupin, kudzu, peanut and rooibos.

In the exercise of systematic review of synthesis papers (Schievano et al. 2023), the use of Leguminous (nitrogen-fixing) crops in agriculture is considered within the context of different agricultural practices, including: 1) crop rotations involving legumes, 2) leguminous cover crops, 3) intercropping, mixed cropping, practices involving two or more crops in proximity Including legumes, 4) agroforestry systems including leguminous perennials, 5) green fallows including legumes, 6) grassland for leguminous forage production and pastures enriched with legumes.

Results were derived from two types of pairwise comparisons: 1) direct comparison, where a farming practice with leguminous crops was compared to the same farming practice, but without leguminous crops; 2) indirect comparison, where a farming practice (either with or without leguminous crops) was compared to the absence of the practice. In this section we provide results from direct comparisons.

3.20.1 Effects on soil organic carbon

The use of leguminous crops in crop rotations, when compared to rotations without leguminous crops, show positive effects on carbon sequestration (higher soil organic carbon) according to the synthesis paper by Powlson et al. 2016.

This was observed in tropical agro-eco-systems, concretely for the Indo-Gangetic Plains. The individual C sequestration rate for crop diversification (including legumes in rotations) was 0.47 ±0.046 Mg C ha-1 yr-1, and was significantly different from conventional practice. The relatively large predicted rate of SOC increase from crop diversification was only based on 4 data comparisons from two studies, showing diverse results. In one study crop diversification comprised a legume replacing rice in the rice–wheat rotation and in the other a maize-wheat rotation was modified to include legumes.

Table 63 Effects of leguminous or N-fixing crops on rates of increase of soil organic carbon. SE: Standard Error of the mean, Nc: number of pairwise comparisons.

Population	Factor and metric	Mean –SE	Mean	Mean + SE	Nc	Effect
Arable crops in Indo-Gangetic plains	Mg C ha^{-1} yr^{-1}	0.37	0.47	0.57	4	Positive

Source: Powlson et al. 2016

3.21 Tillage practices: No tillage and reduced tillage

In the sectoral literature and agronomic practice, a plethora of terms is used to refer to less intensive forms of tillage, compared to conventional tillage, which in turn is sometimes defined in different ways. The EUROSTAT definition[49] of tillage management practices, is the following:

Conventional tillage: refers to the arable land treated by tillage practices involving inversion of the soil, normally with a mouldboard or a disc plough as the primary tillage operation, followed by secondary tillage with a disc harrow;

Conservation tillage (can be assimilated to "Reduced tillage" in the MA review): a system of practices that leaves plant residues (at least 30 %) on the soil surface for erosion control and moisture conservation, normally by not inverting the soil. It comprises more specific tillage practices, such as trip tillage, zonal tillage, tined tillage, vertical tillage, ridge tillage.

Zero tillage (or No tillage in the MA review): a minimum tillage practice in which the crop is sown directly into soil not tilled since the harvest of the previous crop. Weed control is achieved by the use of herbicides and/or appropriate mulching and stubble is retained for erosion control.

[49] https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Glossary:Tillage_practices

Conservation agriculture[50] encompasses a set of complementary agricultural practices which minimise alteration of the composition and structure of the soil (...): permanent soil cover (cover crops, residues and mulches) (...); minimal soil disturbance (through reduced or no-tillage) (...); diversified crop rotations and crop combinations (...). This definition differs from the one in the MA review in the fact that it also includes reduced tillage and not only no- tillage.

In the literature, additional terms are reported (e.g., low tillage, reduced tillage, minimum tillage) and specific definitions are not always consistent. For the purposes of this document, we adopt the Eurostat nomenclature[50], including under the Conservation Tillage class all forms of tillage that are less intensive (i.e., determine less soil disturbance) than conventional tillage but more intensive than no tillage.

3.21.1 Effects on greenhouse gas emissions

No tillage (NT) compared to conventional tillage (CT):

The study by Huang et al. 2018 analyses the effects of no-tillage on CH4, CO2, and N2O, as well as their global warming potential (GWP). They do not provide values for CH4+N2O but only GWP calculated as the sum of CH4+N2O+CO2 (only when fluxes for all three greenhouse gas emissions (GHG) species were reported in each single study).

In addition, Huang et al. 2018 report a positive effect: no-tillage decreased soil CH4 emission by 15.5% and on average had non-significant effect on soil CH4 uptake[51]. They report however significant differences with crops, climate, and soil pH. No-tillage reduced CH4 emission in rice production systems by 22.4% (95% CI: -32.4%, -11.3%; Nc= 48) but the effects on upland crops were not significant. It increased CH4 uptake in wheat by 31.1%. The overall results reported by Huang et al. 2018 have a high share of observations corresponding to rice paddies.

The second selected synthesis paper, J. Feng et al. 2018, includes a lower number of individual studies for rice paddies. This study reports non-significant effect for crops overall. No-tillage significantly increased N2O emission by 10.4% but these effects seem to diminish with long-term duration of NT. Changes in N2O emission between NT and conventional tillage were only significantly different with wheat production, where a 15.2% increase in emission under no-tillage was noted. The net effect of NT (relative to CT) was influenced by several environmental and agronomic factors (climatic conditions, tillage duration, soil texture, pH, and crop species)[52].

For results of N2O+CH4 together, we report both results by J. Feng et al. 2018 and Shang et al. 2021. Results from this last paper are not provided in % but in absolute change (Table 64). For those studies that measured fluxes of all three GHGs, no-tillage exhibited no significant difference in GWP compared to conventional tillage for all crops overall. However, Huang et al. 2018 report that soil pH and crop species significantly affected the difference in GWP. No-tillage decreased GWP by 31.2% in acidic soils and by 24.8% (95% CI: -39.7%, -6.1%; Nc=11) in rice fields. However, fluxes of all three

[50] https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_tillage_practices

[51] Soil absorbs CH4 from the atmosphere through the microbial process of CH4 oxidation by methanotrophs.

[52] Values by Huang et al. 2018 are supported by similar results from other synthesis papers. Values by J. Feng et al. 2018 for no tillage are provided for comparison, as they also report results for reduced tillage + crop rotation, no tillage + crop rotation, reduced tillage + crop straw return and no tillage + crop straw return, both for uplands and paddies that are not reported here.

GHGs and crop yield were reported in few studies comparing no-tillage to conventional tillage, so these results should be interpreted cautiously.

Reduced/conservation tillage (RT) compared to conventional tillage (CT):

We used for quantitative data extraction the synthesis paper by J. Feng et al. 2018 which concludes that the effects of these practices depend on the interaction of no-tillage or reduced tillage with other agronomy practices and land use types.

Overall, reduced tillage not combined with other techniques did not exhibit significant effects on CH_4 and N_2O emissions as compared with conventional tillage. In uplands (non rice/paddy crops), the effects of reduced tillage on CH_4 uptake, N_2O emission and CH_4+N_2O were not significant. In rice paddies, reduced tillage had non-significant effects on N_2O emissions but significantly increased the CH_4 emission by 17% (95% CI: 5%, 31%; Nc=22) and overall CH_4+N_2O by 13.6% (95% CI: 3%, 26%; Nc=22) compared with conventional tillage. The negative effect on CH_4 emissions is based on 22 pairwise comparisons and is explained in the synthesis paper: *"CT incorporated crop residue into deeper soil than reduced tillage, reducing the decomposition of these residues through the protection of the soil matrix (Zhang et al. 2015). Lower soil disturbance and a shallower CH_4 oxidation zone for no-tillage than reduced tillage were conducive to improving CH_4 oxidation (Zhang et al. 2013; Koga et al. 2004)"*.

J. Feng et al. 2018 found that the effectiveness of no-tillage and reduced tillage depended on tillage methods, land use type, and agricultural practices. In rice alternated with another crop during the same year, reduced/conservation tillage led to a not significant increase of CH_4 and GWP of N_2O+CH_4 (Nc=14), while it was significant for double rice, with longer annual flooding time, but this result can be biased as the number of observations is very low (Nc=3). In upland, reduced tillage significantly decreased the overall GWP of CH_4+N_2O under single crop monoculture system (Nc=6) but the effect was not significant under the double crops rotation system (Nc=24). In uplands, when the crop straw was returned, the effect sizes of no-tillage and reduced tillage on CH_4 uptake, N_2O emission, and overall GWP were not significant. But in paddy fields, no-tillage did not affect the overall GWP when crop straw was returned while reduced tillage significantly increased the CH_4 emission and overall GWP when crop straw was returned. The effectiveness of NT/RT on CH_4 and N_2O emissions was influenced by the percentage of basal N fertiliser (PBN), irrigation, and tillage duration.

Table 64. Effect (% change unless otherwise specified) of tillage practices on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

<table>
<tr><th>Technique</th><th>GHG</th><th>System</th><th>Source</th><th>CI_LOW</th><th>Mean</th><th>CI_HIGH</th><th>Nc</th><th>Effect</th></tr>
<tr><td rowspan="7">No tillage compared with conventional tillage</td><td rowspan="5">GWP (CH4+N2O)+CO2</td><td>all</td><td>Huang et al. 2018</td><td>-10.1</td><td>-0.64</td><td>10.1</td><td>64</td><td>Non-significant</td></tr>
<tr><td>wheat</td><td>Huang et al. 2018</td><td>-3.0</td><td>24.9</td><td>59.7</td><td>10</td><td>Non-significant</td></tr>
<tr><td>barley</td><td>Huang et al. 2018</td><td>-2.2</td><td>14.5</td><td>33.3</td><td>31</td><td>Non-significant</td></tr>
<tr><td>maize</td><td>Huang et al. 2018</td><td>-23.5</td><td>-10.2</td><td>5.9</td><td>12</td><td>Non-significant</td></tr>
<tr><td>rice</td><td>Huang et al. 2018</td><td>-39.7</td><td>-24.8</td><td>-6.1</td><td>11</td><td>Positive</td></tr>
<tr><td rowspan="2">GWP (CH4+</td><td rowspan="2">all</td><td rowspan="2">Shang et al. 2021</td><td>-1.1*</td><td>-0.6*</td><td>-0.2*</td><td rowspan="2">55</td><td rowspan="2">Positive</td></tr>
<tr><td colspan="3">*Mg CO2eq/ha/yr</td></tr>
</table>

Technique	GHG	System	Source	CI_LOW	Mean	CI_HIGH	Nc	Effect
		uplands	J. Feng et al. 2018	-19.0	-11.6	-3.6	93	Positive
			Shang et al. 2021	-	-0.015*	-	44	Non-significant
				*Mg CO2eq/ha/yr				
		paddies	J. Feng et al. 2018	-15.1	-5.4	5.0	48	Non-significant
			Shang et al. 2021	-4.8*	-3.2*	-1.5*	11	Positive
				*Mg CO2eq/ha/yr				
	CH4 emissions	Cereals (sigificant share of rice)	Huang et al. 2018	-26.1	-15.5	-3.4	65	Positive
		all (negligible share of rice)	J. Feng et al. 2018	-12.4	-5.9	0.9	196	Non-significant
		wheat	Huang et al. 2018	-2.6	40.2	103.1	10	Non-significant
		maize	Huang et al. 2018	-35.1	-2.0	50.3	7	Non-significant
		rice	Huang et al. 2018	-32.6	-23.0	-11.4	48	Positive
	CH4 uptake	all	Huang et al. 2018	-14	-1	15	56	Non-significant
		wheat	Huang et al. 2018	3.9	31.5	65.8	16	Positive
		barley	Huang et al. 2018	-28.2	-13.6	3.9	29	Non-significant
		maize	Huang et al. 2018	-31.4	-8.5	22.5	11	Non-significant
	N2O emissions	all	Huang et al. 2018	2.5	10.5	18.9	299	Negative
		wheat	Huang et al. 2018	1.0	15.0	31.1	92	Negative
		barley	Huang et al. 2018	-6.1	11.5	32.3	58	Non-significant
		maize	Huang et al. 2018	-6.7	5.0	19.0	110	Non-significant
		rice	Huang et al. 2018	-9.5	11.9	39.1	39	Non-significant
Reduced tillage compared with conventional tillage	GHG (CH4+N2	upland	J. Feng et al. 2018	-28.3	-4.6	22.3	33	Non-significant
		paddies	J. Feng et al. 2018	2.8	13.6	25.8	22	Negative
	CH4 emission	upland	J. Feng et al. 2018	-29.6	-10.2	11.2	33	Non-significant
		paddies	J. Feng et al. 2018	5.0	16.9	30.7	22	Negative
	N2O emission	upland	J. Feng et al. 2018	-8.1	6.1	26.4	33	Non-significant
		paddies	J. Feng et al. 2018	-10.7	3.4	20.1	22	Non-significant

Source: references in the fourth column.

3.21.2 Effects on soil organic carbon

The synthesis paper selected for the extraction of quantitative data on the effects of reduced tillage and no tillage on soil organic carbon is the one from Haddaway et al. 2017. It covers arable soils in boreo-temperate regions and it is based on 351 primary studies, of which 142 from the US, 46 from Canada, 101 from Europe and the rest from other countries. This paper compares tillage management practices classifying them, for analytical purposes, in three intensity classes: no tillage, high intensive tillage and intermediate intensity tillage. Other factors taken into consideration are the depth of tillage, the type of soil, the depth of the soil layer, the latitude and the climatic zone.

The metrics for which effect sizes are reported are SOC concentrations (e.g. g/kg or %) and SOC stock (e.g. Tons C/ha). The difference between the two metrics depends on the bulk density of the soil, so that a similar concentration may not correspond to the same absolute SOC stocks if the density is not the same and/or is not uniform along the soil profile. This makes the assessment not straightforward as bulk density is not always reported in primary studies and, moreover, it may be in turn affected by different tillage intensities. Accordingly, results expressed in SOC stocks tend to be more uncertain than SOC concentration. On the other hand, SOC stock is a more relevant measure when the focus is on quantitative SOC balances to determine the net effect on carbon sequestration and climate change mitigation potential. Here we report results for both SOC concentration (

Table 65) and SOC stocks (

Table 66), for the three comparisons of interventions and for different depths of the studied soil layer.

Table 65 Differences in SOC concentrations (g/kg) for different tillage systems and different soil depths. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Soil depth	CI_LOW	Mean	CI_HIGH	Nc	Effect
No tillage vs. Intensive tillage	0-15 cm	1.41	2.09	2.76	102	Positive
	15-30 cm	-0.86	-0.30	0.26	49	Non-significant
	>30 cm	-0.88	-0.37	0.13	31	Non-significant
No tillage vs. Intermediate tillage	0-15 cm	0.53	1.18	1.83	95	Positive
	15-30 cm	-0.29	0.27	0.84	45	Non-significant
	>30 cm	-0.52	-0.16	0.21	20	Non-significant
Intermediate tillage vs. Intensive tillage	0-15 cm	0.78	1.23	1.66	77	Positive
	15-30 cm	-1.29	-0.89	-0.51	42	Negative
	>30 cm	-0.67	-0.26	0.15	16	Non-significant

Source: Haddaway et al. 2017

For no tillage compared to intensive tillage, a significant mean increase in SOC concentration of 2.09 g/kg (positive effect) was found in the upper layer of the soil (0-15 cm). Results in the lower layers are not significant. Importantly, the positive result for the 0-15 cm soil layer is not affected by factors like latitude or climatic zone, so the result can be considered relevant for the entire Europe (though the type of soil is a relevant factor, see below).

The same applies when no tillage is compared to intermediate tillage: a significant positive effect is found in the upper soil layer with an increase of 1.18 g/kg [0.53 – 1.83], while non-significant effects were detected in at deeper depths. Again, this result holds true regardless of latitude and climatic zones.

When intermediate tillage is compared to intensive tillage, a significant positive result is still found for the upper soil layer (0 – 15 cm) with an average increase of 1.23 g/kg [0.78 –1.66]. Instead, a statistically significant decrease in SOC concentration of -0.89 g/kg [-1.29 – -0.51] was found in the 15-30 cm layer, while at depth >30 cm non-significant effect was detected. Similarly, results do not change with latitude and climatic zones.

Results on SOC stocks were grouped into two classes of soil depth: the upper layer, comprising measurements up to 30 cm depth and the 'full profile' corresponding to measurements taken either in the 0-150 cm profile or in the 30-150 cm profile.

Table 66: Differences in SOC stock (Mg/ha) for different tillage systems and different soil depths. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, NC: number of pairwise comparisons.

Pairwise comparison	Soil layer	CI_LOW	Mean	CI_HIGH	Nc	Effect
No tillage – Intensive tillage	Upper Layer 0-30 cm	0.78	4.61	8.44	29	Positive
No tillage – Intermediate tillage	Full Profile 0-150 cm	-5.04	1.65	8.33	14	Non-significant
Intermediate tillage – Intensive tillage	Upper Layer 0-30 cm	0.63	3.85	7.08	32	Positive
No tillage – Intensive tillage	Full Profile 0-150 cm	-4.48	0.83	6.14	13	Non-significant
No tillage – Intermediate tillage	Upper Layer 0-30 cm	-0.14	1.72	3.59	29	Non-significant
	Full Profile 0-150 cm	-3.07	1.88	6.84	10	Non-significant

Source: Haddaway et al. 2017

The comparison of no tillage vs intensive tillage yields to significantly positive results, when the upper profile is considered, with a mean difference of 4.61 Mg/ha [0.78 - 8.44]. When the full profile is considered, results are more uncertain, and not statistically different from 0 (mean 1.65; C.I. [-5.04 - 8.33]). For the upper layer, results do not change with latitude and climatic zone, whereas where the full profile is considered, latitude was found to be positively correlated with SOC stock, i.e., higher differences were found at northern latitudes.

No tillage shows significant higher SOC stock in the upper profile compared to intermediate tillage too, with a mean difference of 3.85 Mg/ha [0.63 - 7.08]. When the full profile is considered, results are more uncertain, and not statistically different from 0 (mean 0.83; C.I. [-4.48 – 6.14]).

Intermediate tillage compared to intensive tillage has a mean positive but statistically not significant results for both the upper layer and the full profile. Latitude and climatic zones were not significant in both cases.

3.22 Crop residue management

Crop residue management is the handling of stems, leaves, chaff and husks that remain in the fields after crops are harvested for grain, seed or fibber. Main strategies for crop residue management involve residue retention at the surface, residue incorporation into the soil and rice residue burning. This section covers several crop residue management techniques, such as crop residue retention (crop residue or pruning residues are left in the field after harvest), crop residue incorporation into the soil (shallow and deep incorporation), and specific techniques for straw residue management, including additional straw application (beyond residue left by the crop), rice straw burned in paddy rice, and straw returning amended with straw decomposing microorganism inoculants. The impact of straw mulching is addressed as a separate practice in section 3.23 (Mulching). To assess the quantitative effect of the practice on the environment, the used comparator is crop residue removal, no residue return, or crop residue retention without additional straw application.

3.22.1 Effects on GHG emissions

The selected paper is Z. Li et al. 2021. However, given that this paper provides results only for N2O emissions but not for CH4 emissions, we will complement with the results by X. Zhao et al. 2020 for CH4. Results for N2O by X. Zhao et al. 2020 will be also provided for comparison purposes.

According to Z. Li et al. 2021, on average, crop residue retention significantly stimulated N2O emissions by 29.7% (95%CI: 17.7%, 42.7%). In upland soil, N2O emission increased by 46% but it decreased by -18% in paddy soil. Results depended on several factors:

- Climate: Crop residue application significantly stimulated N2O emission by 29.7%, with a significantly higher increase of 35.7% in the temperate zone. In contrast, no significant effect of crop residue application on N2O emission was observed for tropical zones. N2O emission was significantly and positively correlated with latitude, but not with longitude, mean annual temperature and precipitation.
- Soil pH: Compared with the control, crop residue application significantly increased N2O emission by 54.0% when soil pH 5.5–6.5, and by 28.9% for soil pH > 7.5.
- Soil clay content: Crop residue return caused a particularly strong and significant increase in soil N2O emissions except for soil with clay texture, indicating that clay content is an important determinant of the soil N2O emission response to crop residue application.

According to X. Zhao et al. 2020, under crop residues retention compared with residue removed, significant increases in GHGs emissions were observed in China's croplands. Among them, CH4 emission increased by 130.9% 95% CI: 85.2%, 186.1%), and N2O emissions by 12.2% (1.6%, 24.5%).

This effect depended on several factors:

- Crop type: the most sensitive crop species for N2O was maize with a significant increase of 41.0%, while there was a non-significant effect for rice.
- Crop rotation: the increased emission of CH4 was enhanced for regions in which residue retention was adopted with crop rotation. Significant increase of 18.4% in N2O emission was observed in regions where residues retention was adopted with crop rotation ($p < .05$), but it was non-significant without crop rotation.

- Tillage intensity: increases in CH4 and N2O emissions were observed under conventional tillage.
- Mineral N fertiliser amount applied : the highest emissions of CH4 and N2O occurred under higher and medium nitrogen fertiliser input (150-250, and >250 kg N/ha).
- Duration of treatment: over the short term (<5 years), residues retention did not significantly increase N2O emission. Over the medium term (5-10 years), residues retention significantly increased N2O emission. Over the long term (>10 years), residues retention sharply increased CH4 emissions.
- Proportion of residues retained: Compared with 100% in rate of retained residues, reducing the amount of retained parts of residues could lower the increase of CH4 emission. However, more N2O emission were observed under reduced amount of residue retention as compared with 100% amount of residues retained.

Table 67. Effect (% change) of crop residues retention on greenhouse gas emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

GHG	Source	system	CI_LOW	Mean	CI_HIGH	Nc	Effect
N2O	Z. Li et al. 2021	Overall	17.7	29.7	42.7	255	Negative
		Upland	30	46	64	204	Negative
		Paddy	-31	-18	-3.7	51	Positive
		Tropical	-16	2.8	26	67	Non-significant
		Temperate	22	35.7	52	178	Negative
	X. Zhao et al. 2020	China's cropland	1.6	12.2	24.5	117	Negative
		Rice	< -10	-1.6	17.3	NA	Non-significant
		Wheat	0.9	9.4	23.4	NA	Negative
		Maize	24.8	41.0	64.7	NA	Negative
CH4	X. Zhao et al. 2020	China's cropland	85.2	130.9	186.1	92	Negative

Source: references in the second column.

3.22.2 Effects on soil organic carbon

Results are typically provided for soil organic carbon concentration, soil organic carbon stocks and soil carbon change rates. SOC content describes the organic carbon concentration on a weight by weight basis, SOC stocks on a weight by area basis, and C sequestration rates on weight by area and over time.

The selected synthesis paper is Xu et al. 2021. The study includes data from 248 experimental sites reported in 237 primary studies from various regions of the world. The study focuses on different ecosystems, included cropland and grasslands. It explores the impact of crop residue retention and additional straw application on SOC concentration measured at different soil depths, compared to crop residue removal and crop residues retention without additional straw application, respectively (Table 68).

The effect of crop residue management on SOC depended on both the habitat and the portion of soil profile considered. Crop residue retention increased SOC content in grassland by 43% (95% CI: 14%, 79%) compared to crop residue removal, at 0-20 cm soil depth, but not at shallower sections of the soil profile (0-5 and 0-10 cm). In cropland, crop residue retention increased SOC content by 11% (95% CI: 7%, 14%) at 5-20 cm soil depth, by 14% (95% CI: 8%, 20%) at 0-10 cm, by 11% (95% CI: 7%, 15%) at 0-20 cm, and by 11% (95% CI: 6%, 17%) at 20-100 cm.

Additional straw incorporation, at 0-5 cm soil depth, increased SOC content in grassland by 18% (95% CI: 0%, 38%) compared to crop residue retention and no straw addition, but not at 0-20 cm soil depth. In cropland, additional straw application increased SOC content by 23% (95% CI: 13%, 37%) at 20-100 cm, but had non-significant effect at 0-5 cm.

Table 68. Effect (% change) of crop residues management on soil organic carbon (content). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Comparator	Habitat	Soil depth	CI_LOW	Mean	CI_HIGH	Nc	Effect
Crop residue retention	Crop residue removal	Grassland	0-5 cm	-14	-2	12	5	Non-significant
			0-10 cm	-4	11	28	9	Non-significant
			0_20 cm	14	43	79	10	Positive
		Cropland	0-5 cm	-5	10	27	10	Non-significant
			5-20 cm	7	11	14	43	Positive
			0-10 cm	8	14	20	35	Positive
			0-20 cm	7	11	15	57	Positive
			20-100 cm	6	11	17	44	Positive
Additional straw application	Crop residues retention without additional straw application	Grassland	0-5 cm	0	18	38	5	Positive
			0-20 cm	-3	25	60	9	Non-significant
		Cropland	0-20 cm	-14	40	127	5	Non-significant
			20-100 cm	13	23	37	4	Positive

Source: Xu et al. 2021.

3.22.3 Effects on the nutrient balance

Results in the reviewed literature quantify the change in nitrogen use efficiency and nutrient uptake (Annex 1). The selected synthesis papers are the one by L. Xia et al. 2018 for the effect of crop residue retention (straw return with mineral fertilisation) and the one by X. Qin et al. 2021 for crop residue incorporation (straw incorporation). Crop residue management had an overall positive effect on Nitrogen use efficiency: crop residue retention was found to increase nitrogen use efficiency by 14.9% (95% CI: 11.3%, 19.5%) in rice paddies as well as in upland crops globally. While crop residue incorporation was found to increase nitrogen use efficiency by 10% (95% CI: 8%, 12%) for maize in China.

Table 69. Effect (% change) of crop residue management on nutrient use efficiency. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Source	Intervention	Comparator	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
L. Xia et al. 2018	Crop residue retention (Straw return with mineral fertilisation)	Crop residue removal (No straw return with mineral fertilisation)	Overall (rice paddies and upland crops globally)	11.3	14.9	19.5	100	Positive
			Warm temperate climate	5.2	9.0	13.0	33	Positive
			Subtropical climate	13.1	18.2	24.4	66	Positive
X. Qin et al. 2021	Crop residue incorpora-tion (Straw incorpora-tion)	Crop residue removal (No straw return)	Overall (maize in China)	8%	10%	12%	48	Positive
			N application rate > 250 kg ha-1	2%	6%	11%	89	Positive
			N application rate 150 - 250 kg ha-1	7%	11%	14%	20	Positive
			N application rate < 150 kg ha-1	8%	15%	23%	27	Positive
			straw return amount > 9000 kg ha-1	3%	10%	16%	73	Positive
			straw return amount 6000 - 9000 kg ha-1	3%	7%	10%	32	Positive
			straw return amount < 6000 kg ha-1	11%	17%	23%	48	Positive

Source: references in the first column.

Effect of straw return on N use efficiency was higher in subtropical climate (18.2%; 95% CI: 13.1%, 24.4%) than in warm temperate climate (9.0%; 95% CI: 5.2%, 13.0%). Crop residue incorporation increased the most nitrogen use efficiency at low N application rates (< 150 kg ha^{-1}; 15%; 95% CI: 8%, 23%) and at the lowest straw returning amount (< 6000 kg ha^{-1}; 17%; 95% CI: 11%, 23%).

3.22.4 Effects on nutrient leaching and run-off

Results in the reviewed literature quantify the change in leaching of nitrates (NO3-) or nitrogen, and the variation in nitrogen run-off.

One selected synthesis paper, by Z. Li et al. 2021, includes data from 90 primary studies and explores the impact of crop residue return on NO3 leaching in different type of soils and in combination with different type and quantity of fertilisers, compared to no crop residue return (Table 70).

Table 70. Effect (% change) of crop residue return on NO3 leaching. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention			CI_LOW	Mean	CI_HIGH	Nc	Effect
Crop residue return Vs. no residue return	overall		-21	-13	-3	90	Positive
	Soil pH	pH < 5.5	-7	2	12	19	Non-significant
		pH 5.5 - 6.5	-12	3	26	16	Non-significant
		pH 6.5 - 7.5	-45	-32	-12	23	Positive
		pH > 7.5	-2	11	28	24	Non-significant
	Soil type	Silty clay loam soils	-41	-27	-16	13	Positive
		Loamy sand soils	-34	-18	6	33	Non-significant
		Sandy loam soils	-35	-27	-17	4	Positive
		Silty loam soils	-41	-33	-20	10	Positive
		Sandy soils	-5	3	12	12	Non-significant
	Fertiliser compo-sition	Residues and N application	-30	-20	-7	54	Positive
		Residues and NPK application	3	22	54	15	Negative
	Fertiliser form	Residues and Urea application	-26	-14	1	47	Non-significant
		Residues and NH4NO3 application	-32	-21	-7	18	Positive
	Fertiliser applications	Residues and 1 fertiliser application	-51	-44	-39	4	Positive
		Residues and 2 fertiliser application	-36	-19	6	22	Non-significant
		Residues and 3 fertiliser application	-18	-5	9	18	Non-significant
	Residue type	Low C/N residues	-19	8	60	4	Non-significant
		Sawdust	-28	12	239	4	Non-significant
		Barley straw	-17	-5	8	14	Non-significant
		Maize straw	-5	16	52	9	Non-significant
		Rye straw	-34	-17	6	4	Non-significant
		Wheat straw	-30	-21	-9	54	Positive

Source: Z. Li et al. 2021.

Mean response ratio showed that crop residue return decreased NO3 leaching by 13% (95% CI: 3%, 21%) compared to no residue return. Nevertheless, the effect of this farming practice depended on soil pH, type of soil, type and quantity of fertiliser applications, and type of residue.

Crop residue return decreased NO3 leaching only for pH between 6.5 and 7.5 (-32%; 95% CI: -45%, -12%), but it had non-significant effects for other pH values.

Crop residue return decreased NO3 leaching in silty clay loam, sandy loam and silty loam soils by -27% (95% CI: -41%, -16%), -27% (95% CI: -35%, -17%), -33% (95% CI: -41%, -20%), respectively. But not in loamy sand and sandy soils.

Nitrogen fertiliser composition significantly affected the effect size of crop residue application on NO_3^- leaching. Application of NPK fertiliser increased the NO_3^- leaching by 22% (95% CI: 3 - 54), compared to control, whereas it was significantly decreased by 20% (95% CI: 7 - 30) with application of single N fertiliser. Among the different forms of synthetic N fertilisers, NH_4NO_3 significantly decreased NO_3^- leaching by 21% (95% CI: 7 - 32). In contrast, the effect of urea did not significantly change the effect of crop residue application on NO_3^- leaching. The study also shows that when the fertiliser was applied only once during the growing season, NO_3^- leaching was significantly reduced by 44.1% (95% CI: 39 -51).

Only the application of wheat straw residues significantly reduced soil NO_3^- leaching (-21%; 95% CI: -30%, -9%), while the effect of return of other residues with different C:N ratios on NO_3^- leaching was not significant.

3.23 Mulching

Mulching consists on spreading various covering materials on the surface of soil mainly to minimize moisture losses and weed population and to enhance crop yield. The most common mulching materials used in commercial agricultural systems are plastic film and straw. This practice includes several techniques, classified in two main groups: 1) flat planting with mulching; 2) ridge and furrow planting with mulching.

Flat planting with mulching includes: plastic film mulching (black and transparent film mulching), straw mulching (straw or other stubbles), gravel mulching, mulching (several mulching materials aggregated) and biodegradable mulching (rice bran, molecular film, vegetative film, starch-polyester mulch, paper-based mulch, and other degradable films).

Ridge and furrow planting with mulching includes: plastic-covered ridge coupled with furrow planting zone with bare soil, plastic-covered ridge coupled with furrow planting zone with mulching (plastic or straw), and not covered ridge coupled with furrow planting zone with mulching (plastic or straw).

The impact of mulching done with plants (also called living or biological mulching) is addressed as a separate practice in section 3.19 (Cover and catch crops). To assess the quantitative effect of the practice on the environment, the used comparator is no mulching or plastic film mulching.

3.23.1 Effects on GHG emissions

The selected paper is Yu et al. 2021. From this paper, results for CH_4 emissions are available only for paddy fields while for CH_4 uptake only for non-paddy fields or uplands.

In paddy fields, plastic film mulching significantly reduced CH4 emissions by 103.9 kg ha− 1 year− 1 and by -64.2 % (95% CI: -73.4%, -71,7%) compared to those from non-mulched farmlands. This result was mainly due to film mulching greatly increasing the activity of methane-oxidizing bacteria by maintaining aerobic conditions under non-flooded water conditions. It is worth noting that film thickness significantly affected the response of CH4 emissions to mulching, and the use of thick film (>0.01 mm) helped to reduce CH4 emissions from paddy fields. The use of thin film (≤0.01 mm) and thick film (>0.01 mm) significantly ($p < 0.05$) decreased CH4 emissions by 60.9% and 87.4%, respectively.

Table 71. Effect (% change) of mulching on CH4 emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention /comparator	Factors		CI_LOW	Mean	CI_HIGH	Nc	Effect
Plastic and biodegradable film vs. no mulching	Overall	Paddy fields only	-73.4	-64.2	-51.7	42	Positive
	Crop type	Paddy fields	-73.4	-64.2	-51.7	42	Positive
		Rainfed uplands	NA	NA	NA	NA	Not available
		Irrigated uplands	Na	NA	NA	NA	Not available
	Film thickness (mm)	<=0.01	-70.8	-60.9	-47.1	32	Positive
		>0.01	-87.3	-87.4	-97.3	4	Positive
Plastic film vs. no mulching	Film type	Transparent PE	-73.0	-64.0	-50.1	33	Positive
Biodegradable film vs. no mulching	Film type	Biodegradable	-76.3	-66.6	-52.5	6	Positive

Source: Yu et al. 2021.

In rainfed uplands, the soil CH4 uptake was significantly lower in mulched uplands than in non-mulched uplands. However, the negative effect of film mulching on CH4 uptake in irrigated uplands was not significant.

In uplands, the soil CH4 uptake was significantly ($p < 0.05$) reduced by -16.1% (95% CI: -27.8%, -2.9%) following the application of plastic film. CH4 uptake was significantly ($p < 0.05$) reduced by -25.5% in mulched rainfed up- lands compared to non-mulched fields. Moreover, fields covered with a low (<50%) coverage ratio or black PE film exhibited consistently lower CH4 uptake. The response of CH4 uptake to mulching differed significantly ($p < 0.05$) with film thickness. Covering the fields with thick films (>0.01 mm) significantly reduced CH4 uptake by -53.7%. Moreover, authors found no significant relationship between the RR of CH4 emissions/uptake to film mulching and other variables.

Table 72. Effect (% change) of mulching on CH_4 uptake. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention /comparator	Factors		CI_LOW	Mean	CI_HIGH	Nc	Effect
Plastic film vs. no mulching	Overall	Upland only	-27.8	-16.1	-2.9	70	Negative
	Crop type	Paddy fields	NA	Na	NA	NA	Not available
		Rainfed uplands	-32.2	-25.5	-17.9	33	Negative
		Irrigated uplands	-29.3	-6.2	23.9	37	Non-significant
	Coverage ratio	<50%	-32.2	-20.1	-6.2	13	Negative
		50-100%	-28.5	-13.1	5.9	53	Non-significant
	Film type	Transparent PE	-27.8	-13.5	3.4	58	Non-significant
		Black PE	-43.6	-30.7	-15.3	11	Negative
	Mulching method	Ridge-furrow mulching	-27.4	-15.0	1.2	53	Non-significant
		Flat mulching	-50.5	10.0	143.0	4	Non-significant
	Film thickness (mm)	<=0.01	-30.4	-10.2	14.7	42	Non-significant
		>0.01	-60.1	-53.7	-46.5	2	Negative

Source: Yu et al. 2021.

The application of film significantly ($p < 0.05$) increased N_2O emissions by 23.9% compared to those from non-mulched fields, in paddy fields, rainfed uplands and irrigated uplands analysed together. However, the agricultural system, coverage ratio, film type and mulching method significantly ($p < 0.05$) influenced the response of N_2O emissions to mulching. Film mulching significantly ($p < 0.05$) increased N_2O emissions by 89.3% in paddy fields but had no significant effects on rainfed or irrigated uplands. The increase in N_2O emissions was consistent in fields with high coverage ratios (50–100%), transparent and black PE films, and flat mulching. In addition, compared with non-mulching, mulching with thin film (≤0.01 mm) significantly increased N_2O emissions by 29.7%.

Table 73. Effect (% change) of mulching on N_2O emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention /comparator	Factors		CI_LOW	Mean	CI_HIGH	Nc	Effect
Plastic and biodegradable film vs. no mulching	Overall	Paddy fields, rainfed uplands, irrigated uplands	11.3	23.9	38.2	136	Negative
	Crop type	Paddy fields	42.8	89.3	152	39	Negative
		Rainfed uplands	-1.0	6.7	17.4	57	Non-significant
		Irrigated uplands	-10.3	2.0	17.8	38	Non-significant

	Coverage ratio	<50% coverage	-13.3	-3.0	9.7	13	Non-significant
		50-100% coverage	5.1	21.7	40.9	82	Negative
	Mulching method	Ridge-furrow mulching	-8.0	1.7	13.6	52	Non-significant
		Flat mulching	25.5	34.7	45.1	5	Negative
	Film thickness (mm)	<=0.01	11.3	29.7	52.4	82	Negative
		>0.01	-1.4	24.0	57.4	17	Non-significant
Plastic and biodegradable film vs. no mulching	Film type	Transparent PE	7.4	22.8	41.3	89	Negative
		Black PE	11.7	22.4	34.3	24	Negative
Biodegradable film vs. no mulching	Biodegradable		-29.1	-13.7	5.5	9	Non-significant

Source: Yu et al. 2021.

According to the results on this synthesis paper, overall, the effect of film mulching on the total GHG fluxes (N_2O+CH_4 GWP) differed among agricultural systems. In rainfed uplands, film mulching slightly increased GHG fluxes by 95.5 kg CO_2-eq ha−1 year−1. In irrigated uplands, plastic film mulching slightly reduced GHG fluxes by -37.3 kg CO_2-eq ha−1 year−1. In paddy fields, the application of plastic film greatly reduced the GWP of CH_4 emissions by 3532.6 kg CO_2-eq ha−1 year−1 and increased N_2O emissions by 1.42 kg ha−1 year−1 with a results on GHG fluxes of –3109.6 kg CO_2-eq ha−1 year−1.

3.23.2 Effects on ammonia emissions

The selected paper for quantitative data extraction is Abdo et al. 2021. Authors found that mulching has a greater effect on reducing ammonia on maize crops (-29%; 95% CI: -32%,-25%) than on wheat (-26%; 95% CI: -30%, -23%) and rice (-20%; 95% CI: -23%, -18%). The efficiency of mulching on reducing ammonia volatilisation increased when increasing the N mineral fertilisation rate.

Table 74. *Effect (% change) of mulching on ammonia emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.*

Intervention	Crop	Factor		CI_LOW	Mean	CI_HIGH	Nc	Effect
Mulching	Maize	Urea N fertiliser level (kg N/ha)	180-220	-32.4	-28.6	-24.8	NA	Positive
	Wheat		90-120	-5.2	-4.9	-4.6	NA	Positive
			180-220	-29.8	-26.3	-22.6	NA	Positive
	Rice		180-220	-22.8	-20.4	-18.2	NA	Positive
			240-280	-16.4	-14.5	-12.7	NA	Positive

Source: Abdo et al. 2021.

3.23.3 Effects on the nutrient balance

Results in the reviewed literature quantify the change in nitrogen use efficiency and nutrient uptake (Annex 1). The selected synthesis paper is the one by W. Qin et al. 2015, that includes data from 74 primary studies conducted in 19 countries and explores the impact of straw and plastic mulching on nitrogen use efficiency for different type of crops and in combination with different level of water and nitrogen inputs (Table 75).

Table 75. Effect (% change) of mulching on nutrient use efficiency. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Subpractice	Factors		Crop	CI_LOW	Mean	CI_HIGH	Nc	Effect
Straw mulching	overall		Wheat	11.94	17.91	24.88	289	Positive
Plastic mulching	overall		Wheat	12.44	22.39	32.84	72	Positive
Straw mulching	overall		Maize	10.45	21.89	35.32	328	Positive
Plastic mulching	overall		Maize	40.30	58.71	79.10	143	Positive
Straw mulching	water input	low water input < 250 mm	Wheat	11.84	21.22	31.84	NA	Positive
		high water input > 250 mm	Wheat	8.57	15.31	22.65	NA	Positive
		low water input < 370 mm	Maize	4.59	24.49	48.06	NA	Positive
		high water input > 370 mm	Maize	11.94	20.20	29.39	NA	Positive
Plastic mulching	water input	low water input < 250 mm	Wheat	2.45	14.90	28.37	NA	Positive
		high water input > 250 mm	Wheat	18.16	30.20	43.47	NA	Positive
		low water input < 370 mm	Maize	35.20	60.00	92.76	NA	Positive
		high water input > 370 mm	Maize	26.02	40.00	53.88	NA	Positive
Straw mulching	N input	low N input < 120 kg/ha	Wheat	17.32	25.15	33.40	NA	Positive
		high N input > 120 kg/ha	Wheat	8.87	16.08	23.71	NA	Positive
		low N input < 120 kg/ha	Maize	14.05	22.78	33.04	NA	Positive
		high N input > 120 kg/ha	Maize	-12.53	32.28	98.35	NA	Non-significant
Plastic mulching	N input	low N input < 120 kg/ha	Wheat	5.15	15.88	27.42	NA	Positive
		high N input > 120 kg/ha	Wheat	9.07	20.82	33.61	NA	Positive
		low N input < 120 kg/ha	Maize	22.03	35.00	56.96	NA	Positive
		high N input > 120 kg/ha	Maize	34.94	78.00	126.08	NA	Positive

Source: W. Qin et al. 2015.

Mulching had an overall positive effect on nitrogen use efficiency: straw mulching was found to increase nitrogen use efficiency by 17.9% (95% CI: 11.9%, 24.9%) in wheat and by 21.9% (95% CI: 10.5%, 35.3%) in maize; plastic mulching was found to increase nitrogen use efficiency by 22.4% (95% CI: 12.4%, 32.8%) in wheat and by 58.7% (95% CI: 40.3%, 79.1%) in maize.

The mean effects of straw mulching on nitrogen use efficiency of wheat were slightly higher at low water input (+21.2%; 95% CI: 11.8%, 31.8%) than at high water input (+15.3%; 95% CI: 8.6%, 22.6%). On the contrary, mean effects of plastic mulching on NUE of wheat were lower at low water input (14.9%; 95%CI: 2.4%, 28.4%) than at high water input (+30.2; 95% CI: 18.2%, 43.5%). For maize, the mean effect of straw mulching on NUE was independent of water input level, whereas that of plastic mulching was higher at low water input level (+60.0; 95% CI: 35.2%, 92.8%) than at high water input level (+40.0%; 95% CI: 26.0%, 53.9%).

The effects of mulching on nitrogen use efficiency were significantly positive at all N input levels. However, straw mulching did not significantly affect NUE in maize at high N input level.

3.23.4 Effects on nutrient leaching and run-off

Results in the literature quantify the change in soil total nitrogen loss, ammonium nitrogen loss, total phosphorus loss and dissolved phosphorus loss. The selected synthesis paper is the one by R. Liu et al. 2021, that includes data from 85 primary studies and explores the impact of mulching (several materials including plastic film, cereals straw and pruning residues) on nutrient loss in tree crops, compared to no mulching (clean tillage management) (Table 76). Mulching had an overall positive effect on nutrient loss, reducing soil total nitrogen loss by 31.1% (95% CI: 13.9%, 49.0%), soil nitrate nitrogen loss by 86.0% (95% CI: 64.4%, 108.8%), ammonium nitrogen loss reduction by 83.9%, total phosphorus loss by 35.6% (95% CI: 18.9%, 52.5%), and dissolved phosphorus loss by 53.9% (95% CI: 29.6%, 78.3%).

Table 76. Effect (% change) of mulching on nutrient loss. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Soil total nitrogen loss reduction	13.94	31.08	49.00	19	Positive
Soil nitrate nitrogen loss reduction	64.40	86.00	108.80	5	Positive
Ammonium nitrogen loss reduction	NA	83.92	NA	1	Positive
Total phosphorus loss reduction	18.95	35.57	52.48	15	Positive
Dissolved phosphorus loss reduction	29.57	53.91	78.26	8	Positive

Source: R. Liu et al. 2021.

3.24 Grassland conservation and restoration

Grasslands are areas of land predominantly covered by communities of grass-like plants and forbs and may include sparsely occurring trees and shrubs. In the Eurostat classification, percentages of area covered by such canopies are limited to less than 10 % in the case of trees and to less than 20 % in the case of shrubs (i.e., low woody plants capable of reaching heights of up to 5 metres) or shrubs and trees together[53]. However, in the scientific literature reviewed here, grasslands are usually more broadly defined and might not fully meet such canopy limits.

The literature reviewed includes a wide variety of grassland types: savannah, grassy deserts, seasonally flooded grasslands, prairies, meadows, pastures, rangelands, salt-marshes, bioenergy perennial grasslands, calcareous grasslands and wooded grasslands. Fodder crops are not included as grasslands. In addition, for this exercise, grasslands may differ in terms of: management intensity (natural, semi-natural, improved and intensively managed), management history (permanent and temporary), successional stage (old successional and secondary grasslands), biomes and climates (semi-arid, temperate, tropical, Mediterranean, tundra, alpine, subalpine, artic and subarctic).

Grassland conservation refers to the preservation (i.e., no transformation to other land uses) of old successional natural grasslands. Old successional natural grasslands are grasses and forbs communities with no major human-induced structural or functional alterations so that their current management closely resembles historical, endogenous disturbance regimes (e.g., grazing intensity and fire frequency) (Nerlekar & Veldman 2020). In the scientific literature, these ecosystems are also referred as old-growth, ancient, remnant or native grasslands.

Grassland restoration includes active and passive restoration methods usually aiming at enhancing plant community diversity, what is then expected to have cascading positive effects in higher trophic levels. This review includes several restoration methods: sowing of seed mixtures, addition of soil and hay, reintroduction of grazing and burning as active restoration methods, and abandonment of agricultural use as passive restoration method. In this review, we only consider the restoration of former grassland areas degraded by agricultural use (i.e., degradation due to mining or other land uses are excluded). We define secondary grasslands as the herbaceous communities that assemble after destruction (here, by agricultural use) of old successional natural grasslands where the restoration efforts are conducted (Nerlekar & Veldman 2020).

3.24.1 Effects on greenhouse gas emissions

Evidence of impacts of grassland restoration on GHG was found from only one synthesis paper (J. Wu et al. 2020). They report that CH_4 uptake rate was significantly increased when the cropland was converted to pasture, with Hedges' D effect size value of 1.38. However, the metric used does not allow to show the percentage nor the absolute impact.

[53] https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Glossary:Grassland

Table 77. Effect (Hedge's D) of grassland conservation and restoration practices on greenhouse gas emissions by major GHG types. Nc: number of pairwise comparisons.

Technique	GHG	Hedges' D effect size value	Nc	Effect
Conversion to pasture vs. cropland	CH_4 uptake	1.38	6	Positive

Source: J. Wu et al. 2020.

3.24.2 Effects on soil organic carbon

The only synthesis paper that explores the effect of grassland conservation on soil organic carbon content includes European data and found that, in soils of "ex-arable grasslands"[54] SOC stocks were on average lower than in "ancient grassland"[55] soils (Kämpf et al. 2016b). On average, "SOC stocks of ex-arable land did not reach the level of never ploughed ancient soils (...) but were 11% lower". However, the effect was different with time of abandonment. In soils of ex-arable grasslands (average abandonment time: 14 years) SOC stocks were on average only 7.4 t ha^{-1} lower than in ancient grassland soils. Old ex-arable land stored 5.2 t ha^{-1} more SOC than young ex-arable land (mean difference in abandonment time: 13 years).

The effects of grassland restoration (land use changes from crop to pasture or perennial grasses) on soil carbon stocks are extracted from Harris et al. 2015 and Kämpf et al. 2016b.

The data reported by Kämpf et al. 2016b regard the effects of the conversion to temperate grasslands as compared to arable land. Ex-arable grassland as compared to arable land significantly increased SOC by 17%, as average [95% CI: 12%, 22.7%]. However, these results depend on different factors: (i) abandonment time; (ii) Climate; (iii) initial SOC stock. In the first 16 years, SOC in ex-arable land increased evenly but subsequently stagnated at high level (+31%). The proportional gain of carbon was positively correlated with the aridity index of the study site: it increased from 12.5% in humid to 31.7% in semi-arid climate. Absolute values, however, did not differ significantly. The effect of the initial SOC stock on SOC change was negative for both proportional change as well as absolute values. SOC stocks were significantly higher in ex-arable land compared to arable land with initial SOC stocks of 0–25 and 25–50 t ha^{-1}. In contrast, land-use abandonment had non-significant effect on SOC when the initial SOC stock was already 50 t ha^{-1}.

The data reported by Harris et al. 2015 regard the effects of the conversion to perennial bioenergy grassland as compared to Cropland and Perennial Grasses. They report that conversion to perennial bioenergy grasslands as compared to cropland significantly increased SOC by 25.7% ± 6.7% (average time since transition of 5.4 years, with 2 and 16 years as minimum and maximum, respectively). However, when comparing with grassland, SOC decreased by -10.9 ± 4.3% (average time since transition 5.8 years, minimum and maximum 3 and 6), which we also report for comparison.

[54] Ex-arable grasslands is defined as abandonment of formerly ploughed fields.

[55] Ancient grassland was defined as land that is never ploughed.

Table 78 Effect (% change) of grassland conservation on soil organic carbon (stock per hectare). CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Source	Population	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
Grassland conservation								
Ex-arable grassland as compared to ancient grassland	Kämpf et al. 2016b	Temperate grasslands	--	-2.1	-11.4	-20.7	36	Positive
Grassland restoration								
Ex-arable[54] grassland vs. arable land	Kämpf et al. 2016b	Temperate grasslands	-	12.0	17.1	22.7	69	Positive
Perennial bioenergy grasslands vs. cropland	Harris et al. 2015	Bioenergy perennial grasslands		17.9	25.7	33.9	63	Positive
Perennial bioenergy grasslands vs. grassland	Harris et al. 2015	Bioenergy perennial grasslands	-	-14.5	-10.9	-7.3	43	Negative
Ex-arable grassland vs. arable land	Kämpf et al. 2016b	Temperate grasslands	Abandonment time					
			> 16 years	18.1	31.0	44.1	11	Positive
			> 8-16 years	20.0	26.8	34.6	15	Positive
			> 4-8 years	10.9	15.9	21.4	16	Positive
			0-4 years	-0.5	7.2	15.2	10	Non-significant
Ex-arable grassland vs. arable land	Kämpf et al. 2016b	Temperate grasslands	Global aridity index					
			humid	5.0	12.4	20.4	19	Positive
			moderately humid	11.0	19.4	27.9	18	Positive
			sub-humid	18.0	31.1	43.4	6	Positive
			semi-arid	17.1	31.6	46.4	14	Positive
Ex-arable grassland vs. arable land	Kämpf et al. 2016b	Temperate grasslands	Initial SOC stock					
			> 75 t/ha	-1.1	3.7	9.0	21	Non-significant
			> 50-75 t/ha	-4.0	0.2	4.5	6	Non-significant
			> 25-50 t/ha	10.5	20.1	29.9	20	Positive
			0-25 t/ha	23.5	32.9	42.4	22	Positive

Source: references in the second column.

3.24.3 Effect on nutrient leaching and runoff

One synthesis paper (Y. Zhang et al. 2021), explores the effect of grassland restoration on the soil N/I ratio. Nitrate (NO_3^-) is readily lost via leaching or transformation into gaseous nitrogen during nitrification and denitrification. The soil ratio of nitrification to immobilization (N/I) reflects the potential risk of nitrogen loss from soils and is therefore positively correlated with amounts of nutrient loss by leaching. Soils with a high N/I ratio have greater potential of nitrogen loss through leaching, runoff, or denitrification than those with a low N/I ratio. Thus, the N/I ratio has been used to evaluate the risk of nitrogen loss. The authors reported that converting cropland to grassland decreased the soil N/I ratio by 45% (Nc=5).

Table 79 Effects (% change) grassland restoration practices on Soil nitrification to immobilization ratio. SE: Standard Error of the mean, Nc: number of pairwise comparisons..

Metric	Population	Mean-SE	Mean	Mean+ SE	Nc	Effect
Soil nitrification to immobilization ratio	Grasslands (vs. cropland)	-57.3	-45	-23.3	5	Positive

Source: Y. Zhang et al. 2021

3.25 Grassland management

Following the previous definition of grasslands as included in this literature review, the interventions here considered as grassland management with relevant effects on the climate and nutrients are:

- Use of soil organic amendments: Soil organic amendments are materials of plant or animal origin that can be added to soil, such as manures, biosolids, green wastes, and composts, to improve the soil quality in terms of its structure and biochemical function. This includes also biochar, which is charcoal produced by pyrolysis of biomass in the absence of oxygen (as defined in the corresponding section 3.8 of this document).
- Use of Enhanced-efficiency fertilisers (EEF) : Enhanced-efficiency fertilisers (EEF) are different types of fertilisers or products associated to fertilisers, which have been developed to better synchronize fertiliser nitrogen (N) release with crop uptake, offering the potential for enhanced N-use efficiency (NUE) in crops and reduce losses (as defined previously in 3.5). This section report results that are specific for the use of EEF on grassland, including the use of different types of EEF, such as nitrification inhibitors, urease inhibitors and polymer-coated fertilisers.
- Use of fertilisers
- Increasing grass/forb species richness

3.25.1 Effects on greenhouse gas emissions

The effects of grassland management practices on greenhouse gases are diverse across specific techniques and gases.

Use of soil organic amendments:

One synthesis paper explores the effect of the application of biochar to grasslands soils on GHG emissions, showing that soil amendments with biochar have non-significant effect in reducing N2O emissions as compared to not amending (Cai & Akiyama 2017).

Table 80. Effect (% change) of soil organic amendments as compared to unmanaged grassland on soil GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Metric	CI_HIGH	Mean	CI_LOW	Nc	Effect
Biochar as compared to No amendments	Soil GHG emissions: soil N2O emission	+19	-15	-39	3	Non-significant

Source: Cai & Akiyama 2017.

Use of Enhanced-efficiency fertilisers (EEF):

Regarding the effect of the application of different inhibitors associated to fertilisers on N2O emissions in the framework of grassland management, overall the addition of these products resulted in a reduction of the emissions of this green-house gas. The selected paper for numerical data extraction in this impact is Cai & Akiyama 2017. This paper explores the effects of grassland fertilisation using various nitrification inhibitors such as: Dicyandiamide, N-[n-butyl] thiophosphoric triamide or Pyrazole derivates. Different forms of these additives as well as combined applications of them are explored. Results are provided for soil N2O emissions and are valid for grasslands.

Table 81. Effect (% change) of the use of Enhanced-efficiency fertilisers as compared to unmanaged grassland on soil GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Metric	CI_HIGH	Mean	CI_LOW	Nc	Effect
Dicyandiamide in liquid form as compared to No inhibitor	Soil GHG emissions: soil N2O emission	-47	-52	-56	105	Positive
Dicyandiamide coated with zeolite as compared to No inhibitor	Soil GHG emissions: soil N2O emission	-33	-39	-44	6	Positive
N-[n-butyl] thiophosphoric triamide in liquid form as compared to No inhibitor	Soil GHG emissions: soil N2O emission	+4	-23	-53	4	Non-significant
N-[n-butyl] thiophosphoric triamide and Dicyandiamide in liquid form as compared to No inhibitor	Soil GHG emissions: soil N2O emission	-41	-47	-53	19	Positive
N-[n-butyl] thiophosphoric triamide and	Soil GHG emissions:	-46	-52	-56	4	Positive

Pairwise comparison	Metric	CI_HIGH	Mean	CI_LOW	Nc	Effect
Dicyandiamide coated with zeolite as compared to No inhibitor	soil N2O emission					
Pyrazole derivatives in liquid form [PD] as compared to No inhibitor	Soil GHG emissions: soil N2O emission	+34	8	-11	10	Non-significant

Source: Cai & Akiyama 2017.

The selected article also explores the effect of using Dicyandiamide as inhibitor taking into consideration different modifying factors as the type of excreta used for fertilisation or the deposition season. The numerical results considering these factors are shown in the table below.

Table 82. Effect (% change) of use of enhanced-efficiency fertilisers as compared to unmanaged grassland on soil GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Metric	Factor	CI_HIGH	Mean	CI_LOW	Nc	Effect
Dicyandiamide as compared to No inhibitor	Soil GHG emissions: soil N2O emission	Excreta type					
		dairy cattle urine	-47	-52	-56	91	Positive
		artificial urine	-44	-53	-60	17	Positive
		beef cattle urine	+261	63	-13	2	Non-significant
Dicyandiamide as compared to No inhibitor	Soil GHG emissions: soil N2O emission	Deposition time					
		Summer	-15	-25.9	-35.3	5	Positive
		Autumn	-48.3	-53.9	-59.1	58	Positive
		Winter	-1.1	-35.7	-52.8	11	Positive
		Spring	-42.7	-50.1	-57.8	27	Positive

Source: Cai & Akiyama 2017.

Use of fertilisers:

The effects of grassland fertilisation on GHG emissions are overall negative according to the retrieved synthesis papers examining CH4 uptake by soil and N2O emissions when Nitrogen, Phosphorous or the two combined are applied (Schievano et al. 2023).

Among the available meta-analyses reporting negative results on GHG emissions, we selected these two for quantitative data extraction: L. Zhang et al. 2020 and J. Wang et al. 2018.

Results are provided for soil GHG emissions measured as soil CH4 uptake and N2O emissions (Table 83). The data reported by L. Zhang et al. 2020 regard the effects of grassland fertilisation on CH4 uptake as compared to no fertilisation, covering different types of fertiliser combination, such as phosphorous (P) fertilisation alone, nitrogen (N) fertilisation alone or N and P fertilisation combined (N+P). The data reported by J. Wang et al. 2018 cover the effect of N fertilisation on N2O emissions as compared with no fertilisation.

Table 83. Effect (% change) of the use of fertilisers as compared to unmanaged grassland on GHG emissions for the specified metrics. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Reference	Pairwise comparison	Metric	CI_HIGH	Mean	CI_LOW	Nc	Effect
L. Zhang et al. 2020	P fertilisation as compared to No fertilisation	Soil GHG emissions: Soil CH4 uptake	+16	10	+4	7	Positive
L. Zhang et al. 2020	N+P fertilisation as compared to No fertilisation	Soil GHG emissions: Soil CH4 uptake	-8	-14	-20	7	Negative
L. Zhang et al. 2020	N fertilisation as compared to No fertilisation	Soil GHG emissions: Soil CH4 uptake	-21	-27	-33	7	Negative
J. Wang et al. 2018	N fertilisation as compared to No fertilisation	Soil GHG emissions: soil N2O emission	+349	271	+207	82	Negative

Source: references in the first column.

3.25.2 Effects on soil organic carbon

Use of soil organic amendments:

One revised meta-analysis found that organic amendment addition as compared to no amendment to rangeland has a positive effect on soil organic carbon (Gravuer et al. 2019). The dataset includes experiments and field measurements performed taking into account some particular factors, such as application rate (50 Mg/ha and 10 Mg/ha) and N content in amendment (3.6 % on dry weight basis and 1.8 % on dry weight basis). Results vary according to these particular factors.

Soil organic amendments significantly increased SOC. The overall increase was 33.4% (95% CI: 21.4%, 45%) for rangelands in arid, semiarid, and Mediterranean climates, based on 244 data points from primary studies.

Table 84 provides the main numerical results regarding the effect of this management practice on soil organic carbon SOC expressed in percentage of change as compared to the control. Population information such as geographical location, pedo-climatic conditions and other factors are provided.

Table 84. Effect (% change) of organic amendments as compared to unmanaged grassland on Carbon sequestration for the metric SOC. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Population	CI_HIGH	Mean	CI_LOW	Nc	Effect
Soil organic amendment as compared to No amendment	Rangelands in arid, semiarid, and Mediterranean climates.	45	33.4	21.4	244	Positive
	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (1.8 %) and Application rate (10 Mg/ha).	3	-20.1	9.9	7	Non-significant

Pairwise comparison	Population	CI_HIGH	Mean	CI_LOW	Nc	Effect
	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (1.8 %) and Application rate (50 Mg/ha).	38.9	9.4	23.2	7	Positive
	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (3.6 %) and Application rate (50 Mg/ha).	45.2	16.1	30.8	7	Positive

Source: Gravuer et al. 2019

3.25.3 Effects on nutrient leaching and runoff

Use of soil organic amendments:

Amendment addition to rangeland soils increases N and P runoff (Gravuer et al. 2019). The data reported by Gravuer et al. 2019 regard the effects of soil amendments as compared to no amendment. The dataset includes experiments and field measurements performed taking into account some particular factors, such as application rate (50 Mg/ha and 10 Mg/ha) and N content in amendment (3.6 % on dry weight basis and 1.8 % on dry weight basis). Results are consistently negative across all these particular factors.

Table 85. Effect (% change) of amendment addition as compared to unmanaged grassland on nutrients runoff. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Metric	Population	CI_HIGH	Mean	CI_LOW	Nc	Effect
Nutrients runoff: Nitrate runoff	Rangelands in arid, semiarid, and Mediterranean climates.	+307.1	140.3	+64.3	57	Negative
Nutrients runoff: Nitrate runoff	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (1.8 %) and Application rate (10 Mg/ha).	+134.6	58.2	+25.2	7	Negative
Nutrients runoff: Nitrate runoff	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (1.8 %) and Application rate (50 Mg/ha).	+292.1	133.4	+62.6	7	Negative
Nutrients runoff: Nitrate runoff	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (3.6 %) and Application rate (50 Mg/ha).	+492.6	227	+104.6	7	Negative

Metric	Population	CI_HIGH	Mean	CI_LOW	Nc	Effect
Nutrients runoff: Phosphorous runoff	Rangelands in arid, semiarid, and Mediterranean climates.	+423.6	223.5	+117.4	72	Negative
Nutrients runoff: Phosphorous runoff	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (1.8 %) and Application rate (10 Mg/ha).	+276.8	100	+36.1	7	Negative
Nutrients runoff: Phosphorous runoff	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (1.8 %) and Application rate (50 Mg/ha).	+681.4	274.3	+111.4	7	Negative
Nutrients runoff: Phosphorous runoff	Rangelands in arid, semiarid, and Mediterranean climates: N content in amendment (3.6 %) and Application rate (50 Mg/ha).	+1234.8	488.2	+194.8	7	Negative

Source: Gravuer et al. 2019

3.26 Grazing

For the purpose of this exercise, grazing is defined as a method of animal husbandry in which livestock are allowed to feed outdoors consuming wild vegetation. The grazing livestock species included in the synthesis papers reviewed are mainly cows, but goats, sheep and domesticated horses and yaks are also included.

Results from the literature review explore:

- grazing as compared to no grazing. No grazing refers to areas with grazing exclusion or where only wild herbivores graze in natural conditions.
- different grazing intensities Grazing intensities are classified, as reported by authors, in light (or extensive), moderate and heavy (or intensive). They are compared to no grazing or lower grazing intensity as control. It has been assumed that the authors of the original individual papers report grazing intensity based on their knowledge of the systems studied, meaning that, for instance, the grazing intensity considered light in one study system can be high in another. Therefore, grazing intensities across individual studies and synthesis papers cannot be compared. The review of the impacts of grazing intensities includes spatial and temporal comparisons.
- and rotational grazing. Rotational grazing comprises grazing regimes that incorporate periods of planned rest (McDonald et al. 2019). In the scientific literature it is also refereed as strategic-rest grazing. For the exploration of the impacts of rotational grazing, nearby areas with no grazing or continuous grazing are used as control.

3.26.1 Effects on greenhouse gas emissions

Only one synthesis paper (Tang et al. 2019) with global coverage was retrieved on the effects of different grazing regimes on GHG emissions. They report the effects of light, moderate and intensive/heavy grazing compared to no grazing. Only heavy grazing was found to have a positive effect on N2O emissions and a negative effect on CH4 uptake by soil (decrease in the rate of uptake).

- Light/extensive grazing as compared to No grazing had non-significant effect on Soil CH4 uptake (6.5; 95% CI: -12.9%, 30.3%).
- Moderate grazing as compared to No grazing had non-significant effect on soil CH4 uptake (3.9; 95% CI: -12.2%, 23.4%).
- Heavy/intensive grazing as compared to No grazing significantly decreased soil CH4 uptake (-30.3; 95% CI: -40.4%, -19.6%).
- Light/extensive grazing as compared to No grazing had non-significant effect on soil N2O emissions (-23.3; 95% CI: -60.7%, 50.9%).
- Moderate grazing as compared to No grazing had non-significant effect on soil N2O emissions (-22.9; 95% CI: -57.4%, 40%).
- Heavy/intensive grazing as compared to No grazing significantly decreased soil N2O emissions (-39.1; 95% CI: -62.7%, -3.3%).

The main findings regarding the effects of Grazing as compared to No grazing on GHG emissions are:

- Grazing with a duration < 5 years as compared to No grazing had non-significant effect on soil CH4 uptake (-3.9%; 95% CI: 22.3%, 38.8%).
- Grazing with a duration of 5 - 10 years as compared to No grazing significantly decreased soil CH4 uptake (-23.6%; 95% CI: -40.9%, -1.3%).
- A grazing duration of 10 years or more, as compared to No grazing, significantly decreased soil CH4 uptake (-35.1%; 95% CI: -56.5%,-3.3%).
- Grazing (< 5 years) as compared to No grazing had non-significant effect on soil N2O emission (-18.6%; 95% CI: -54.8%, 49.1%).
- Grazing for 5 - 10 years, as compared to No grazing, significantly decreased soil N2O emission (-42.8; 95% CI: -54.8%, -27.6%.)
- Grazing for 10 years or more, as compared to no grazing, significantly decreased soil N2O emission (-70.9; 95% CI: -85.5%, -42.3%.)

Findings also indicate that such results are significant in dry grassland (precipitation < 400 mm/year), while in areas with precipitation above 400 mm/year the effects on GHG fluxes are not significant. According to the authors, overall, “grazing results in inhibition of GHG emissions but at the cost of plant productivity and soil fertility”. Soil CH4 uptake and N2O emission were reduced much more under heavier grazing, longer grazing duration or less precipitation.

Table 86. Effect (% change) of grazing management practices on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	GHG	Factor	CI_LOW	Mean	CI_HIGH	Nc	Effect
Low intensity grazing vs. no grazing	CH4 uptake	-	-12.9	6.5	+30.3	15	Non-significant
	N2O	-	-60.7	-23.3	+50.9	8	Non-significant
Medium intensity grazing vs. no grazing	CH4 uptake	-	-12.2	3.9	+23.4	20	Non-significant
	N2O	-	-57.4	-22.9	+40	10	Non-significant
High intensity grazing vs. no grazing	CH4 uptake	-	-40.4	-30.3	-19.6	29	Negative
	N2O	-	-62.7	-39.1	-3.3	16	Positive
Grazing vs. no grazing	CH4 uptake	< 5 years	-22.3	3.9	+38.8	8	Non-significant
		5-10 years	-40.9	-23.6	-1.3	10	Negative
		≥ 10 years	-56.5	-35.1	-3.3	5	Negative
	N2O	< 5 years	-54.8	-18.6	+49.1	4	Non-significant
		5-10 years	-54.8	-42.8	-27.6	14	Positive
		≥ 10 years	-85.5	-70.9	-42.3	5	Positive
Grazing vs. no grazing	CH4 uptake	< 400 mm	-26.0	-15.4	-5.4	51	Negative
		≥ 400 mm	-33.3	-14.0	11.8	13	Non-significant
	N2O	< 400 mm	-56.8	-41.5	-22.1	27	Positive
		≥ 400 mm	-38.6	-7.9	39.3	13	Non-significant

Source: Tang et al. 2019.

3.26.2 Effects on soil organic carbon

Light grazing compared to no grazing showed non-significant effect on SOC.

Moderate grazing compared to no grazing showed a negative effect on SOC, but the effect depended on the soil layer. In the selected paper for data extraction (Lai & Kumar 2020), the authors report non-significant effect on SOC in the upper soil layer (0-10 cm) and a negative effect in the 0-30 cm.

Heavy grazing compared to non-grazed or less intensively grazed areas has a negative effect on soil organic carbon. In the selected paper for data extraction (Lai & Kumar 2020), the authors found that heavy/intensive grazing as compared to no grazing significantly decreased SOC by -10.8% (95% CI: -17.7%, -3.8%) in the top soil layer (0-10 cm depth) and by -22.5% (95% CI: -33.9%, -10.2%) in the below layer (10-30 cm soil depth). Given that they give values classified by soil depth, we also provide the results from other synthesis papers for comparison purposes. Tang et al. 2019 report a decrease in SOC of -10.2% (95% CI: -17.6%, -3%), in line with the top soil layer value of Lai & Kumar 2020. Byrnes et al. 2018 report an average decrease of -13.9% (95% CI: -20.5%, -6.8%). Therefore, in case of further information on the soil layers, the value of the top soil layer seems appropriate.

Grazing in general (no intensity specified) showed a negative effect by most synthesis papers. In the selected synthesis paper (M. He et al. 2020), the authors report an average decrease of SOC of -4.1% (95% CI: -4.2%, -4.0%). Results depended on several factors like the mean annual temperature, precipitation and altitude.

Rotational grazing has a positive effect on SOC (+28.4%; 95% CI:+10.5%, 50.7%) when compared to continuous grazing and non-significant effect when compared to no grazing, according to the only available synthesis paper (Byrnes et al. 2018).

Table 87. Effect (% change) of grazing management practices on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Source	Soil depth	CI_LOW	Mean	CI_HIGH	Nc	Effect
Light grazing vs. no grazing	Several	-	-	-	-	-	Non-significant
Moderate grazing vs. no grazing	Lai & Kumar 2020	0-10 cm	-5.2	1.9	9.2	NA	Non-significant
		10-30 cm	-28.7	-16.4	-3	NA	Negative
Heavy grazing	Lai & Kumar 2020	0-10 cm	-17.7	-10.8	-3.8	NA	Negative
		10-30 cm	-33.9	-22.5	-10.2	NA	Negative
	Tang et al. 2019	-	-17.6	-10.2	-3	18	Negative
	Byrnes et al. 2018	-	-20.5	-13.9	-6.8	59	Negative
Grazing vs. no grazing	M. He et al. 2020	-	-4.2	-4.1	-4.0	380	Negative
Rotational grazing vs. continuous grazing	Byrnes et al. 2018	-	10.5	28.4	50.7	44	Positive
Rotational grazing vs. no grazing	Byrnes et al. 2018	-	-5.8	7.3	20.9	15	Non-significant

Source: references in the second column.

3.27 Peatland management

A peatland is a type of wetland with a naturally accumulated layer of peat at the surface due to organic matter production exceeding decomposition. Peatlands are generally classified as bogs and fens. Bogs are fed mainly by rain and snow, while fens develop in landscape depressions and are fed with surface and/or ground water. Other names referring to peatlands found in this review are swamp, and wet heath, though the search string included more synonyms. Carbon-rich soils are included in this definition. These soils have and organic layer usually with >15 % of organic matter. However, the synthesis papers included in this review do not clearly define what they consider carbon-rich soils and thus we rely on authors' knowledge and criteria to include individual studies in their reviews. This review focuses on peatlands under agricultural use and thus, includes mostly drained peatlands.

This section includes interventions towards an improved and more sustainable management of peatlands. A variety of interventions have been proposed such as low tillage, ban of burning, paludiculture, etc. In this literature review we have found evidence for the following management interventions:

- No fertilisation. Fertilisation, and in particular N inputs in wetlands, might promote greenhouse gas emissions by changing element cycling. Here, no fertilised peatlands are compared to fertilised ones.
- Grazing: the direction and magnitude of grazing impacts on ecosystem properties depends on grazing management such as stocking density, grazing duration and grazer species. Here, grazed peatlands (comprising different management intensities and levels of drainage) are compared to mowed peatlands.
- No burning. Burning is a routinely vegetation management practice conducted in some in some areas with the objective of stimulating the growth of grasses, maintaining these areas in young productive successional stages and increasing heterogeneity at the landscape scale and thus, supporting biodiversity. However, peatland burning releases carbon to the atmosphere and banning this practice has been suggested as a measure to reduce GHG emissions.
- Low intensity farming comprises a wide variety of farming practices analysed together that refer mainly to the maintenance of permanent and extensive grasslands and fallowing in peatlands. They include extensive livestock grazing, mowing and the lack of fertiliser inputs.

3.27.1 Effects on greenhouse gas emissions

The impacts of peatland management on GHG emissions found in this review differ depending on the management technique and on the GHG.

N fertilisation, compared to no fertilisation, has non-significant effect on CH4 emissions but increases N2O emissions (Table 88). Nitrogen input significantly promoted N2O emission in peatlands compared to other wetlands. The external nitrogen enhanced microbial activity and triggered a priming effect that further facilitated the release of available nitrogen. Thus, nitrogen input has a greater effect on N2O emission from peatlands than other wetlands (M. Chen et al. 2020). Hedge's D or Standardized Mean Difference is 2.44 (95% CI: 1.86, 3.09).

Low intensity farming (here defined as the maintenance of permanent and extensive grasslands and fallowing) has non-significant effect on CH4 and N2O emissions compared to high intensity farmed peatlands (Haddaway et al. 2014).

Grazing has non-significant effect on CH4 emissions in peatlands compared to mowing (Haddaway et al. 2014).

Table 88. Effect (% change and Hedge's G) of peatland management practices on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Source	GHG	Units	CI_LOW	Mean	CI_HIGH	Nc	Effect
N fertilisation vs. no fertilisation	M. Chen et al. 2020	CH_4	Hedge's G effect size	-1.6	0.6	4.3	11	Non-significant
		N_2O	Hedge's G effect size	1.9	2.4	3.1	67	Negative
Low intensity farming	Haddaway et al. 2014	CH_4	mg CH_4 m^{-2} h^{-1}	-0.011	-0.004	0.004	4	Non-significant
		N_2O	mg $N_2O m^{-2}$ h^{-1}	-0.05	0.14	0.34	5	Non-significant
Grazing vs. mowing	Haddaway et al. 2014	CH_4	mg CH_4 m^{-2} h^{-1}	-14.0	-5.39	3.23	3	Non-significant
		N_2O	NA	NA	NA	NA	NA	NA

Source: references in the second column.

3.28 Peatland conservation

Conservation refers to the preservation (i.e., no transformation to other land uses) of natural non-disturbed peatlands compared to either degraded or restored peatlands. The degraded peatlands found in this review are mainly affected by alterations in their water regime due to drainage and/or conversion to cropland or pasture. The restored peatlands found in this review have been subject to different interventions including the re-establishment of peatland vegetation, water supplementation or passive restoration by abandonment of agricultural use. The estimation of peatland conservation impacts is based on the spatial comparison between preserved natural peatlands (intervention) and nearby degraded or restored peatlands (comparator).

3.28.1 Effects on greenhouse gas emissions

The conservation of natural peatlands compared to degraded peatlands has been reported to have different effects on GHG depending on the gas and the cause of degradation.

When the cause of degradation is conversion to cropland and pasture, the selected synthesis paper, (Tan et al. 2020) reported not significant results for CH_4 and N_2O emissions. For net ecosystem CO_2 exchange, Tan et al. 2020 report non-significant effect when compared to cropland but a positive effect (CO_2 sink) when compared to pastures (+130%; 95% CI: 8%, 392%; Nc=12). Given that Tan et al. 2020 report % values for degraded peatlands as compared to values of natural peatland (conservation of natural peatland), we have recalculated those values in order to express them as % values for conserved natural peatland as compared to degraded peatland. This conversion has been done using the formula ($-\alpha/(1+\alpha)$, being α the percentage change divided by 100). For example, a 100% increase is converted to -50% decrease ($-1.00/(1+1.00)=-0.5$).

When compared to drained peatland, Tan et al. 2020 reported non-significant effect on CH_4 emissions but the synthesis paper selected for data extraction on this intervention (Haddaway et al. 2014) reported negative effects in boreo-temperate lowland peatlands. According to Haddaway et al. 2014, drained peatlands may release 0.126 mg CH_4 less than undrained peatlands per square metre per hour; equating to an approximate annual additional release of 11.0 kg CH_4 ha-1 (95% CI = 22.3, 0.263) and a 100-year global warming potential difference of 276 kg CO_2 equivalents ha-1, although this result is only marginally significant (p=0.055). Similarly, when degradation is due to drainage, conservation of natural peatlands has a positive effect (i.e., decrease) on N_2O emissions, according to Haddaway et al. 2014, and non-significant effect according to another synthesis paper (Tan et al. 2020). Drained peatlands release, on average, 0.008 mg N_2Om-2 h-1 (95% CI=0.001, 0.016) relative to undrained peatlands (p=0.033). This corresponds to an approximate annual difference of 0.701 kg N_2O ha-1 (95% CI=0.088, 1.4) and a 100-year global warming potential difference of 209 kg CO_2 equivalents ha-1 (Haddaway et al. 2014).

Haddaway et al. 2014 found a non-significant result in net ecosystem exchange CO_2 for drained versus undrained peatlands across the only three studies analysed (all of which were from Canada). Regarding ecosystem respiration CO_2, drained peatlands released, on average, 125 mg CO_2 m-2 h-1 (95% CI=6.71, 243) more CO_2 through respiration than undrained peatlands, approximating to an annual difference of 11,000 kg CO_2 ha-1 (95% CI=0.588, 21.3, Nc=10) resulting from drainage (p=0.038). Studies varied in their findings.

Table 89. Estimated GHG changes of conservation of natural peatlands as compared to different land use changes. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	GHG	Source	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Peatland conservation vs. cropland [cropland vs. natural peatland]	CH_4	Tan et al. 2020	%	-12*	56*	178*	43	Non-significant
	N_2O	Tan et al. 2020	%	-53*	-31*	1*	32	Non-significant
	Net ecosystem exchange (CO_2)[56]	Tan et al. 2020	%	-12*	51*	159*	14	Non-significant
Peatland conservation vs. pasture [pasture vs. natural peatland]	CH_4	Tan et al. 2020	%	-66*	-3*	170*	19	Non-significant
	N_2O	Tan et al. 2020	%	-81*	-50*	35*	12	Non-significant
	Net ecosystem exchange (CO_2)[57]	Tan et al. 2020	%	8*	130*	392*	12	Positive
Peatland conservation vs. drained	CH_4	Haddaway et al. 2014	mg CH_4 m^{-2} h^{-1}	-0.003	0.126	0.254	9 studies	Negative
			kg CH_4 ha^{-1} yr^{-1}	0.26	11	22.3		

[56] CO_2 Net ecosystem exchange can be either positive or negative. The reported changes refer to the % change of absolute values.

[57] CO_2 Net ecosystem exchange % change refers to absolute values, in this case, of the CO_2 sink effect.

Pairwise comparison	GHG	Source	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
[undrained vs drained]			kg CO2 eq. ha^{-1} (100yr)	6.5	276	557.5		
	N2O	Tan et al. 2020	%	-71*	-31*	67*	15	Non-significant
		Haddaway et al. 2014	mg N2O m^{-2} h^{-1}	-0.016	-0.008	-0.001	5 studies	Positive
			kg N2O ha^{-1} yr^{-1}	-1.4	-0.7	-0.088		
			kg CO2 eq. ha^{-1} (100yr)	-417	-209	-26		
	Ecosystem respiration CO2	Haddaway et al. 2014	mg CO2 m^{-2} h^{-1}	-243	-125	-6.71	10 studies	Positive
			kg CO2 ha^{-1} yr^{-1}	-588	-10,950	-21,287		
	Net ecosystem exchange CO2	Haddaway et al. 2014	mg CO2 m^{-2} h^{-1}	-431	-116	200	3 studies	Non-significant

Source: references in the third column. (*) results marked with an asterisk have been converted as explained in the text.

3.28.2 Effects on soil organic carbon

The conservation of natural peatlands, compared to degraded peatlands converted to croplands, has a positive effect on SOC. Xu et al. 2019a found significant negative effects on SOC for the cultivation of wetlands with peat soil, as compared to the conservation of natural wetlands with peat soil (-42.4%, $p < .001$). This effect was not significant for other types of wetlands. We have converted the numbers to calculate the % change that implied the conservation of natural wetland as a % of the SOC of the degraded cultivated peatland, according to what explained in section 3.28.1 (e.g. from -42.4% to 73.5% using the formula - 0.424/(1+0.424)=0.736).

Table 90. Effect (% change) of peatland conservation on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Source	Population	CI_LOW	Mean	CI_HIGH	Nc	Effect
Peatland conservation vs. cultivated peatlands	Xu et al. 2019a	Wetlands with peat soil	37*	73.5*	120.4*	57	Positive
Peatland conservation vs. restored peatlands	Xu et al. 2019b	Wetlands with peat conditions in Northern hemisphere	-13*	15*	52*	19	Non-significant

Source: references in the second column. (*) results marked with an asterisk have been converted as explained in the text.

Compared to restored peatlands, the conservation of natural peatlands has non-significant effect on soil organic carbon, according to 1 synthesis paper (Xu et al. 2019b). This synthesis paper indicates that restoration of wetlands with peat conditions, as compared with natural wetlands with peat conditions (that we have identified with peatlands conservation) have no significant lower SOC (-13%; 95% CI: -34, 14.5%). In the same way as in the previous section, percentage changes have been recalculated to express peatland conservation as a % of cultivated or restored peatlands.

3.29 Peatland restoration

Restoration methods have been classified in:

Management of water regime: Restoration by managing water regime refers to re-allocating a part of the natural water regime back to these soils/ecosystems, resulting in the full or partial recovery of water inputs and inundation by removing draining, ditching, precipitation exclusion, flooding, dams or groundwater extraction.

Several restoration methods pooled together: When different restoration methods are analysed together in the reviewed synthesis papers. This pool includes active revegetation, full or partial recovery of water inputs and passive restoration by cessation of human disturbance (i.e., agricultural abandonment and prohibited grazing).

The estimation of peatland restoration impacts is based on the spatial and temporal comparisons between restored peatlands (intervention) and degraded peatlands (comparator). Spatial comparisons are those conducted simultaneously between restored peatlands and nearby degraded peatlands. Temporal comparisons are those conducted in the same peatland before and after restoration.

3.29.1 Effects on greenhouse gas emissions

Peatland restoration by managing water regime has different effects depending on the GHG. The selected paper for CH4 (Huang et al. 2021) analyses the difference of higher and lower water tables. As the reference scenario is not indicated to be a peatland in natural status and the comparison is done in space (different sites) under the same time period, we assume that the effect of increasing the water table is just the same with opposite sign. According to this synthesis paper, the estimated mean value of changes in CH4 emissions is a significant increase of 26 mgCO2-eq m^{-2} h^{-1} (95% CI: 35, 20).

We also report for comparison the results by Haddaway et al. 2014, that focus only on boreo-temperate lowland peatland systems, while Huang et al. 2021 use a higher number of studies at the global scale, differentiating arctic, boreo-temperate, and tropical climatic zones. The increases in CH4 emissions found by Haddaway et al. 2014 are considerably lower (i.e. 6 mgCO2-eq m−2 h−1) as compared to the 26 found in the previous paper (Huang et al. 2021). This is probably due to the difference in regions and climates considered. The selected paper (Haddaway et al. 2014) includes results from 3 studies from Finland with 2 intervention and control observations each, and 1 study from Canada with 11 observations. The results from Canada are low and determine the low value of the overall result, while the results of 2 of the 3 Finnish papers are much higher (around 20 and 37 mgCO2-eq m−2 h−1). Regarding the possible climatic influence, Huang et al. 2021 found that the average sensitivity of CH4 emissions change to unit water-table change was smaller (less change) in tropical than boreal and temperate peatlands, while results for artic zones depended on the methodology for coefficient estimation. Another factor that can affect the results is the height of the water table: Huang et al. 2021 also quantified the sensitivities of GHG fluxes to the magnitude of water-table increase and found that the overall average sensitivity to a 1 cm water-table increase was 2.9 (95% CI: 3.6, 2.2) mgCO2-eq m−2 h−1 for CH4.

Regarding CO2 emissions, the restoration of drained peatlands by managing the water regime has been reported as positive for ecosystem respiration and net ecosystem exchange of CO2 (=ecosystem respiration –vegetation CO2 uptake). According to Huang et al. 2021, the water-table increase would induce a decrease in CO2 emissions from respiration exceeding the decrease of vegetation CO2 uptake. The estimated mean value of net ecosystem exchange of CO2 is -62 mgCO2 m−2 h−1 (955CI: -47% to -77%), meaning a net decrease of CO2 emissions (or an increased sink) for a water table becoming higher. The decreases in terms of global warming potential (in CO2 equivalents), offsets the increase in CH4. Therefore, the measure would be overall positive for reducing GHG emissions. Individual values have a large variability, varying from +497 to -1,234 mgCO2 m−2 h−1 across sites. The paper also quantified the sensitivities of GHG fluxes to the magnitude of water-table increase and found that the overall average sensitivity to a 1 cm water-table increase was -4.1 (95% CI: -3.3, -5.0) mgCO2 m−2 h−1 for CO2 net ecosystem exchange. The average sensitivity for total GHG (net ecosystem exchange CO2 + net methane fluxes) was -1.6 (95% CI: -0.8, -2.3) mgCO2-eq m−2 h−1cm−1 based on a subset of experiments that measured both net ecosystem exchange and CH4.

Haddaway et al. 2014 analysed several pairwise comparisons for restoration: drained and restored vs. undrained, restored vs. unrestored, and restored vs. natural. However, authors only found sufficient data allowing for meaningful meta-analysis for the comparison restored vss. unrestored and for 3 outcome metrics: CH4 emissions, CO2 ecosystem respiration emission, and dissolved organic carbon concentration. Ecosystem respiration, results significantly higher (125 mgCO2 m^{-2} h^{-1}) in undrained compared to drained peatlands in the two selected synthesis papers. However, for net ecosystem exchange CO2 release, Haddaway et al. 2014 reported a non-statistically significant effect.

Table 91. Effects of peatland restoration on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	GHG	Population	Source	Units	CI_LOW	Mean	CI_HIGH	Nc	Effect
High vs. low water table	CH4	Peatlands (mesocosmos and field experiments)	Huang et al. 2021	kgCO2-eq ha^{-1} yr^{-1}	1754	2277.6	3066	532	Negative
				mgCO2-eq[58] m^{-2} h^{-1}	20	26	35		
Restored vs unrestored	CH4	Boreo-temperate lowland peatlands	Haddaway et al. 2014	mg CH4 m-2 h-1 (CO2-eq)	0.052 (1.3)	0.248 (6.2)	0.446 (11.15)	17	Negative
Undrained vs drained	CH4	Boreo-temperate lowland peatlands	Haddaway et al. 2014	mg CH4 m-2 h-1 (CO2-eq)	-0.003 (-0.075)	0.126 (3.15)	0.254 (6.35)	49	Non-significant

[58] mg CO2-eq=mg CH4 x 25.

Pairwise comparison	GHG	Population	Source	Units	CI_LOW	Mean	CI_HIGH	Nc	Effect
High vs. low water table	Net ecosystem exchange CO_2	Peatlands (mesocosmos and field experiments)	Huang et al. 2021	mgCO_2 m^{-2} h^{-1}	-77	-62	-47	376	Positive
High vs. low water table	Ecosystem respiration CO_2	Peatlands (mesocosmos and field experiments)	Huang et al. 2021	mgCO_2 m^{-2} h^{-1}	-102	-125	-147.5	407	Positive
Restored vs. unrestored	Ecosystem respiration CO_2	Boreo-temperate lowland peatlands	Haddaway et al. 2014	mgCO_2 m^{-2} h^{-1}	-155	48.7	253	3 studies	Non-significant
Undrained vs drained	Net ecosystem exchange CO_2	Boreo-temperate lowland peatlands	Haddaway et al. 2014	mgCO_2 m^{-2} h^{-1}	-431	-116	200	3 studies	Non-significant
Undrained vs drained	Ecosystem respiration	Boreo-temperate lowland peatlands	Haddaway et al. 2014	mgCO_2 m^{-2} h^{-1}	-6.7	-125	-243	10 studies	Positive

Source: references in the third column.

3.29.2 Effects on soil organic carbon

Peatland restoration (with several restoration methods pooled together in the analyses) has a positive effect on soil organic carbon, compared to peatlands converted to cropland, according to one synthesis paper reviewed (Xu et al. 2019b).

The paper looked at the effects of wetlands in general on SOC. Because many studies did not report whether a wetland was a peatland, they classified a wetland as peatland if the following conditions were met: the study clearly indicated that the wetland was a peatland, the soil had a high SOC content (>300 g/kg) and a low bulk density (<0.3 g/cm3), or the soil type was Histosol. All the studies were conducted in the northern hemisphere with a latitude range from 1.6° to 59.2°, a mean annual precipitation range from 383 mm to 2570 mm, and a mean annual temperature range from −6.7 °C to 26.7 °C. The results suggest that the hydrological conditions of cultivated wetlands, soil depth, vegetation type, peat condition and restored age were important influential factors affecting SOC after wetland restoration, with the first 2 being the most important influential factors.

The SOC in restored wetlands was significantly higher (+34%; 95% CI: 6%, 68%) than in cultivated wetlands identified as peatlands. Their results also suggest that the SOC lost in peatlands was easier to recover than that in wetlands with no peat.

Table 92. Effect (% change) of peatland restoration on soil organic carbon. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Population	CI_LOW	Mean	CI_HIGH	Nc	Effect
Peatland restoration vs. peatland converted to cropland	Wetlands with peat conditions in Northern hemisphere	6	34	68	16	Positive

Source: Xu et al. 2019b

Note: The comparison of the effect of SOC of peatland restoration with natural peatland (not significant) is not included here as it is included under peatland conservation practice.

3.30 Wetland management

Wetlands are areas where water is the primary factor controlling the environment and the associated plant and animal life. The Ramsar Convention on Wetlands adopts a broad definition of wetlands. It includes lakes and rivers, underground aquifers, swamps and marshes, wet grasslands, peatlands, oases, estuaries, deltas and tidal flats, mangroves and other coastal areas, coral reefs, and all human-made sites such as fish ponds, rice paddies, reservoirs and salt pans. However, this review does not include all these wetland types either because they are not well represented in the scientific literature, because they are not relevant in a European context or because they are not under agricultural use. Besides, this review excludes peatlands and carbon rich soils, as their conservation, restoration and management are assessed in separate sections. This review focuses on wetlands under agricultural use and thus, includes mostly drained wetlands.

The wetland management practice includes interventions towards an improved and more sustainable management of wetlands. Specific interventions have been proposed such as low tillage, ban of burning, paludiculture, etc. In the literature review we have found evidence for the following management interventions:

- No fertilisation. Fertilisation, and in particular N inputs in wetlands, might promote greenhouse gas emissions by changing element cycling. Here, no fertilised wetlands are compared to fertilised ones.
- Grazing. The direction and magnitude of grazing impacts on ecosystem properties depends on grazing management, such as stocking density, grazing duration and grazer species. Here, grazed wetlands are compared to wetlands with grazing exclusion or where only wild herbivores graze in natural conditions.

3.30.1 Effects on greenhouse gas emissions

Results by L. Liu & Greaver 2009 indicate that "N addition, ranging from 30 to 400 kg N /ha /year, significantly increased CH4 emission by an average of 95% when averaged across grassland, wetland and anaerobic agricultural systems. This response ratio did not differ among the three ecosystem types. Data were not sufficient to estimate an emission factor for non-agricultural ecosystems" (e.g. wetland). Overall, CH4 uptake was reduced, but not statistically significant for grasslands and drained wetlands.

According to the authors (L. Liu & Greaver 2009) "Nitrogen addition, ranging from 10 to 562 kg N /ha/yr, significantly increased N_2O emission by an average of 216% across all ecosystems", and for wetlands by 207% (95% CI: 78%, 431%).

93. Effect (% of change) of wetland fertilisation on greenhouse gas emissions by major GHG types. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

GHG	Wetland type	CI_LOW	Mean	CI_HIGH	Nc	Effect
CH4 emissions	Wetlands only	NA	NA	NA	NA	Not available
	Wetlands (undrained) (Nc=11) + anaerobic agricultural systems (Nc=13) + grassland (Nc=2)	39.9	97.2	178.2	26	Negative
CH4 uptake	Drained wetlands	-63	-37	9	6	Non-significant
N2O	Wetlands	78	207	431	19	Negative

Source: L. Liu & Greaver 2009.

3.31 Wetland conservation

Conservation refers to the preservation (i.e., no transformation to other land uses) of natural non-disturbed wetlands compared to either degraded or restored wetlands.

- The degraded wetlands found in this review are mainly affected by alterations in their water regime due to drainage and/or conversion to cropland or pasture, or by their conversion into aquaculture ponds or other constructed wetlands.
- The restored wetlands found in this review have been subject to different interventions including water flow re-establishment, revegetation or addition of a sediment layer.

The estimation of wetland conservation impacts is based on the spatial comparison between preserved natural wetlands (intervention) and nearby degraded or restored wetlands (comparator).

3.31.1 Effects on greenhouse gas emissions

Only one synthesis paper (Tan et al. 2020) was found on the impacts of the conservation of wetlands on (soil) GHG emissions. It compares natural coastal wetlands and riparian wetlands (wetlands on river banks) with different types of degraded wetlands: (1) constructed artificial wetland; (2) wetland converted to cropland; (3) drained wetland; (4) wetland converted to pasture; (5) wetland converted to aquaculture pond. Given that Tan et al. 2020 report values for degraded peatlands expressed as % of values of natural peatland (conservation of natural peatland), we have recalculated those values in order to express them as % values for conserved natural peatland as compared to degraded peatland. This conversion has been done using the formula ($-\alpha/(1+\alpha)$, being α the percentage change divided by 100). For example, a 100% increase is converted to -50% decrease ($-1.00/(1+1.00)=-0.5$).

The conservation of natural wetlands has different effects on GHG emissions, depending on the cause of degradation.

CH_4 emission: A negative effect of the conservation of natural riparian wetlands compared to wetland degraded due to drainage is observed (CH_4 emissions increase by 317%; 95% CI: 144%, 614%). Compared to wetland converted to cropland, we can assume a weak negative effect overall, with non-significant effect for coastal wetlands and a negative effect for natural riparian wetlands (245%; 95% CI: 117%, 400%). Lower CH_4 emissions are observed only for the conservation of natural coastal wetland as compared to the conversion to aquaculture pond. Non-significant effect was observed when degradation is due to conversion to pasture of riparian wetlands, or due to constructed artificial wetlands as compared to conserved coastal wetlands.

N_2O emission: Tan et al. 2020 report a weak positive effect of the conservation of wetlands as compared to their conversion to cropland (i.e. lower N_2O emissions, by -78%; 95% CI: -19%, -94% for natural riparian wetlands, but non-significant effect for natural coastal wetlands). A negative effect was observed when natural riparian wetlands are converted to pasture (N_2O decreases with the conversion by -80%; 95% CI:-4%, -96%, so that conservation would increase N_2O by 400%; 95% CI: 4%, 2400%); non-significant effect was observed for the rest of the pairwise comparisons.

Net ecosystem CO_2 exchange: the conservation of natural wetlands has positive effects (i.e. higher CO_2 sequestration capacity) for natural riparian wetlands as compared to drained wetland (that leads to an increase of net CO_2 fluxes by 395%; 95% CI: 225%, 645%), and wetlands converted to pasture (398%; 95% CI: 95%, 1154%). As well, the conservation of natural coastal wetlands as compared to constructed wetlands has a positive effect (70%; 95% CI: 8%, 163%). Non-significant effects were observed for any type of natural wetland as compared to croplands.

Table 94. Effect (% of change) of conservation of natural wetlands as compared to different land use changes on GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Comparison	GHG	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect of conservation
Natural coastal wetland vs. cropland [cropland vs. natural coastal wetland]	CH_4 flux	% (g C $m^{-2}year^{-2}$)	-40*	122*	733*	27	Non-significant
	N_2O flux	% (g N $m^{-2}year^{-2}$)	-92*	-58*	1133*	22	Non-significant
	Net ecosystem CO_2 exchange	% (g C $m^{-2}year^{-2}$)	-78*	65*	1128*	4	Non-significant
Natural riparian wetland vs. cropland [cropland vs. natural riparian wetland]	CH_4 flux	% (g C $m^{-2}year^{-2}$)	117*	245*	400*	24	Negative
	N_2O flux	% (g N $m^{-2}year^{-2}$)	-19*	-78*	-94*	23	Positive
	Net ecosystem CO_2 exchange	% (g C $m^{-2}year^{-2}$)	-47*	2*	100*	11	Non-significant
Natural riparian wetland vs. pastures [pastures vs. natural riparian wetland]	CH_4 flux	% (g C $m^{-2}year^{-2}$)	-71*	-25*	285*	15	Non-significant
	N_2O flux	% (g N $m^{-2}year^{-2}$)	4*	400*	2400*	11	Negative
	Net ecosystem	% (g C $m^{-2}year^{-2}$)	95*	398*	1154*	4	Positive

Comparison	GHG	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect of conservation
	CO_2 exchange[59]						
Natural riparian wetland vs. drained wetland [Drained wetland vs. natural riparian wetland]	CH_4 flux	% (g C $m^{-2}year^{-2}$)	144*	317*	614*	10	Negative
	N_2O flux	% (g N $m^{-2}year^{-2}$)	-63*	-29*	37*	16	Non-significant
	Net ecosystem CO_2 exchange	% (g C $m^{-2}year^{-2}$)	225*	395*	645*	6	Positive
Natural coastal wetland vs. constructed wetlands [Constructed wetlands vs. natural coastal wetland]	CH_4 flux	% (g C $m^{-2}year^{-2}$)	-72*	12*	355*	14	Non-significant
	N_2O flux	% (g N $m^{-2}year^{-2}$)	-50*	9*	138*	15	Non-significant
	Net ecosystem CO_2 exchange	% (g C $m^{-2}year^{-2}$)	8*	70*	163*	9	Positive
Natural coastal wetlands vs. aquaculture ponds [Aquaculture ponds vs. natural coastal wetlands]	CH_4 flux	% (g C $m^{-2}year^{-2}$)	-99*	-93*	-43*	13	Positive
	N_2O flux	% (g N $m^{-2}year^{-2}$)	-33*	-7*	32*	10	Non-significant
	Net ecosystem CO_2 exchange	% (g C $m^{-2}year^{-2}$)	NA	NA	NA	NA	NA

Source: Tan et al. 2020. (*) results marked with an asterisk have been converted as explained in the text.

3.31.2 Effects on soil organic carbon

One synthesis paper (Xu et al. 2019a) analyses the effects on SOC of the conservation of wetlands compared with cultivated wetlands. In this synthesis paper, wetlands can be considered peats (wetlands that are peatland, or the soil had a high SOC content (> 300 g/kg) and a low bulk density (< 0.3 g/cm3), or the soil type was histosol), While for wetlands in general, including peatlands, the cultivation led to a significant decrease in SOC of 26.3% ($p < .001$)), for the category "rest of wetlands" or "no peat wetlands", non-significant effect was reported as compared to cultivated wetlands. However, as some of the primary studies also include peatlands and are analysed together, caution needs to be taken to interpret this result.

Compared to restored (by adding a sediment layer) coastal wetlands, the conservation of wetlands has a positive effect on carbon sequestration (i.e., increase of soil organic carbon), according to one synthesis paper (Ebbets et al. 2020). These authors compare restored wetlands with nearby reference natural wetlands. They differentiate an initial or "short" period for the first 15 years after wetland

[59] The change refers to absolute values of net ecosystem exchange, while the net ecosystem exchange for wetlands is negative (ln(-NEEi/-NEEc) where (NEEi/NEEc) is always positive). Therefore a positive change implies higher CO_2 sequestration.

recovery, and a long period, for after 15 years. They report that, "Based on mean RRs for the short recovery period of 15 years (...), soil OC (...) showed the greatest differences between restored and reference sites (68% (...)). For the long recovery period (greater than (...) 15 years), the absolute value of the mean RRs de creased for all response groups, indicating a greater similarity between restored and reference sites. (...) soil OC remained 36% lower (...) at restored sites than reference sites." Therefore, for the estimation of the benefits from wetland conservation as compared to restored wetlands, we deduct that there is an increase of +211% of SOC as compared to restored wetlands in the first 15 years after restoration, while in the longer term, natural wetlands still have, on average, 55% more SOC than restored wetlands

Table 95. Effect (% change) of peatland conservation on soil organic carbon. SE: Standard Error of the mean. Nc: number of pairwise comparisons.

Pairwise comparison	Population		Mean -SE	Mean	Mean + SE	Nc	Effect
Wetland conservation vs. restored wetlands	Coastal wetlands in the northern Gulf of Mexico	≤15 years	182	211	243	111	Positive
Wetland conservation vs. restored wetlands	Coastal wetlands in the northern Gulf of Mexico	>15 years	34	55	80		Positive
(Original comparison: restored vs. natural wetlands)	Coastal wetlands in the northern Gulf of Mexico	≤15 years	-71	-68	-64	111	Negative
(Original comparison: restored vs. natural wetlands)	Coastal wetlands in the northern Gulf of Mexico	>15 years	-45	-36	-25		Negative

Source: Ebbets et al. 2020

3.32 Wetland restoration

Restoration methods found in this review are classified into three main classes: management of invasive plant species, management of water regime and a general class with several restoration methods pooled together.

- Restoration by managing invasive plant species includes the use of chemical (herbicides) and mechanical (tilling, disking, mowing, crushing, cutting and uprooting) control methods, as well as burning, grazing and replacement of invasive plants with natives.
- Restoration by managing water regime includes the recovery of the duration, frequency and/or depth of inundation, waterlogging or environmental flow release. The creation of new wetlands is also considered here.

- When different restoration methods are analysed together in the reviewed synthesis papers, we report them as several restoration methods pooled together. This pool includes active revegetation, enhancement of in-wetland and structural heterogeneity, habitat creation, passive restoration, restoration of hydrological dynamics, restoration of water quality, addition of soil amendments or wildlife management (i.e. reintroduction of native species and/or elimination of exotic ones).

3.32.1 Effects on greenhouse gas emissions

Only one synthesis paper (Wails et al. 2021) has been found reporting a positive effect ($p \leq 0.1$) on abiotic carbon fluxes (CO2 and CH4 emissions-sequestration from soil and water) of removal of two common and pervasive invasive plant species (*Spartina alterniflora* or smooth cordgrass and *Phragmites australis* or common reed) in coastal wetlands (including swamps, mud flats, intertidal zones, salt marshes, estuaries, and similar marine, inshore habitats). The study compared non-invaded wetlands in original status pre-infestation with invaded wetlands, and also restored or managed wetlands (post-infestation) with invaded wetlands. Results correspond to *P. Australis* invasions. It was found that abiotic C fluxes (emissions) and C pools (stocks) measured from soil (no data was available for C fluxes from water) were higher in invaded systems relative to managed areas, though overall results were only marginally significant ($p \leq 0.1$). When examining C fluxes and C pools measured amongst biotic sources (i.e. invertebrates, microbes, and plants), no differences were detected between managed-invaded comparisons.

There are, however, some caveats that need to be taken into account. The metric used was Hedges' G, therefore the % change could not be calculated. Moreover, the synthesis paper includes under C fluxes the CO2 fluxes and CH4 fluxes, but no evidence was found of an aggregation of CO2 and CH4 results from the same experiments with a conversion of units between both (e.g. a 1% change in net CO2 emissions is not comparable to a 1% change in net CH4 emissions in terms of Global Warming Potential). Results are also not provided differentiated in CO2 fluxes and CH4 fluxes.

Table 96. Effect (Hedge's G) on the differences in Carbon fluxes of wetland conservation. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Comparison	GHG	Description	CI_LOW	Mean	CI_HIGH	Nc	Effect
Managed - Invaded	C flux (CO2 flux &CH4-flux)	Overall Species: *P. australis* Source: Soil	-1.1	-0.6	-0.1	26	Positive ($p \leq 0.1$)
			(Hedge's g)				

Source: Wails et al. 2021.

3.32.2 Effects on nutrient leaching and runoff

According to two synthesis papers retrieved, wetland restoration or wetland creation, compared to no wetlands or degraded wetlands, have a positive effect on nitrate and phosphorus leaching and run-off (i.e., decrease of nutrient leaching and run-off).

One synthesis paper (Carstensen et al. 2020) assesses the efficiency of the potential of free-water-surface constructed wetlands as a mitigation measure treating drainage water before it enters streams for reducing nitrogen and phosphorus losses from agricultural areas. The results showed that free-water-surface wetland significantly reduced the nitrate and phosphorus loss from drainage systems, but variation in the reported removal efficiencies was large. Free water surface significantly reduced nitrate loading by 41% within a range from – 8 to 63% (95% CI: 29%, 51%). The CI varied from 29 to 51%. The average total phosphorus removal efficiency was 33%, ranging from − 103% to 68% (95% CI: 19%, 47%).

Similarly, another synthesis paper (Land et al. 2016) found that created and restored wetlands (when several restoration methods pooled together) remain appropriate and potentially sustainable ecological engineering approaches for removing nutrients from treated wastewater and urban and agricultural runoff. The overall average summary effect represents a median total N removal ratio (R) of about 0.63. This means that the median total N load reduction, or removal efficiency, is 37 % (95 % CI: 29 to 44 %). Removal efficiency of total nitrogen in wetlands was positively correlated with average annual air temperature and negatively correlated with hydraulic loading rate. The total nitrogen removal rate was positively correlated with the inflow concentration and was also found to be positively correlated with hydraulic loading. The overall average summary effect represents a median total P removal ratio (R) of about 0.54, which can be recalculated to a median TP removal efficiency of 46 % (95 % CI: 37 to 55 %). The total phosphorus removal rate was positively correlated with concentration at inlet and hydraulic loading rate. In contrast, the total phosphorus removal rate was negatively correlated with wetland area, especially for wetlands smaller than 2*10^4 m2.

Table 97. Effect (% of change) in nutrients leaching and run-off of wetland creation or restoration as compared to no wetland. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Pairwise comparison	Source	Population	CI_HIGH	Mean	CI_LOW	Nc	Effect
N in inflow vs. N in outflow (wetland vs. no wetland)	Carstensen et al. 2020	Drainage systems with free water surface constructed wetlands	-51	-41	-29	42	Positive
	Land et al. 2016	Created and restored wetlands	-44	-37	-29	112	Positive
Total P in inflow vs. P in outflow (wetland vs. no wetland)	Carstensen et al. 2020	Drainage systems with free water surface constructed wetlands	-47	-33	-19	41	Positive
	Land et al. 2016	Created and restored wetlands	-55	-46	-37	146	Positive

Source: references in the second column

3.33 No irrigation

No-irrigation corresponds to rainfed (or non-irrigated) agriculture, where the soil water available to plants comes mainly from rainfall (Molden et al. 2007). Water-saving irrigation practices aim to satisfy crop water requirements while improving the timing and reliability of water deliveries to minimize water use (Perry et al. 2017). Conversely, non-water-saving irrigation practices are standard techniques of irrigation that supply water to the crop in order to increasing crop productivity, with no intention of minimising water use. They rely on various techniques such as flooding or use of sprinklers (European Parliamentary Research Service (EPRS) 2019).

3.33.1 Effects on greenhouse gas emissions

The selected synthesis paper (X. Gu et al. 2022) includes data reported in 34 primary studies from various regions of the world. The study focuses on rice, and quantifies the effects of different water management practices on CH4 emissions from paddy fields, compared to continuous long-term flooding. As can be observed in Table 98, no irrigation reduced CH4 emissions by -151.45% (95% CI: -232.3%, 70.0%)[60], compared to continuous long-term flooding.

Table 98. Effect (% change) of no irrigation on CH4 emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Comparator	CI_LOW	Mean	CI_HIGH	Nc	Effect
No irrigation	Continuous long-term flooding	-232.3	-151.15	-70	4	Positive

Source: X. Gu et al. 2022.

3.33.2 Effects on nutrient leaching and run-off

The selected synthesis paper (Sharma & Chaubey 2017) explores the effects of no irrigation on nitrate leaching in maize compared to non-water-saving irrigation practices. These results come from two peer reviewed journal articles discussing irrigation effect on nitrate loss. The studies related with irrigation effect were conducted in two types of soil categories; sandy loam soil and silty loam soil. The data represents identical soil types that had a similar range of precipitation and fertilisation rate. Nevertheless, these experiments were not conducted over the same location. No irrigation was found to significantly decrease nitrate leaching compared to non-water-saving irrigation practices. A typical effect size was not calculated, but median and relative quartile values are reported (Table 99) for different treatments and local conditions.

Table 99. Effect of no irrigation on nitrate leaching. Median and quartiles (Q1 and Q3) of difference in nitrate leaching difference between irrigation and no irrigation practices according to soil texture and fertilisation management.

Intervention	Comparator	Crop	Factor	Median (KgN/ha)	Q1 (KgN/ha)	Q3 (KgN/ha)	Effect
No irrigation	Non water-saving irrigation practices	Maize	Silty loam soil - irrigation –N fertiliser 248-296 kg/ha	117.11	136.91	106.39	Positive
			Silty loam soil - no irrigation – N fertiliser 248-296 kg/ha	47.01	60.21	39.59	
			Sandy soil - irrigation – N fertiliser 168-224 kg/ha	84.12	143.51	77.53	
			Sandy soil - no irrigation – N fertiliser 150-280 kg/ha	47.84	59.38	17.32	

Source: Sharma & Chaubey 2017.

[60] Dryland rice soils can be sinks of methane (Singh et al. 1998).

3.33.3 Effects on the nutrient balance

Results in the literature quantify the change in plant nitrogen use efficiency, the metric used being the partial factor productivity of applied nitrogen (i.e. the grain yield production per kilogram of nitrogen input in production).

The selected synthesis papers are B. Y. Liu et al. 2020 and B.-Y. Liu et al. 2021 for the effect of no irrigation in winter wheat and maize, respectively, compared to non-water-saving irrigation practices.

No irrigation had an overall negative effect on nitrogen use efficiency compared to non-water-saving irrigation practices, decreasing nitrogen partial factor productivity by -19.36% (95% CI: −20.8%, -17.4%) in winter wheat and by 20.0% (95% CI: −30.3%, −11.1%) in maize.

In winter wheat, the decrease in nitrogen partial factor productivity depended on the level of precipitations (i.e., stronger negative effect at lower precipitation levels), nitrogen content in the soil (i.e., stronger negative effect at lower nitrogen values), nitrogen application rate (i.e., stronger negative effect at intermediate nitrogen values), and irrigation amount and frequency (i.e., stronger negative effect at intermediate amount and frequency values).

Table 100. Effect (% change) of no irrigation vs. non water-saving irrigation practices on partial factor productivity of nitrogen. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Crop and source	Factor		CI_LOW	Mean	CI_HIGH	Nc	Effect
Maize *Source:* B.-Y. Liu et al. 2021	Overall		-11.14	-20.00	-30.29	29	Negative
Winter wheat *Source:* B. Y. Liu et al. 2020	Overall		-17.39	-19.36	-20.84	395	Negative
	mean annual precipitation (mm)	<=625	-20.05	-22.24	-24.62	202	Negative
		>625	-7.25	-10.79	-14.64	60	Negative
	seasonal precipitation (mm)	<140	-18.33	-21.88	-25.55	139	Negative
		>140	-15.88	-17.91	-19.95	231	Negative
	annual precipitation (mm)	<=550	-24.03	-29.92	-36.02	23	Negative
		>550	-9.41	-13.07	-16.81	16	Negative
	Initial soil nitrogen content (g kg-1)	<=0.85	-21.53	-26.28	-31.09	23	Negative
		>0.85	-18.77	-21.03	-23.28	189	Negative
	Nitrogen application rate(kg N ha-1)	<150	-9.99	-14.22	-18.53	79	Negative
		150-180	-22.12	-24.20	-26.34	31	Negative
		>180	-18.00	-20.04	-22.16	235	Negative
	Irrigation amount (mm)	<80	-13.25	-15.28	-17.23	127	Negative
		80-160	-20.24	-22.96	-25.76	162	Negative
		>160	-14.16	-18.01	-22.09	106	Negative
	Irrigation times	1	-11.21	-13.53	-16.01	116	Negative
		2	-20.46	-23.25	-26.04	157	Negative
		>2	-16.16	-20.14	-24.19	110	Negative

Source: references in the first column.

3.34 Water-saving irrigation practice in flooded lands

Water-saving irrigation practices aim to reduce water use in irrigated fields. In flooded lands, specifically in rice paddy fields, the common feature of water-saving irrigation techniques is to keep the field in a water-free or thin water layer condition at a certain growth stage, by introducing short term drainage of flooding water during the rice cultivation (Qiu et al. 2022; Yagi et al. 2020). Conventional continuous flooding corresponds to flooding rice paddies throughout the growing season (Climate and Clean Air Coalition 2014). In this review, different water-saving irrigation practices applied in flooded lands are compared to continuous flooding. Water-saving irrigation practices in flooded lands in this review include:

- water-saving irrigation practices that vary according to the number of drainages applied during the rice growing season (single or multiple), irrigation water amount, and irrigation schedule (e.g., alternate wet and dry irrigation, intermittent irrigation, mid-season drainage, controlled irrigation, moist irrigation, shallow irrigation and deep storage, thin and wet irrigation);
- drip irrigation;
- dry cultivation (non-flooded rice, but might be irritated);
- and drill seeding methods (rice is continuously flooded from the 3-6 leaf stage to final drain for harvest).

3.34.1 Effects on greenhouse gas emissions

The selected synthesis paper (X. Zhao et al. 2019) includes data reported in 32 individual of primary studies from various regions of the world. It quantifies the effects of different water-saving irrigation practices on CH4 and N2O emissions from paddy fields, compared to continuous flooding (Table 101).

Water-saving irrigation practices in rice paddies reduced area-scaled CH4 emissions by -51.5% (95% CI: -56.8%, -44.5%) compared to continuous flooding, and global warming potential (GWP, based on both CH4 and N2O emissions) by -39.1% (95% CI: -48.1%, -28.2%). However, water-saving irrigation practices in rice paddies increased area-scaled N2O emissions by 84.4% (95% CI: 53.2%, 128.7%) compared to continuous flooding. In addition, the changes in yield-scaled emissions are provided in the synthesis paper.

Table 101. Effect (% change) of water-saving irrigation practices in flooded lands on GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention vs. comparator		CI_LOW	Mean	CI_HIGH	Nc	Effect
Water-saving irrigation practices in flooded land vs. continuous flooding	CH4 emissions	-56.82	-51.50	-44.55	172	Positive
	N2O emissions	53.17	84.40	128.57	97	Negative
	Global warming potential (N2O and CH4)	-48.06	-39.10	-28.16	97	Positive

Source: X. Zhao et al. 2019.

3.34.2 Effects on soil organic carbon

The selected synthesis paper (Livsey et al. 2019) includes data from 12 primary studies conducted in Asia. It explores the impact of water-saving irrigation practices (alternate wet and dry or mid-season drainage) on soil organic carbon (SOC) concentration in rice paddies, compared to continuous flooding (Table 102). Authors found a consistently negative effect of alternate wet, dry, and mid-season drainage on SOC. The mean percentage change of SOC concentration was −5.2% (95% CI: −8.9%, −1.3%).

Table 102. Effect (% change) of water-saving irrigation practices in flooded lands on soil organic carbon (SOC) concentration. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Comparator	CI_LOW	Mean	CI_HIGH	Nc	Effect
Water-saving irrigation practices in flooded lands	Continuous flooding	−8.9	−5.2	−1.3	NA	Negative

Source: Livsey et al. 2019.

3.34.3 Effects on ammonia emissions

The selected synthesis paper (Qiu et al. 2022) includes data from 74 primary studies and explores the impact of different water-saving irrigation practices in rice paddies on ammonia (NH3) emission compared to continuous flooding (Table 103).

The effect of water-saving irrigation practices on NH3 emission was found to be either positive (i.e., decreased emission) or non-significant, depending on the type of practice: alternate wet and dry reduced NH3emission by 21.0% (95% CI: −26.9%, −14.7%), and controlled irrigation, instead, did not significantly influenced NH3 emission (−0.5%; 95% CI: −9.3%, 9.2%).

Table 103. Effect (% change) of water-saving irrigation practices in flooded lands on NH3 emission. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Comparator	Type of sub-practice	CI_LOW	Mean	CI_HIGH	Nc	Effect
Water-saving irrigation practices in flooded lands	Continuous flooding	Alternate wet and dry	-26.88	-21.04	-14.68	17	Positive
		Controlled irrigation	-9.29	-0.50	9.15	13	Non-significant

Source: Qiu et al. 2022.

3.34.4 Effects on the nutrient balance

The selected synthesis paper (Qiu et al. 2022) includes data from 74 primary studies and explores the impact of different water-saving irrigation practices in rice paddies on nitrogen use efficiency compared to continuous flooding (Table 104).

The effect of water-saving irrigation practices on nitrogen use efficiency was found to be either positive (i.e., increased use efficiency) or non-significant, depending on the type of practice:

— Both alternate wet and dry and controlled irrigation had a positive effect on nitrogen use efficiency by 0.47% (95% CI: 0.07%, −0.87%) and by 1.06% (95% CI: 0.46%, −1.66%), respectively.

— Dry cultivation did not significantly influence nitrogen use efficiency (−1.64%; 95% CI: −3.32%, 0.05%).

Table 104. Effect (% change) of water-saving irrigation practices in flooded lands on nitrogen use efficiency. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention vs. comparator	Type of sub-practice	CI_LOW	Mean	CI_HIGH	Nc	Effect
Water-saving irrigation practices in flooded lands vs. continuous flooding	Alternate wet and dry	0.07	0.47	0.87	27	Positive
	Controlled irrigation	0.46	1.06	1.66	13	Positive
	Dry cultivation	-3.32	-1.64	0.05	4	Non-significant

Source: Qiu et al. 2022.

3.34.5 Effects on nutrient leaching and run-off

The selected synthesis paper (Qiu et al. 2022) includes data from 74 primary studies and explores the impact of different water-saving irrigation practices in rice paddies on nitrogen leaching and runoff compared to continuous flooding (Table 105).

The effect of water-saving irrigation practices on nitrogen leaching and runoff was found to be either positive (i.e., reduced leaching and runoff) or non-significant, depending on the type of practice and on the metric:

— Alternate wet and dry had a positive effect on both nitrogen leaching and runoff and the mean percentage change was −14.3% (95% CI: −4.9%, −22.7%) and −33.2% (95% CI: −22.6%, −42.1%), respectively.

— Controlled irrigation did not significantly influence nitrogen runoff loss (−19.5%; 95% CI: −45.1%, 17.9%) but had a positive effect on nitrogen leaching loss (−49.2%; 95% CI: −52.9%, −45.1%).

— Moist irrigation reduced nitrogen runoff by 66.3% (95% CI: −84.7%, −25.3%).

— Shallow irrigation and deep storage did not significantly influence both nitrogen leaching and runoff.

— Thin and wet irrigation significantly decreased nitrogen leaching by 35.2% (95% CI: −48.8%, −17.8%).

Table 105. Effect (% change) of water-saving irrigation practices in flooded lands on N leaching and runoff. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention vs. comparator	Type of sub-practice	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
Water-saving irrigation practices in flooded lands vs. continuous flooding	Alternate wet and dry	N runoff	-42.1	-33.2	-22.6	21	Positive
		N leaching	-22.7	-14.3	-4.9	15	Positive
	Controlled irrigation	N runoff	-45.3	-19.5	17.9	6	Non-significant
		N leaching	-52.9	-49.2	-45.1	24	Positive
	Moist irrigation	N runoff	-84.7	-66.3	-25.3	3	Positive
		N runoff	-31.6	52.1	238.9	3	Non-significant

Intervention vs. comparator	Type of sub-practice	Metric	CI_LOW	Mean	CI_HIGH	Nc	Effect
	Shallow irrigation and deep storage	N leaching	-54.7	-27.4	16.4	3	Non-significant
	Thin and wet irrigation	N leaching	-48.8	-35.2	-17.8	5	Positive

Source: Qiu et al. 2022.

3.35 Water-saving irrigation practice in non-flooded lands

Water-saving irrigation practices in non-flooded lands aim to satisfy crop water requirements while improving the timing and reliability of water deliveries to minimize water use (Perry et al. 2017). Conventional non water-saving irrigation practices are standard techniques of irrigation that supply water to the crop in order to increase crop productivity, with no intention of minimizing water use (European Parliamentary Research Service (EPRS) 2019).

The water-saving irrigation practices in non-flooded lands included in this review are:

- aerated irrigation: irrigation practice characterised by the delivery of aerated water directly to the root zone by subsurface drip irrigation (Du, Niu, Gu, Zhang, Cui, et al. 2018);
- deficit irrigation: an irrigation practice whereby water supply is reduced below maximum levels and mild stress is allowed with minimal effects on yield (Kijne et al. 2003);
- optimised irrigation period: irrigation during crucial stages of the growing season (e.g. G. He et al. 2017; Quemada et al. 2013a),
- partial root-zone drying: a modified version of deficit irrigation where approximately half of the root system is in a dry state, while the remaining half is irrigated (e.g. Sonawane & Shrivastava 2022);
- reduced irrigation amount: reducing the amount of water used for irrigation (e.g. Du, Niu, Gu, Zhang, Cui, et al. 2018; G. He et al. 2017);
- drip irrigation: water is dripped onto the soil at very low rates (2-20 litres/hour) from a system of small diameter plastic pipes fitted with outlets called emitters or drippers; Water can be applied at the surface (surface drip irrigation) or underground (subsurface drip irrigation) (Brouwer et al. 1988);
- irrigation using water from other source: irrigation using reclaimed, brackish, saline, or treated waste water rather than fresh water(e.g. X. Liu et al. 2023).

These practices are compared either to non-water saving irrigation practices in non-flooded lands, such as sprinkler irrigation, furrow irrigation, centre pivot irrigation, full or supplemental irrigation; or to an alternative water-saving irrigation practice in non-flooded lands.

3.35.1 Effects on greenhouse gas emissions

The selected synthesis paper (Kuang et al. 2021) includes data reported in 74 primary studies from various regions of the world. The study focused on croplands, and it quantifies the effects of drip irrigation on N2O emissions, compared to furrow or sprinkler irrigation (Table 106). Water-saving irrigation practices (i.e., drip irrigation) reduced N2O emissions by 32% (95% CI: 20%, 42%) when compared to furrow irrigation and by 46% (95% CI: 30%, 58%) when compared to sprinkler irrigation.

Table 106. Effect (% change) of water-saving irrigation practices in non-flooded lands on GHG emissions. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Comparator	GHG	CI_LOW	Mean	CI_HIGH	Nc	Effect
Drip irrigation	Furrow irrigation	N2O emissions	20.00	32.00	42.00	68	Positive
	Sprinkler irrigation	N2O emissions	30.00	46.00	58.00	26	Positive

Source: Kuang et al. 2021.

3.35.2 Effects on the nutrient balance

The two selected synthesis papers report contrasting results depending on the subpractice, water reduction, and irrigation level used as comparator. According to Du, Niu, Gu, Zhang, & Cui 2018, sub-optimal water input has a negative effect (i.e. decrease of plant nitrogen uptake) on tomatoes. It decreased nitrogen use efficiency (NUE) by -18.7 % (95% CI: -21.7%,-15.7%). Sub-optimal water input decreased yield by 35.3% and water use efficiency by only 5%. The optimal inputs of water and N are the minimum inputs of water and N that produced the highest yield. Sub-optimal water input would be defined as the water inputs that were lower than the optimal water input. NUE (kg kg−1) was calculated as the tomato yield (kg ha−1) divided by the input of N fertiliser to the soil (kg ha−1) (Partial factor productivity of N).

J. Gu et al. 2020 compared reduced water to full irrigation amount in greenhouse vegetables in China. They reported a non-significant overall effect on NUE. For small/moderate water reductions (lower than 20% of the full irrigation water) nitrogen use efficiency increases (ranging from 76% to 157%). However, reducing water inputs had small effects on vegetable yields (ranging from −8% to 3%) and non-significant when the proportion of reduced water was less than 40%. This finding suggested that full irrigation water amounts were greater than the vegetable requirements.

Table 107. Effect (% change) of water-saving irrigation practices in non-flooded lands on nitrogen use efficiency. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention and	Sources	Population	Water reduction	CI_LOW	Mean	CI_HIGH	Nc	Effect
Sub-optimal water input vs. optimal water input	Du, Niu, Gu, Zhang, & Cui 2018	Tomatoes Global scale	Overall	-21.7	-18.7	-15.7	NA	Negative
Reduced irrigation amount vs. full irrigation water	J. Gu et al. 2020	Greenhouse vegetables China	Overall	-12.3	12.3	45.4	39	Non-significant
			20–40%	-31.5	-3.8	33.1	23	Non-significant
			<20%	5.4	76.2	194.6	12	Positive

Source: references in the second column.

3.35.3 Effects on nutrient leaching and run-off

The selected synthesis paper (Quemada et al. 2013b) includes data reported in 74 primary studies from various regions of the world. The study focused on cereals and vegetables, and it quantifies the effects of different non water-saving irrigation practices on nitrate leaching, compared to non-water-saving irrigation practices (i.e., irrigation following crop needs).

The effect of water-saving irrigation practices on nitrate leaching was found generally positive (i.e., decreased nitrate leaching). The size of the positive effect depended on the type of water-saving irrigation practice (Table 108,, adjusting water application to crop needs reduced nitrate leaching by 77.3% (95% CI: −87.9%, −61.0%); an improved irrigation schedule by 59.2% (95% CI: −70.0%, −46.2%); deficit irrigation by 51.2% (95% CI: −71.0%, −30.0%); and improved irrigation technologies by 23.2% (95% CI: −42.1%, −1.6%).

Table 108. Effect (% change) of water-saving irrigation practices in non-flooded lands on Nitrate leaching. CI_LOW: confidence interval bottom, CI_HIGH: confidence interval top, Nc: number of pairwise comparisons.

Intervention	Comparator	CI_LOW	Mean	CI_HIGH	Nc	Effect
Adjust water application to crop needs	Excessive irrigation	-87.86	-77.29	-61.02	24	Positive
Improved irrigation schedule	Suboptimal timing of application, same water amount	-69.97	-59.19	-46.17	25	Positive
Deficit irrigation	Crop needs	-70.98	-51.25	-29.90	16	Positive
Improved irrigation technologies	Less efficient system for water delivery, same water amount	-42.10	-23.19	-1.63	12	Positive

Source: Quemada et al. 2013b.

Conclusions

This document presents a comprehensive collection of numerical coefficients quantifying the environmental impacts of sustainable farming practices, focusing on greenhouse gas emissions, carbon sequestration, and nutrient losses.

Extracted from about 200 scholarly synthesis articles, primarily meta-analyses, these circa 1000 coefficients offer valuable insights into the effects of various farming practices, with relevance to assess the likely impacts of CAP interventions on the environment and the climate. By synthesizing existing meta-analyses and applying a systematic review framework, the study ensures a rigorous and transparent approach for informing policy. The collection gathers coefficients quantifying the effects of 35 agricultural management options, ranging from single farming practices to cropping systems or conservation and restoration actions. This valuable resource enables informed policy decisions at various levels, being targeted intervention design or assessing the agricultural policy environmental performance.

However, it is important to acknowledge some limitations of this approach. Firstly, as with every literature review, these analyses are performed at a specific moment in time and new papers can be published after our review. Secondly, the search strings used in the systematic review were tailored to specific farming practices, as defined within the document, but not targeted to particular impacts. In that sense, this document is not complete as it focuses on the impacts found at synthesis literature level for a selected set of practices.

Finally, it is important to note that this work is based on synthesis literature, which does not exclude the possibility of primary studies documenting links between specific farming practices and environmental impacts not accounted for at the synthesis level. There may be instances where primary studies investigate the relationship between specific farming practices and environmental impacts, but these linkages have not been included in the synthesis literature due to a lack of meta-analyses or systematic reviews addressing them.

Despite these limitations, this work offers a robust foundation assessing sustainable agricultural practices and contributing to the European Union's sustainability objectives; while further research and primary literature analysis is encouraged to ensure that policymakers, researchers, and practitioners have access to the most current and robust information.

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List of abbreviations and definitions

CAP	Common Agricultural Policy
CH4	Methane (CH_4)
CI	Confidence Interval
CI_LOW	Lower confidence interval
CI_HIGH	Upper confidence interval
DCD	Dcyandiamide (nitrification inhibitor)
DIN	Total dissolved inorganic nitrogen
DMI	Dry Matter Intake
DMPP	3,4-Dimethylpyrazole phosphate (nitrification inhibitor)
FP	Farming practice(s)
g	grams
GAEC	Good Agricultural and Environmental Conditions
GNB	Gross Nitrogen Balance
HQ	Hydroquinone (urease inhibitor)
GPB	Gross Phosphorus Balance
JRC	Joint Research Centre
kg	kilograms
LCA	Life Cycle Assessment
MS	Member Sates
N	Nitrogen
N2O	Nitrous oxide (N_2O)

NA	Not available
Nc	Number of samples or number of pairwise comparisons
NH4+-	Ammonium (NH_4^+)
NBTP	N-(n-butyl) thiophosphoric triamide (urease inhibitor)
NO3-	Nitrates (NO_3^-)
NR	Nitrogen recovery (metric of NUE, see Annex 1)
NU	Nitrogen uptake
NUE	Nitrogen use efficiency
PFPN	Partial factor productivity of nitrogen
PMEF	Performance and Monitoring Evaluation Framework
PPD	Phenyl-phosphoryl-diamine
RR	Response ratio
SE	Standard Error
SMD	Standardized mean difference
SMR	Statutory Management Requirements
UAA	Utilised Agricultural Area
yr	Year

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Annexes

Annex 1. Metrics considered for the analysis of the effects of farm practices on the nutrient balance

The Gross Nutrient Balance, including the Gross Nitrogen Balance and the Gross Phosphorus Balance, is a common indicator used in agriculture-related frameworks to estimate the surplus of nitrogen and phosphorus[61]. This surplus results in polluting and climate changing emissions, mainly: gaseous emissions of nitrogen in the form of NH3, NOx and N2O and losses of phosphorus and nitrogen to the water (leaching and runoff) that lead to eutrophication problems.

Even though it is a relevant policy relevant indicator[62], synthesis papers do not provide results on nutrient balance or surplus but on other parameters related to it. Therefore, additional considerations are needed to describe which results from available synthesis papers can be used to derive meaningful quantitative coefficients applicable to it.

Eurostat[63] defines the gross nutrient balance simply as the sum of nutrient input minus the sum of nutrient outputs.

The inputs include:

- Consumption of Fertilisers,
- Gross Input of Manure (nutrient excretion), and
- Other Inputs.

The outputs are:

- Removal of nutrients with the harvest of crops,
- Removal of nutrients through the harvest and grazing of fodder, and
- Crop Residues removed from the field.

While the retrieved synthesis papers do not explicitly report results in terms of gross nutrient balance, they report quantitative data on one or more of the components listed above and can therefore be informative. The main impacts retrieved in the systematic review of meta-analyses that are considered influencing the gross nutrient balances are:

– Nutrient excretion (nitrogen and phosphorus)
– Plant nutrient uptake (nitrogen use efficiency, nitrogen uptake, nitrogen partial factor productivity)

A net nitrogen balance is further calculated subtracting the nitrogenous emissions from the GNB. Several impacts from the meta-analysis review are linked to nitrogen losses in the environment: air

[61] Eurostat (2013). Nutrient Budgets – Methodology and Handbook. Version 1.02. Eurostat and OECD, Luxembourg.

[62] Communication from the Commission to the Council and the European Parliament - Development of agri-environmental indicators for monitoring the integration of environmental concerns into the common agricultural policy {SEC(2006) 1136} COM/2006/0508 final https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX%3A52006DC0508

[63] https://ec.europa.eu/eurostat/cache/metadata/en/aei_pr_gnb_esms.htm

pollutants (NH3 and NO emissions), GHG emissions (N2O emissions), nutrients leaching and run-off (nitrogen leaching and run-off), soil nutrients (soil total nitrogen).

A1.1 Nitrogen and phosphorus excretion

The most simple and straightforward way to calculate the change in gross nutrient balance is when changes in all the inputs and the outputs are known. Some synthesis papers, however, only provide the change in some N input or output. This does not allow to calculate the change in gross nutrient balance, but if the quantitative amount of the elements of a nutrient balance are known, it can allow to calculate its change.

In the case of nitrogen and phosphorus excretion, if a feeding technique results in lower (or higher) nutrient excretion from livestock, assuming that the other components of the gross nutrient balance do not change and that the change in manure excretion does not affect the manure trade in the region, then the decreases (or increases) in the gross nutrient surplus can be calculated.

A1.2 Plant nutrient uptake and nitrogen use efficiency

Some synthesis papers provide information on how farming practices affect the plant nutrient uptake and the efficiency of fertilisation using different metrics. From a qualitative point of view, an increase of the plant nitrogen uptake or of the nitrogen use efficiency will allow to determine a decrease of the nitrogen surplus in Gross N balance (assuming N inputs remain constant). However, most often they do not allow to directly quantify the change of the N balance. Some of the metrics used for N use efficiency are:

- Nitrogen uptake by the plants (NU): it is the amount of nitrogen in the crop biomass. It can be considered proportional to the N output with crop harvest in first approximation. This metric in absolute number (or in % if the output reference value is known) would be enough to quantify the change in gross nutrient balance, provided that the N inputs do not change.

- Nitrogen use efficiency (NUE): it is defined as total nitrogen outputs divided by total nitrogen inputs[64]. However, different definitions of NUE are used. It gives an indication of the relative utilization of nutrients applied to agricultural production system. The change of this metric affects directly the N uptake by the crop and indirectly the amount released in the environment.

Other two metrics related to nitrogen use efficiency were retrieved in the review of meta-analyses:

- Partial factor productivity of N (PFPN) or yield divided by N applied as fertiliser (used by Quemada et al. 2013a, Ding et al. 2018, B.-Y. Liu et al. 2021, Du, Niu, Gu, Zhang, & Cui 2018 and J. Gu et al. 2020): the changes in this metric can be assimilated to those of N uptake, assuming that the N fertilisation is constant and that the protein content of the crop yield is constant.

$$PFPN = {(Yield_f - Yield_0)}/{NF}$$

[64] Eurostat Statistics Explained. Archive:Agri-environmental indicator - gross nitrogen balance. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_gross_nitrogen_balance&oldid=328298#Nitrogen_use_efficiency

where $Yield_f$ is the crop yield with fertiliser, $Yield_0$ the non-fertilised crop yield and *NF* is the nitrogen fertiliser applied.

- Nitrogen fertiliser recovery (NR): N fertiliser recovery is calculated as the difference between N uptake in harvested crop with fertiliser minus N uptake without fertiliser divided by the fertiliser amount:

$$NR = {(NU_f - NU_0)}\Big/{NF}$$

where *NR* is the nitrogen fertiliser recovery, NU_f is the nitrogen uptake by the crop with fertiliser, NU_0 the N uptake by the non-fertilised crop and *NF* is the nitrogen fertiliser applied.

As the NUE was corrected based on the N uptake of a blank without fertiliser, it is also referred to as blank-corrected NUE (NUE(bc)).

Comparing the % changes produced by the farm practice with the two metrics, NU and NR, the following equivalences apply:

$$\%\ change\ with\ NU = {(NU_{f2} - NU_f)}\Big/{NU_f}$$

$$\%\ change\ with\ NR = {(NU_{f2} - NU_f)}\Big/{(NU_f - NU_0)}$$

Where NU_{f2} is the nitrogen uptake by the crop with fertiliser with the intervention or farm practice, NU_f is the nitrogen uptake by the crop with fertiliser used as comparator, and NU_0 the nitrogen uptake in the comparator without fertiliser (NU_0). Thus, the percentage change with NR would be a higher amount (as the denominator would be much lower) and is probably less useful for policy analysis and simulations as in most cases the nitrogen uptake without fertiliser (NU_0) is not known. For this reason, we will give priority to synthesis papers providing values for the N uptake metric.

A1.3 Nitrogen gaseous emissions, nitrogen leaching and soil accumulation

Some synthesis papers provide effects on some or all of the N losses, as reported above. The most important N losses, as estimated by the CAPRI model[65] at European scale for 2012 are: N leaching (32%); N accumulation in the soil (30%) and ammonia emissions (24%), then with a minor weight N runoff (8%), N2O emissions (4%) and other N gaseous emissions (2%).

Ammonia and nitrates are among the most important components, but their changes often compensate each other when a farm practice particularly targets one of them. Therefore, if the FP reduces ammonia emissions with e.g. nitrification inhibitors, it may result in an increase of nitrates and there would be only a small net impact on the N balance. By contrast, if the FP reduces nitrogen input by e.g. precision farming, it will reduce both, and, therefore, may have a relatively higher impact on the N balance. Therefore, we will assume that, if the farm practice results in a decrease in ammonia and nitrates leaching in a similar amount, this indicates that the practice decreases the gross nitrogen surplus.

[65] Approximate results from the CAPRI model (https://www.capri-model.org/), 2012 output results from (2021 CAPRI MTO baseline).

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